02

Imler-Jacquez, Sandra R -FS

From:	BRENT THOMPSON <dockrs21@gmail.com>
Sent:	Wednesday, December 11, 2019 12:02 PM
To:	FS-comments-southwestern-santafe
Subject:	Encino Vista Landscape Restoration Project

Hello,

I have some comments & concerns with aspects of the proposed project.

--I am cautiously optimistic that work within the footprint of proposed project can alleviate some concerns regarding possible catastrophic wildfire happening within the footprint. The prescription of prescribed fire is especially encouraging.

--In the Purpose & Need, it is stated that Roads are in poor condition and likely contributing to erosion in area; No new roads should be built to accomplish anything related to the project. Furthermore, any roads deemed necessary should be rehabilitated to accommodate increased traffic before any heavy equipment is on the roads. Reduction of roads within the project area, and forest in general, will thereby reduce siltation and infill of streams, as well as increase the quality of available water supply to the forest and adjoining communities.

--No new roads should be constructed. No information is given as to how many actuall miles of "temporary roads" other than '5-10 miles', nor is any information given as to how these potential roads would be reclaimed/rehabilitated when project is complete, nor what timeline this restoration would happen in. The ranger district should administratively close the roads within the project area during the winter time to lessen the impact that careless forest users cause to open roads during htis timeframe.

-- I applaud the FS for using alternative means ('small ruminant animals - GOATS) to control less desirable vegetation. However, no prescription is given for number, duration, time of year of animals to be used within the project area. These animals should be as minimally as possible in numbers and rotated on a schedule that does not allow them to denude the forest of all vegetation.

--Re: Conifer removal w/in meadows & aspen should be done by sawyers only and felled trees should be bucked up for removal by hand crews. No machinery should be allowed to trample within these precarious ecotones.

--No tree removal should occur during the MSO and goshawk breeding timeframes; especially within known PACs or other sensitive areas. Tree removal should be minimized during MBTA-related migratory birds also. Timber harvest within slopes >40% should not be allowed, nor within Wilderness or Study Areas. Comments regarding the guideline to 'Strive to retain trees greater than 24 inches is too vague and arbitrary! The current comments should be retained and possibly have wording added to retain conifer spp. at >21 in. DBH, and oak spp. at >15ft tall (or the like) as these types of tree tend to be taller than wider as they age.

-- possibly fencing off (or other means of prohibiting livestock/recreation users) areas of Riparian & meadows should be considered. Using natural vegetation within the area to construct fences should be considered as this can accomplish the fencing-out goal, as well as a means to remove woody biomass from the vertical structure. After a pre-determined time of allowing the riparian/meadows/aspen to regenerate, the "natural" fencing could be set afire as a prescribed/pile burn.

--The FS should consider implementing phase-out of all petroleum based lubricants for chainsaws and feller-bunchers, as they use large amounts of this ecologically degrading material. This project should also consider stipulating this requirement for any sub-contracted company that may be brought on to do vegetation removal work. Nowak et al (2019).
Nowak, P., Kucharska, K., & Kamiński, M. (2019). Ecological and Health Effects of Lubricant Oils Emitted into the Environment. *International journal of environmental research and public health*, *16*(16), 3002.

--No mention is made regarding soil disturbance during peak Jemez Mountains Salamander surface activity. There should be prescriptions added to account for this Endangered species.

--No best management practices mentioned regarding the minimization of amphibian Chytrid fungus spread due to heavy equipment use. a protocol regarding this should be implemented as well.

--no mention of eradication efforts related to invasive plant species. Please add prescriptive language regarding invasive/exotic plant species.

Thank you,
Brent.

03

Imler-Jacquez, Sandra R -FS

From: Dennis Smith <smithd00@hotmail.com>
Sent: Wednesday, December 11, 2019 12:56 PM
To: FS-comments-southwestern-santafe
Subject: Encino Vista Landscape Restoration Project
Attachments: encino vista comments.pdf; ATT00001.txt

Mr. Dennis Smith
PO Box 146
#159, CR 430
Coyote, NM 87012

Mr. Rich Nieto, District Ranger
Coyote Ranger District
HC-78, Box 1
Coyote, NM 87012

Mr. Nieto,

I am writing to provide comments on the Encino Vista Landscape Restoration Project Proposal dated November 2019. I request that as planning proceeds, more specific description with maps be provided to show where specific actions are planned. I would like to see where temporary roads are planned, where specific silvicultural treatments are planned and where areas planned to be burned without allowing silvicultural treatments are.

I am curious to hear how the USFS plans to use goats to thin Gamble oak. I would like to know why areas would be burned without allowing thinning beforehand. I am wary of the use of masticators to thin areas instead of allowing fuelwood harvesting by the public. Prior use of masticators was not well planned or executed and resulted in over-thinning of areas that should not have been thinned.

I request that any proposed actions planned that are in proximity to private land be discussed with the landowner prior to implementation. Simply publishing a document or having a public meeting are not sufficient in this. Direct contact with the affected property owner should be attempted if possible.

If you have questions or would like to discuss these comments, please contact me at 505-365-1409 or smithd00@hotmail.com.

Sincerely,

Dennis Smith

U.S.A. RANCH
usa.ranch@yahoo.com
Cornelio Salazar
Owner/Sole Proprietor

136 County Road 194-Cañones, NM 87516
PO Box 642 Abiquiu, NM 87510 (575) 638-5434

Dec. 16, 2019

RE: Rito Encino Comments

As community members from Cañones, NM. And Grazing Allotment owners on the Youngsville + Mesa del medio Allotments, and heirs of the Juan Bautista Valdez land Grants WE TOTALLY OPPOSE this Project for the following reasons;

(1) The Cañones creek watershed does not need Anymore diruptions from its natural state. erosion, ETC. ETC. Back in the 1960's The forest Service clearcut vast areas, without any NEPA or EIS Assesments as to any longterm effects this would have on the water shed. TO this day we are still seeing the effects of The unplanned logging operation. The silt and deeper cutacts in the meadows on the creek bottoms. This means some of our most valuable grazing has been lost. Then the Elk population is playing a huge factor in in all this areas.

(2) The NEPA And EIS studies Are far from realization because they do not take in effect the longterm effects of this Plan on the Community. The cultural traditional And social economic values will be Affected. The Pueblo ruins that are in this area "Mesita del Pueblo" the trail Into the mogote canyon is A National Trail All of these areas stand to be ruined with burn scars. not to mention the Added silt and erosion. The trout habitats in the Cañones Creek + Chihaureños Creek will be severely damaged.

(3) As Allotment holders the forest Service will defer our grazing due to burn scars and rehabilitation. The widening of roads to create fire breaks will scar the land. As there Are countless roads Aside from the main roads and four wheeler or Atv roads we do not need Anymore roads. The forest Service personel is Adamant that A control burn will not get away. but if it does we As the Agricultural Community have to deal with washed out bridges diversion dams erosion loss of Irrigation for our crops →

RE Rito Encino Dec 14, 2019

(4) last but not least. The forest Service say this will open up more area for grazing of elk. We do not need more area for the elk population, what we need is to Physically remove the elk herds out of our traditional grazing areas. Any benefit we do to the area the Elk are the first ones to take advantage. we have vested rights in these allotments and as Owners of the allotments and community members our concerns should be taken in to consideration. Also the name of this such project should be "Rito Encino" something we can relate to. this other way it seems foreign as can be.

Sincerely,

Camilo Salazar

Imler-Jacquez, Sandra R -FS

From:	Tom & Carlyn Jervis <Jervidae@cybermesa.com>
Sent:	Wednesday, December 18, 2019 4:20 PM
To:	FS-comments-southwestern-santafe
Cc:	Joanna Hatt
Subject:	Encino Vista Landscape Restoration Project
Attachments:	Scoping Encino Vista Restoration.pdf

Rich Nieto, District Ranger
Coyote Ranger District

Dear Mr. Nieto:

Attacherd pelase find the comments of Sangre de Cristo Audubon Society on the proposed Encino Vista Landscape Restoration Project.

Thank you foir the opportunity to comment on this project. Please keep us informed about the status and development of the project.

Sincerely,

Thomas Jervis, President
Sangre de Cristo Audubon Society
Santa Fe
505-988-1708

Sangre de Cristo Audubon Society

December 14, 2019

Rich Nieto, District Ranger
Coyote Ranger District
USDA Forest Service
Santa Fe National Forest
HC-78 Box 1
Coyote, NM 87012

Dear Mr. Nieto:

Many of Sangre de Cristo Audubon Society's 1,400 members watch birds and recreate in the Jemez Mountains and have a longstanding interest in the management of the Forest. We support your efforts to restore forested landscapes in the Encino Vista area and encourage broader scale forest restoration projects on your district.

We have reviewed the Purpose and Need Statement for the Encino vista restoration project and have a number of comments:

In general, we feel that thinning projects are often too concentrated on ***conditions on the Forest***. We believe that the emphasis should be on proper ***function*** of the ecosystem processes that maintain the resiliency of the forest system. Silvicultural prescriptions that manage for particular conditions, while appropriate for specific projects and treatments, will fail if the processes that support the ecosystem are not allowed to function. This is particularly true in light of changing climate. A desired condition which is appropriate for today's climate may not be appropriate in the future, but if ecological processes are intact, the forest will sustain its resiliency.

That said, we support the need to create conditions for the resilience of these forests and feel the proposed action is appropriate if managed properly. It would be helpful for the public to know what further specific actions are planned or contemplated to further this broader goal. Restoration is desirable, restoration of natural ecological function in the surrounding forest should be the real goal.

In carrying out this work, we would hope that you would also consider the following in developing the details of the project:

- Improving stream function to accommodate processes of stream meander, stable stream morphology, and floodplain development.
- Managing recreational access to minimize impacts on the function of watersheds, riparian and cultural areas, and wildlife.
- Managing grazing to minimize impacts on watersheds, rangeland, riparian and cultural areas, and wildlife.

We have a number of specific concerns with respect to the proposal:

1. We are primarily concerned about nesting birds. Direction for management and protection of migratory birds and their habitats within the continental United States exists in several forms.

- The Migratory Bird Treaty Act (MBTA) enacted in 1918 established Federal prohibition, unless permitted by regulations, to pursue, hunt, take, capture, kill any migratory bird, any part, nest, or egg of any such bird.
- Executive Order (EO) 13186 signed January 10, 2001 directed Federal agencies to avoid or minimize adverse impacts (to the extent practical) on migratory bird resources when conducting agency actions (among many items within the "Federal Agency Responsibilities" section of the EO).
- Pursuant to the EO, agencies were to develop Memorandum of Understanding (MOU) to strengthen and promote migratory bird conservation and collaboration with the U.S. Fish and Wildlife Service. The original 2008 MOU was extended and signed in 2016.
- Bald and Golden Eagle Protection Act (1940 as amended) protects eagles from actions of anyone (or entity) which would "take" eagles to the point of causing nest failure or reduce productivity (unless you or your entity have obtained a permit issued by the Secretary of the Interior).

There have not been specific USFS policies provided to direct migratory bird analyses into the NEPA process. However, the Southwestern Regional Office (R3 USFS) direction on migratory bird analysis is as follows:

1) Analyze effects to Species of Concern which are developed by the local (State) Partners In Flight Office with an emphasis on "high priority species".
2) Analyze effects of project action on Important Bird Areas (IBA's).
3) Analyze effects of project actions to important overwintering areas on USFS lands.

While we appreciate the attention paid to Mexican Spotted Owl (*Strix occidentalis lucida*) and Northern Goshawk (*Accipiter gentilis*) in this initial Proposed Action, we are troubled by the lack of attention paid to other species. The New Mexico Avian Conservation Partners (Partners in Flight) Birds of Conservation Concern (Primary threat list status SC1) that can be reasonably expected to be found in the project area include:

Flammulated Owl *Psiloscops flammeolus*
Grace's Warbler *Setophaga graciae*
Juniper Titmouse *Baeolophus ridgwayi*
Lewis' Woodpecker *Melanerpes lewisii*
Mexican Spotted Owl *Strix occidentalis lucida*
Pinyon Jay *Gymnorhinus cyanocephalus* (nesting colonies are of particular concern)
Virginia's Warbler *Leiothlypis virginiae*
Woodhouse's Scrub-Jay *Aphelocoma woodhouseii*
Scaled Quail *Callipepla squamata*

In the absence of comprehensive survey data of the area, these species should be presumed to exist in the project area and would therefore fall under R3 USFS item 1 above.

A project that cuts live trees or shrubs during the nesting season will result in the total failure of all nests in that vegetation. Inasmuch as most of the trees in an area will be cut during the restoration activities, compliance with R3 USFS direction suggests that restoration/thinning work should not occur during the peak of the nesting season, specifically April 15 through August. This is also the primary season for reproduction of all wildlife so this restriction will have benefits for mammalian, piscine, and herpetological fauna as well.

Even those trees and shrubs that are not cut will be disturbed, all resulting in reduced nesting success by many neotropical migrant songbirds. Quite apart from violation of the Migratory Bird Treaty Act, this is another example where managing for desired conditions can disrupt natural ecological processes (reproduction) that are essential to proper ecological function. Since the period also includes the peak of the fire season, avoiding the use of mechanical equipment in treatment areas during this period reduces the likelihood of ignition at a sensitive time.

We are concerned about using goats to manage Gambel's oak. If goats were deployed during the nesting season, they would eat up a lot of understory vegetation (important for shrub nesters) and possibly the nests themselves. The use of goats for shrub control should also be restricted to periods outside of the nesting season.

2. We are also concerned that proposed changes in the Forest Plan will relax standards for Mexican Spotted-owl management. While this may be in an effort to reduce fuel loads, it appears to reduce protection and indeed degrade habitat. For example, on Page 18 the proposal states: "Within PACs, combinations of thinning trees up to ***17.9 inches*** d.b.h., mechanical fuel treatment and prescribed fire should be used to abate fire risk to owl nest/roost habitats and improve habitat structure in select protected activity center outside the 100-acre core area." The prior language states: "Use combinations of thinning trees ***less than 9 inches*** in diameter, mechanical fuel treatment and prescribed fire to abate fire risk in the remainder of the selected protected activity center outside the 100-acre "no treatment" area." Larger trees are an essential component of Mexican Spotted-owl habitat. Removal of trees in the 9-17.9" size class will inevitably degrade habitat for Mexican Spotted-owls. There is no justification for this change.

3. We are concerned with soil compaction in treatment areas. Skidding of whole trees and/or collection of fuelwood by large numbers of individual pickup trucks can lead to excessive soil compaction in large areas. Soil compaction retards recovery of desirable grasses, forbs, and shrubs that are important for wildlife and can advance the establishment and spread of noxious weeds. While the use of small fuelwooders to perform restoration work has some social benefits, we would prefer to see the use of tracked feller-bunchers in conjunction with forwarding equipment to remove both the fuelwood and the slash (see concern below) from the treatment areas. A central location where fuelwood could be collected by individuals would result in a more controlled area of compaction that could be remediated at the end of the project.

4. We are worried that the use of many individual pickup trucks over a large area will result in the establishment of a large number of "social" roads that are difficult to obliterate. These roads tend to have a life of their own beyond the project lifetime and

result in continuing disturbance of wildlife, the poaching of remnant snags (see concern below), and attendant erosion.

5. We are concerned that the slash resulting from the thinning will remain on the ground for long periods of time prior to burning. Large quantities of green slash are likely to attract bark beetles, particularly in case of drought. This will lead to increased and unnecessary mortality in the remaining trees. If forwarding of the entire trees is not used as suggested above, we would encourage the piling and burning of slash as the project proceeds followed by a broadcast burn at the end of the project.

6. We are disturbed by the low number of snags in the Santa Fe National Forest generally. Snags are extremely important for many species of birds and other wildlife. There is a propensity on the part of fuelwooders to cut snags in the mistaken belief that they are "lightning rods" that ignite fires. Many snags also make particularly nice firewood on account of their pitch content. We urge your to make a concerted effort to conserve existing snags through education of the personnel involved and if necessary by the marking of snags and snag recruit trees. Also, we note that dense duff can lead to the death of otherwise healthy mature trees during broadcast burns. While this is one means of snag recruitment, we do not support it.

7. We are apprehensive about the treatment of old, large trees in this project. We urge you to be more specific as to the treatment of old, large trees in the analysis to ensure that these trees, which are essential for wildlife and future forest resilience, are protected.

8. We are concerned about the management of Piñon-Juniper savannas and woodlands. "Management of Piñon-Juniper vegetation has been hindered, especially where ecological restoration is a goal, by inadequate understanding of the variability in historical and modern ecosystem structure and disturbance processes that exists among the many different environmental contexts and floristic combinations of Piñon, Juniper and associated species...For example, "persistent woodlands" may still be within their historical range of variability, whereas degraded woodlands would be strong candidates for restoration to pre-1900 conditions."[1] "The first step in effective restoration is to identify and then modify the cause of degradation. If our land uses are found to be responsible for tree invasions or density increases, and if restoration is to have lasting value, it is essential to change the land uses that led to the need for restoration."[2] We strongly urge you approach the "restoration" of these woodlands and savannas with humility and care, cognizant of the centuries of land uses that have led to the conditions that are found on the Forest.

We thank you for the opportunity to comment on this project. Getting it right at the beginning is important.

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[1] Romme, W.H., and others. 2008. Historical and modern disturbance regimes, stand structures, and landscape dynamics in Piñon-Juniper vegetation of the western US. Colorado Forest Restoration Institute, Colorado State University, Fort Collins, CO.

[2] Baker, W.L., and D.J. Shinneman. 2004. Fire and restoration of piñon-juniper woodlands in the western United States: a review. *Forest Ecology and Management* 189: 1–21.

Sincerely,

Tom Jervis, President

Joanna Hatt, Conservation Chair

109 Daybreak
Santa Fe, NM 87507

06

Imler-Jacquez, Sandra R -FS

From:	Emmy Koponen <emmykoponen@gmail.com>
Sent:	Wednesday, December 18, 2019 9:48 PM
To:	FS-comments-southwestern-santafe
Subject:	Encino Vista Landscape Restoration Project

Good day. I am submitting my demand for an EIS to be held before granting permission for this large " restoration project". Never will sampling's survive and replace the existing biodiversity that their elders helped sustain.
I would like the FS to uphold the law. Thank you. Emmy Koponen

--
necessity is the mother of invention...Be Kind.

05

Imler-Jacquez, Sandra R -FS

From: Melissa-Roxanne Velasquez <mvelasqu.colostate.edu@gmail.com>
Sent: Wednesday, December 18, 2019 8:39 PM
To: FS-comments-southwestern-santafe
Subject: Encino Vista Landscape Restoration Project
Attachments: Juan Bautista Land Grant Advisory Group (working committee) Comment.pdf

Community Comment submitted on behalf of the Juan Bautista Valdez Land Grant Advisory Group (working committee).

JUAN BAUTISTA LAND GRANT ADVISORY GROUP

(WORKING COMMITTEE) supported by the Juan Bautista Land Grant, the Cañones, NM community base; and associated organizations

BOX 7 Cañones, NM 87516

JBLG ADVISORY GROUP COMMITTEE RE: VELASQUEZ, MELISSA ROXANNE

CC: JUAN BAUTISTA LAND GRANT

ADVISORY GROUP MEMBERS: BACA, JIMMY SANTIAGO (Chicano Poet and Writer); BOIES, MARYLOU (Land and Properties Representative); BRITT, JOHN & LESLIE, (Photographer and Private Landowner Representatives); BROTHERS (St. Michael's Monastery); DRENNAN, BRYAN (Private Landowner); FRANK, DARRYL (Private Landowner Representative); GALLEGOS, DENNIS (Acequia de Arriba Representative); GALLEGOS, INEZ (Community Water Association President); GARCIA, FINIANO (Buena Vista Ranch); GARCIA SR, PETER (Polvadera Creek Acequia and Mesa del Medio); LUCERO, LEVI (Private Landowner Representative); MARTINEZ, CY (Land Grant Representative); SALAZAR, CORNELIO (USA Ranch and Livestock Representative); SALAZAR, ELUID (Eluid's Art Design Construction); SALAZAR, LORENZO, (USA Ranch and Livestock Representative); SALAZAR, LUPITA (Querencia Creations and Agricultural Representative); SOLARIUS, NICHOLAS (Marine Corps Veterans Representative); VALDEZ, ANGELA (Community Representative); VALDEZ, ARTURO (Livestock and Mesa del Medio Grazing Allotment); VELASQUEZ, ISAIAH (Acequia Mayordomo and Mesa del Medio Grazing Allotment); VELASQUEZ, MELISSA ROXANNE (JBVLG Advisory Group Committee Lead); and VIGILSR, NORMAN (Livestock and Mesa del Medio Grazing Allotment)

ADVISORY GROUP ORGANIZATIONS: ACEQUIA DE ARRIBA; JUAN BAUTISTA LAND GRANT; POLVADERA ACEQUIA; MESA DEL MEDIO GRAZING ASSOCIATION; ST MICHAELS MONASTERY

Collaborated Comments from Community Meetings held December 2019

December 12, 2019

Via First Class and Electronic Mail to comments-southwestern-santafe@usda.gov subject line Encino Vista Landscape Restoration Project and USDA Forest Service Coyote Ranger District c/o Richard Nieto, District Ranger HC-78, Box 1 Coyote, NM 87012

Dear Richard Nieto:

The Juan Bautista Land Grant Advisory Group (working committee) appreciates the opportunity to provide comments on the Encino Vista Landscape Restoration Project for the Proposed Implementation phase prior to the drafting of the Environmental Analysis or Environmental Impact Statement. The Encino Vista proposed alternative calls for a proposed treatment area of about 128,400 acres on the Santa Fe National Forest, Coyote Ranger District. The purpose and need for action summarize that there is "a need to increase forest ecosystem sustainability and resiliency to insects, disease, and climate change by shifting forest composition and structure toward desired conditions with the historic (or natural) range of variability for each forest type." In addition, "a need to reduce the risk of uncharacteristic wildfire, to improve species habitat, and overall watershed conditions."[1]

Actions mentioned to be considered in detail within the draft document include:

Reduced stand densities;

Reintroduction of fire on the landscape;

Revitalization of meadows and aspen stand;

Promotion a diverse forest structure for a variety of wildlife species;

Improved watershed conditions; and

Efforts to significantly reduce the risk of catastrophic wildlife and its aftermath.[2]

The Cañones community has held several community meetings regarding this project, in addition to the regularly scheduled land grant meeting, in which this project was also discussed. A working committee was formed, and community comments have been taken from at least 40-50 members of the community. They represent various interests within the community, and all comments were considered as part of our working committee's position. For a more thorough response to this purpose and need for action, we would request for an extended deadline.

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[1] Encino Vista Landscape Restoration Project, Purpose and Need for Action and Proposed Action Coyote Ranger District, November 2019 (8-9).

[2] Encino Vista Landscape Restoration Project, Purpose and Need for Action and Proposed Action Coyote Ranger District, November 2019 (9).

Primarily, there is a need to state in this responsive document that there are several community members who oppose this project in its entirety. There are major concerns that the historical management by the Forest Service has led to its current ecological state. Many individuals have shared that many times these concerns have been brought to the US Forest Service in less publicized meetings, interactions, dealings with staff members, district rangers, and they have been largely ignored. There is a consensus that the public is not being involved in management decisions in a manner that reflects the dynamics of small rural communities; that documents of this nature should be publicized and distributed in the Spanish language, as many individuals only speak that language; that land grants and acequias have not been seen as organized governmental bodies that merit their comments to be incorporated into major federal actions, and/or in projects such as these; that individuals have seen Forest service staff turnover at alarming rates on this district, (especially district rangers); that this community was witness to the "Mesita del Pueblo" (Tsi' pinouinge), historically land grant, (and in the community's possession), aggressively acquired by the US Forest Service as we encounter trespassers on private lands (as a result of issuance of visitation permits); and that the level of trust needed to be maintained with managing a public resource has not been developed or cultivated among the people. Land-based rural communities around the district should be considered in the methods in which they communicate opposed to the bureaucratic language and literature so prevalent in federal agencies, in an effort to be able, and willing, to provide the level of feedback needed to respond in projects such as these.

The community meetings held in Cañones have provided us with a lot of comments and feedback on the Forest Service in general and this specific project. Prior to proceeding to proposed measures, we ask the following regarding this project:

1.) A level of trust be established between US Forest Service (Coyote Ranger District) and the Cañones community. The community was (for the most part) uninformed regarding this project. The first community notice received from many individuals and permittees came after the initial pre-scoping meeting. That being said, the consensus is that the leader of this initiative should work to develop a level of trust with the community before embarking on projects of this scope. The National Environmental Policy Act (NEPA) requires federal agencies to considers social, cultural and economic aspects. The community is interested in knowing if this project is truly one that intends to protect and sustain the forests' natural resources, associated fish and wildlife habitats, and the social, cultural, and economic practices of the surrounding rural communities, or if it's being driven by mandated acreage targets set forth by upper management within the federal agencies programs. Fire is the predominant tool mentioned as the most likely response to the above-mentioned purpose and need for action, surpassing all other methods, including most silvicultural methods that might be less invasive. Is there an example where a project such as this has yielded positive results, and, if so, can that be available for public review and comparison?

2.) After thoroughly reviewing the document and agreeing in consensus, the community would like to see this project complete an Environmental Impact Statement and not shortcut through an Environmental Assessment. The community agrees, that more attention to details is necessary as this project could have not only "significant" but "catastrophic" implications to its main watershed if not managed in a way conducive and considerate to this ecosystem and associated habitats for aquatic wildlife and water quality downstream. The community agrees that the threshold of a potential significant impact has been reached with the controversy of environmental effects, primarily to river ecosystems, but also to fish and wildlife species associated habitats and to the social, cultural, and economic aspects of this community and its livelihood.

3.) Scientific documents and data analyses of conditions that have merited the purpose and need for action should be cited or referenced in the documents at length. The effects by the proposed action on the environment and associated ecosystems- and on the aesthetic, historic, cultural, economic, social, and health- are of concern.

4.) A funding and contingency plan should be prepared in concert with the proposed project in case of catastrophic consequences. This contingency plan should be made available to the public for review. The management decisions made caring for this public resource should not be viewed as a least cost-effective analysis decision, but in a resulting decision that ensures the protection of the public resource and is amicable to all users of the forest system.

5.) Rangeland Grazing should be incorporated into a segment of the purpose and need for action as a "valid" multiple user of the forest with permitted allotment access. Range grazing pastures that will be improved, are to be identified and mapped, as well as discussed, in the document to ensure the opportunity to comment (livestock allotment permittees) on pasture improvements. As one of the larger cultural and economic interests of the area, the community is centered on farming and ranching as a land-based community with rights and access to national forest land grazing. This traditional practice of grazing forest lands occurred prior to forest service being established in the early 1900's. It is vital that rangeland grazing be at the forefront of this initiative.

6.) Consideration should be given to some of the oldest forms of governance in Cañones. These lands were historic land grants, awarded by the King of Spain to the Spanish families that settled here. The land grant is a functioning body of families whose origination and genealogy in the area dates back to the early Spanish settlement, and well before many of these traditional lands were acquired by the federal government.

7.) Cultural resource management surveys and associated protection should be identified as a "priority" within the landscape plan, considering the area is so culturally diverse and a main geographical area of ancestral Tewa lands, native American culture. Archaeological survey data should be thoroughly completed and made available to the public for review.

8.) Cañones Watershed should be granted protection as a "priority" and with the least significantly impactful methods. Allow areas to be cleaned out through wood hauling, controlled thinning in incremental phases and as smaller niches, rather than large acreage prescribed burn tracts. Less prescribed burning in the watershed (and pile and burn methods) so as to not contaminate water quality or increase sediment flow in streams.

9.) Include accountability and collaboration with local governmental bodies, the use of local resources, and local personnel and contractors, with simplified ways of participation, with oversight representation by local community in the performance of defined work scope.

10.) Water quality, of the area's streams and acequias should be considered as a "significant" impact and one that merits concern. Studies about prescribed fire impact on macroinvertebrate communities in select river systems, in conjunction with findings from studies of wildfire in Yellowstone National Park USA, have shown that as fire produces large quantities of fine debris and increases run-off of ground litter materials, it reduces taxonomic richness and diversity and increases dominance of Chironomidae and Baetis spp.[3] As a community, we appreciate the taxonomic biodiversity of our river's ecosystem and the multiple attributes such as the richness and abundance of species, the phylogenetic diversity and the presence of different evolutionary lineages, and the functional diversity with a variety of growth forms and resource use strategies. We appreciate this because it signifies to us the health conditions of the river ecosystem.

11.) Water quantity, in streams below, should be "strongly considered" in a manner to prevent severe flooding during regular rain events or natural flood events, especially with the potential for an increase in quantity of water through potentially thinned and open canopies. Some past studies have shown low-intensity prescribed fires have little or no influence on stream flows (D. H. Van Lear, Douglass, Cox, & Augspurger, 1985), but many other studies also observed an increase in stream run-off (Ursic, 1970; Schindler et al., 1980). "The influence of fire on hydrology can be expressed indirectly by the

[3] L. E. Brown et al., 2015; L. E. Brown, Johnston, Palmer, Aspray, & Holden, 2013; Minshall et al., 2001.

changes of vegetation, ground cover, soil and environmental factors that affect water cycles. In general, as prescribed fires intensify and consume more of the forest floor, there are "effects" on stream hydrology similar to wildfires or forest harvesting" (Baker, 1988; Shuren, 2003).

12.) Prepare a detailed timeline (in specification and in increments) available to the public for review. The timeline should include phases to identify and measure impacts from areas within the project and effects outside of project boundaries). One study on the "Effects of Prescribed Forest Fire on Water Quality and Aquatic Biota," concluded that timing was essential; for example, one month after a prescribed burn, and during a dry period with almost no precipitation, the first high intensity rainfalls post-burn introduced elevated DOC levels and nutrients into the downstream water and "adversely" affected water quality. The timeline scope needs to have a planned (proactive not reactive) set of strategies that would address any scope creep and its impact on project and stakeholders.

13.) A careful consideration should be given to Threatened and Endangered Species currently listed, once listed, or species of concern within the project boundaries, i.e.; Mexican Spotted Owl, Northern Goshawk, Rio Grande Cutthroat trout, and/or other fish and wildlife species who might "potentially" share the same habitat space.

Proposed Action:

In the proposed action you mention 77,106 acres of broadcast and pile burning and subsequent maintenance burning in conjunction with or independent of uneven and even aged silvicultural system methods; (Group and or individual tree selection on up to 39,720 acres and prescribed burning and a combination of intermediate thinning and or pre-commercial thinning, and or prescribed burning on up to 26,480 acres); Un-even and even aged silvicultural system methods in conjunction with broadcast and pile burning on up to 22,200 acres; prescribed burning without prior silviculture treatments on up to 10,907 acres with additional maintenance burning occurring on a 5-20 year rotation for all prescribed burning areas; an allocation approximately of 22,225 acres for old growth characteristics; construction of 5 to 10 miles of temporary roads, road infrastructure improvement, utilization of small ruminant animals to control Gamble oak, amending the Santa Fe National Forest Plan for the MSO habitat, and amending the 1987 Santa Fe National Forest Plan for northern goshawk.[4]

[4] "Encino Vista Landscape...", 10.

Primary Concerns:

1.) **Timeline of the above actions.** Considering that the percentage of what will be prescribed burn, and the percentage non prescribed burn cannot be determined until more than likely MSO and Northern Goshawk survey data has been completed; there is no timeline of activities in the purpose and need for action for the project at length.
2.) **Specifications of the above actions.** It is not clear the exact acreage and in what particular areas the prescribed burning will take place. Is there data on the actual location activities of prescribed burning. Where have current conditions departed from the desired conditions? Is there survey data on these actual locations?
3.) **Rationale behind prescribed burning without prior silvicultural treatments on up to 10,907.00.** What is the rationale behind prescribed burning with no exact delineation? Will less impactful silvicultural treatments be done prior to burning?
4.) **Cañones watershed.** Prescribed burning near aquatic streams has been shown in some studies to have detrimental effects to the water quality and aquatic wildlife. The Cañones creek is key habitat for the Rio Grande Cutthroat trout which has in the recent decade finally been de-listed. The community is concerned about ash-filled streams from prescribed burns, loss of sediments, and impacts on water quality. According to some studies mentioned earlier, utilizing fire to treat near riparian areas can potentially be detrimental to the benthic and macroinvertebrate communities. Some studies have shown prescribed burning resulting in streams that have significantly lower taxonomic richness and diversity. There is no current hydrological data cited with a reference and approach to river ecosystem protection. The Cañones watershed draws water from a sizeable area, and during regular rain events, the Cañones creek and Polvadera creeks can triple in size. The incremental phasing of the project is vital to control unintended higher velocity flow that can increase sediment in the water and detrimental erosion.
5.) **Temporary roads map available to the public.** Is it possible to see the exact road maps that will be constructed to facilitate this project?
6.) **Road Improvement in project area**. There are many existing Forest Roads in need of repair. Will these roads be identified in this document for maintenance and improvement?
7.) **Ruminant Animals Containment Plan.** Are you planning on enclosure facilities; how will ruminant animals be controlled on the landscape? Are there examples from other projects where this has been a successful tool?
8.) **MSO Amendment**. What survey analysis has been completed up to this point for Mexican Spotted Owl recovery? Have locations of habitat been mapped within the boundaries of the proposed Encino Landscape Restoration Plan? Do you have current survey data available to the public as an addition to this proposed project? If so, are there current habitat mappings? We refer to page 15 of the document Purpose and Need for Action. Will these guidelines be adhered to in the proposed project? Highest densities of Mexican spotted owls have been shown to occur in mixed-conifer forests

that have experienced minimal human disturbance. Will areas within these project boundaries be reserved for this protection.

9.) In response to amend the **1987 Santa Fe National Forest Plan** to add clarifying language for northern goshawk management with current survey analysis available for public review. Owl and goshawk breeding time occurs during the summer months (March 1 to August 31). Do these intensity burns affect habitat and or breeding cycles for T&E Species? More existing data on wildlife species should be reflected prior to implementation of this project. Currently there is a lack of field sampling and current data analysis. Will this data be available prior to project implementation and available for public review?

10.) The Forest Service acknowledges in the purpose and need for action that fuelwood is an important resource. Fuelwood should be available to the communities prior to and during project implementation free of charge to accelerate the cleaning of the forest dead and down resources. It is a benefit to small communities with low income residents and assists in the cleaning of potential fire hazards. Local people should be hired in forest activities, and the process in which they participate needs to be simplified to encourage a greater pool of candidates. An example might be collaborating with local loggers and logging operations to mark or fell dangerous trees and pile them to be harvested by community members for fire wood or timber or allotting both community members and or loggers certain plots to harvest valuable timber.

Proposed Forest Plan Amendment:

Sufficient monitoring data on aquatic, fish, and wildlife species to provide an adequate comment.

Concluding Remarks:

The community is located within the Northern Rio Grande National Heritage Area (NRGNHA), which "is a place recognized by the United States Congress for its contribution to the American experience." Our region is defined by how communities interact with their environments; our land-based communities live in communion with our ecosystems. For many of us who live in this region, our food, water, if not a good portion of our incomes come from working the land, or harvesting from what used to be our common lands in the mountains. Primarily for rural communities, any issues that could arise from mismanagement of this project that can potentially negatively impact the ecosystem will have a socioeconomic impact on our community as a whole.

The community is a water and land-based culture and, yes, we can agree, we are always concerned about catastrophic events, such as major wildfire. But loss of biodiversity in fire, either through prescribed plans, or one that is naturally caused, could potentially have devastating effects on the entire riparian ecosystem, and all those that depend on it. The livelihood of many community members is tied to this ecosystem. We rely on a clean and healthy river for our acequias, gardens, and livestock, as well as appreciate the natural processes that occur within these ecosystems. We advocate for an environmental impact statement that is considerate of this specific culturally historic geographical area rich in fish and wildlife diversity; which protects the historical and cultural traditions of the community and its people; which considers the community as a collaborative decision maker prior to project implementation.

As each decade approaches, it seems that the approach to management of the public resource shifts. Fire suppression has been a traditional tool, and to reintroduce fire to the landscape, (and its associated implementation methods), is not just a programmatic change at the forest service level, it is a psychological change that involves acceptance by the community, who accesses the resource. There are always risks associated with the management of public resources. Community participation in the decision-making process and a collaborative approach to planning can work to strengthen the relationship and lead to a more neutral understanding between the forest service and the rural community. However, the community does not find that the level of trust is sufficient enough, at this point in time. We take the position of asking the Forest Service to conduct a more detailed review of the project, including timelines, and significant impacts that could potentially arise, prior to proceeding.

P.O. Box 87

Abiquiu, NM 87510

December 19, 2019

Mr. Rich Nieto

Coyote Ranger District

HC-78 Box 1

Coyote, NM 87012

Dear Mr. Nieto,

I am concerned with the effects of the Rito Encino Project proposal by the Santa Fe National Forest. I am a landowner in Canones and concerned that this proposal will have a negative impact on the land along the Canones Creek.

Sincerely,

Delfinia Gallegos

Delfinia Gallegos

Imler-Jacquez, Sandra R -FS

From: Jonathan Glass <jonathan@courseofhumanevents.org>
Sent: Thursday, December 19, 2019 6:51 PM
To: FS-comments-southwestern-santafe
Subject: Encino Vista Landscape Restoration Project

December 19, 2019

Via email to: *comments-southwestern-santafe@usda.gov*

Ref: comments on Encino Vista Landscape Restoration Project

Dear Santa Fe National Forest,

The Encino Vista project proposal is an unwelcome surprise. Please consider the following:

> Encino Vista is the largest project ever proposed by SFNF. Burn smoke from the proposed action will adversely affect the health of people in communities up to hundreds of miles away.
>
> Announcement of the project and its scoping period did not even make it to SFNF's main news release email list used for far smaller projects and far more minor news.
>
> There has been no mention of the Encino Vista Project in newspapers or other news, including Santa Fe National Forest's own online news feed.
>
> At least several people and organizations well known by SFNF likely not to be in accord with a project such as Encino Vista were not notified about the project in advance of scoping.
>
> Roughly a million people live within about a hundred miles of the up to hundred-plus thousand acres proposed to be burned.
>
> The Encino Vista is a major federal project which will certainly affect the quality of the human environment - particularly the air we all breathe. As such, it feels unconscionable for SFNF not to prepare an Environmental Impact Statement for public consideration before proceeding.
>
> Burning millions of tons of vegetation across tens of thousands of acres will be irreversible in the lifetimes of many or all New Mexicans and be a significant contributor to the state's worsening air pollution and our planet's climate crisis. Such a project should be undertaken only with great care prior analysis, if at all. The scoping document's stated purpose ignores the highly questionable scientific basis of fuels treatments on forests and presents nothing resembling a cost-benefit analysis that the public deserves before contemplating such an impactful project.

Please put the Encino Vista project on hold pending release of an Environmental Impact Statement which demonstrates a clear, positive cost-benefit of the undertaking backed by the best available science.

Please also make this and all other public comments submitted in relation to this project publicly available in unredacted form as soon as possible.

Thank you for your attention.

Sincerely,

Jonathan Glass

422 Abeyta St
Santa Fe, NM, 87505
(505) 227-8473

Imler-Jacquez, Sandra R -FS

From:	Lupita Salazar <justlupitasalazar@gmail.com>
Sent:	Thursday, December 19, 2019 9:46 PM
To:	FS-comments-southwestern-santafe
Subject:	Encino Vista Landscape Restoration Project

12/19/19

To Whom it May Concern,

My name is Ana Maria Salazar. I am a young farmer and run a local non-profit for youth in art and agriculture. I am from and currently reside in Cañones, one of the villages that will be very much affected by the actions taken in the proposed Encino Vista Landscape Restoration Project. I am writing to oppose the Encino Vista Landscape Restoration Project, in the hopes that you will take my concerns and those of others in my community into consideration as you propose actions.

I am concerned that there is not a defined proposed start date for the project, or a general timeline in which the project is to be carried out in the proposal. I am advocating for a longer timeline to properly study and attain the data necessary to properly treat the diverse landscape mapped out in the proposal. If the forest service could work on small land parcels bit by bit, not only would it be safer to our watershed, but it would also protect the diversity of the forest. As cutting and clearing diminishes diversity for a certain time period, working to restore smaller plots preserves the diversity of the rest of the forest. The surrounding older forests protect the particular restoration plot as it regenerates after treatment with adequate vegetation to protect the soil from erosion, and help hold water.

The proposed area of "treatment," especially the canyons that our community is most concerned about is prime habitat for a variety of species, yet there is no mention in the Restoration Project of any surveys to understand exactly what species are currently living and breeding in the "treatment" area, or how they would be protected, as per the Forest Service Plan and its proposed amendments if they are endangered species. There are various species that call this ecosystem home, such as elk, bear, various birds, and reptiles, is there data on what creatures live and coexist here with each other and humans and livestock? I would like to request more study to take into consideration how the actions of the forest service could impact our land based community in the long term both economically and socially, as our fate is tied to that of the lands and waters that we rely on.

Our particular watershed draws water from a large area, when we experience heavy rains, our river can triple in size. If our watershed is further damaged, we could experience river flows of higher velocities, which will cause detrimental erosion. As well as potentially bringing down ash from the fires and killing agricultural crops, as occurred with the Polvadera fire on the other side of our canyon. This particular project is a larger area, and closer to our village, and the canyons steeper, so any negative effects of the treatment could prove devastating to our community. We are concerned over the quality of water that can be compromised, our natural springs that could potentially be damaged, and our lands, animals, and crops that could be wiped out in the incident of a devastating flood. What practices is the Forest Service willing to enact to protect our watershed from potential damage? And how can we hold them accountable? Without our water we cannot live here, threatening our watershed threatens our existence.

As our irrigation water source, the acequias are the veins that have brought life to our village and ranches for hundreds of years. Inadequate protections for the top of the watershed could result in destruction of our acequias and lack of water to provide our crops for our animals and families.

I am concerned about the effects of the proposed treatment, especially burning on our river water quality. What efforts can be enacted to protect our waters from flooding due to lack of vegetation, excessive ash in our water from burning, and intense erosion resulting in sediment in our river and acequias, and eventually our fields and gardens? What compensation is available if our crops are ruined due to flooding, or if there are losses of homes, barns, or any other structures?

There was no mention about any hydrological study in the proposal, or studies to see what kinds of plant and animal species reside in these ecosystems, and how they are to be protected. The previously endangered cutthroat trout is one of the residents of our rivers, and the residue from fire can harm their habitat as well as the other species that reside in the rivers, and downstream in the acequias. There is no mention in the plan of how our rivers will be protected from the residues of the proposed treatments.

I am in agreement that there needs to be treatment to clean certain areas of our watershed yet I oppose the Encino Vista Landscape Restoration Project. I advocate for more thorough study of the area, ecosystems, and hydrology to help make a more informed plan. I also hope you take into consideration the devastation that fire can have on our entire watershed and community when it is used in treatment, and use it only as a last resort, rather than a go-to. I write so my own concerns can be heard and I also write in solidarity and agreement with the comments and suggestions of the Juan Bautista Land Grant Advisory Group.

Gracias

~Ana Maria Salazar

Imler-Jacquez, Sandra R -FS

From: Norman Vigil <norman.vigilsr@outlook.com>
Sent: Thursday, December 19, 2019 7:49 AM
To: Imler-Jacquez, Sandra R -FS
Subject: Encino Vista Landscape Restoration Project #54965

Follow Up Flag: Follow up
Flag Status: Flagged

Sandra,

The following comments are being submitted to the proposed Encino Vista Restoration Project proposal from the Coyote Ranger District.

After attending the community meetings at Canoñes, it was apparent that the major obstacle to any watershed improvement plans is the lack of trust between the USFS and community members. There are many reasons but mostly this has occurred over decades of interaction or lack off. In this particular proposal the local acequias and land grant were not notified formally. This indicates a basic lack of knowledge of both entities as major components of the community. Hopefully this will change in the future but the damage has been done.

1. A great deal of emphasizes is being placed on wildlife species as indicated by the amount of references and length of discussion in the proposal document at the expense of the other land uses (livestock grazing, Acequias, local customs and culture).
2. Support of the proposed treatment will only come when details of the project are outlined. There is a genuine concern with reintroduction of fire into an ecosystem in which fire has been suppressed for decades. A recommendation is to move forward with small treatments showing success before a large scale project is proposed. This approach might build the trust and confidence needed as opportunities arise in the future. Another recommendation is use silviculture treatments until the forestlands are in a "natural state" to allow fire as a tool.
3. Contingency plans should be part of the planning process in the event a prescribed burn gets out of control which could have a major impact on the grazing operations not to mention other land uses.
4. The USFS should clearly commit financial resources for implementation of the plan. Past performance indicates a lack of financial resources to many treatment plans on the shelf. At a minimum the financial plan should identify sources of funding or the approach to funding the proposed treatments. The current staff at all levels indicate a willingness for treatment but past history indicates a continual change in personnel with differing approaches and commitment. Almost impossible to build trust with a revolving change in personnel. Again, you started off on the wrong foot with the community of Canoñes.

Submitted by Norman Vigil. Mesa del Medio permittee.

Norman Vigil
575-684-0042
505-967-8760
Norman.vigilsr@outlook.com
P.O. Box 623
Canjilon NM, 87515

Imler-Jacquez, Sandra R -FS

From: Sam Hitt <sam@wildwatershed.org>
Sent: Thursday, December 19, 2019 9:18 PM
To: FS-comments-southwestern-santafe
Subject: Encino Vista Landscape Restoration Project
Attachments: Scoping Comments_EnciinoVista_181219.pdf

Part one comments attached

I want you to act as if your house is on fire.
Because it is.
Greta Thunberg

Sam Hitt
WILD WATERSHED
48 Old Galisteo Way
Santa Fe, NM 87508
505-438-1057
sam@wildwatershed.org

December 19, 2019

Rich Nieto, District Ranger
USDA Forest Service
Santa Fe National Forest
Coyote Ranger District
HC-78, Box 1
Coyote, NM 87012

RE: comments to the **Encino Vista Landscape Restoration Project: Purpose and Need and Proposed Action**

submitted electronically via comments-southwestern-santafe@usda.gov

Dear Mr. Nieto,

The following are comments to the Encino Vista Landscape Restoration Project: Purpose and Need and Proposed Action (project) issued November 19, 2019 and located on the Coyote Ranger District, Santa Fe National Forest (SFNF). Please accept these comments on behalf of the Santa Fe Forest Coalition, Wild Watershed and the Center for Biological Diversity. The 30-day comments period ends December 19, 2019 making these comments timely.

The **Santa Fe Forest Coalition** is an all volunteer nonprofit that educates the public, the media and policy makers on critical issues concerning forest and wildlife preservation in New Mexico. Member groups include Wild Watershed, Once a Forest, Multiple Chemical Sensitivities Taskforce, La Cueva Guardians, Tree Huggers Santa Fe and others. **Wild Watershed** is an all volunteer organization focused on aquatic conservation and wilderness preservation. These comments supplement and are in addition to other public comments that these groups may submit.

The **Center for Biological Diversity** is a non-profit environmental organization with over 61,000 members, and 1.6 million activist-supporters nationwide who value wilderness, biodiversity, old growth forests, and the threatened and endangered species which occur on America's spectacular public lands and waters. Many of the Center's members and supporters frequently use and enjoy the spectacular landscapes of the Santa Fe National Forest for recreation, sustenance, nature study, and spiritual renewal.

These comments are constrained by the minimal 30-day comment period. The SFNF has offered no justification for limiting public involvement in scoping to such a degree. Due to lack of time important issues may have been overlooked and the full implication of others unrealized. Therefore, these comments are filed under protest.

1. LEGAL BACKGROUND

A. NEPA OBLIGATIONS

Under the National Environmental Policy Act (NEPA), every federal agency that takes a major federal action "significantly affecting the quality of the human environment" is required to create a detailed statement discussing: (i) the environmental impact of the proposed action; (ii) any adverse environmental impacts that cannot be avoided; (iii) alternatives to the proposed action; (iv) the relationship between the short-term uses of man's environment and the maintenance and enhancement of long-term productivity; and (iv) any irreversible and irretrievable commitments of resources which would be involve in the proposed action should it be implemented.[1] When, as here, any significant environmental impacts may result from the proposed action, the agency <u>must</u> complete a meticulous environmental impact statement (EIS).[2]

B. NFMA OBLIGATIONS

The National Forest Management Act (NFMA) imposes a substantive duty on the Forest Service to "provide for diversity of plant and animal communities . . ." 16 U.S.C. § 1604(g)(3)(B). This statutory intent is attained in NFMA's 2005 implementing regulations that guide implementation of the 1987 SFNF Plan. It requires the Forest Service to:

> document how the best available science was taken into account in the planning process; evaluate and disclose substantial uncertainties in that science; evaluate and disclose substantial risks associated with plan components based on that science and document that the science was appropriately interpreted and applied.

36 C.F.R. § 219.11(a)(1)-(4). The Forest Service may satisfy the 2005 regulations' requirements through the use of "independent peer review, a science advisory panel, or other review methods to evaluate the consideration of science in the planning process." *Id.* § 219.11(b).

2. THE PROJECT FAILS TO DISCLOSE AND ANALYZE CLIMATE IMPACTS

The project fails to disclose and analyze the important role forests and woodlands play in sequestering atmospheric carbon. It is well established that removing carbon dioxide from the atmosphere is crucial to stabilizing the rapidly warming climate. The failure to discuss project impacts to the climate undermines the public participation goals of NEPA and deprives the decision-maker of necessary information, contrary to 40 C.F.R. §§ 1502.1, 1503.1-.4, 1505.2, and 1506.6.

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[1] 42 U.S.C. § 4332(2)(C)(i)–(v).

[2] *Sierra Club v. Van Antwerp*, 661 F.3d 1147, 1153 (D.C. Cir. 2011) (citing *Sierra Club v. Peterson*, 717 F.2d 1409, 1415 (D.C. Cir. 1983)); *see also* 40 C.F.R. §§ 1508.11, 1508.27.

The 2018 Intergovernmental Panel on Climate Change's (IPCC) special report[3] found that the single biggest source of carbon emissions from the land use sector is global deforestation and forest degradation. In addition to identifying the extreme urgency of achieving significant emissions reductions by 2030, the IPCC report highlights the important role of land conservation. Increased forest protection could account for approximately *half* of the climate change mitigation needed to keep global temperature rise to 1.5 degrees Celsius or less.[4]

Unfortunately the SFNF has consistently ignored these findings. The scoping letter for the massive Encino Vista project fails to even mention the impacts of removing millions of trees on the rapidly warming and drying climate of the southwest.

Disregarding these potentially dire impacts is negligent in light of the following: 1) more logging occurs in U.S. forests than in any other nation in the world, making the U.S. the largest global problem in terms of carbon emissions from logging;[5] 2) forests and other natural systems if protected in the U.S. could offset as much as 21% of total U.S. greenhouse gas emissions;[6] and 3) If all tree cutting ceased on national forests, the rate of carbon storage on those lands would increased by an average of 30 percent over the next five decades.[7]

Federal lands, including national forests, must be quickly mobilize to preserve carbon stocks.[8] Urgently needed is a shift in federal subsidies away from logging/thinning/burning toward investments in resilient, carbon-rich ecosystems that provide wildlife habitat and steady sources of clean water. In addition to enhancing the carbon sequestration potential of U.S. public lands, sensible conservations practices will preserve interconnected wildlife habitat as species adapt to a rapidly changing climate.

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[3] Special Report on Global Warming of 1.5°C at https://www.ipcc.ch/sr15/download/.

[4] Erb, K.H., et al. 2018. Unexpectedly large impact of forest management and grazing on global vegetation biomass. Nature 553: 73-76. Griscom, B.W., et al. 2017. Proceedings of the National Academy of Sciences, Vol. 114, pp. 11645-50.

[5] Hansen, M.C., et al. 2013. High-resolution global maps of 21st-century forest cover change. Science 342: 850-53; Prestemon, J.P., et al. 2015. The global position of the U.S. forest products industry. U.S. Forest Service, e-Gen. Tech. Rpt. SRS-204.

[6] Tackling Climate Change: A Climate Change Adaptation and Carbon Dioxide Removal Landscape Analysis (Sierra Club, Feb. 2019) Attachment A hereto, at p. 14

[7] Depro, B.M., B.C. Murray, R.J. Alig, A. Shanks. 2006. Public land, timber harvests, and climate mitigation: quantifying carbon sequestration potential on U.S. public timberlands. Forest Ecology and Management 255: 1122-1134.

[8] *The United States Mid-Century Strategy for Deep Decarbonization*, p, 15 listing the need to "[q]uickly scale up forest restoration and expansion on federal lands" as a "Long-term U.S. Mid-Century Strategy Priority"; p. 70: "Federal lands will play an important role in preserving carbon stocks and providing early action."; and p. 82 listing "quickly mobilizing federal lands" as a "Priority for Policy, Innovation, and Research" towards achieving 2050 goals."

Without acknowledging the climate stabilizing benefits of preserving and rewilding, this project calls for more than 200 square miles of public and private forests to be treated by either cutting trees, deliberate burning or both to purportedly reduce the risk of unmanaged wildfire. This strategy is faulty and incomplete for several reasons.

First, the assumption that logging/thinning/burning will reduce the severity of wildfires is not universally supported. Cutting trees causes a substantial net loss of forest carbon storage, and a net increase in carbon emissions relative to not cutting. In addition, logged areas tend to experience higher severity fire than unlogged areas (Bradley *et al.* 2016).[9] Using over three decades of fire severity data from relatively frequent-fire pine and mixed-conifer forests throughout the western United States, Bradley *et al.* found that "burn severity tended to be higher in areas with lower levels of protection status (more intense management), after accounting for topographic and climatic conditions;"

Second, increased vegetation treatment operations will reduce forest carbon stocks in the short term without guaranteeing increased carbon sequestration in the future. Vegetation reduction projects will definitely increase carbon emissions in the near-term, releasing carbon through cutting timber, burning slash, and in the milling and manufacturing process. Likewise deliberately set fires will release additional carbon.

Third, the scoping letter failed entirely to address the issue of whether the putative future emission reductions from thinning will occur at all. Although unacknowledged, the project seems to be trading certain increases in net carbon emissions for uncertain future reductions. As highlighted by the 2018 IPCC report, global greenhouse gas (GHG) emissions must be cut approximately in half over the next decade to avoid catastrophic harms from climate change. These targets require increasingly steep reductions in emissions over the coming decade. Yet this is precisely the time period during which the carbon emitted from these proposed treatments will increase atmospheric CO2 levels without any guarantee of reduced emissions in the longer term.

[9] Bradley, C. M., C. T. Hanson, and D. A. DellaSala. 2016. *Does increased forest protection correspond to higher fire severity in frequent-fire forests of the western United States?* Ecosphere 7(10):e01492. 10.1002/ecs2.1492 at 7,9.

Fourth, the notion that dense, long-unburned forests must be "thinned" through logging operations prior to reintroducing fire is simply not scientifically supported, and is directly contradicted by a wealth of scientific data.[10]

The SFNF must quantitatively disclose and analyze the impacts of GHG emissions using guidance provided by the Council on Environmental Quality (CEQ). Effects include both the potential effects of the proposed action as indicated by assessing GHG emissions and the effects of climate change on the proposed action and its environmental impacts. The Global Climate Change Prevention Act of 1990, sections 6701(b)5 and (c)3, requires that all federal agencies analyze climate change effects in decision-making and propose alternatives that mitigate the adverse effects of climate change.

In addressing GHG emissions, the Forest Service must include a comparison of estimated net GHG emissions and carbon stock changes that are projected to occur with and without the proposed actions. According to the CEQ, finding that a land management action represent only a small fraction of global emissions is not an appropriate basis for deciding whether or to what extent to consider climate change impacts under NEPA. CEQ also notes that monitoring is particularly appropriate to confirm the effectiveness of mitigation. Unfortunately, the Santa Fe National Forest has been woefully deficient in monitoring the impacts of its 1987 Land Management Plan.

In conclusion, the project ignores NFMA's requirement to base decisions on the best available science and NEPA's requirement to address allegedly insufficient information. To the degree that impacts to the climate are highly uncertain or involve unique or unknown risks, then an EIS is clearly required. 40 C.F.R. §1508.27(b)(5). When there is incomplete or unavailable information concerning reasonably foreseeable climate impacts, the agency must include a summary of the existing information in the EIS and an evaluation of the impacts based on such information. 40 C.F.R. §1502.22.

[10] *See* Keifer, M.B., 1998. Fuel load and tree density changes following prescribed fire in the giant sequoia-mixed conifer forest: the first 14 years of fire effects monitoring. In: Proceedings of the Tall Timbers Fire Ecology Conf., vol. 20. pp. 306–309; Stephens, S.L., Finney, M.A., 2002. Prescribed fire mortality of Sierra Nevada mixed conifer tree species: effects of crown damage and forest floor combustion. For. Ecol. Manage. 162, 261–271; Fulé, P.Z., Cocke, A.E., Heinlein, T.A., Covington, W.W., 2004. Effects of an intense prescribed forest fire: is it ecological restoration? Restoration Ecology 12, 220–230; Schwilk, D.W., Knapp, E.E., Ferrenberg, S.M., Keeley, J.E., Caprio, A.C., 2006. Tree mortality from fire and bark beetles following early and late season prescribed fires in a Sierra Nevada mixed-conifer forest. Forest Ecology and Management 232, 36–45; van Mantgem, P.J., J.C.B. Nesmith, M. Keifer, and M. Brooks. 2013. Tree mortality patterns following prescribed fire for Pinus and Abies across the southwestern United States. Forest Ecology and Management 289: 463-469; van Mantgem, P.J., A.C. Caprio, N.L. Stephenson, and A.J. Das. 2016. Does prescribed fire promote resistance to drought in low elevation forests of the Sierra Nevada, California, USA? Fire Ecology 12: 13-25; van Mantgem, P.J., N.L. Stephenson, J.J. Battles, E.K. Knapp, and J.E. Keeley. 2011. Long-term effects of prescribed fire on mixed conifer forest structure in the Sierra Nevada, California. Forest Ecology and Management 261: 989-994.

3. THE PROJECT FAILS TO PROTECT THE MEXICAN SPOTTED OWL

The Forest Service has failed for more than two decades to obtain critical information needed to conserve the Mexican spotted owl (MSO). As a result there continues to be exceptional uncertainty when assessing both region-wide and site-specific impacts. First and foremost, the Forest Service has failed to acquire baseline information on MSO population trends. Second, the Forest Service has failed to acquire any information on the cause-effect relationship between the large-scale clearing and burning and MSO population trends. NEPA's regulations require that an EIS be prepared if the environmental impacts of an agency action are likely to be highly uncertain. 40 C.F.R. § 1508.27(b)(5).

Despite the clear need for caution, this project calls for an unprecedented level of habitat disruption to both protected activity centers (PACs) and restricted habitat (steep slopes), far more than allowed by the 1996 SFNF Plan's standards and guidelines. This is unwarranted given studies which suggest that most MSO populations have either declined in the recent past or are still declining. Further, some evidence suggests that owls may be slow to re-colonize areas where such declines have occurred (Seamans and Gutiérrez 2006, Stacey 2010, Willey and Willey 2010).

The extreme level of uncertainty coupled with evidence of a declining population argue strongly against rolling back the binding 1996 standards and guidelines. The proposed project-specific forest plan amendment would invalidate the current programmatic MSO Biological Opinion (BiOp) for the SFNF. This BiOp assumes the implementation of the 1996 standards and guidelines including rigorous population trend monitoring. In the absence of region-wide long-term population trend monitoring, a separate BiOp would be required for the project to evaluate whether the proposed landscape-level clearing and burning will jeopardize the owl population and/or adversely modify its critical habitat.

4. THE PROJECT FAILS TO PROTECT OLD GROWTH FORESTS

NFMA imposes on the Forest Service a duty to ensure that any specific project in the forest complies with the "land resource management plan of the entire forest," in this case the SFNF Plan. 16 U.S.C § 1604(i).

The SFNF Plan's old growth standards begin with an admission of uncertainty, followed by a commitment to learn and identify old growth in all project planning:

> Old growth is not well understood in the Southwest. Consequently, as knowledge is gained the characteristics and inherent values of old growth stands will be better defined. Site specific identification of old growth will occur during ecosystem area analysis or project planning. (SFNF Plan p. 67)

As noted earlier, NEPA's regulations requires that an EIS be prepared if the environmental impacts of an agency action are likely to be highly uncertain. 40 C.F.R. § 1508.27(b)(5). The project's impact to old growth are clearly rife with uncertainty.

It is unclear what, if any, knowledge has been gained of old growth's characteristics and values over the course of the implementation of the SFNF Plan. It is not disclosed how project-level knowledge will be gained to better define "the characteristics and inherent values of old growth stands." This would include how the SFNF Plan's parameters for determining old growth has been refined for this project. For example: number of live trees in the main canopy; variation in tree diameters; dead trees (standing snags and downed logs); tree decadence; number of tree canopies; total basal area; and total percent canopy cover.

Only the bare minimum of 20 percent of the project area—the floor established by the SFNF Plan—is being managed for old growth. Managing for minimums gives no room for error and errors are inevitable given the acknowledged uncertainty and unprecedented scale and intensity of proposed activities. Managing for minimums and allowing discretionary cutting of trees up to 23.9 d.b.h. is clearly inconsistent with the SFNF Plan that requires projects to "strive to create or sustain as much old growth compositional, structural, and functional flow as possible over time at multiple-area scales."

It is unclear how old growth can be sustained as required by the SFNF Plan when as much as 30 percent of remainder trees left after aggressive clearing die in prescribed fires and more from wind throw in newly opened stands. Also, *Ips* beetle populations increase dramatically in untreated slash during dry conditions often overwhelming old growth ponderosa pines.

5. THE PROJECT FAILS TO PROTECT SOUTHWESTERN WHITE PINE

At the northern limits of its distribution, the Southwestern white pine (*Pinus strobifomis*) population may be exhibiting unique resistance to white pine blister rust as a result of widespread hybridization with limber pine (*Pinus flexilis*). Hybridization can increase genetic diversity and generate novel allelic combinations. Novel combinations may exhibit both resistance to the devastating blister rust and facilitate adaptive evolution to ongoing and future climate change. There is evidence that this hybrid zone is shifting northward in response to the warming climate.[11]

Removing individuals that are genetically resistant before it can be determined their value in countering the disease and adapting to climate change would be a significant loss to regional biodiversity.

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[11] Menon M, Landguth E, Leal-Saenz A, et al. Tracing the footprints of a moving hybrid zone under a demographic history of speciation with gene flow. Evol Appl. 2019;00:1–15. https://doi.org/10.1111/eva.12795

Unfortunately, the Forest Service has a long history of ignoring evolutionary processes such as natural selection. In its formative years the agency encouraged land owners along the eastern seaboard to cut down all American chestnuts before they were killed by an exotic blight. As a result genetically resistant trees that may have allowed the species to survive and adapt were lost.[12] A more recent example is salvage logging of beetle killed white bark pine in the northern Rockies. In this case, resistance and adaption is threatened by both clearing dead and surviving *Pinus albicaulis* and large-scale replanting of non-resistant trees.[13]

The standards of the SFNF Plan (replacement page 69a) must be met requiring a minimum of 120 Southwestern white pine remain per acre following clearing and burning. However, preserving all individuals of this unique and relatively uncommon species is biologically warranted and needed to meet NFMA's biological diversity mandate.

6. THE PROJECT FAILS TO PROTECT ROADLESS AREAS

This project failed to identify protection of inventoried roadless areas (IRAs) as a potential issue. No information was presented concerning the delineation, location and potential impact to IRAs.

National forest roadless lands are heralded for their conservation values. Those values are described at length in the preamble of the Roadless Area Conservation Rule (RACR) and in the Final Environmental Impact Statement (FEIS) for the RACR.[14] They include: high quality or undisturbed soil, water, and air; sources of public drinking water; diverse plant and animal communities; habitat for threatened, endangered, proposed, candidate, and sensitive species and for those species dependent on large, undisturbed areas of land; primitive, semi-primitive non-motorized recreation; reference landscapes; natural appearing landscapes with high scenic quality; traditional cultural properties and sacred sites; and other locally identified unique characteristics (e.g., uncommon geological formations, unique wetland complexes, exceptional hunting and fishing opportunities).

Roadless lands are responsible for higher quality water and watersheds. Anderson et al. 2012[15] assessed the relationship of watershed condition and land management status, and found a strong spatial association between watershed health and protective designations. DellaSalla et al. 2011[16]

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[12] Kelly, A.R. *Chestnut surviving blight.* Science 40 (1924): 292-93

[13] Six, D., C. Vergobbi and M. Cutter. 2018. Are survivors different? Genetic-based selection of trees by mountain pine beetle during climate change-driven outbreaks in a high-elevation pine forest. Frontiers in Plant Science

[14] 66 Fed. Reg. at 3245-47 and Final Environmental Impact Statement, Vol. 1, 3–3 to 3–7, available at http://www.fs.usda.gov/roaddocument/roadless/2001roadlessrule/finalruledocuments.

[15] Anderson, H. Mike et al., 2012. Watershed Health in Wilderness, Roadless, and Roaded Areas of the National Forest System. The Wilderness Society, Washington DC. http://wilderness.org/resource/watershed-health-wilderness-roadless-and-roaded-areas-national-forest-system.

[16] DellaSala, D., J. Karr, and D. Olson. Roadless areas and clean water. Journal of Soil and Water Conservation, vol. 66, no. 3. May/June 2011.

found that undeveloped and roadless watersheds are important for supplying downstream users with high-quality drinking water, and that developing those watersheds comes at significant costs associated with declining water quality and availability. Protecting and connecting undeveloped areas is also an important action agencies can take to enhance climate change adaptation.

It is also likely that there are substantial "unroaded" areas that could become inventoried roadless lands and recommended for wilderness designation in the future. These lands play an important ecological role in ensuring the persistence of species, providing connectivity and ensuring watershed functionality.

Therefore, the project planning team must identify, delineate and quantify unroaded lands and take the required hard look to determine if planned clearing and burning activities may have significant impacts. We strongly oppose any developments in unroaded portions of the project area until potential impacts can be comprehensively disclosed and analyzed.

In summary, the cumulative effects of clearing and burning thousands of acres over many decades in unroaded, lightly-roaded and IRAs eligible for wilderness must be analyzed and disclosed in an EIS.

7. THE PROJECT FAILS TO USE THE BEST AVAILABLE SCIENCE

As noted earlier, NFMA's 2005 regulations that guide implementation of the 1987 SFNF plan requires the Forest Service use and document the best available science. 36 C.F.R. § 219.11(a)(1)-(4). Please consider the following issues when using scientific information to prepare an EIS for this project. For more detail see the attached comments by Dr. Dominick DellaSala to the Santa Fe Forest Plan/DEIS currently in review. The references cited below are in his comments.

Biased Fire Scar Sampling

Fire scar estimates to determine <u>fire return intervals</u> are often extrapolated over large areas instead of using multiple lines of evidence to calculate <u>fire rotation intervals</u> (Odion et al. 2014a, Moritz et al. 2018).

There are significant uncertainties with extrapolating fire scar point sampling data over large landscapes with the goal of reconstructing historic fire regimes for comparisons to contemporary conditions (Baker 2017). They include sample-site selection bias, lack of tree scars in fire-killed trees (thereby underestimating high severity occurrence), and inappropriate extrapolation of site-specific data to draw landscape-level inferences (Baker 2017).

One result is a bias toward short fire return intervals which initiates a cascade of errors, the most obvious are: 1) forests historically were predominately open; 2) contemporary forest conditions are overly dense; and 3) there is a need for aggressive mechanical treatments to return to an idealize past forest structure.

Records from paleo-ecology reduce sampling bias and records high severity fires that other methods miss. The paleo-record from charcoal sediments shows that when wet periods are followed by successive droughts, large fires occur, including high severity patches (Meyer 2010). Variability in fire return intervals results in high levels of fire-mediated biodiversity (i.e., pyrodiversity begets biodiversity, DellaSala and Hanson 2015). These benefits are not possible with a strategy exclusively based on low-severity managed fires.

Closed Canopy Conditions Arbitrarily Defined

The LANDFIRE model currently in use is arbitrarily constrained to define closed canopy conditions in the mixed conifer and ponderosa pine frequent fire Ecosystem Response Units as woody cover exceeding 30%. No empirical evidence is provided for this decision. The result is that extreme openness is used to determine the desired reference/baseline condition and contemporary departure indices for alternative analyses.

Closed canopy forests in some cases currently exceed 70%. Dramatic reductions in cover constitutes a major change that will have significant impacts to species requiring closed canopy conditions. Large interspaces will be created across the landscape to meet this arbitrarily defined "open" reference condition, creating novel ecosystems that do not comport with the need to maintain ecological integrity or diversity.

Ecosystem Response Unit Classification Is Flawed

The outdated Ecosystem Response Unit (ERU) method of classifying vegetation is unreliable and inaccurate. ERUs are based on an idealized view from the 1950s of what vegetation at a particular site could potentially become independent of human influence.

The ERU methodology fails on two counts: 1) it runs counter to Forest Service policy set forth in FSH 1909.12 which mandates the use of 'natural range of variation,' a scientifically credible approach that recognize forests as dynamic ecosystems subject to change over time; and 2) does not acknowledge human influence as a universal shaper of ecosystem structure and function.

The project is embedded within the Colorado Rockies Forest Ecoregion that has been heavily exploited by humans for centuries and continue to be exploited today. In addition, human-caused GHG emissions are rapidly warming the planet with increasingly dire consequences for Rocky Mountain forest ecosystems. Climate change alone invalidates ERUs as a credible management tool. In addition, it is now nearly impossible to imagine potential vegetation independent of human influence, let alone develop accurate predictive models for future desired conditions.

Additional problems with ERUs include: 1) the tendency to manage for an idealized vegetation type that may not be possible at a specific site due to the changing climate; and 2) formulating desired conditions without planning for the distribution of seral stages of development based on what is actually present on the landscape. These seral states often provide essential habitat for many species.

Low Probability of Treatment Success

It is incorrectly assumed that fuel treatments are always effective in lowering wildfire intensity. In fact, it is extremely unlikely that a wildfire will encounter a treated area during the 10-15 year period following treatment when fuels are lowest (Schoennagel et al. 2017). Simply increasing the area treated does not change these odds appreciably given one cannot accurately predict when and where a fire will occur and many areas are inaccessible (Schoennagel et al. 2017).

8. THE PROJECT MUST BE CONSISTENT WITH SFNF PLAN

As noted earlier NFMA requires that any action taken at the project-specific level comply with the Forest Land and Resource Management Plan. 16 U.S.C. Sec. 1604(i). Forest Service procedures also require consistency with the Forest Land and Resource Management Plan (FSM 1922.12 and FSH 1909.12). The following measure must be met to ensure the project is consistent with the 1987 Santa Fe National Forest Plan (SFNFP).

SFNFP requires that canopy cover of mid-aged (VSS 4)[17], mature (VSS 5) and old (VSS 6) ponderosa pine forests be managed for an average canopy cover of 40 percent or greater. For mixed conifer forests the canopy cover averages are one-third 60 percent and two-third 40 percent or greater for mid-aged forest (VSS 4), 50 percent or greater for mature forests (VSS 5) and 60 percent or greater for old forest (VSS 6). Average canopy cover for spruce-fir is one-third 60 percent or greater and two-thirds 40 percent or greater for mid-aged forest (VSS 4) and 60 percent or greater for mature and old forests (VSS 5 and 6).

The SFNFP's canopy cover standards apply to all forest and woodland communities not already protected as Mexican spotted owl habitat (USDA Forest Service 1996:91). These canopy cover minimums protect the Northern Goshawk (*Accipiter gentiles*), a raptor morphologically adapted to dense forests that studies using radio telemetry consistently demonstrate selects habitats with high canopy closure (Austin 1993; Beier and Drennan 1997; Boal et al. 2001; Bright-Smith and Mannan 1994; Drennan and Beier 2003; Hargis et al. 1994 and Stephans 2001). Please indicate the methods used to identify the VSS classes in the project area that meet these canopy cover requirements.

The SFNFP requires the project to "identify and manage dispersal (Goshawk) post-family fledging areas (PFA) and nest habitat at 2 to 2.5 miles spacing across the landscape" (USDA Forest Service 1996:92). The SFNFP links VSS, tree density and tree age to the "site quality of the ecosystem management area" (USDA Forest Service 1996:92).

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[17] VSS is Vegetative Structural Stage. Canopy cover is the percentage of ground area shaded by overhead foliage (Daubenmire 1959 cited in Ganey and Block 1994:21) measured by the vertical crown projection of the upper, mid and lower canopies (USDA Forest Service 1996:92).

The SFNFP also lists "dozer piling" as the least preferred treatment for woody debris and wisely "limits dozer use for piling or scattering of logging debris so that the forest floor and herbaceous layer is not displaced or destroyed" (USDA Forest Service 1996:94). Maintaining the organic surface soil layers where ectomycorrhizae fungi are concentrated—mobilizing nutrients and providing food for Goshawk prey—is critically important to sustaining healthy forest ecosystems (Reynolds et al. 1992:31). Please indicate site-specific measures that will be taken to limit dozer piling.

The SFNFP says "no treatments should occur in a stand managed for old growth once the stand has achieved minimum structural characteristics of old growth" (SFNFP, p. 69).[18] To determine old growth please indicate the methods used for determining the age of trees in the main canopy; the size, height and number of standing dead trees; the size, length and pieces of down dead trees; the number of decadent trees; the number of tree canopies; and the total percent of canopy cover and how this site-specific data will be used in the "quantitative models" specified in the SFNFP (USDA Forest Service 1996:95).

In addition, please document how the SFMLRP is "incorporating natural variation . . . into management prescriptions" . . . maintaining "all species of native trees". . "allowing natural canopy gap processes to occur" . . . (USDA Forest Service 1996:89) and "monitoring management practices within designated peregrine falcon habitat" (SFNFP, p. 62) . . . provide ". . . adequate perch and roost trees for raptors . . . within a 200 foot wide stand along . . . major ridges" (SFNFP, p. 66) . . . coordinate timber activities in turkey nesting areas "to minimize impacts between April 20 and June 10" (SFNFP, p. 72) . . . locate log landing areas to the extent practical "outside . . . threatened and endangered species habitat" (SFNFP, p. 73) . . . maintain adequate cover "within 8 chains (530 feet) of actively used elk wallows, licks, and seeps" (SFNFP, p. 73) and, finally, protect "trails, blaze trees, and trail markers" during timber harvest activities (SFNFP, p. 74).

9. THE PROJECT FAILS TO PROTECT MYCORRHIZAL FUNGI

The project does not mention of the critical role that mycorrhizal fungi networks play in sustaining forests. No protection is proposed for mycorrhizal networks from vegetation clearing and burning, roads and livestock grazing. These omissions undermine the environmental protection purpose of NEPA and the biodiversity mandate of NFMA.

Mycorrhizal networks play important roles in mitigating the impacts of climate disruption to forest ecosystems. They facilitate regeneration of migrant species that are better adapted to

[18] Old growth is defined on p. 69a of the Forest Plan by cover type for a range of live trees in main canopy, variation in tree diameters, dead trees, tree decadence, number of tree canopies, total basal area and total canopy cover.

warmer climates and primed for resistance against insect attacks.[19] To achieve these benefits all of the parts and processes of highly interconnected forest ecosystems must be preserved and protected.

Mycorrhizal fungi distribute photosynthetic carbon by connecting the roots of the same or different tree species in a network allowing each to acquire and share resources. Large mature trees become the hubs of the network and younger trees the satellite nodes.

Mycorrhizal networks transmit water, carbon, macronutrients, micronutrients, biochemical signals and allelochemicals from one tree to another, usually from a sufficient tree to a tree in need. This type of source-sink transfer has been associated with improved survivorship, growth and health of the needy recipient trees in the network.

Recognition of kin is also evident between established large hub trees and their seedlings and saplings. Hub trees shuttle their kin more micro-elements and support more robust mycorrhizal networks providing them with a competitive advantage. However, hub trees also share resources with strangers, suggesting these evolutionary mechanisms exist not just for individual species but also at the community level.

Injury to a tree from defoliation by an insect herbivore or by physically removing foliage results in the transmission of defense signals through the connecting mycorrhizal mycelium to neighboring trees. These neighbors respond with increased defense-gene expression and defense-enzyme activity, resulting in increased pest resistance.

In Douglas-fir, sudden injury to a hub tree not only increases defense enzymes of healthy neighbors but elicits a rapid transfer of photosynthate carbon to a healthy neighbor. This suggest that the exchange of biochemicals between trees elicits meaningful changes in the senders' and receivers' behavior that enables the community to achieve greater stability in the face of a changing climate.[20]

The complete omission of any consideration of mycorrhizal networks is a symptom of a single minded vision of the future that is inconsistent with the unpredictability of climate-driven change. Instead, forest managers should use scenario building models to explore an envelope of probable futures that becomes wider the further forward one projects.[21] In this more multifaceted

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[19] Song, Y.Y. Simard, S.W., Carroll, A., Mohn, W.W. and Zheng, R.S. 2015. Defoliation of interior Douglas-fir elicits carbon transfer and defense signalling to ponderosa pine neighbors through ectomycorrhizal networks *Nature / Sci. Rep. 5, 8495; DOI:10.1038/srep08495 (2015).* Attached

[20] ibid. Song et. al. 2015

[21] Lempert, R. J. 2002. A new decision sciences for complex systems. Proceedings of the National Academy of Sciences USA 99:7309–7313 and Parrott, L. and W. S. Meyer. 2012. Future landscapes: managing within complexity. Frontiers in Ecology and the Environment 10:382–389.

approach based on complex systems science, managers quantify the likelihood of each scenario and then address the ranges of uncertainties in the ecological, social, and economic dimensions.[22]

10. THE PROJECT FAILS TO PROTECT PUBLIC HEALTH

The scoping document calls for more than 77,000 acres to be periodically burned using low-intensity prescribed fires that produce high particulate smoke emissions. This would expose affected citizens to far more smoke particulates over time than emissions produced by an infrequent high intensity wildfire.

There is no known safe level of exposure to small particulate matter in smoke (< 2.5 microns in size) below which health impacts are not observed. A significant portion of the population, possibly even a majority, is at increased risk of harm from exposure. A Health Impact Assessment must be prepared to disclose and analyze potential impacts to the disadvantaged rural communities adjacent to the project area that will be most directly impacted.

11. QUESTIONS

Public involvement during the short 30 day comment period was limited. One meeting was held in a remote location and no public notification was published in a newspaper of record. The scoping document was devoid of critically important information.

A fundamental purpose of NEPA mandates that "federal agencies **shall** to the fullest extent possible . . . encourage and facilitate public involvement in decisions which affect the quality of the human environment." 40 C.F.R. § 1500.2(d). Since that did not occur we submit the following questions:

A. PURPOSE AND NEED AND NATIONAL ENVIRONMENTAL POLICY ACT

- ✳ Why isn't protecting lives and property the primary purpose of this project? Making vulnerable homes fire-safe and clearing flammable vegetation immediately around structures are proven strategies.
- ✳ Will measures to protect soils, water quality and wildlife habitat be mandatory and enforceable if they are proposed in an Environmental Assessment as opposed to an Environmental Impact Statement? Please explain the role of mitigation measures in each document.

B. ROADLESS FORESTS AND ROAD IMPROVEMENT

- ✳ How many inventoried roadless areas exist in this area? Are they be proposed for Wilderness in the new forest plan?
- ✳ Constructing temporary roads will increase human caused fires in this area. Does the SFNF have the capacity of responding to this increase?

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[22] Filotas, E., Parrott, L., Burton, P.J., Chazdon, R.L., Coates, K.D., Coll, L., Haeussler, S., Martin, K., Nocentini, S., Puettmann, K.J., Putz, F.E., Simard, S.W., and Messier, C. (2014). **Viewing forests through the lens of complex systems science** *Ecosphere 5:art1*

* Will unneeded roads be obliterated to protect water quality and wildlife habitat and prevent the spread of invasive plants and access by arsonists and poachers?
* How will ATVs be effectively restricted during project activities?

C. CLIMATE DISRUPTION

* Is the Forest Service allowed to discuss the role that human emissions play in creating a hotter and drier climate in the Southwest? If so, why was the climate not discussed in the scoping document?
* Is current climate science being used to analyze the impacts of clearing trees and annual burning?
* Why isn't climate change mentioned as the primary driver of larger and more frequent high-severity fires?
* Why is the aim of this project to restore past forest structure instead of working with natural succession and evolutionary processes to help the forest adapt to a warmer and drier climate?

D. WILDLIFE AND ANCIENT FORESTS

* How will wildlife corridors be maintained in areas cleared and annually burned? Have corridors been identified in the project area?
* Will clearing and burning be restricted in the spring to protect breeding bird nests and other wildlife? If not, please explain why.
* Old growth aspen is important breeding bird habitat. Clearing and burning conifers in the understory will cause significant harm. Will bird populations in old growth aspen habitat be monitored to determine impacts? If not, please explain why.
* Why are the threats of high severity fire to Mexican spotted owl habitat highlighted while it's benefits and the adaptability of the owl to burned forest habitat not discussed? Does the SFNF monitor the Mexican spotted owl population? If so, what are the current trends?
* Why is retaining the minimum allowed old growth the aim of this project when the forest plan requires as much old growth be managed as possible?
* Preservation of old growth and fuel reduction have conflicting aims. How will old growth forests with their dense multistoried and high canopy cover be maintained on a minimum of 20% of the project area?

E. CLEARING TREES AND ANNUAL BURNING

* How many live trees will remain after the initial clearing and burning? How many remainder trees are expected to die in prescribed fires, bark beetles outbreaks and wind throw in newly opened stands?
* Will the legally required regeneration standards for remainder trees be monitored? If so will that data publicly be available?
* Will the size of burned debris piles be limited to protect soils and discourage invasive plants from becoming established?

* Why are protection measures for the vulnerable Southwestern white pine population not discussed?
* Will on-going livestock grazing impede the goal of restoring low-severity fire regimes?

* Reference conditions are mentioned as being used to establish a desired forest structure. Please identify the reference sites in the project's Colorado Rockies Forest ecoregion.

Respectfully Submitted,

/s/ Sam Hitt

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Imler-Jacquez, Sandra R -FS

From: Sam Hitt <sam@wildwatershed.org>
Sent: Thursday, December 19, 2019 9:24 PM
To: FS-comments-southwestern-santafe
Subject: Encino Vista Landscape Restoration Project

Attachment available until Jan 18, 2020

Click to Download

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179.4 MB

Part two comments attached

I want you to act as if your house is on fire.
Because it is.
Greta Thunberg

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September 30, 2019
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Submitted by: Dominick A. DellaSala, Ph. D, Conservation Scientist

Re: **Comments on the Santa Fe National Forest Draft Land Management Plan and Draft Environmental Impact Statement (DEIS)**

Please accept these comments for the public record regarding the Santa Fe National Forest Draft Land Management Plan and DEIS. I am a conservation scientist with over 30 years-experience in forest ecosystems, including documenting the importance of fire-mediated biodiversity in dry pine and mixed conifer forests of the West (DellaSala and Hanson 2015[1]). My relevant expertise also includes developing robust conservation strategies for land managers to accommodate wildfires for ecosystem benefits while reducing fire risks to communities. I am submitting the enclosed publications as pdfs in support of my comments, including how extensive logging has increased fire severity in forests (Bradley et al. 2016), limitations of forest thinning in reducing fire intensity (DellaSala et al. 2018), livestock grazing and climate change cumulative impacts on national forests (Beschta et al. 2012), fire ignitions associated with roads (Ibisch et al. 2016), climate change effects on fire regimes (Abatzoglou and Williams 2017), and ecological importance of high severity burn patches in dry pine/mixed conifer forests including New Mexico (DellaSala and Hanson 2019), among other relevant peer reviewed publications. My comments are meant to improve the Forest Service's approach to forest-fire resilience in the Santa Fe National Forest (SFNF) and surroundings with the intent of showing how the agency can and must do better with respect to using the best available science along with involving scientists with a biodiversity perspective on wildfire and not just a fuel centric perspective dominated by fuel management.

The SFNF encompasses 1.6 million acres (nearly the size of Yellowstone National Park) of diverse conifer forests, woodlands, riparian forests, native grasslands and shrublands that make up the scenic beauty and quality of life for surrounding communities, including unmatched recreation, clean water, hunting and fishing, and iconic wildlife species. The SFNF includes nationally significant roadless areas; designated and proposed Wilderness and Wild and Scenic rivers; tribal-cultural sites; and essential habitat for large carnivores, ungulates, and at-risk wildlife such as Mexican Spotted Owl, Southwestern Willow Flycatcher, Jemez Mountain

[1] Note – a copy of the book – a very large pdf – can be purchased here https://www.sciencedirect.com/book/9780128027493/the-ecological-importance-of-mixed-severity-fires. For the purpose of these comments, I included the relative chapter, however, these included editing notes as the publisher did not provide chapter pdfs.

salamander, Rio Grande cutthroat trout, and New Mexico meadow jumping mouse. These and many other species in the project area require intact areas periodically maintained by wildfires of low and mixed severity effects on vegetation that also include occasional large and small patches of high severity fire effects. The SFNF's low elevation forests are predominately influenced by frequent fires of low severity with fire-flare ups that often kill overstory trees (site and landscape heterogeneity). During drought cycles and under extreme fire weather these flare ups can include small and large high severity patches that are important ecologically (DellaSala and Hanson 2019). Upper elevation spruce-fir forests are on centuries long *fire rotation intervals* (landscape scale) where high severity fire effects are characteristic (Margolis et al. 2002) and climatic factors are the top-down driver of fire behavior, not fuels (see Bessie and Johnson 1995). This variability is not appropriately recognized, planned for, or even properly analyzed in the DEIS, which mostly emphasizes mechanical treatments designed to move substantial amounts of closed canopy forests into low fuel condition conducive of low-severity fire effects lacking diversity/heterogeneity at site or landscape levels.

Much of the Santa Fe National Forest biodiversity is distributed along elevation gradients with changes in life zones prominent from valley bottoms and foothills to montane and alpine. Thus, a primary objective of the DEIS should be to maintain landscape connectivity that accommodates biological diversity across life zones and for focal species, species of conservation concern, and at-risk species and ecosystems. The DEIS is deficient in analyzing how connectivity is being impacted specifically by habitat fragmentation in the project area and surroundings (cumulative effects) caused by roads, extensive thinning and forest canopy reductions, ski area development, mining, livestock grazing and infrastructure, off highway vehicles (OHVs), and other developments. Connectivity cannot simply be maintained at the coarse-filter level via vegetation management and very general site-specific measures incorrectly presented as a fine filter approach. Connectivity maintenance requires proper analysis (species-specific trigger points and population viability analysis, see Noon et al. 2003, Schultz et al. 2013) to meet the best available scientific information (BASI) requirement of the 2012 forest planning rule. None of the alternatives in the DEIS meet the BASI requirement for connectivity (Box 1 and Box 2).

Box 1. Ecological integrity. The quality or condition of an ecosystem when its dominant ecological characteristics (e.g., composition, structure, function, connectivity, and species composition and diversity) occur within the natural range of variation and can withstand and recover from most perturbations imposed by natural environmental dynamics or human influence (36 CFR 219.19).

Box 2. Connectivity. Ecological conditions that exist at several spatial and temporal scales that provide landscape linkages that permit the exchange of flow, sediments, and nutrients; the daily and seasonal movements of animals within home ranges; the dispersal and genetic interchange between populations; and the long distance range shifts of species, such as in response to climate change.

Planning deficiencies regarding integrity and connectivity are summarized as follows:

- Connectivity is inadequately addressed by an emphasis on vegetation management in Ecological Response Units (mostly coarse filter), general site-specific measures (inadequate fine filter), and some road closures/decommissioning. Notably, in a comprehensive analysis of biodiversity strategies in a changing climate, connectivity (site-specific structural features, landscape intactness, corridors) was identified as the single most important strategy for enabling plants and wildlife to adapt to climate change and is critical to achieving climate resilient ecosystems (Haber and Nelson 2015). These authors recommend far more measures for maintaining connectivity than what was provided in the DEIS.
- There are substantial roads (6,900 miles) throughout the SFNF, many of which are leaking sediments into streams and pose a barrier and mortality risk to wildlife (vehicle collisions). Roads, cattle, and logging/thinning all affect the biological and physical environment of focal species, at-risk species, and species of conservation concern and this requires fine-scale analysis ("trigger points," and population viability analysis (PVA); as in Noon et al. 2003, Schulz et al. 2013) along with stepped up conservation (see conservation requirement of the planning rule below) that must be analyzed at the appropriate scale using BASI to take a hard look at connectivity and not just providing unsupported claims that vegetation management actions satisfy this requirement.
- The DEIS must analyze connectivity to maintain viable populations of focal species, at-risk species, and species of conservation concern (i.e., via PVA and trigger points) especially in a changing climate and in the context of both direct and indirect cumulative effects (e.g., analyze habitat fragmentation as the antithesis of connectivity).
- A connectivity analysis needs to incorporate cumulative impacts (e.g., livestock, thinning, roads), importance of intact areas (especially connecting life zones along gradients for species movements), and barriers to terrestrial and aquatic focal species, at-risk species, and species of conservation concern along with specific measures for reconnecting habitat. Examples of connectivity analyses include identification of species-specific road density thresholds (generally >1 mi/square mile is problematic for aquatic species), identification and protection of ungulate migration corridors (e.g., deer and elk winter and summer range movements) and large carnivore travel routes (especially along riparian areas) (i.e., the Forest Service must follow approaches in Haber and Nelson 2015).

Maintaining the mixture of fire severity effects on the SFNF is key to meeting the diversity requirements of the 2012 forest planning rule (see section on diversity of plant and animal communities), including mixed-severity fires that produce high-severity patches having unique ecological functions (DellaSala and Hanson 2019). The DEIS is deficient in this regard as it over emphasizes low-severity fire at the expense of mixed-severity fire effects (including high severity patches) essential to ecological processes, ecological conditions, and ecological integrity (Box 1, 3, 4, 5).

Box 3. The selected set of ecological conditions and key ecosystem characteristics related to the composition, structure, ecological processes, and connectivity of plan area ecosystems (terrestrial, riparian, and aquatic), provide the basis for monitoring ecosystem integrity (36 CFR 219.8(a)(1)) and the diversity of plant and animal communities (36 CFR 219.9).

Box 4. System drivers, including dominant ecological processes, disturbance regimes, and stressors, such as natural succession, wildland fire, invasive species, and climate change; and the ability of the terrestrial and aquatic ecosystems on the plan area to adapt to change (§ 219.8)

Box 5. Ecological conditions. The biological and physical environment that can affect the diversity of plant and animal communities, the persistence of native species, and the productive capacity of ecological systems. Ecological conditions include habitat and other influences on species and the environment. Examples of ecological conditions include the abundance and distribution of aquatic and terrestrial habitats, connectivity, roads and other structural developments, human uses, and invasive species.

The DEIS conflicts with the above planning rule requirements in the following ways:

- Alternative 3 (natural process alternative) is erroneously dismissed for Alternative 2 (preferred alternative) that relies on far more mechanical treatments than natural processes. More natural process features from Alternative 3 need to be incorporated into the final plan. Ostensibly, the main reason for the Forest Service rejecting Alternative 3 stems from inaccuracy problems inherent to LANDFIRE, fire scar analysis sampling biases, and inappropriate reference conditions tied to Forest Service research publication GTR-310 that have led to an over-reliance on mechanical treatments to achieve novel ecosystems devoid of most small trees with remaining trees prone to blowdown.
- The DEIS assumes high-severity fire patches are an anomaly of contemporary fire systems and therefore does not properly analyze positive contributions of high-severity patches in contributing to diverse ecosystems (DellaSala and Hanson 2019), particularly high elevation areas that experience characteristic high-severity fires (the predominant fire regime) on long fire rotation intervals.
- High-severity patches are ecological diverse habitats (DellaSala and Hanson 2019) and are important as foraging habitat for raptors such as Northern Goshawks and Mexican Spotted Owls (see Lee 2018), woodpeckers and songbirds (Hutto et al. 2015), small mammals and ungulates (Bond 2015), and may play a role in snowshoe hare/lynx dynamics. This needs to be acknowledged and the at-risk species tables in the DEIS

adjusted to include positive effects of high-severity fires on wildlife instead of all negative effects as incorrectly noted in the DEIS.

- The DEIS does not include sufficient actions for limiting the spread of invasive species via vector management of livestock (maximum permitted stocking rate of 11,400 AUMs is not sufficiently mitigated), roads, and OHVs. Improved foraging habitat for cattle through thinning and infrastructure changes under the preferred alternative is inadequate for addressing the chronic invasive species problems across the SFNF that will accumulate (cumulative effects) over space and time through active management (thinning entries), continued grazing especially in a changing climate (see Beschta et al. 2012) and road developments (see Ibisch et al. 2016 for a review of road impacts, including spread of invasive species).
- The DEIS is completely inadequate in addressing the critical habitat needs and population dynamics of threatened Mexican Spotted Owls (MSO), which require site specific and region-wide population monitoring and not just knowledge of habitat availability. Notably, a federal judge on September 11 enjoined all “timber management actions” in eleven national forests in New Mexico and Arizona for failing to survey and protect the MSO. The SFNF through Endangered Species Act section 7 consultation must engage in specific and region-wide population monitoring to ensure the MSO population is recovering and its habitat protected from thinning and other project actions (e.g., effects of livestock grazing on prey species).

Overall, the DEIS and supporting documents do not meet the BASI requirement of the 2012 forest planning rule with respect to *accurate, reliable, and relevant issues being considered* (Box 6). There are incorrect reference conditions tied to the Forest Service research publication GTR-310 extrapolated from a completely different region, accuracy problems inherent with the LANDFIRE program at the SFNF scale, uncertainties with fire return interval estimates using fire scar sampling, and arbitrary determinations regarding closed canopy forest conditions that has led to an over emphasis on mechanical treatments to achieve desired open forest canopy conditions at the expense of plant and wildlife diversity.

Box 6. § 219.3 Role of science in planning. The responsible official shall use the best available scientific information to inform the planning process required by this subpart. In doing so, the responsible official shall determine what information is the most accurate, reliable, and relevant to the issues being considered. The responsible official shall document how the best available scientific information was used to inform the assessment, the plan decision, and the monitoring program as required in §§ 219.6(a)(3) and 219.14(a)(4). Such documentation must: identify what information was determined to be the best available scientific information, explain the basis for that determination, and explain how the information was applied to the issues considered. (emphasis added)

The DEIS does not sufficiently meet the *conservation* requirement of the 2012 forest planning rule (Box 7).

Box 7. Conservation. The protection, preservation, management, or restoration of natural environments, ecological communities, and species. Conserve. For purposes of subpart A, § 219.9, to protect, preserve, manage, or restore natural environments and ecological communities to potentially avoid federally listing of proposed and candidate species.

Noted deficiencies in the *conservation* requirement include:

- Lack of preservation and protection of natural environments (especially roadless areas, closed canopy mature forests, riparian areas, critical MSO habitat), ecological communities, focal species, at-risk species, and species of conservation concern. Alternative 2, for instance, emphasizes extensive mechanical treatments that may cause irreparable harm to MSO, focal species, at-risk species, and species of conservation concern through major reductions in canopy closure and understory vegetation. Extensive thinning of forest canopies may constitute an adverse effects determination in section 7 consultation for the MSO (and prey species) that uses closed canopy forests for nesting and may use severely burned areas for foraging (see Lee 2018).

Importantly, the DEIS presents a questionable analysis of fire emissions derived from assumptions in the LANDFIRE program and does not include an appropriate analysis of the emissions from logging (in-boundary and transportation/manufacturing of wood products), livestock grazing and infrastructure, and road building/maintenance. An emissions analysis related to all project activities is necessary to properly assess air quality impacts to surrounding communities and CO_2 contributions to climate change with an alternative chosen that minimizes emissions related specifically to forest plan activities (direct, indirect, cumulative emissions impacts).

In sum, I am requesting that the Forest Service modify or include a new alternative that meets the following requirements:

- Identification and protection of specific connectivity areas (e.g., roadless areas, intact riparian and watersheds) for achieving viable populations of focal species, species of conservation concern, and at-risk species in a changing climate (see Noon et al. 2003, Schulz et al. 2013, Haber and Nelson 2015, especially Table 1). Such areas should be protected from mechanical treatments especially habitat of at-risk species (e.g., MSO).
- Consistent with the guidelines for connectivity in Haber and Nelson (2015:Tables 1 and 2), it is essential for the forest plan to identify key characteristics of connectivity (also

Haber and Nelson 2015:Table 3), including composition, structure, process/function at scale: site, landscape, corridors, riparian areas, and wildlife travel routes.

- An analysis and mitigation of how conditions on the SFNF and surrounding areas (logging, roads, development, grazing especially in riparian areas) affect connectivity (cumulative effects analysis).
- Substantial reduction in livestock AUMs and increase in riparian, native meadows, and aspen grove protections, restoration and invasive species containment. This should include opportunities for local conservation groups to purchase grazing leases from willing sellers with the allotments and AUMs permanently retired by the Forest Service. Livestock should be removed from riparian areas and curtailed in areas with native plant communities.
- Accuracy determination and field verification (and error correction) of LANDFIRE and forest canopy determinations, particularly in relation to site-specific reference conditions and ecologically appropriate definitions of closed canopy forests; the >30% closed canopy definition in the DEIS is arbitrary and has resulted in excessive canopy reduction measures to achieve "open" conditions.
- Use of multiple lines of evidence (e.g., see Odion et al. 2014a, Moritz et al. 2018) in estimating historic fire regimes and recognition/ analysis of the importance of mixed-severity fire effects on plant and wildlife diversity, including small and large patches of high-severity fire effects characteristic of drought cycles, fire flare ups, and upper elevation forests.
- A substantial reduction in mechanical treatments that are otherwise resulting in novel forest conditions that lack integrity and climate resilience because of the over-emphasis on open forest conditions that retain few trees. Forests opened by excessive thinning lack understory shrubs, forbs and small trees that contribute to climate resilience (see Baker and Williams 2015, Baker 2018); small trees may also have mature/old growth characteristics because of slow growth rates and more of them need to be maintained as an important understory cohort for future old-growth development (e.g., by creating small gaps and leaving many more tree cohorts).
- A focus on community wildfire risk reduction through partnerships with private landowners that emphasize defensible space measures for homes instead of extensive thinning in the backcountry.
- A substantial reduction in livestock grazing including large no-grazing zones that more aptly address cumulative effects of livestock, infrastructure, and climate change (see Beschta et al. 2012).
- A reduction in project related carbon dioxide emissions by a project level comparison of emissions alternatives.

My detailed comments and supporting publications follow this signature page.

Sincerely,

Dominick A. DellaSala, Ph. D
Independent Conservation Scientist

UNCERTAINTIES OF FIRE SCAR METHODOLOGY, REFERENCE CONDITIONS, AND FAILURE TO MEET BASI REQUIREMENTS OF THE PLANNING RULE

The 2012 planning rule requires forest plans to meet the best available scientific information (BASI) standard during planning assessments. Ryan et al. (2018) provide specifics on how best to meet this standard illustrated in their Figure 2:

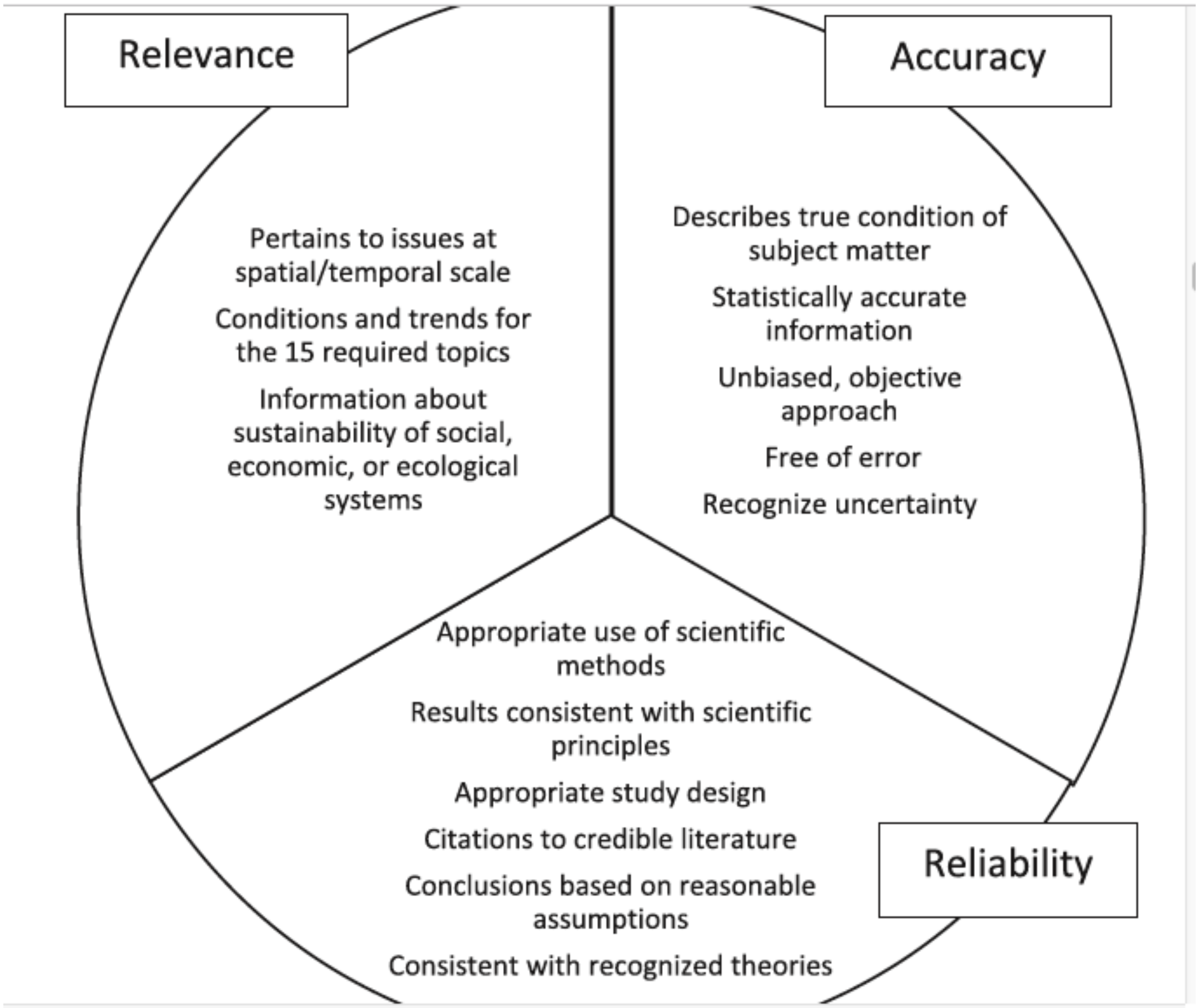
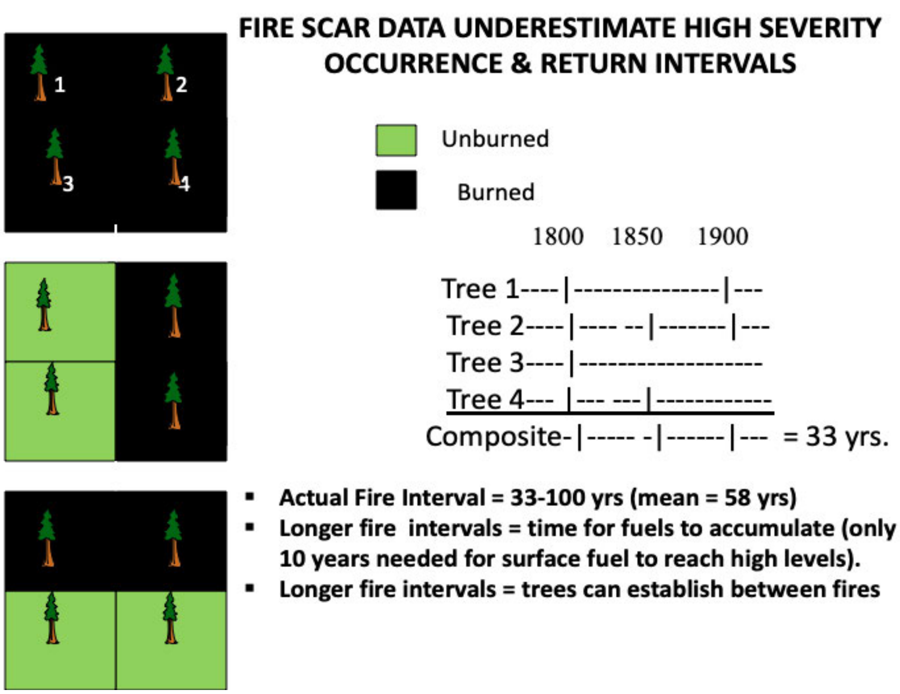


Figure 2 (from Ryan et al. 2018). Criteria for determining best available scientific information (BASI). Source: Forest Service Handbook 1909.12.07.12

According to Ryan et al. (2018) "the definition of BASI is contained in the "zero code" chapter of the handbook and specifies three primary criteria: accuracy, reliability, and relevance (FSH 1909.12.07.12), in addition to referencing the Data Quality Act (PL 106–554) for guidance on evaluating available information (Figure 2). Available is defined as information that currently exists in a form useful for the planning process without further data collection, modification, or validation (FSH 1909.07.01).

Based on the BASI standard above (especially Ryan et al. 2018: Figure 2), there are two main problems with the DEIS fire assumptions: (1) over reliance on fire scar estimates used to determine fire return intervals that are then extrapolated over large areas instead of the more appropriate use of multiple lines of evidence used to calculate *fire rotation intervals* (landscape scale; see Odion et al. 2014a, Moritz et al. 2018); and (2) accuracy and reliability problems with use of LANDFIRE to estimate reference and contemporary conditions in forest plan analyses (discussed below).

Fire return intervals are biased - While local sampling is important for estimating fire return intervals at the stand level, there are significant uncertainties with extrapolating fire scar point sampling data over large landscapes often used by researchers to re-construct historic fire regimes for comparisons to contemporary conditions (Baker 2017). They include sample-site selection bias, lack of tree scars in fire-killed trees (thereby underestimating high severity occurrence), and inappropriate extrapolation of site-specific data to draw landscape-level inferences (Baker 2017). The hypothetical figure below illustrates the inherent sampling bias of grouping individual fire scar data to construct composite fire interval (mean CFI).



In sum, variability in CFI estimates is masked whenever the mean return interval is used (instead of the range or scope-of-inference is inappropriately extrapolated from sites to large areas and whenever measures of central tendency (rather than the range) are used in fire return intervals.

This results in a cascade of errors beginning with a bias toward very short fire return intervals (i.e., because the mean and not the range was used), conclusions that historic conditions were predominately open forests (especially when open is arbitrarily defined using LANDFIRE, see below), conclusions that contemporary forest conditions are way out of bounds, and, the inappropriate need for aggressive mechanical treatments. To fix this problem, the best estimator of fire intervals at landscape scales is to use the fire rotation interval (Baker 2017).

Baker (2017) notes that fire rotations at the landscape scale can be derived from:

1. Areas burned in recent fires from agency fire records or records from remotely sensed data.
2. Historical areas burned reconstructed from scarred trees or plot locations.
3. Historical areas burned reconstructed using a ratio method and scarred-tree or plot records, or comparable data in a table or graph.

The Forest Service must provide information on the fire rotations using methodologies in Baker (2017) supplemented wherever possible with the paleo-ecology literature that can be used to reduce sampling bias associated with shorter sampling timelines and lack of high severity detectability from fire scar extrapolations. For instance, Baker (2017) goes through each source of bias in tree-ring reconstructions and shows that using corrected estimators actually yields longer fire rotation periods for dry pine/mixed conifer areas. Note that Figure 3 and Figure 4 in Baker (2017) show the diversity of fire rotations (longer intervals) in the Santa Fe area and the S2 Table has individual estimates for New Mexico. The sampling bias in fire-scar data must be disclosed as the DEIS is based mainly on fire-scar interpolation from plots to landscapes thereby compounding errors.

To correct for sampling bias, the Forest Service must account for variability in fire-free intervals using more robust methodologies, disclose whether there are historic accounts of fires in the DEIS area beyond just fire-scars, and include paleo-ecology studies from comparable sites to illustrate variability in fire regimes over longer time intervals. Significant discrepancies and debate among researchers about fire scar sampling must be disclosed (e.g., see Odion et al. 2016 response to Stephens et al. 2016 and Moritz et al. 2018).

As an example, a key fire-history study for the Santa Fe watershed is Margolis and Balmat (2009). These researchers indicate that the historical low-severity fire rotation in this watershed for dry pine forests was estimated at 39.80 years. They define frequent fire as < 25 years. Using their definition means that the Santa Fe watershed would not qualify as a frequent-fire regime, as this is a sufficient mean number of years between surface fires to allow understory fuels including shrubs and small trees to accumulate levels that would certainly enable the occurrence of some mixed and high-severity fires and some dense forests overtime. Moreover, this longer period corresponds with the paleo-record from charcoal sediments showing that when wet periods are followed by successive droughts, large fires, including patches of high severity, do indeed occur (Meyer 2010).

It is important to accommodate this variability in fire return interval estimates as heterogeneity in the ensuing burn severity patches at the landscape scale results in high levels of biodiversity (i.e., pyrodiversity of fire severity patches begets biodiversity, DellaSala and Hanson 2015). Notably, even slight differences in fire-return intervals are consequential. Baker (2017) reports that understory fuels in dry forests recover after fires in 7-25 years. If mean fire-return intervals were <25 years, understory fuels would be limited. However, if the interval was >25 years, as reported by Margolis and Balmart (2009), then shrubs and small trees would recover across the landscape and excessive thinning to shift forest to more open-canopy forests with minimal small tree and shrub cover would be inappropriate at large spatial scales.

The role of shrubs and understory vegetation is also a key ecosystem component in dry forests allowing for nutrient cycling and below-ground processes, water absorption and retention, provision of wildlife habitat, pollination and other ecosystem services. Spatial heterogeneity in fire-return intervals at landscape scales is a key indicator of resilience as it allows for both fire refugia (longer return intervals) and fire-maintenance (short return intervals). It is essential to manage for this variability at the site and landscape scale to accommodate wildlife that require a range of severity effects on vegetation: low, moderate and high severity. In other words, when it comes to fire, nature is complex (e.g., mixed severity) while management tends for uniformity (mainly low severity) typically at the expense of fire-mediated biodiversity.

The following Baker (2017) observations about fire interval estimators need to be addressed in the DEIS:

> "Dry-forest landscapes until recently were thought to have historically been primarily old growth forests, with a history of frequent low-severity fire, across their extent (e.g. [72]), but this has been refuted by GLO reconstructions and early aerial photographs (Table 6), paleoecological evidence [24], and early forest-reserve reports and other evidence [63 , 73]. Even in Arizona, which had abundant old forests with frequent fire (Fig 3), denser forests and high severity fire were extensive at certain times and in certain places, as on Black Mesa and parts of the Mogollon Plateau [60 , 73]. It is sensible to restore low-severity fire to its former dominance in the parts of dry-forest landscapes with a history of primarily low-severity fire, historically averaging about 34% of western dry-forest landscapes (Table 6). Estimated mean PMFI/FRs [population mean fire interval/fire rotation] here provide a guide for restoration and management of low-severity fire in extant old-forest parts of landscapes. For most dry-forests today, which are not old, using frequent fire (PMFI/FR <25 years) in restoration is not supported, and fuels do not need to be substantially reduced, because historical PMFI/FRs naturally allowed historical shrubs and small trees to fully recover after fires. Restoration of low-severity fire is still

> needed. The most appropriate approach, given likely long but uncertain mean rates of historical low-severity fire, is for most dry forests today to receive at most one prescribed fire, followed by managed fire for resource benefit, with the goal of mimicking mean historical PMFI/FRs and variability in fire (fire-size distributions, unburned area) as forests reach old age."

Thus, based on Baker (2017) and the problems noted in estimating fire return intervals, the DEIS needs to greatly scale back thinning except where thinning of small trees is needed to re-introduce fire nearest homes.

Problems with GTR-310 reference conditions - The DEIS tiers to GTR-310 (Restoring Composition and Structure in Southwest Frequent Fire Forests). However, GTR-310 does not even align with the geographic scope of the SFNF, as the SFNF is within the Colorado Rockies Forest Ecoregion yet GTR-310 is predominately within the Arizona Mountain Forest Ecoregion, which has a different climate, soil types, historical conditions, and fire regime. Extrapolating from one region to another is inappropriate (Moritz et al. 2018) and thus GTR-310 cannot be relied on for project-specific descriptions or actions.

The DEIS relies upon General Technical Report 310 as a primary source for desired conditions in the SFNF. This is inappropriate because none of the reference studies were from the Sangre de Cristo Mountains, and the two locations in the Jemez Mountains represent just 12 acres of sampled forest. The DEIS should instead rely on site-specific reference conditions and exercise caution when extrapolating fire regimes and forest structures from one geographic location to another given differences in vegetation, fire rotation intervals, elevation gradients, regional climate, and the influence of a rapidly changing climate on contemporary and future fire conditions (see Moritz et al. 2018). Thus, applying the "Flagstaff fire model" derived from GTR-310 is completely inappropriate for the SFNF.

ACCURACY PROBLEMS WITH LANDFIRE NEED FULL DISCLOSURE AND CORRECTION

Fire regime condition class (FRCC) and LANDFIRE vegetation models and maps are used by the Forest Service in planning assessments. These approaches are useful at large spatial scales (national) but they have well known accuracy problems at the project level that need full disclosure, cross validation with field data, and error correction.

For instance, Swetnam and Brown (2010) examined accuracy of LANDFIRE and FRCC assessments in Utah for similar vegetation types as in the DEIS planning area (Box 7).

Box 7. "LANDFIRE map data were found to be ~58% accurate for BpS and ~60% accurate for existing vegetation types. Results suggest that limited sampling of age-to-size relationships by different species may be needed to help refine reference condition definitions used in FRCC assessments, and that more empirical data are needed to better parameterize FRCC vegetation models in especially low-frequency fire types."

Zhu et al. (2006) tested the vegetation mapping protocol of LANDFIRE and likewise *concluded mapping accuracies of 60% or better at 30-m spatial resolution*. Notably, such low accuracy determinations do not comport with Ryan et al. (2018) summary of BASI criteria (their Figure 2 above) and the intent of the Data Quality Act.

Helmbrecht and Blankenship (2016) tested the ability of LANDFIRE to accurately reflect the true or accepted geographic location of features finding problems with errors in feature locations, source data, precision of field measurements, and data entry. Problems in map unit assignments may arise through "*errors or limitations in remote sensing data, field plots, statistical modeling, processing logic, or a combination of these and other factors*" (emphasis added). This is especially the case for forest vegetation and fuels data depending on the age of the source data. For instance, LANDFIRE data are updated every two years but by the time the data are delivered to the user, the data are 3 or more years out of date.

To correct for these problems, Helmbrecht and Blankenship (2016) recommend (and the DEIS should as well) include the following:

1. update for landscape changes that have occurred since the LANDFIRE version,
2. calibrate to local data and knowledge,
3. improve the thematic agreement (accuracy),
4. change the spatial or thematic resolution (e.g. lump or split map units),
5. modify the map unit classification,
6. create additional data versions that reflect temporal variability (e.g. peat soils being available for burning in drought situations, or exotic annual grasses being present in wet years but not dry years),
7. facilitate comparative analysis by creating data versions (e.g. analyzing pre- and post-treatment effects or comparing treatment alternatives),
8. analyze future conditions (e.g. modifying data to represent future conditions under a climate change scenario).

In Northern Idaho, Hyde et al. (2015) evaluated two LANDFIRE fuel loading raster options: (1) Fuels Characteristic Classification Systems (LANDFIRE-FCCS); and (2) Fuel Loading Model (LANDFIRE-FLM) vs. measured fuel loadings for a 20,000-ha mixed conifer study area. They found that the LANDFIRE-FCCS layer showed 200% higher duff loadings relative to measured loadings that led to 23% higher total mean consumption and emissions when modeled in FOFEM. The LANDFIRE-FLM layer showed lower loadings for total surface fuels relative to measured data, especially in the case of coarse woody debris that led to 51% lower mean total consumption and emissions when modeled in FOFEM. Additionally, LANDFIRE-FLM consumption was *59% lower relative to that on the measured plots, with 58% lower modeled emissions*. The authors concluded that these differences in fuel loadings led to significant differences in consumption and emissions depending upon the data and model chosen. The DEIS therefore needs to disclose how errors in fuel loading consumption were addressed in emissions determinations regarding wildfires and how these errors were corrected.

In the Sierra Nevada region, Odion and Hanson (2006) found FRCC *was not able to accurately predict occurrence of high-severity fire* (Box 8).

Box 8. "We found that the proxy for fire suppression effects, Condition Class, was not effective in identifying locations of high-severity fire. Condition Classes 2, 3, and 3+ in the McNally fire all had similar fire severity proportions. When the same Condition Class criteria were applied to the other two fires, we found that fire severity generally decreased rather than increasing from Condition Class 2 to 3+. In short, Condition Class identified nearly all forests as being at high risk of burning with a dramatic increase in fire severity compared to past fires. Instead, we found that the forests under investigation were at low risk for burning at high-severity, especially when both spatial and temporal patterns of fire are considered. The lack of an observed relationship between Condition Class and fire severity suggests that exogeneous forces such as weather, climate, topography, and neighboring vegetation (for example, pyrogenic shrubs) largely determine fire-severity patterns in forests."

Vogelmann et al. (2014) recognized four major potential sources of error associated with field plot data used in LANDFIRE:

1. Errors occur frequently in the identification of species and measurement of vegetation structure in the field (for example, in the data for one prototype field plot, a misplaced decimal point indicated a shrub height of 60 feet).
2. The vegetation on some field plots has undoubtedly changed between the time the field data were collected and when the imagery was acquired.
3. Geo-location errors in plot and imagery data result in inaccurate characterization of some imagery pixels.
4. The assignment of plots to specific vegetation classes will have errors associated with the wide array of opinions among professional field ecologists regarding the field classification of any given field plot.

To correct for these problems, Vogelmann et al. (2014) suggest (and the DEIS should follow) that the Forest Service conduct a suite of accuracy assessment methods for LANDFIRE, ranging from mostly qualitative assessments (such as the critical inspection of products, consultation with regional experts, and comparisons with existing data sets) to more quantitative analyses (such as cross-validation assessments, traditional accuracy assessments at the superzone level, and select evaluations at local levels). These combined approaches will provide the Forest Service with the accuracy information necessary to facilitate the appropriate use of the data in the DEIS.

Cruz and Alexander (2010) note additional problems with related fire modeling summarized in their abstract. The Forest Service needs to disclose errors associated with use of these models in the DEIS:

Abstract. To control and use wildland fires safely and effectively depends on creditable assessments of fire potential, including the propensity for crowning in conifer forests. Simulation studies that use certain fire modelling systems (i.e. NEXUS, FlamMap, FARSITE, FFE-FVS (Fire and Fuels Extension to the Forest Vegetation Simulator), Fuel Management Analyst (FMAPlus!), BehavePlus) based on separate implementations or direct integration of Rothermel's surface and crown rate of fire spread models with Van Wagner's crown fire transition and propagation models are shown to have a significant underprediction bias when used in assessing potential crown fire behavior in conifer forests of western North America. The principal sources of this underprediction bias are shown to include: (i) incompatible model linkages; (ii) use of surface and crown fire rate of spread models that have an inherent underprediction bias; and (iii) reduction in crown fire rate of spread based on the use of unsubstantiated crown fraction burned functions. The use of uncalibrated custom fuel models to represent surface fuelbeds is a fourth potential source of bias. These sources are described and documented in detail based on comparisons with experimental fire and wildfire observations and on separate analyses of model components. The manner in which the two primary canopy fuel inputs influencing crown fire initiation (i.e. foliar moisture content and canopy base height) is handled in these simulation studies and the meaning of Scott and Reinhardt's two crown fire hazard indices are also critically examined.

DellaSala et al. (2015) further summarize the problems with fuel models and simulations not comporting with field data and resulting in over-emphasis of efficacy of fuel reduction treatments and these uncertainties need to be addressed in the DEIS as follows:

"Fuel reduction also has been overpromised to be effective, using questionable logic and unvalidated models. First, fire intensity in most forest types is much more strongly affected by wind than by fuel. High fire-line intensity, the primary fire characteristic that promotes crown fires, is the product of the energy released by burning fuel and the rate of spread of fire (Alexander, 1982). Energy release by fuel varies over perhaps a 10-fold range, however, whereas rate of spread can vary over more than a 100-fold range; thus a high rate of spread caused by strong winds can easily overcome the limited reductions in fuel that are feasible (Baker, 2009). This was confirmed by a recent analysis of the 2013 Rim Fire in California, which concludes: "Our results suggest that even in forests with a restored fire regime, wildfires can produce large-scale, high-severity fire effects under the type of weather conditions that often prevail when wildfire escapes initial suppression efforts. . . . During the period when the Rim fire had heightened plume activity... no low severity was observed [in thinned areas], regardless of fuel load, forest type, or topographic position" (Lydersen et al., 2014, p. 333). Second, common fire models used to show that forests would be fire-safe after fuel reductions have an underprediction bias and are not validated. These flawed models include NEXUS, FlamMap, FARSITE, FFE-

FVS, FMAPlus, and BehavePlus (Cruz and Alexander, 2010; Alexander and Cruz, 2013; Cruz et al., 2014). The underprediction bias means that these models often predict that fuel reductions would reduce or eliminate the potential for crown fires in forests, when in fact fuel reductions do not achieve this effect. Fixing these models would be difficult and has not yet occurred (Alexander and Cruz, 2013). Also, these models have not been sufficiently tested and validated using a suite of actual fires, in which case they would likely be shown to fail (Cruz and Alexander, 2010). Alternative validated models are available and could be further developed, but they are not being used (Cruz and Alexander, 2010). Further, studies of tree mortality in thinned areas following fire do not typically take into account the mortality caused by the logging itself before the fire, leading to further biased results."

As further noted by DellaSala et al. (2015) "these concerns should raise red flags about the effectiveness of fuel treatments, as well as issues regarding liability and responsibility. Imagine if a company sold airplanes with identified flawed designs and without adequate test flights, which then crashed. There are thus sound scientific reasons to closely scrutinize government wildland fuel-reduction programs. Meanwhile, we need to be honest and warn the public that living within or adjacent to natural forests prone to burn is inherently hazardous. Only treating fuels in the immediate vicinity of the homes themselves can reduce risk to homes, not backcountry fuel reduction projects that divert scarce resources away from true home protection (Cohen, 2000; Gibbons et al., 2012; Calkin et al., 2013; Syphard et al., 2014)."

Closed Canopy Conditions Arbitrarily Defined - the DEIS arbitrarily defines closed canopy conditions in the mixed conifer-frequent fire and ponderosa pine ERUs as when *woody cover exceeds 30%* (DEIS: Figure 14, Figure 16), using LANDFIRE to determine the reference/baseline condition and contemporary departure indices for alternative analyses. The preferred alternative is based on moving closed canopy forests into desired open canopy conditions over 50 years. Closed canopy forests in some cases currently exceed 70% overstory cover and thus extensive thinning in the preferred alternative constitutes a major change in overstory cover impactful to species requiring closed canopy conditions. Large interspaces will be created across the landscape with substantial reductions in canopy cover and percent of forests in closed conditions to meet this arbitrarily defined "open" reference condition, creating novel ecosystems that do not comport with the ecological integrity or diversity requirements of the planning rule.

Importantly, Scott (2008) documented seven potential shortcomings with the canopy and fuel related provisions of LANDFIR, including:

- canopy cover values are too high,

- data discontinuities exist within and between map zones,
- canopy bulk density values are too low for use in FARSITE,
- canopy base height is too high to generate crown fire,
- treelist data sources may not be best for canopy fuel calculations
- alternative canopy fuel calculation programs may produce different results
- Refreshing and calibrating LANDFIRE data is needed

Scott (2008) reported that the dead fuel moisture model is especially sensitive to errors in canopy cover and concludes:

"Moreover, canopy cover mapping errors may lead to significant indirect fire modeling effects. Because canopy cover is a keystone variable, these indirect effects are difficult to quantify. If canopy cover is overestimated, LANDFIRE may subsequently map the incorrect fuel model, incorrect CBD, incorrect CBH, etc., all of which can strongly affect fire modeling outputs in a geospatial fire analysis."

"Because it is used as an independent variable, the importance of an accurate canopy cover layer in the LANDFIRE process should not be underestimated."

THE DEIS NEEDS TO RECOGNIZE THE ECOLOGICAL IMPORTANCE OF MIXED-SEVERITY FIRES, INCLUDING LARGE AND SMALL HIGH SEVERITY PATCHES FOR POSITIVE CONTRIBUTIONS TO PLANT AND WILDLIFE DIVERSITY

While low elevation pine and mixed conifer forests are predominately maintained by frequent fires of low severity effects on vegetation, there are occasional canopy flare ups and high-severity patches related to local fire-weather conditions, slope, aspect, elevation, and vegetation condition. This variability in fire effects needs to be maintained under the diversity requirement of the planning rule. Instead, the DEIS includes no analysis of the positive effects of mixed-severity fire influences on plant and wildlife communities, especially in upper elevation forests where fires are on centuries long rotation intervals (landscape scale) and produce diverse ecosystem effects including the creation of complex early seral forests (Swanson et al. 2011).

Notably, high-severity fire patches generate "biological legacies" (large live and dead trees, logs, shrubs) essential to forest succession and the maintenance of complex early seral forest conditions (Swanson et al. 2011, DellaSala et al. 2014, DellaSala and Hanson 2015, DellaSala and Hanson 2019). Large and small high severity patches provide important foraging habitat for Mexican Spotted Owls (federally listed, Lee 2018), Northern Goshawks (at-risk species), ungulate foraging habitat (Bond 2015), snowshoe hare/lynx dynamics, woodpeckers (including at-risk species: Lewis's woodpecker) and songbirds (Hutto et al. 2015), bats (Chambers and Saunders 2013), and boreal owls (at-risk species) in upper elevation spruce-fir forests. The DEIS inappropriately and arbitrarily assumes high-severity patches constitute wildlife habitat degradation (e.g., DEIS Volume 1: Tables 51, 57; "catastrophic fire analysis" p. 244).

Using LANDFIRE, the DEIS inappropriately assumes that current fire return intervals are highly departed from reference conditions (86%) as is fire severity leading to what the DEIS claims is a departure from NRV (DEIS Volume 1:89). However, based on a study of high-severity patches in dry pine and mixed conifer forests across the West, including New Mexico, large (>400 ha) high-severity fire patches have not been increasing since the 1990s (DellaSala and Hanson 2019.

From DellaSala and Hanson (2019):

Over the entire time series, 1984-2015, there was a significant increasing trend in the combined total area of CESF [complex early seral forests] patches >400 ha in each year (τ = 0.407, p = 0.001), but no trend in patch size (τ = 0.009, p = 0.802). However, when the data were analyzed by time periods, there was only a significant difference in the annual area of CESF habitat created by high-severity fire relative to the earliest time period (1984-1991), but no significant differences were detected among time periods since the early 1990s (Table 1, Figure 2). With regard to size of individual large CESF patches, there were no significant differences detected among time periods (Table 2).

Table 1. Critical values ($q_{0.05, 4}$), absolute difference between mean of ranks ($|R_A\text{-}R_B|$), standard errors (SE), and test statistics (q) to assess statistical significance, at α = 0.05, of any differences between the four time groups ("1" = 1984-1991, "2" = 1992-1999, "3" = 2000-2007, and "4" = 2008-2015) for total annual area of CESF patches >400 ha using the Nemenyi non-parametric test for multiple comparisons between groups with an equal sample size (n = 8 years for each time group). Statistical significance of levels of q are shown as "Y" (significant) or "N" (not significant).

Time group comparison	$q_{0.05,4}$	$\lvert R_A\text{-}R_B \rvert$	SE	q	Significant? ($q > _{0.05,4}$?)
1-2	3.63	45.0	26.53	1.70	N
1-3	3.63	108.0	26.53	4.07	Y
1-4	3.63	107.0	26.53	4.03	Y
2-3	3.63	63.0	26.53	2.37	N
2-4	3.63	62.0	26.53	2.34	N

3-4	3.63	1.00	26.53	0.04	N

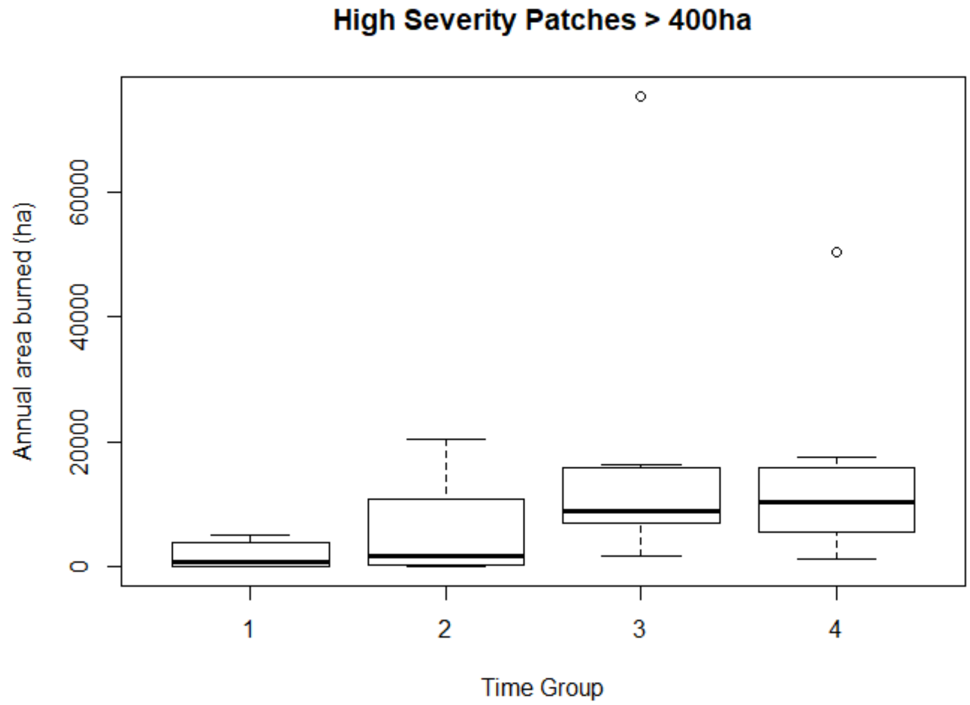


Figure 2 from DellaSala et al. Annual area of large patches (>400 ha) of high-severity fire in the four time periods ((“1” = 1984-1991, “2” = 1992-1999, “3” = 2000-2007, and “4” = 2008-2015).

Thus, the DEIS claims about uncharacteristically severe fires, for which mechanical treatments are based upon, cannot be substantiated by empirical data (including from New Mexico) and thus the DEIS does not meet the BASI requirements.

Importantly, Hutto et al. (2016) recommended that managers maintain ecological integrity of western dry pine and mixed-conifer forests through a more informed approach to the importance of mixed and high-severity fires. Here is their abstract:

Abstract. *We use the historical presence of high-severity fire patches in mixed-conifer forests of the western United States to make several points that we hope will encourage development of a more ecologically informed view of severe wildland fire effects. First, many plant and animal species use, and have sometimes evolved to depend on, severely burned forest conditions for their persistence. Second, evidence from fire history studies also suggests that a complex mosaic of severely burned conifer patches was common historically in the West. Third, to maintain ecological integrity in forests born of mixed-severity fire, land managers will have to accept some severe fire and maintain the integrity of its aftermath. Lastly, public education messages surrounding fire could be modified so that people better understand and support management designed to maintain ecologically appropriate sizes and distributions of severe fire and the complex early-seral forest conditions it creates.*

DellaSala et al. (2017) recommend that managers include mixed-severity effects in dry pine and mixed conifer forests to achieve ecological integrity and plant diversity. And while much of the

DEIS project area can be assumed to be in a xeric pine condition, mixed-severity fire effects, including large and small high-severity patches are indeed characteristic, need to be maintained, and are being grossly underestimated in ecological importance throughout the DEIS. Thus, the DEIS does not meet the BASI requirements of the planning rule as well as the diversity, ecological processes, ecological conditions, and integrity provisions as noted.

BIASED APPROACHES, AREAS OF AGREEMENT & DISAGREEMENT NEED TO BE ACKNOWLEDGED AND CORRECTED

Bias: The DEIS needs to clearly state scientific disputes (disagreements) and avoid biased perspectives on fire as generally noted by Iftekhar and Pannell (2015) and Moritz et al. 2018 (below). The following biased perspectives are inherent in the DEIS:

- Action bias – tendency to take actions even when it is better to delay action (in this case the impacts of aggressive thinning and roads may be more significant than effects of fire on ecosystems given uncertainties of treatment effectiveness as noted).
- Framing effect – tendency to respond differently to alternatively worded but objectively equivalent descriptions of the same item (use of catastrophic fire terminology in the DEIS that fails to account for ecosystem benefits of mixed-severity fires, including periodic flare-ups of high severity patches).
- Reference-point bias – tendency to overemphasize a pre-determined benchmark for a variable when estimating the level of that variably (i.e., over-reliance on fire scar sampling in the DEIS rather than presenting more robust and multiple lines of evidence).
- Satisficing rule – tendency to stop searching for a better decision (i.e., a NEPA based range of alternatives) once a decision that seems sufficiently good is identified.
- Loss aversion – tendency to value losses more highly than similar gains (i.e., managing wildfire of moderate-high intensity for ecosystem benefits instead of avoiding it by mechanical thinning and fire suppression as in the DEIS).
- Limited reliance on systematic learning – tendency to use information from past successful efforts rather than using information from both successful and failed efforts via extensive and well-funded ecosystem monitoring (adaptive management and learning is not possible without well-funded monitoring).

The best way to avoid these biases is to use multiple lines of evidence in re-constructing fire regimes, not rely mainly on fire scars, and conduct well-funded monitoring studies that fully assess project effects on species of conservation concern and ecological and cultural values. Multiple lines of evidence and monitoring are discussed in Odion et al. 2016 and Moritz et al. (2018) in the Common Ground Report (see below).

Areas of Agreement/Disagreement (Common Ground): I participated as one of the respondents in the so-called "Common Ground" report and am thoroughly familiar with the report's findings. The DEIS should pay particular attention to the following key findings in relation to areas of agreement, uncertainty, and disagreement and adjust project actions accordingly.

Areas of Agreement (high certainty):

- The role of changing climatic conditions is increasingly important in influencing fires.
- Multiple fire ecology and fire history research can be useful.
- Heterogeneity of fire effects, including patterns of patches created by fires of all severities, is important to forest resilience to future fires.
- Generalized models of historical fire regimes vary by ecoregion and forest type.
- Even within the same ecoregion and forest type, there is variation in historical fire regimes among differing environmental gradients.
- Historically, some degree of low-, moderate-, and high-severity fire has occurred in all forest types, but in substantially different proportions and patch size distributions at different locations.
- Classification of historical fire regimes according to forest types can be coarse; thus, failure to recognize variation of historical fire regimes *within* forest types can lead to overgeneralization and oversimplification of landscape conditions.
- Presence of roads, road density and railways, livestock grazing, invasive species, and mining can alter fire regimes. Even a single one of these influences can have profound effects on vegetation and fire behavior conditions. When present in combinations, cumulative effects will arise that may push ecosystems past tipping points (Paine et al. 1998, Lindenmayer et al. 2011).
- Knowledge of historical range of variability (HRV) is useful but does not dictate land management goals. HRV findings from one area may or may not have relevance elsewhere.
- Recent trends in many western forest regions of more large fires and more area burned are linked to recent climatic trends of hotter droughts and longer, more severe fire seasons.
- Respondents who emphasized the longer time scales of charcoal records noted that most areas of predominantly low-severity fires showed some incidence of moderate- or high-severity fire over longer time frames.
- It is desirable to use multiple methods to reconstruct historical fire regimes. More can be learned using multiple approaches and considering data from diverse temporal and spatial scales.

- Importance of local context in management of fire-prone landscapes underscores the need to move away from oversimplified narratives that encourage application of fire research beyond its original scope of inference. Note: the scope of inference is of particular concern here as over reliance on fire scar sampling for landscape scale interpolation has inherent biases and uncertainties.

Areas of Disagreement (high uncertainty):

- Fire regime inferences from historical and modern tree inventory data, simulation models, and other approaches have levels of uncertainty.
- Whether large, high-severity fires have increased to a significant and measurable degree in all forest types *in comparison to historical fire regimes* (i.e., prior to modern fire suppression) remains debatable.
- Fuel treatments are urgently needed across nearly all forests remains debatable.
- Fuel treatments should be focused around communities and plantations; but hazard fuel reduction elsewhere remains debatable.
- There is high uncertainty about where and when fuel treatments are beneficial.
- Commonly used vegetation classification schemes as a suitable basis for generalizing about fire regimes remains debatable. Known geographic variation in fire regimes within forest types argues for improved forest and fire regime classifications.
- Tree-ring evidence sometimes supports conclusions that contrast with those derived from landscape-scale inventory and monitoring data using different sampling frames creates uncertainty.
- General applicability of "thinning and prescribed burning remedies" to offset human influences is debatable.
- Human impacts on forest successional conditions in moist and cold forests remains debatable.
- Extent to which landscape tipping points have been reached as a result of high-severity fires is debatable.
- Effectiveness of fuel treatments under projected climate futures and associated more extreme fire weather is uncertain.
- Interpretation of any research evidence and the scope of related inferences is limited by scaling (uncertainty) and sampling concerns associated with the methods, and these limitations apply to all research methods.
- All methods for reconstructing historical fire regimes are necessarily indirect and have degrees of uncertainty. They may include, but are not limited to, interpreting evidence of past fires or the extent of fire-dependent ecosystems from historical documents, land surveys, aerial photographic reconstructions, fire-scar and growth-release data from tree rings, tree age and death dates from tree-ring data, climatic data linked with past fires,

charcoal and pollen deposits, current characteristics of stands (i.e., structure, species, and stand age distribution), fire perimeter mapping, historical timber survey data, and use of statistical distributions for modeling stand-replacing fire.

ROAD IMPACTS AND ROADLESS AREA IMPORTANCE NEED TO BE ANALYZED TO COMPLY WITH CONNECTIVITY REQUIREMENTS OF THE PLANNING RULE

Roads – Given the extensive and cumulative impacts of roads on ecosystem processes, wildlife, water quality, and fire ignitions (see below), *a minimum road density analysis* needs to be conducted to assure the public that there are no excessive roads and that more roads can and should be decommissioned and obliterated rather than improving and building more roads. The DEIS needs to provide a transportation plan analysis to fully assess road-related fire ignitions associated with improved access and to come up with an alternative that reduces them.

Simply improving culverts and surfacing primitive dirt roads with poor drainage also may not be enough to improve water quality. Notably, the DEIS provides no information on Clean Water Act 303d water-quality limited streams and how project-related impacts will be minimized to comply with state and federal water quality standards[2]. Water quality must be assessed in relation to road improvements, greater road access, thinning impacts, and road-stream intersections.

In sum, the DEIS needs to fully disclose road-related impacts as follows:

- Roads and thinning contributions to soil erosion and sediment inputs affecting water-quality even when roads are improved.
- Probability of human-caused wildfire ignitions associated with improved road access (see Balch et al. 2017 for human-caused ignitions, pdf enclosed).
- Fragmentation and degradation of wildlife habitat at road densities > 1 mi/sq mi, particularly impacts to large carnivores and aquatics.
- Spread of invasive species and their effects on fire regimes.
- Likelihood of mass-wasting events on steep erosive slopes along the road prism.

2Particularly in relation to EPA standards see https://nepis.epa.gov/Exe/ZyNET.exe/00001O9W.TXT?ZyActionD=ZyDocument&Client=EPA&Index=1986+Thru+1990&Docs=&Query=&Time=&EndTime=&SearchMethod=1&TocRestrict=n&Toc=&TocEntry=&QField=&QFieldYear=&QFieldMonth=&QFieldDay=&IntQFieldOp=0&ExtQFieldOp=0&XmlQuery=&File=D%3A%5Czyfiles%5CIndex%20Data%5C86thru90%5CTxt%5C00000001%5C00001O9W.txt&User=ANONYMOUS&Password=anonymous&SortMethod=h%7C-&MaximumDocuments=1&FuzzyDegree=0&ImageQuality=r75g8/r75g8/x150y150g16/i425&Display=hpfr&DefSeekPage=x&SearchBack=ZyActionL&Back=ZyActionS&BackDesc=Results%20page&MaximumPages=1&ZyEntry=1&SeekPage=x&ZyPURL

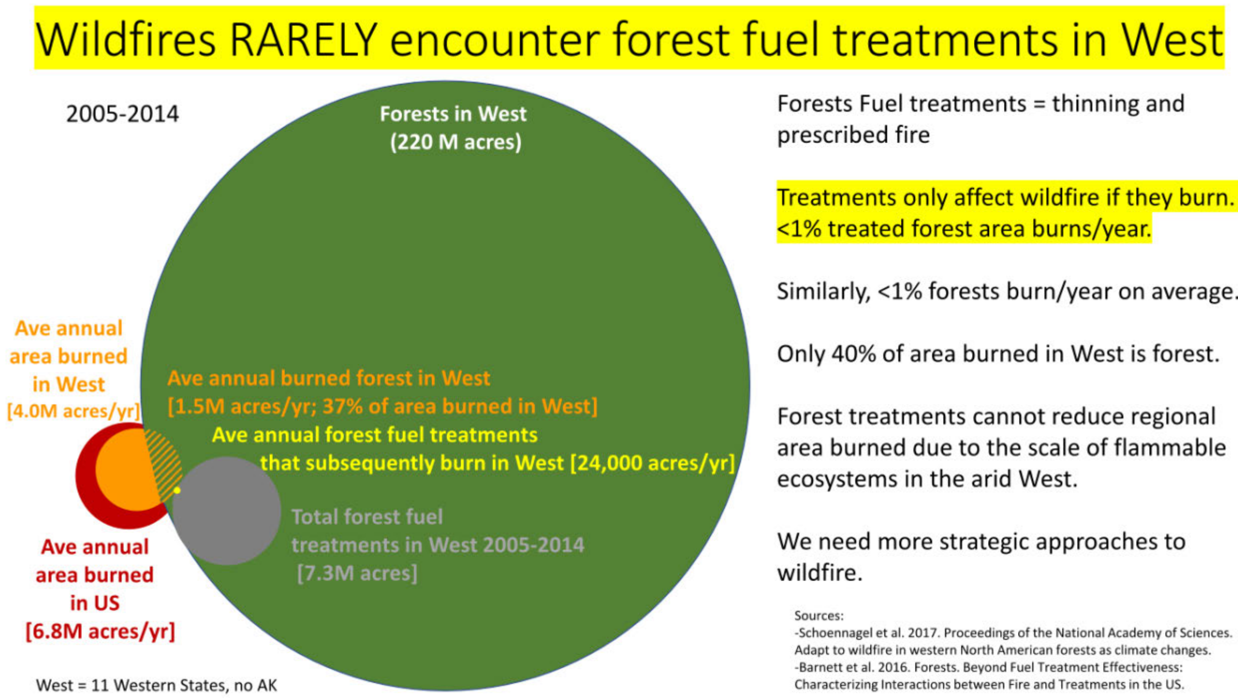
Ibisch et al. (2016) provide a global synthesis of road-related impacts including: wildlife mortality (vehicle collisions); poaching pressure; sediment increases (runoff); chemical contamination; carbon emissions; spread of invasive species; fire ignitions; and habitat fragmentation among others. These impacts can extend out to 1 km on either side of the road prism. Thus, road impacts need to be fully addressed and properly mitigated to assess planned extensive road upgrades and access.

Roadless Areas - The ecological importance of roadless areas is well-documented in the literature (Strittholt and DellaSala 2001, Loucks et al. 2003, Crist et al. 2005, Ibisch et al. 2016) and emphasized in landmark Forest Service policies such as the Roadless Conservation Rule[3] and Interior Columbia River Basin strategy[4]. At a minimum, the DEIS needs to disclose any treatments proposed in inventoried roadless areas and low density roaded areas (<1 mi/sq mi) and must avoid thinning in these areas because of their high conservation value, particularly as relatively unfragmented blocks of wildlife habitat. Roadless areas and low-density roaded areas are of considerable importance to ecosystem integrity (as defined by the 2012 planning rule) as they are often at the headwaters of watersheds essential in maintaining water quality and terrestrial and aquatic ecosystem integrity (DellaSala et al 2011). Roadless areas also tend to be of much lower priority for fuels reduction given their fire regimes are less altered by suppression and they lack the ignition problems associated with roaded areas (e.g., see Roadless Conservation Rule, Columbia River Basin strategy, DellaSala and Frost 2001).

LIMITATIONS OF THINNING ON FIRE BEHAVIOR IN A CHANGING CLIMATE NEED TO BE RECOGNIZED AND CORRECTED

The figure below illustrates uncertainties of relying on thinning to reduce fire intensity given that the period of when fuels are lowest is generally short lived and fires rarely encounter thinned sites when fuels are lowest (Schoennagel et al. 2017). The extremely low probability of fire and thinned site co-occurrence invalidates the DEIS assumptions about lowering fire intensity. Simply increasing the area thinned does not change these odds appreciably given one cannot accurately predict when and where a fire will occur and many areas are inaccessible (Schoennagel et al. 2017).

[3] https://www.fs.usda.gov/roadmain/roadless/2001roadlessrule
[4] https://www.fs.fed.us/r6/icbemp/html/ICBEMP_Frameworkmemorandum-and-strategy_2014.pdf



Moreover, the DEIS needs to disclose the difference between prescribed fire that is applied at the stand level (where impacts to soils can be dispersed and limited) vs. pile burning to consume slash that can cause localized soil damage (excessive soil heating) facilitating the spread of invasive plants and delaying forest succession (especially if livestock grazing also occurs, Besctha et. al 2012).

Excessive opening of the tree canopy can also lead to higher wind penetrance and rapid fire spread, particularly if thinning is conducted on steep slopes and in remote areas with limited access making fine fuel consumption via pile burning impractical. In a warming climate where more extreme fire weather is likely, thinning is even less likely to alter fire behavior (Abatzoglou and Williams 2017, Schoennagel et al. 2017).

CUMULATIVE IMPACTS OF LIVESTOCK GRAZING IN A CHANGING CLIMATE NEED TO BE FULLY ANALYZED AND GREATLY REDUCED

Livestock grazing and associated infrastructure in combination with climate change are causing extensive cumulative effects in the SFNF that are not properly analyzed or mitigated by the DEIS. The DEIS acknowledges that livestock have contributed to degradation of ecosystem resilience (DEIS Volume 1:5) but the alternatives contain numerous contradictions stating, for example, that the DEIS (Volume 1:13) "aims to provide *healthy* forested and non-forested lands that would supply forage for both livestock and wildlife" (Volume 1:13) and that it will "provide sustained multiple uses, products, and services in an *environmentally acceptable manner* (including timber, livestock forage, recreation opportunities, and leasable and locatable minerals) (emphasis added, DEIS Volume 1:16), all the while maintaining grazing at ecologically unacceptable levels (maximum of 11,400 AUMs).

The DEIS (Volume 1:37) recognizes that livestock grazing is "*not a natural process*" (emphasis added), yet, continues grazing under all planning alternatives even though it is inconsistent with ecological processes, ecological integrity, and ecological condition requirements of the planning rule (as noted in the boxes above). None of the alternatives meet these requirements given the high level of grazing maintained.

Importantly, the DEIS does not meet the BASI requirement of the planning rule by failing to analyze cumulative impacts of livestock from roads, infrastructure, and especially climate change. Besctha et al. (2012) noted livestock use affects a far greater proportion of BLM and Forest Service lands than do roads, timber harvest, and wildfires combined by altering vegetation, soils, hydrology and wildlife species composition and abundance "*in ways that exacerbate the effects of climate change on these resources*" (emphasis added). Livestock also contribute to greenhouse gas emissions globally (18% of the total anthropogenic emissions) and in the SFNF, thus, the DEIS needs to analyze livestock-related emissions.

Beschta et al. (2012) recommended large areas free of livestock use to "help initiate and speed the recovery of affected ecosystems as well as provide benchmarks or controls for assessing the effects of grazing versus no grazing at significant spatial scales in a changing climate."

The DEIS analyzed and dismissed Alternative 3 (lower livestock use) and dismissed a no grazing alternative as out of scope. However, Alternative 2 is deficient in meeting the ecological integrity, ecological condition, and ecological processes requirement of the planning rule. Therefore, the Forest Service needs to develop a new alternative or modify Alternative 2 to meet the specific recommendations of Beschta et al. (2012: Table 2) as follows.

Beschta et al. (2012:Table 2). Priority areas for permanently removing livestock and feral ungulates from Bureau of Land Management and US Forest Service lands to reduce or eliminate their detrimental ecological effects

- Watersheds and other large areas that contain a variety of ecotypes to ensure that major ecological and societal benefits of more resilient and healthy ecosystems on public lands will occur in the face of climate change
- Areas where ungulate effects extend beyond the immediate site (e.g., wetlands and riparian areas impact many wildlife species and ecosystem services with cascading implications beyond the area grazed)
- Localized areas that are easily damaged by ungulates, either inherently (e.g., biological crusts or erodible soils) or as the result of a temporary condition (e.g., recent fire or flood

disturbances, or degraded from previous management and thus fragile during a recovery period).

- Rare ecosystem types (e.g., perched wetlands) or locations with imperiled species or communities (e.g., aspen stands and understory plant communities, endemic species), including fish and wildlife species adversely affected by grazing and at-risk and/or listed under the ESA
- Non-use areas (i.e., ungrazed by livestock) or exclosures embedded within larger areas where livestock grazing continues.
- Such non-use areas should be located in representative ecotypes so that actual rates of recovery (in the absence of grazing impacts) can be assessed relative to resource trend and condition data in adjacent areas that continue to be grazed.
- Areas where the combined effects of livestock, wild ungulates, and feral ungulates are causing significant ecological impacts.

Notably, Ratner et al. (2018) document extensive impacts of livestock grazing on aspen groves in in Utah and their findings are generally applicable west-wide and therefore to the DEIS. These researchers found livestock significantly suppressed aspen sprout growth and trampled soils in study plots. They noted that livestock tended to concentrate in aspen groves due to forage availability and shading, even on allotments where livestock grazing is “controlled” and under “moderate” grazing. Ratner et al. (2018) recommended reducing livestock pressure via exclosures at least until aspen height exceeds browsing height and this will require periodic repetition (exclosures) to ensure proper aspen regeneration. At a minimum, exclosures should include entire aspen clonal areas and this needs to be incorporated into the DEIS.

Finally, the DEIS needs to allow for permanent allotment retirement and significantly reduced livestock grazing. This needs to include an analysis of the cumulative effects of *livestock grazing and climate change* and emissions related to livestock use, roads, and infrastructure. The DEIS (Volume 1:31) only allows for continuation of even vacant or understocked allotment and therefore should be modified or a new alternative developed to permanently retire vacant or understocked allotments and allow for voluntary buyout of grazing leases by conservation groups.

RIPARIAN AREAS NEED MORE EFFECTIVE PROTECTION, CONSERVATION, AND RESTORATION ESPECIALLY FROM LIVESTOCK AND THINNING TREATMENTS

The DEIS (Volume 1:153) correctly notes that although riparian areas occupy < 3% of the landscape, they support ~ 80% of the forest’s plant and animal diversity, including several at-risk species (e.g., Mexican Spotted Owl, Lewis’s Woodpecker, Arizona willow, Jemez Mountain salamander, masked and water shrew, New Mexican meadow jumping mouse, Northern leopard frog, Rio Grande chub, cutthroat trout, and sucker). Hubbard (1977; cited in Kauffman et al.

1984) report that 16-17% of the entire breeding avifauna of temperate North America reside in just 2 New Mexico river valleys and 77% of 166 nesting birds in the southwest depend on riparian habitat (Johnson et al. 1977 cited in Kauffman et al. 1984). Thus, riparian areas need stepped up conservation measures, especially protection from livestock grazing, given their exceptional importance in southwestern dry ecosystems.

Riparian areas also congregate livestock that have a strong preference for stream-side areas and wet montane meadows with high forage production. Livestock degrade this important wildlife habitat type in many ways, including soil compaction, spread of invasive species, stream-bank erosion, hydrological alterations, water quality and stream temperature degradation, and trampling effects (Kauffman et al. 1984, Besctha et al. 2012).

Kauffman et al. (1984) list several ways livestock grazing impacts can be reduced in riparian areas that should be readily adopted by the DEIS:

- Exclusion of livestock grazing;
- Alternative grazing schemes (e.g., late season – after bird nesting);
- Salting, alternative water sources, fencing, range riders to keep livestock out;
- Instream structures (e.g., trash catchers, gabions, small rock dams, individual boulder placement, rock jetties, and silt log drops) for increasing water table in areas of former wet meadows as well as improving fish habitat;
- Combining rest rotation with check dams (although the rest-rotation system may increase trailing and trampling damage, causing streambank erosion and instability);
- Selection of cattle with a preference for upland areas over riparian (cattle are known to have group-specific preferences)

Because of the disproportionate use of wildlife in riparian areas (especially at-risk species) and the extensive livestock damage in the area, the DEIS should incorporate the best elements from Alternative 3 with some notable additions as follows:

- Double the objectives in Alternative 2 (DEIS Volume 1:Table 3, p. 58) for restoring composition and structure in riparian vegetation.
- Within the riparian management zone, move toward desired conditions for vegetation types that are outside of or trending away from natural range of variability by restoring the composition and structure of 30 miles of stream every 10 years. Actions that could improve riparian areas would include removing invasive plant species, stabilising stream channels, planting native species, promoting natural revegetation of bare ground, and redirecting other uses (e.g., providing other watering sources or closing areas to camping – note this redirection needs to include redirecting cattle and not just “other uses”).

- Complete aquatic restoration on priority projects on 60 miles of aquatic habitat (e.g., increasing pool quantity, providing stream cover, removing or installing fish barriers, restoring beaver populations, treating invasive aquatic species, etc.) every 10 years to benefit aquatic species.
- Every 10 years, restore native fish species to 40 miles of streams where nonnative fish are absent and where natural or human-made fish barriers exist.
- Further reductions in road densities throughout the forest and avoidance of permanent or temporary roads, particularly those that parallel or cross streams.
- Additionally, an emphasis on beaver reintroduction is complimentary with the above improvements.
- The DEIS should include large no-grazing riparian zones where cattle are fenced out and permanently removed to allow riparian recovery and reintroduction of beaver populations.

FOCAL SPECIES, AT-RISK SPECIES, SPECIES OF CONSERVATION CONCERN NEED TO BE MONITORED AND HABITAT PROTECTED FROM THINNING AND GRAZING

The DEIS (Volume 2:312) states that “the 2012 Rule does not require or prohibit monitoring of population trends of focal species. Instead, it allows the use of any *existing or emerging approaches for monitoring the status of focal species* that are supported by current science” (emphasis added). However, the DEIS is deficient in meeting the BASI requirement of the planning rule as it inadequately monitors population viability of species and does not provide enough habitat protection measures for focal species, species of conservation concern, and at-risk species. Specifically, the DEIS needs to meet the BASI requirement for these species with respect to connectivity (Haber and Nelson 2015), PVA (Noon et al. 2003), and species-specific “trigger points” (Schulz et al. 2012).

The DEIS largely bases management of these species on coarse-filter approaches. The DEIS site specific measures are very general and insufficient as a fine-filter approach.

Importantly, The Committee of Scientists (COS 1999[5]) stated, ‘‘Habitat alone cannot be used to predict wildlife populations’’ and ‘‘diversity is sustained only when individual species persist; the goals of ensuring viability and providing for diversity are inseparable. For this reason, the fine-filter species assessment is critical." To meet the BASI requirements, therefore, the Forest Service must appropriately provide fine-filter approaches following recommendations of the COS (1999), Noon et al. (2003) and Schultz et al. (2012) as follows.

[5]COS (1999) https://www.fs.fed.us/emc/nfma/includes/cosreport/Committee%20of%20Scientists%20Report.htm

Noon et al. (2003) note: "to assess the viability of species, at least three assumptions must hold true: (1) attributes that define the coarse filter (i.e., dominant vegetation types) are sufficient and reliable surrogates for habitat and can effectively predict the occurrence of a given species; (2) managing coarse-filter attributes will address the factor(s) currently limiting abundance, density, and persistence of each species; and (3) the spatial resolution of the coarse filter matches the scale at which given species respond to environmental heterogeneity. Although these assumptions may be valid for some species in many circumstances, especially species that are small-bodied, abundant, and tightly linked to a particular vegetation community, the likelihood that the assumptions are met for all, or even most, species in an assemblage is low. For that reason, landscape planning employs "fine-filter" assessments, which are based on direct measures of the status and trends of individual species or on models of population viability to evaluate the needs of species at risk of decline. Noon et al. (2003) report numerous prediction errors associated with coarse-filter approaches that need supplementation with species-specific analyzes. For instance, forest planning needs to include PVA methods in its monitoring and adaptive management approach to better ensure coarse-filter requirements are representative of the community of interest."

Similarly, Schultz et al. (2012) indicate monitoring plans must include species-specific trigger points that initiate a review of management actions and provisions to ensure species-specific (fine filter) monitoring will be well funded and implemented. The *trigger points must be enforceable and ensure* that specific project actions cease should they further impair the viability of select species (especially the case for at-risk and listed species).

Schultz et al. (2012) note the 2012 planning rule requires "at least, some amount of direct species measurement may be needed to assess the effectiveness of the ecological conditions provided under the coarse-filter approach in achieving the goal of conserving the biological diversity of the area (USFS 2012:124)."

Schultz et al. (2012) provide recommendations for incorporating more specific fine-filter monitoring lacking in the DEIS, as summarised:

- Focusing on distribution, rather than traditional measures of population size and growth rate, which greatly increases the efficiency of broad-scale monitoring programs.
- Advancements in wildlife monitoring, based on detection/non-detection data, including the use of sign surveys, genetic evaluation, and historical presence–absence survey data decrease the cost of monitoring changes in distribution, which can be inferred from the proportion of sample units at which the species is detected.
- Area occupied by a species can be used as an effective measure of a species' spatial distribution.

- Temporal and spatial patterns in detection/non-detection monitoring data allow inference to changes in animal abundance, the single most influential parameter that provides insights into likelihood of species persistence.

The methods above recommended by Noon et al. (2003) and Schultz et al. (2012) along with connectivity measures recommended by Haber and Nelson (2015) should be applied to all 36 at-risk species, all 32 species of conservation concern, and all 7 focal species in the project area.

Mexican Spotted Owl (MSO) - The Santa Fe National Forest contains 198,888 acres of designated critical habitat for this owl. MSO requires dense conifer forests for nesting; however, will forage in recently severely burned areas (Lee 2018). The main factor involved in the decline of this species has been habitat destruction from logging; severe fire is not necessarily a habitat loss (Lee 2018), yet the DEIS assumes this to be the case. Large and small patches of severe burns juxtaposed with fire refugia for nesting may provide optimal habitat for MSO (Lee 2018). And while much is not known about how thinning affect MSO and its prey, declines in habitat and prey species have been noted for Northern Spotted Owl (see Odion et al. 2014b for review and analysis) and California Spotted Owl (Stephens et al. 2014). For all three subspecies of owls, removal of large trees (before/after fire) and reducing canopy cover (e.g., below 60% for NSO) constitutes habitat degradation that has been linked to nest occupancy failures (Lee 2018).

Thus, at a minimum, thinning units need to be dropped from MSO critical habitat and demographic monitoring implemented for this at-risk species.

FIRE EMISSIONS ARE OVER-ESTIMATED USING LANDFIRE AND EMISSIONS FROM PROJECT ACTIVITIES NEED TO BE ANALYSED FOR DIRECT, INDIRECT, AND CUMULATIVE EFFECTS

The DEIS pays an inordinate amount of attention to emissions from wildfires yet includes no analysis of emissions from livestock grazing, livestock infrastructure and transport, thinning and road development and maintenance. Therefore, the DEIS is deficient in assessing cumulative impacts of emissions and air quality to the surrounding community from project activities.

With respect to fire emissions, the DEIS needs to pay attention to the literature on wildfire emissions from related studies in dry pine and mixed conifer forests as follows.

For instance, Mitchell (2015: chapter 10 in DellaSala and Hanson 2015) has an excellent summary of ineffectiveness of thinning and reduction of carbon stores from thinning.

"While such treatments [referring to thinning and prescribed burning] can sometimes be effective in reducing fire severity, if and when fires occur in thinned areas (Rhodes and Baker, 2008), they can come at the expense of carbon storage. The majority of carbon stored in leaves, leaf litter, and duff is typically consumed by high-severity wildfire and often constitutes the majority of the carbon emissions during the a given fire, yet most of the carbon stored in forest biomass (stem wood, branches, and coarse, woody debris) remains unconsumed even by high-severity wildfires. Consequently, fuel removal via forest thinning almost always reduces carbon storage more than the additional carbon that a stand is able to store when made more resistant to wildfire. For this reason, removing large amounts of biomass to reduce the fraction by which other biomass components are consumed via combustion is inefficient (Mitchell et al., 2009). Fuel reduction treatments that involve the removal of overstory biomass (i.e., intermediate-sized and large trees) are, perhaps unsurprisingly, the most inefficient methods of reducing wildfire-related carbon losses because they remove large amounts of carbon for only a marginal reduction in expected fire severity (Figure 10.2)."

"The majority of carbon stored in montane forest ecosystems of western North America remains unconsumed, even in high-severity wildfires. Large carbon stores in the bole biomass of large forest trees are not consumed, and the substantial proportion of carbon stored in forest soils is only slightly consumed. Most of the carbon emissions in a wildfire are from combustion of litter, duff, and woody debris. In the 2002 Biscuit Fire, CFs for total forest biomass (i.e., trees, snags, shrubs, woody fuels, litter, duff, and soil), weighted according to their respective prefire biomass, were 0.13, 0.15, and 0.21 for low-, medium-, and high-severity fires, respectively. Such factors can be even lower among stands with a higher proportion of carbon storage in bole biomass that likewise remains unconsumed in high-severity wildfires, such as Sitka spruce (P. sitchensis)/Western Hemlock (T. heterophylla) forests in the coast range of the Pacific Northwest (Smithwick et al., 2002;Mitchell et al., 2009). The application of fuel treatments can be effective in reducing fire severity and carbon emissions, but such treatments come at the cost of a net reduction in carbon storage relative to fire alone (Mitchell et al., 2009)."

In a recent global study of pyrogenic carbon emissions, Jones et al. (2019) concluded that "large wildfires convert a significant fraction of the burned vegetation biomass into pyrogenic carbon that can be stored on site for centuries to millennia and this stored carbon is underestimated in emissions calculations. The amount of carbon emitted globally from wildfires is in fact buffered by pyrogenic carbon production resulting in burned landscapes becoming a significant carbon sink." The value of this sink is not even reported in the DEIS nor is it estimated in LANDFIRE and it needs to be in the forest plan. Here is the Jones et al. (2019) abstract, the pdf is attached.

Abstract

Landscape fires burn 3–5 million km2 of the Earth's surface annually. They emit 2.2 Pg of carbon per year to the atmosphere, but also convert a significant fraction of the burned vegetation biomass into pyrogenic carbon. Pyrogenic carbon can be stored in terrestrial and marine pools for centuries to millennia and therefore its production can be considered a mechanism for long-term carbon sequestration. Pyrogenic carbon stocks and dynamics are not considered in global carbon cycle models, which leads to systematic errors in carbon accounting. Here we present a comprehensive dataset of pyrogenic carbon production factors from field and experimental fires and merge this with the Global Fire Emissions Database to quantify the global pyrogenic carbon production flux. We found that 256 (uncertainty range: 196–340) Tg of biomass carbon was converted annually into pyrogenic carbon between 1997 and 2016. Our central estimate equates to 12% of the annual carbon emitted globally by landscape fires, which indicates that their emissions are buffered by pyrogenic carbon production. We further estimate that cumulative pyrogenic carbon production is 60 Pg since 1750, or 33–40% of the global biomass carbon lost through land use change in this period. Our results demonstrate that pyrogenic carbon production by landscape fires could be a significant, but overlooked, sink for atmospheric CO_2.

We repeat from above our concerns about problems with LANDFIRE fire emissions as follows.

In Northern Idaho, Hyde et al. (2015) evaluated two LANDFIRE fuel loading raster options: (1) Fuels Characteristic Classification Systems (LANDFIRE-FCCS); and (2) Fuel Loading Model (LANDFIRE-FLM) vs. measured fuel loadings for a 20,000 ha mixed conifer study area. They found that the LANDFIRE-FCCS layer showed 200% higher duff loadings relative to measured loadings that led to 23% higher total mean consumption and emissions when modeled in FOFEM. The LANDFIRE-FLM layer showed lower loadings for total surface fuels relative to measured data, especially in the case of coarse woody debris that led to 51% lower mean total consumption and emissions when modeled in FOFEM. Additionally, LANDFIRE-FLM consumption was *59% lower relative to that on the measured plots, with 58% lower modeled emissions*. The authors concluded that these differences in fuel loadings led to significant differences in consumption and emissions depending upon the data and model chosen. The DEIS therefore needs to disclose how errors in fuel loading consumption were addressed in emissions determinations regarding wildfires and how these errors were corrected.

CONCLUSIONS AND NEED FOR GREATLY IMPROVED PREFERRED ALTERNATIVE

Based on the above analysis, deficiencies in the DEIS, and need for an improved or new alternative to better meet the BASI and planning rule requirements, I am requesting that the SFNF revise the DEIS to include the following actions.

- **Prioritize community wildfire safety and fire-risk reduction, including home-hardening, defensible space, additional road closures/decommissioning to reduce ignitions, and identification/maintenance of community evacuation routes.** The most prudent means of community fire protection is to *work from the home-out* rather than the *wildlands-in* (emphasis added) according to retired Forest Service researcher Jack Cohen

(2000; also see Youtube interviews[6]) and related home fire-risk reduction work (Syphard et al. 2013, 2014). Community and fire-fighter safety actions should be directed at home protection and anthropogenic fire-ignitions along high-use roads (especially ingress/egress; see Balch et al. 2017). As noted above, research demonstrates that there is a very low (<1%) probability of thinned areas encountering a fire when fuels are lowest (Schoennagel et al. 2017). Therefore, it is imperative that the Forest Service strategically direct limited resources at protecting homes rather than extensive thinning in the backcountry that does nothing for home protection.

- **Reduce human-caused wildfire ignitions (see Balch et al. 2017) associated with road access**. The Forest Service needs to conduct project-specific transportation plans to determine the probability of human-caused fire ignitions in relation to road densities, road improvements, and increased human access along improved roads. These plans should address a broad scope of road-related impacts and choose an alternative based on minimal road access.
- **Protect high value conservation areas from logging/thinning/road improvements.** The DEIS needs to fully disclose impacts of road improvements and thinning on low-density (<1 mi/sq mile) and inventoried roadless areas (see below) and make clear how late-successional (closed canopy) forests within the project area will be maintained and restored to levels comparable to historic or documented reference conditions.
- **Disclose limitations and uncertainties of fire-scar sampling, importance of fire-free periods to shrub and tree recruitment and include more robust fire occurrence/ severity estimators that account for variability in fire-free and frequent-fire intervals.** The DEIS primarily relies on fire-scar sampling to determine the dominant fire regime present yet does not disclose uncertainties and limitations in sampling approaches (i.e., confidence levels). Notably, paleo-ecology studies conducted over longer timelines (millennia) than fire scar sampling show high variability in fire regimes related primarily to regional and local microclimatic factors (slope, aspect, elevation) over time (Meyer 2010). Large fires historically included high severity patches during alternating cycles of wet followed by droughts (Margolis et al. 2011). This is particularly important as extreme fire-weather (top-down driver) is known to over-ride bottom up influences (fuels) on fire behavior in the Rockies (Bessie and Johnson 1995, Schoennagel et al. 2004) and elsewhere (Abatzoglou and Williams 2017). The effect of global heating and increased likelihood of regional droughts may (Margolis et al. 2011) or may not (Parks et al. 2016, Margolis et al. 2017) increase fire severity. This uncertainty is most significant and must be analyzed to determine the need for and limitations of extensive fuels treatments based predominately on assumptions regarding frequent-fire regimes that may become increasingly unlikely in a rapidly changing climate. Additionally, variability in fire return

6 National Fire Protection Association presentations by Jack Cohen - https://www.youtube.com/watch?v=vL_syp1ZScM; https://www.youtube.com/watch?v=RqKFDDBGd5o

(point/plot scale) and fire rotation (landscape scale) intervals accounts for longer fire-free periods that allow for shrub and small tree recruitment, including both dense and open forest conditions (see below). Thus, the DEIS needs to fully disclose its characterization of a low-severity fire regime, and "open" forest conditions (reference sites) with respect to heterogeneity and in relation to tree canopy mortality, shrub and small tree densities. Notably, even low severity systems have occasional fire-flare ups that kill dominant overstory trees and allow for sufficient shrub and small tree recruitment (see Baker 2017).

- **Substantially reduce livestock grazing in riparian areas and high value conservation areas.** Stepped up conservation and restoration need to be in the forest plan, including large no-grazing zones (exclosures), additional riparian and wet meadow/spring protections, road obliteration, invasive species removals, and beaver reintroductions.
- **More fully disclose and avoid impacts to at-risk species like the Mexican Spotted Owl (MSO).** There is no discussion of importance of mixed-severity wildfires in maintaining foraging habitat for spotted owls (Lee 2018, pdf enclosed). Instead, the DEIS incorrectly assumes, without site-specific data on owl occupancy or region-wide population trends, that wildfire (mostly high severity) degrades MSO habitat. However, Lee (2018) conducted a meta-analysis of fire effects on all three owl subspecies concluding that mixed-severity fire, including patches of large severity, was not the main cause of owl nest abandonment; pre- and post-fire logging was the predominant factor. Also, full disclosure of incidental take under the Endangered Species Act is required and the Forest Service needs to conduct population monitoring to assess MSO demographics and region-wide population trends.
- **Analyze and maintain connectivity especially for at risk, focal, and species of conservation concern.** The forest plan needs to properly analyze connectivity as noted herein including PVA, trigger points, and species/landscape specific measures that properly integrate coarse and fine-filter approaches under the BASI and connectivity requirements of the 2012 forest planning rule and the noted literature cited herein.
- **Reduce emissions from logging and roads.** A stated intent of the DEIS is to provide for resilience to climate change yet there is no requirement of an analysis of project-related emissions from tree clearing and road improvements. Notably, emissions from wildfires are typically much lower than landscape-level logging projects aimed at reducing wildfires (e.g., see Mitchell et al. 2009, Campbell et al. 2016, Law et al. 2018 as examples of appropriate methodologies). Project-specific alternatives must be developed to minimize emissions with alternatives selected that produce the lowest emissions. Alternatives should be compared in CO_2 equivalents, including the social cost of carbon[7].

[7]See https://19january2017snapshot.epa.gov/climatechange/social-cost-carbon_.html

- **Provide a cost-benefits analysis of managing wildfires for ecosystem benefits by working with fire under safe conditions.** The DEIS must disclose project-related costs of thinning, prescribed fire, and road improvements in comparison to managing fire for ecosystem benefits as a viable alternative (e.g., refer to the Cohesive Wildland Fire Management Strategy for wildfire ecosystem benefits[8] and 2012 forest planning rule regarding ecosystem integrity, vegetation diversity, and wildfire maintenance). Thus, it must be disclosed under what conditions will wildfires be managed for ecosystem benefits vs. suppressed so that when fires do eventually occur appropriate actions are taken based on pre-fire response planning and the Forest Service is accountable for implementing those actions accordingly.
- **Thinning to create open canopy forests at the expense of closed canopy forests needs to be greatly reduced and more strategically (surgically) applied.** The over-reliance on thinning stems from accuracy problems noted in the LANDFIRE program, biased fire scar fire estimates, inappropriate extrapolations from the Forest Service research publication GTR-310, and a failure to recognize site-specific and landscape heterogeneity. Thus, thinning treatments need to be greatly scaled back and strategic in application (mostly nearest homes).
- **In limited cases where thinning occurs, forest canopies need to be more fully maintained for closed canopy species associates by**: (1) stops and gaps (explain for the general reader) in thinning to for increased site heterogeneity; (2) retention of much more basal area (as compared to site-specific reference sites) especially around tree cohorts to make them wind firm; (3) retention of old/mature trees on site (based on increment core analysis and not just diameter at breast height); (4) in cases where tree thinning is necessary within the drip line of large mature trees, girdle those trees and leave standing on site as biological legacies; (5) retain more shrubs, forbs, and native grasses by reducing the interval between successive prescribed fires to allow for understory recruitment; and (6) fell and tip large trees in stream-side areas to create in-stream structures rather than thin and remove those trees from the site.
- **"Surgically" applied thinning treatments should be limited to the most drastically altered forests,** most notably, pine plantations in the Jemez and spruce/fir clearcuts on the eastern side of the SFNF.
- **Restoration and conservation measures should be greatly increased to address the following needs not sufficiently met in the DEIS**: (1) beaver reintroduction in riparian areas; (2) large livestock exclosures especially in riparian areas, wet meadows, and aspen groves; (3) road closures and road obliterations to provide connectivity; (4) defensible space within a narrow buffer (~60 feet) around homes; (5) ingress/egress routes for community protection; (6) increases in invasive species removal and containment; and

[8] See https://www.forestsandrangelands.gov/strategy/

(7) identification and protection of site and landscape specific habitat for focal species, species of conservation concern, and at risk species.

- **Compartmentalize the SFNF into fire management units** to determine when to suppress fire for community safety vs. working with fire for ecosystem benefits.[9]
- **Conduct a minimum road access analysis** and decommission/obliterate more roads to reduce impacts to water quality, wildlife habitat and human-caused fire ignitions.

In closing, while I respect the ability of the Forest Service to apply BASI to forest planning decisions on the Santa Fe National Forest, I remain greatly concerned that the noted inadequacies in the DEIS have not met the BASI standard. Instead the preferred alternative will (1) fragment and degrade important wildlife habitat; (2) jeopardize at-risk species (MSO), focal species, and species of conservation concern; (3) degrade water quality (mainly from roads, livestock, tree thinning), impact mature forests and riparian areas (along with wildlife and cultural values); and (4) uses methodologies (e.g., LANDFIRE, fire scar sampling, GTR-310) inappropriate to the SFNF. There is a heavy reliance on fire-scar sampling without disclosure of biases and uncertainties and thinning in stands that may possess old growth characteristics by moving them increasingly into open canopy conditions that lack overstory and understory structures. The efficacy of Alternative 2 mechanical treatments is highly uncertain because of the likelihood that the region's fire regimes will increasingly shift to larger burns due primarily to climate change (Abatzoglou and Williams 2017) and the extremely low odds that thinned sites will encounter a fire when fuels are lowest (Schoennagel et al. 2017).

Additionally, and contrary to what is often claimed by the Forest Service, insect and disease outbreaks are not associated with increased fire intensity. Insect-fire studies, including analysis of outbreaks and fire intensity in the Rockies and elsewhere (Romme et al. 2006, Kauffman et al. 2008, Bond et al. 2009, Black et al. 2011, Six et al. 2014, Hart et al. 2015, Meigs et al. 2016, Talucci and Krawchuck 2019) have repeatedly shown that there is no coupling of increased fire intensity with insect outbreaks. Instead, outbreaks may actually lower fire intensity once the needles of dead trees fall to the ground (within 1-3 years) as canopy fuels and therefore crown fires become highly unlikely. Dead trees also do not contribute to fire spread as they do not fall all at once nor result in accumulation of fine fuels (fine fuel accumulation is associated with logging). Dead trees are keystone legacies that provide essential habitat for cavity nesting birds, denning mammals, and numerous other wildlife. Their role in forest ecosystems needs to be better disclosed and maintained.

While wildfire clearly can be devastating to human communities, it is not an ecological catastrophe as often claimed. The Forest Service needs to develop better supported consensus

[9]see https://www.fs.fed.us/rmrs/publications/framework-developing-safe-and-effective-large-fire-response-new-fire-management; https://www.fs.fed.us/rmrs/publications/spatial-optimization-operationally-relevant-large-fire-confine-and-point-protection

alternatives that focus first and foremost on community protection where there is strong scientific agreement (see Moritz et al. 2014, Schoennagel et al. 2017, Moritz et al. 2018).

Impact of anthropogenic climate change on wildfire across western US forests

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Increased forest fire activity across the western continental United States (US) in recent decades has likely been enabled by a number of factors, including the legacy of fire suppression and human settlement, natural climate variability, and human-caused climate change. We use modeled climate projections to estimate the contribution of anthropogenic climate change to observed increases in eight fuel aridity metrics and forest fire area across the western United States. Anthropogenic increases in temperature and vapor pressure deficit significantly enhanced fuel aridity across western US forests over the past several decades and, during 2000–2015, contributed to 75% more forested area experiencing high (>1 σ) fire-season fuel aridity and an average of nine additional days per year of high fire potential. Anthropogenic climate change accounted for ∼55% of observed increases in fuel aridity from 1979 to 2015 across western US forests, highlighting both anthropogenic climate change and natural climate variability as important contributors to increased wildfire potential in recent decades. We estimate that human-caused climate change contributed to an additional 4.2 million ha of forest fire area during 1984–2015, nearly doubling the forest fire area expected in its absence. Natural climate variability will continue to alternate between modulating and compounding anthropogenic increases in fuel aridity, but anthropogenic climate change has emerged as a driver of increased forest fire activity and should continue to do so while fuels are not limiting.

wildfire | climate change | attribution | forests

Widespread increases in fire activity, including area burned (1, 2), number of large fires (3), and fire-season length (4, 5), have been documented across the western United States (US) and in other temperate and high-latitude ecosystems over the past half century (6, 7). Increased fire activity across western US forests has coincided with climatic conditions more conducive to wildfire (2–4, 8). The strong interannual correlation between forest fire activity and fire-season fuel aridity, as well as observed increases in vapor pressure deficit (VPD) (9), fire danger indices (10), and climatic water deficit (CWD) (11) over the past several decades, present a compelling argument that climate change has contributed to the recent increases in fire activity. Previous studies have implicated anthropogenic climate change (ACC) as a contributor to observed and projected increases in fire activity globally and in the western United States (12–19), yet no studies have quantified the degree to which ACC has contributed to observed increases in fire activity in western US forests.

Changes in fire activity due to climate, and ACC therein, are modulated by the co-occurrence of changes in land management and human activity that influence fuels, ignition, and suppression. The legacy of twentieth century fire suppression across western continental US forests contributed to increased fuel loads and fire potential in many locations (20, 21), potentially increasing the sensitivity of area burned to climate variability and change in recent decades (22). Climate influences wildfire potential primarily by modulating fuel abundance in fuel-limited environments, and by modulating fuel aridity in flammability-limited environments (1, 23, 24). We constrain our attention to climate processes that promote fuel aridity that encompass fire behavior characteristics of landscape ignitability, flammability, and fire spread via fuel desiccation in primarily flammability-limited western US forests by considering eight fuel aridity metrics that have well-established direct interannual relationships with burned area in this region (1, 8, 24, 25). Four metrics were calculated from monthly data for 1948–2015: (*i*) reference potential evapotranspiration (ETo), (*ii*) VPD, (*iii*) CWD, and (*iv*) Palmer drought severity index (PDSI). The other four metrics are daily fire danger indices calculated for 1979–2015: (*v*) fire weather index (FWI) from the Canadian forest fire danger rating system, (*vi*) energy release component (ERC) from the US national fire danger rating system, (*vii*) McArthur forest fire danger index (FFDI), and (*viii*) Keetch–Byram drought index (KBDI). These metrics are further described in the *Materials and Methods* and *Supporting Information*. Fuel aridity has been a dominant driver of regional and subregional interannual variability in forest fire area across the western US in recent decades (2, 8, 22, 25). This study capitalizes on these relationships and specifically seeks to determine the portions of the observed increase in fuel aridity and area burned across western US forests attributable to anthropogenic climate change.

The interannual variability of all eight fuel aridity metrics averaged over the forested lands of the western US correlated significantly ($R^2 = 0.57$–0.76, $P < 0.0001$; Table S1) with the logarithm of annual western US forest area burned for 1984–2015, derived from the Monitoring Trends in Burn Severity product for 1984–2014 and the Moderate Resolution Imaging Spectroradiometer (MODIS) for 2015 (*Supporting Information*). The record of standardized fuel aridity averaged across the eight metrics (hereafter, all-metric mean) accounts for 76% of the variance in the burned-area record, with significant increases in both records for 1984–2015 (Fig. 1). Correlation between fuel aridity and forest fire area remains highly significant ($R^2 = 0.72$, all-metric mean) after removing the linear-least squares trends for each time series for 1984–2015, supporting the mechanistic relationship between fuel aridity and

Significance

Increased forest fire activity across the western United States in recent decades has contributed to widespread forest mortality, carbon emissions, periods of degraded air quality, and substantial fire suppression expenditures. Although numerous factors aided the recent rise in fire activity, observed warming and drying have significantly increased fire-season fuel aridity, fostering a more favorable fire environment across forested systems. We demonstrate that human-caused climate change caused over half of the documented increases in fuel aridity since the 1970s and doubled the cumulative forest fire area since 1984. This analysis suggests that anthropogenic climate change will continue to chronically enhance the potential for western US forest fire activity while fuels are not limiting.

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See Commentary on page 11649.

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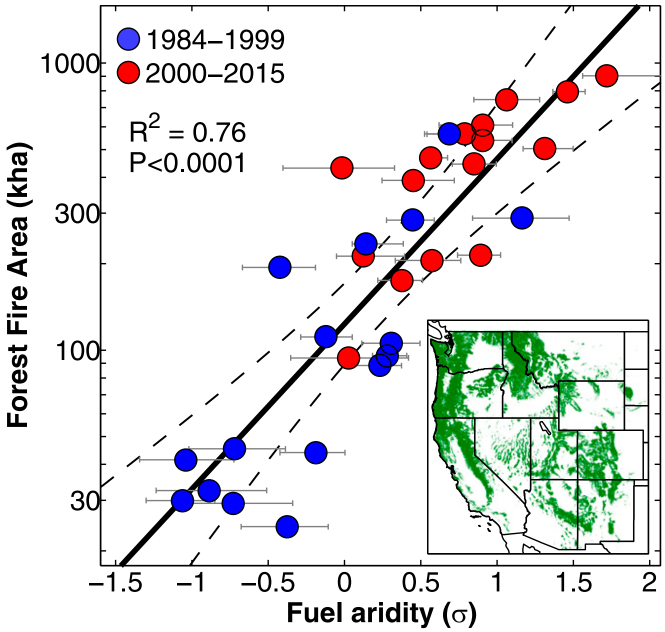

Fig. 1. Annual western continental US forest fire area versus fuel aridity: 1984–2015. Regression of burned area on the mean of eight fuel aridity metrics. Gray bars bound interquartile values among the metrics. Dashed lines bounding the regression line represent 95% confidence bounds, expanded to account for lag-1 temporal autocorrelation and to bound the confidence range for the lowest correlating aridity metric. The two 16-y periods are distinguished to highlight their 3.3-fold difference in total forest fire area. *Inset* shows the distribution of forested land across the western US in green.

forest fire area. It follows that co-occurring increases in fuel aridity and forest fire area over multiple decades would also be mechanistically related.

We quantify the influence of ACC using the Coupled Model Intercomparison Project, Phase 5 (CMIP5) multimodel mean changes in temperature and vapor pressure following Williams et al. (26) (Fig. S1; *Methods*). This approach defines the ACC signal for any given location as the multimodel mean (27 CMIP5 models) 50-y low-pass-filtered record of monthly temperature and vapor pressure anomalies relative to a 1901 baseline. Other anthropogenic effects on variables such as precipitation, wind, or solar radiation may have also contributed to changes in fuel aridity but anthropogenic contributions to these variables during our study period are less certain (22). We evaluate differences between fuel aridity metrics computed with the observational record and those computed with observations that exclude the ACC signal to determine the contribution of ACC to fuel aridity. To exclude the ACC signal, we subtract the ACC signal from daily and monthly temperature and vapor pressure, leaving all other variables unchanged and preserving the temporal variability of observations. The contribution of ACC to changes in fuel aridity is shown for the entire western United States; however, we constrain the focus of our attribution and analysis to forested environments of the western US (Fig. 1, *Inset*; *Methods*).

Anthropogenic increases in temperature and VPD contributed to a standardized (σ) increase in all-metric mean fuel aridity averaged for forested regions of +0.6 σ (range of +0.3 σ to +1.1 σ across all eight metrics) for 2000–2015 (Fig. 2). We found similar results with reanalysis products (all-metric mean fuel aridity increase of +0.6 σ for two reanalysis datasets considered; *Methods*), suggesting robustness of the results to structural uncertainty in observational products (Figs. S2–S4 and Table S2). The largest anthropogenic increases in standardized fuel aridity were present across the intermountain western United States, due in part to larger modeled warming rates relative to more maritime areas (27). Among aridity metrics, the largest increases tied to the ACC signal were for VPD and ETo because the interannual variability of these variables is primarily driven by temperature for much of the study area (28). By contrast, PDSI and ERC showed more subdued ACC driven increases in fuel aridity because these metrics are more heavily influenced by precipitation variability.

Fuel aridity averaged across western US forested areas showed a significant increase over the past three decades, with a linear trend of +1.2 σ (95% confidence: 0.42–2.0 σ) in the all-metric mean for 1979–2015 (Fig. 3*A*, *Top* and Table S1). The all-metric mean ACC contribution since 1901 was +0.10 σ by 1979 and +0.71 σ by 2015. The annual area of forested lands with high fuel aridity (>1 σ) increased significantly during 1948–2015, most notably since 1979 (Fig. 3*A*, *Bottom*). The observed mean annual areal extent of forested land with high aridity during 2000–2015 was 75% larger for the all-metric mean (+27% to +143% range across metrics) than was the case where the ACC signal was excluded.

Significant positive trends in fuel aridity for 1979–2015 across forested lands were observed for all metrics (Fig. 3*B* and Table S1). Positive trends in fuel aridity remain after excluding the ACC signal, but the remaining trend was only significant for ERC. Anthropogenic forcing accounted for 55% of the observed positive trend in the all-metric mean fuel aridity during 1979–2015, including at least two-thirds of the observed increase in ETo, VPD, and FWI, and less than a third of the observed increase in ERC and PDSI. No significant trends were observed for monthly fuel aridity metrics from 1948–1978.

The duration of the fire-weather season increased significantly across western US forests (+41%, 26 d for the all-metric mean) during 1979–2015, similar to prior results (10) (Fig. 4*A* and Table S2). Our analysis shows that ACC accounts for ~54% of the increase in fire-weather season length in the all-metric mean (15–79% for individual metrics). An increase of 17.0 d per year of high fire potential was observed for 1979–2015 in the all-metric mean (11.7–28.4 d increase for individual metrics), over twice the rate of increase calculated from metrics that excluded the ACC signal (Fig. 4*B* and Table S2). This translates to an average of an additional 9 d (7.8–12.0 d) per year of high fire potential during 2000–2015 due to ACC.

Given the strong relationship between fuel aridity and annual western US forest fire area, and the detectable impact of ACC on fuel aridity, we use the regression relationship in Fig. 1 to model

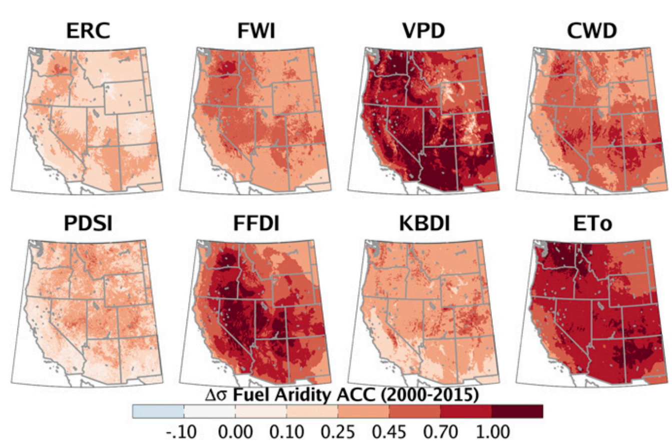

Fig. 2. Standardized change in each of the eight fuel aridity metrics due to ACC. The influence of ACC on fuel aridity during 2000–2015 is shown by the difference between standardized fuel aridity metrics calculated from observations and those calculated from observations excluding the ACC signal. The sign of PDSI is reversed for consistency with other aridity measures.

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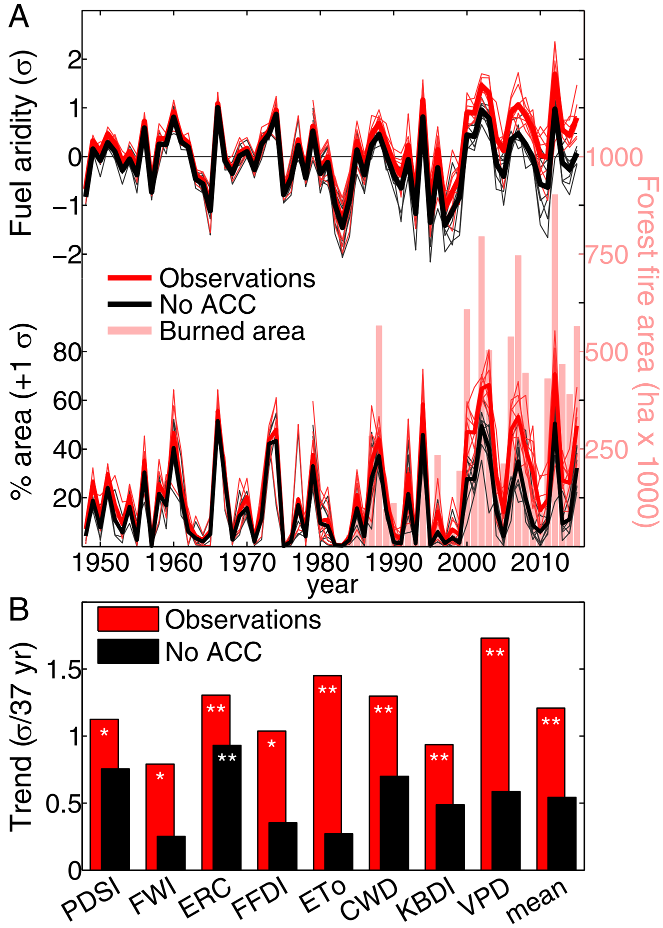

Fig. 3. Evolution and trends in western US forest fuel aridity metrics over the past several decades. (*A*) Time series of (*Upper*) standardized annual fuel aridity metrics and (*Lower*) percent of forest area with standardized fuel aridity exceeding one SD. Red lines show observations and black lines show records after exclusion of the ACC signal. Only the four monthly metrics extend back to 1948. Daily fire danger indices begin in 1979. Bold lines indicate averages across fuel aridity metrics. Bars in the background of *A* show annual forested area burned during 1984–2015 for visual comparison with fuel aridity. (*B*) Linear trends in the standardized fuel aridity metrics during 1979–2015 for (red) observations and (black) records excluding the ACC signal (differences attributed to ACC). Asterisks indicate positive trends at the (*) 95% and (**) 99% significance levels.

the contribution of ACC on western US forest fire area for the past three decades (Fig. 5 and Fig. S5). ACC-driven increases in fuel aridity are estimated to have added ~4.2 million ha (95% confidence: 2.7–6.5 million ha) of western US forest fire area during 1984–2015, similar to the combined areas of Massachusetts and Connecticut, accounting for nearly half of the total modeled burned area derived from the all-metric mean fuel aridity. Repeating this calculation for individual fuel aridity metrics yields ACC contributions of 1.9–4.9 million ha, but most individual fuel aridity metrics had weaker correlations with burned area and thus may be less appropriate proxies for attributing burned area. The effect of the ACC forcing on fuel aridity increased during this period, contributing ~5.0 (95% confidence: 4.2–5.9) times more burned area in 2000–2015 than in 1984–1999 (Fig. 5*B*). During 2000–2015, the ACC-forced burned area likely exceeded the burned area expected in the absence of ACC (Fig. 5*B*). A more conservative method that uses the relationship between detrended records of burned area and fuel aridity (2) still indicates a substantial impact of ACC on total burned area, with a 19% (95% confidence: 12–24%) reduction in the proportion of total burned area attributable to ACC (Fig. S5).

Our attribution explicitly assumes that anthropogenic increases in fuel aridity are additive to the wildfire extent that would have arisen from natural climate variability during 1984–2015. Because the influence of fuel aridity on burned area is exponential, the influence of a given ACC forcing is larger in an already arid fire season such as 2012 (Fig. 5*A* and Fig. S5*C*). Anthropogenic increases in fuel aridity are expected to continue to have their most prominent impacts when superimposed on naturally occurring extreme climate anomalies. Although numerous studies have projected changes in burned area over the twenty-first century due to ACC, we are unaware of other studies that have attempted to quantify the contribution of ACC to recent forested burned area over the western United States. The near doubling of forested burned area we attribute to ACC exceeds changes in burned area projected by some modeling efforts to occur by the mid-twenty-first century (29, 30), but is proportionally consistent with mid-twenty-first century increases in burned area projected by other modeling efforts (17, 31–33).

Beyond anthropogenic climatic changes, several additional factors have caused increases in fuel aridity and forest fire area since the 1970s. The lack of fuel aridity trends during 1948–1978 and persistence of positive trends during 1979–2015 even after removing the ACC signal implicates natural multidecadal climate variability as an important factor that buffered anthropogenic

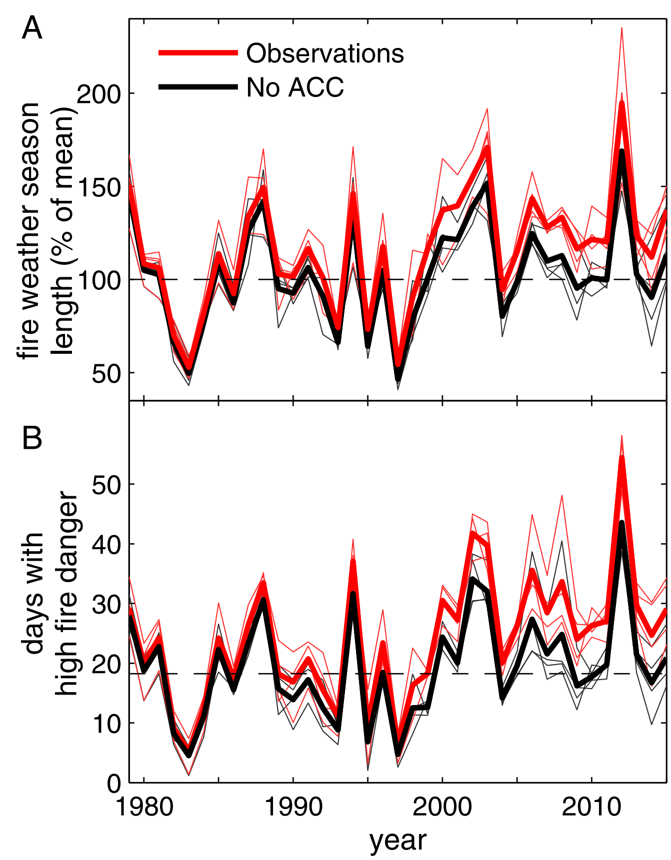

Fig. 4. Changes in fire-weather season length and number of high fire danger days. Time series of mean western US forest (*A*) fire-weather season length and (*B*) number of days per year when daily fire danger indices exceeded the 95th percentile. Baseline period: 1981–2010 using observational records that exclude the ACC signal. Red lines show the observed record, and black lines show the record that excludes the ACC signal. Bold lines show the average signal expressed across fuel aridity metrics.

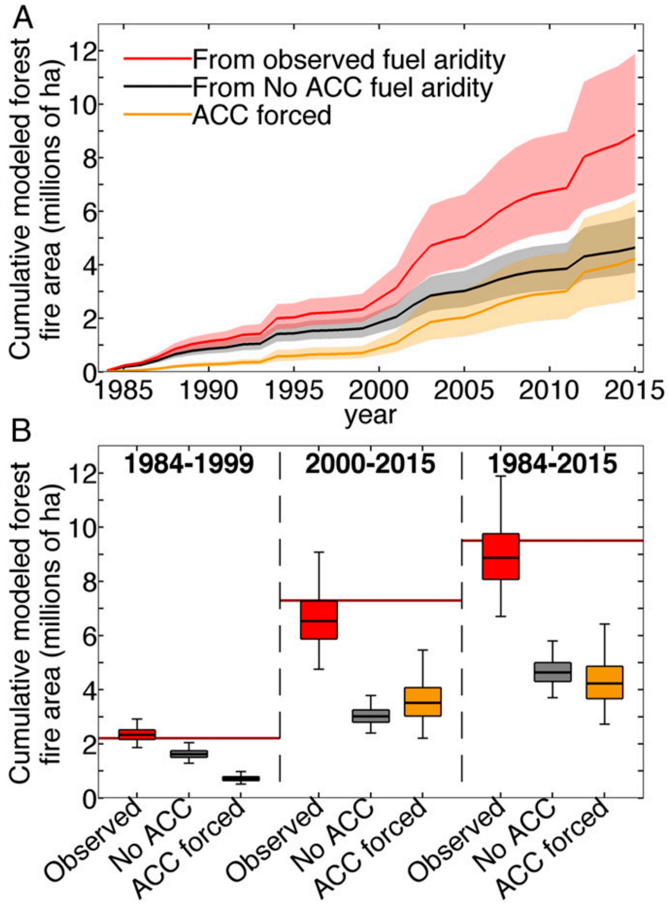

Fig. 5. Attribution of western US forest fire area to ACC. Cumulative forest fire area estimated from the (red) observed all-metric mean record of fuel aridity and (black) the fuel aridity record after exclusion of ACC (No ACC). The (orange) difference is the forest fire area forced by anthropogenic increases in fuel aridity. Bold lines in *A* and horizontal lines within box plots in *B* indicate mean estimated values (regression values in Fig. 1). Boxes in *B* bound 50% confidence intervals. Shaded areas in *A* and whiskers in *B* bound 95% confidence intervals. Dark red horizontal lines in *B* indicate observed forest fire area during each period.

effects during 1948–1978 and compounded anthropogenic effects during 1979–2015. During 1979–2015, for example, observed Mar–Sep vapor pressure decreased significantly across many US forest areas, in marked contrast to modeled anthropogenic increases (Fig. S6) (34). Significant declines in spring (Mar–May) precipitation in the southwestern United States and summer (Jun–Sep) precipitation throughout parts of the northwestern United States during 1979–2015 (Fig. S7 *A* and *B*) hastened increases in fire-season fuel aridity, consistent with observed increases in the number of consecutive dry days across the region (10). Natural climate variability, including a shift toward the cold phase of the interdecadal Pacific Oscillation (35), was likely the dominant driver of observed regional precipitation trends (36) (Fig. S7 *B* and *D*).

Our quantification of the ACC contribution to observed increases in forest fire activity in the western United States adds to the limited number of climate change attribution studies on wildfire to date (37). Previous attribution efforts have been restricted to a single GCM and biophysical variable (14, 16). We complement these studies by demonstrating the influence of ACC derived from an ensemble of GCMs on several biophysical metrics that exhibit strong links to forest fire area. However, our attribution effort only considers ACC to manifest as trends in mean climate conditions, which may be conservative because climate models also project anthropogenic increases in the temporal variability of climate and drought in the western United States (34, 38, 39). In focusing exclusively on the direct impacts of ACC on fuel aridity, we do not address several other pathways by which ACC may have affected wildfire activity. For example, the fuel aridity metrics that we used may not adequately capture the role of mountain snow hydrology on soil moisture. Nor do we account for the influence of climate change on lightning activity, which may increase with warming (40). We also do not account for how fire risk may be affected by changes in biomass/fuel due to increases in atmospheric CO_2 (41), drought-induced vegetation mortality (42), or insect outbreaks (43).

Additionally, we treat the impact of ACC on fire as independent from the effects of fire management (e.g., suppression and wildland fire use policies), ignitions, land cover (e.g., exurban development), and vegetation changes beyond the degree to which they modulate the relationship between fuel aridity and forest fire area. These factors have likely added to the area burned across the western US forests and potentially amplified the sensitivity of wildfire activity to climate variability and change in recent decades (2, 22, 24, 44). Such confounding influences, along with nonlinear relationships between burned area and its drivers (e.g., Fig. 1), contribute uncertainty to our empirical attribution of regional burned area to ACC. Our approach depends on the strong observed regional relationship between burned area and fuel aridity at the large regional scale of the western United States, so the quantitative results of this attribution effort are not necessarily applicable at finer spatial scales, for individual fires, or to changes in nonforested areas. Dynamical vegetation models with embedded fire models show emerging promise as tools to diagnose the impacts of a richer set of processes than those considered here (41, 45) and could be used in tandem with empirical approaches (46, 47) to better understand contributions of observed and projected ACC to changes in regional fire activity. However, dynamic models of vegetation, human activities, and fire are not without their own lengthy list of caveats (2). Given the strong empirical relationship between fuel aridity and wildfire activity identified here and in other studies (1, 2, 4, 8), and substantial increases in western US fuel aridity and fire-weather season length in recent decades, it appears clear from empirical data alone that increased fuel aridity, which is a robustly modeled result of ACC, is the proximal driver of the observed increases in western US forest fire area over the past few decades.

Conclusions

Since the 1970s, human-caused increases in temperature and vapor pressure deficit have enhanced fuel aridity across western continental US forests, accounting for approximately over half of the observed increases in fuel aridity during this period. These anthropogenic increases in fuel aridity approximately doubled the western US forest fire area beyond that expected from natural climate variability alone during 1984–2015. The growing ACC influence on fuel aridity is projected to increasingly promote wildfire potential across western US forests in the coming decades and pose threats to ecosystems, the carbon budget, human health, and fire suppression budgets (13, 48) that will collectively encourage the development of fire-resilient landscapes (49). Although fuel limitations are likely to eventually arise due to increased fire activity (17), this process has not yet substantially disrupted the relationship between western US forest fire area and aridity. We expect anthropogenic climate change and associated increases in fuel aridity to impose an increasingly dominant and detectable effect on western US forest fire area in the coming decades while fuels remain abundant.

Methods

We focus on climate variables that directly affect fuel moisture over forested areas of the western continental United States, where fire activity tends to be flammability-limited rather than fuel- or ignition-limited (1) (study region shown in Fig. 1, *Inset*). There are a variety of climate-based metrics that have been used as proxies for fuel aridity, yet there is no universally preferred metric across different vegetation types (24). We consider eight frequently used fuel aridity metrics that correlate well with fire activity variables, including annual burned area (Fig. 1 and Table S1), in western US forests.

Fuel aridity metrics are calculated from daily surface meteorological data (50) on a 1/24° grid for 1979–2015 for the western United States (west of 103°W). Although we calculated metrics across the entire western United States, we focus on forested lands defined by the climax succession vegetation stages of "forest" or "woodland" in the Environmental Site Potential product of LANDFIRE (landfire.gov). Forested 1/24° grid cells are defined by at least 50% forest coverage aggregated from LANDFIRE. We extended the aridity metrics calculated at the monthly timescale (ETo, VPD, CWD, and PDSI) back to 1948 using monthly anomalies relative to a common 1981–2010 period from the dataset developed by the Parameterized Regression on Independent Slopes Model group (51) for temperature, precipitation, and vapor pressure, and by bilinearly interpolating NCEP–NCAR reanalysis for wind speed and surface solar radiation. We aggregated data to annualized time series of mean May–Sep daily FWI, KBDI, ERC, and FFDI; Mar–Sep VPD and ETo; Jun–Aug PDSI; and Jan–Dec CWD. We also calculated the aridity metrics strictly from ERA-INTERIM and NCEP–NCAR reanalysis products for 1979–2015 covering the satellite era (*Supporting Information*).

Days per year of high fire potential are quantified by daily fire danger indices (ERC, FWI, FFDI, and KBDI) that exceed the 95th percentile threshold defined during 1981–2010 from observations after removing the ACC signal. Observational studies have shown that fire growth preferentially occurs during high fire danger periods (52, 53). We also calculate the fire weather season length for the four daily fire danger indices following previous studies (10).

The ACC signal is obtained from ensemble members taken from 27 CMIP5 global climate models (GCMs) regridded to a common 1° resolution for 1850–2005 using historical forcing experiments and for 2006–2099 using the Representative Concentration Pathway (RCP) 8.5 emissions scenario (Table S3 and *Supporting Information*). These GCMs were selected based on availability of monthly outputs for maximum and minimum daily temperature (T_{max} and T_{min}, respectively), specific humidity (*huss*), and surface pressure. Saturation vapor pressure (e_s), vapor pressure (e), and VPD were calculated using standard methods (*Supporting Information*). A variety of approaches exist to estimate the ACC signal (26). We define the anthropogenic signals in T_{max}, T_{min}, e, e_s, VPD, and relative humidity by a 50-y low-pass-filter time series (using a 10-point Butterworth filter) averaged across the 27 GCMs using the following methodology: For each GCM, variable, month, and grid cell, we converted each annual time series to anomalies relative to a 1901–2000 baseline. We averaged annual anomalies across all realizations (model runs) for each GCM and calculated a single 50-y low-pass-filter annual time series for each of the 12 mo for 1850–2099. We averaged each month's low-pass-filtered time series across the 27 GCMs and additively adjusted so that all smoothed records pass through zero in 1901. The resultant ACC signal represents the CMIP5 modeled anthropogenic impact since 1901 for each variable, grid cell, and month (*Supporting Information*).

We bilinearly interpolated the 1° CMIP5 multimodel mean 50-y low-pass time series to the 1/24° spatial resolution of the observations and subtracted the ACC signal from the observed daily and monthly time series. We consider the remaining records after subtraction of the ACC signal to indicate climate records that are free of anthropogenic trends (26).

Annual variations in fuel aridity metrics are presented as standardized anomalies (σ) to accommodate differences across geography and metrics. All fuel aridity metrics are standardized using the mean and SD from 1981 to 2010 for observations that excluded the ACC signal. Although the selection of a reference period can bias results (54), our findings were similar when using the full 1979–2015 time period or the observed data (without removal of ACC) for the reference period. The influence of anthropogenic forcing on fuel aridity metrics is quantified as the difference between metrics calculated with observations and those calculated with observations that excluded the ACC signal. Area-weighted standardized anomalies and the spatial extent of western US forested land that experienced high (>1 σ) aridity are computed for each aridity metric. Annualized burned area as well as aggregated fuel aridity metrics calculated with data from ref. 50 and the two reanalysis products are provided in Datasets S1–S3.

We use the regression relationship between the annual western US forest fire area and the all-metric mean fuel aridity index in Fig. 1 to estimate the forcing of anthropogenic increases in fuel aridity on forest fire area during 1984–2015. Uncertainties in the regression relationship due to imperfect correlation and temporal autocorrelation are propagated as estimated confidence bounds on the anthropogenic forcing of forest fire area. This approach was repeated using a more conservative definition of the regression relationship, where we removed the linear least squares trend for 1984–2015 from both the area burned and fuel aridity time series before regression to reduce the possibility of spurious correlation due to common but unrelated trends (Fig. S5). Statistical significance of all linear trends and correlations reported in this study are assessed using both Spearman's rank and Kendall's tau statistics. Trends are considered significant if both tests yield $P < 0.05$.

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SYNTHESIS & INTEGRATION

Transitioning western U.S. dry forests to limited committed warming with bet-hedging and natural disturbances

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Abstract. Historical evidence suggests natural disturbances could allow more forest persistence, than expected from models, over 40 yr of transition to the net-zero emissions needed to limit warming to <2.0°C (e.g., Paris Agreement). Forests must ultimately equilibrate with committed warming from accumulated emissions. Historical dry-forest landscapes were heterogeneous from large, infrequent disturbances (LIDs) that reduced tree density and basal area, followed by slow, variable tree regeneration and recovery for 1–3 centuries. These together effectively provided bet-hedging through stand- and landscape-level heterogeneity that enhanced resistance and resilience to a diversity of unpredictable subsequent disturbances. Recent disturbances have not yet exceeded historical variability in rates and patterns, but could cause mortality of ~26–51% of dry-forest area in the transition. This also means 1/2 to 3/4 of dry-forest area could escape most mortality and the mortality area could also have substantial forest persistence. Projections are unavailable for droughts or beetle outbreaks, but they recently caused about 3–4 times as much tree mortality as did moderate- to high-severity fires. Mortality could reduce forest area if new trees do not regenerate, but 24 studies showed recent regeneration after high-severity fires was slow, but indistinct from historical variability. Survival of smaller trees provided regeneration after beetle outbreaks and droughts. Regeneration in general was projected by 2060 to decline by ~10% in one study and increase by 50% in another. If openings from disturbances increased, some grasslands and shrublands could be restored, increasing landscape heterogeneity and resistance to disturbance spread. Given these trends and our limited ability to prevent LIDs, I suggest (1) refocusing restoration to increase bet-hedging resilience to droughts and beetle outbreaks by retaining small trees and diverse tree species, (2) expanding development of fire-safe landscapes to protect people and infrastructure from unavoidable increased fire, (3) enabling more managed fire to restore and enhance stand- and landscape-scale bet-hedging, and (4) accepting that LIDs will revise resistance, resilience, and adaptation, which enhance forest persistence, particularly if post-disturbance survivors are not logged and trees are not planted. Natural disturbance and slow recovery, if bet-hedged to increase resistance and resilience, could enable substantial forest persistence.

Key words: adaptation; beetle outbreaks; bet-hedging; climate change; disturbances; droughts; dry forests; fire; natural recovery; resilience; succession.

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Introduction

Since the 2015 Paris Agreement, the world plans to reduce emissions to limit warming to much less than 2.0°C, possibly 1.5°C, and it is worthwhile to focus on how major ecosystems may transition to this more limited level of warming that is now a global commitment. Extensive disappearing climates and ecosystem changes and the need for widespread assisted migration by the mid- to late 21st century under continuing moderate- to high emissions (e.g., Rehfeldt et al. 2014) are less likely. Understanding is now needed of impacts of more limited warming for specific ecosystems. Here, I review how bet-hedging and natural-disturbance processes (Baker and Williams 2015) could help transition current dry-forest landscapes in the western United States to limited committed warming. Bet-hedging uses small trees, large trees, and diverse trees to hedge against diverse disturbances. Committed warming occurs because once emissions are reduced so they are at net zero (emissions balanced by fixation), the long persistence of emitted CO_2 in the atmosphere and high oceanic heat capacity cause global temperatures to remain elevated for centuries near where they are at net zero (Collins et al. 2013, Mauritsen and Pincus 2017).

Dry forests are major montane ecosystems (Fig. 1), covering ~25.5 million ha of the western United States (Baker 2015). Dry forests include (1) dry pine forests most often dominated by ponderosa pine (*Pinus ponderosa*) or similar pines with relatively few associated trees, and (2) dry mixed-conifer forests with pines plus several other trees (e.g., *Abies concolor*, *Abies grandis*, *Populus tremuloides*, *Pseudotsuga menziesii*). Dry-forest landscapes historically also included grasslands and shrublands, as well as younger forests (Fig. 2), some of which were seral stages after high-severity fires in dry forests, although others were more persistent (Baker 2017*a*).

To keep committed warming below 2.0°C across dry forests of the western United States, emissions may need to be net zero by A.D. 2050 when 80% of projections show 2.0°C of warming would be reached with current emissions (Karmalkar and Bradley 2017). However, globally committed warming of well below 2.0°C that might allow 2.0°C of committed warming across dry forests of the western United States could also be achieved if net-zero emissions are reached by A.D. 2060 after rapid near-term reductions (Sanderson et al. 2016). The 2.6 representative concentration pathway (RCP), the lowest scenario of the Fifth Assessment Report (AR5) from the Intergovernmental Panel on Climate Change, was thought to feasibly constrain warming to <2.0°C (IPCC 2015), but this now appears unlikely (Sanderson et al. 2016). The next IPCC report (AR6), with newer scenarios congruent with 1.5–2.0°C of committed warming, is not due until 2022. However, Shared Socio-Economic Pathways (SSPs) that are being developed suggest a 1.9 RCP could feasibly constrain warming to 1.5°C (Rogelj et al. 2018). Updated global carbon-emissions accounting and pathways make 1.5°C feasible (Tokarska and Gillett 2018, Van Vuuren et al. 2018). Thus, net-zero emissions by 2060 are needed and feasible to avoid rising above 1.5–2.0°C (Tanaka and O'Neill 2018). Therefore, I consider A.D. 2060, ~40 yr, as the main period for transitioning dry forests, after which further, slower adjustment to committed warming continues.

No projections yet exist for extent of climate loss (current climate moves elsewhere or is changed) or its effects on tree populations in dry forests for pathways leading to net-zero emissions by 2060, but perspective is still possible now. Projections of climate loss in dry forests, primarily from bioclimate models, were mostly for A.D. 2060–2100 and/or RCPs of medium to high emissions (Table 1). Loss of climate would likely be lower than in RCP 2.6 (Table 1), but specific projections are lacking. Nonetheless, by 2015, total human-induced global warming was 0.93°C (Millar et al. 2017), about 1/2 to 2/3 of the way to 1.5–2.0°C, suggesting that effects that will occur are well underway. Here, I synthesize what might ensue in dry forests based on recent trends in natural disturbances, tree mortality, and tree regeneration, aided by projections and scenarios to 2060 for low or modest emissions, where available. Further refinement will be needed, but substantial evidence is available now that can provide useful perspective.

Also, bioclimate models do not reveal ecological effects, since they usually lack demography, dispersal, or natural disturbance, and mostly

Fig. 1. Dry forests covered about 25.5 million ha of the western United States, including about 12.6 million ha of dry pine forests and 12.9 million ha of dry mixed-conifer forests. Data are Landfire biophysical settings, which predict historical vegetation (http://www.landfire.gov).

only show how the climate of an ecosystem may change, not effects (Campbell and Shinneman 2017). Climate loss is expected to move upward from lower-elevation and northward from southerly trailing edges of dry-forest ranges, and tree mortality may follow, but unpredictably. Adult ponderosa may be most vulnerable in interior populations (var. *scopulorum*) and less in Pacific populations (var. *ponderosa*) of ponderosa pine, but vulnerability in dry forests may be heterogeneous in general (McCullough et al. 2017). These models are generally only for

Fig. 2. Historical dry-forest landscapes included forests as well as openings with grasslands and shrublands, as shown here in this Whitman Cross photograph from 1897 looking south at Mesa Verde (on the skyline), southwestern Colorado, across a ponderosa pine landscape with Gambel oak (*Quercus gambelii*) shrublands and montane grasslands. Reproduced from a scanned print of the original photograph (Cross 297) at the U.S. Geological Survey Denver Library, Photographic Collection, Denver, Colorado.

adult trees, but tree regeneration may ultimately control tree persistence and expansion (Bell et al. 2014, Dobrowski et al. 2015, Petrie et al. 2017). Natural disturbances (droughts, beetle outbreaks, wildfires, and diseases) will likely cause the tree mortality as climate is lost. Forest resilience could be exceeded and a tipping point (Reyer et al. 2015) crossed. However, inertia from long tree life spans, changing disturbances, and tree survival and regeneration might allow more forest persistence (Campbell and Shinneman 2017).

Here, I first review the historical roles of large, infrequent disturbances (LIDs), post-disturbance legacies, and slow natural recovery in dry forests. Then, I review recent natural disturbances, tree regeneration, and how persistence of tree populations in dry forests to warming could be aided by bet-hedging. Emergence of climates at higher elevations may offset losses in current ranges, if dispersal succeeds (Campbell and Shinneman 2017), but is not addressed here.

Historical Variability in Natural Disturbance and Recovery in Dry-Forest Landscapes

Large, infrequent disturbances historically accomplished most renewal in dry-forest landscapes

Historical dry-forest landscapes included open, low-density stands with large, old trees and a history of low-severity fires, but probabilistic landscape-scale studies found these open forests over only about 34%, on average, of dry-forest area (Baker 2017*a*). The other 66% historically had more diverse stand structures (examples in Table 2, reviews in Odion et al. 2014, Hanson et al. 2015). Historical forests were often younger, denser, and had been burned in fires varying in intensity and severity, as described explicitly in Hessburg et al. (2007:19): "Instead, area was dominated by forest structures that were intermediate between new and old forests, i.e., by pole to medium sized, rather than large trees…. This observation suggested that before any extensive management had occurred, the influence of fire in the dry forest was

Table 1. Projected losses of current dry-forest climates for individual species that occur in current dry forests of the western United States, based on bioclimate and process-based (only Mathys et al. 2017) models.

Emissions level/location	Species	Change (%)†	Date	Emissions scenario or RCP‡	Author(s)
Low					
Arizona–New Mexico Plateau	*Pinus ponderosa*	−58.0	2075–2100	2.6	Mathys et al. (2017)
North America	*Pseudotsuga menziesii*	−22.0	2075–2100	2.6	Mathys et al. (2017)
Idaho Batholith	*Pseudotsuga menziesii*	−19.0	2075–2100	2.6	Mathys et al. (2017)
Wyoming Basin	*Pseudotsuga menziesii*	−1.0	2075–2100	2.6	Mathys et al. (2017)
Medium–high					
North America	*Abies concolor*	−13.4§	2071–2100	A2/B2 mean	McKenney et al. (2007)
North America	*Abies grandis*	−49.6§	2071–2100	A2/B2 mean	McKenney et al. (2007)
North America	*Picea pungens*	−51.2§	2071–2100	A2/B2 mean	McKenney et al. (2007)
North America	*Pinus jeffreyi*	−68.6§	2071–2100	A2/B2 mean	McKenney et al. (2007)
North America	*Pinus ponderosa*	−40.4§	2071–2100	A2/B2 mean	McKenney et al. (2007)
North America	*Pinus ponderosa* var. *ponderosa*	−45.0	2060	6.0	Rehfeldt et al. (2014)
North America	*Pinus ponderosa* var. *scopulorum*	−77.0	2060	6.0	Rehfeldt et al. (2014)
North America	*Populus tremuloides*	−24.7§	2071–2100	A2/B2 mean	McKenney et al. (2007)
North America	*Pseudotsuga menziesii*	−31.5§	2071–2100	A2/B2 mean	McKenney et al. (2007)
North America	*Pseudotsuga menziesii* var. *glauca*	−35.0	2060	6.0	Rehfeldt et al. (2014)
North America	*Pseudotsuga menziesii* var. *menziesii*	−18.0	2060	6.0	Rehfeldt et al. (2014)
High					
Southwestern USA	*Picea pungens*	−81.0	2070–2099	A2	Notaro et al. (2012)
Southwestern USA	*Pinus ponderosa*	−47.0	2070–2099	A2	Notaro et al. (2012)
Southwestern Colorado	*Populus tremuloides*	−52.0	2060	6.0/8.5 mean	Rehfeldt et al. (2015)
Southwestern USA	*Pseudotsuga menziesii*	−50.0	2070–2099	A2	Notaro et al. (2012)
North America	*Pseudotsuga menziesii*	−59.0	2075–2100	8.5	Mathys et al. (2017)
Southwestern USA	All needleleaf evergreen trees	−100.0	2099	A2	Jiang et al. (2013)

Note: Area outside current climates may also emerge with some new area of suitable dry-forest climates, not shown here.
† The change (%) is relative to the present.
‡ Emissions scenarios are A2 (High emissions), B1 (Low), and B2 (Low–Medium); RCP = representative concentration pathway, which is the change in radiative forcing (W/m^2) in 2100 relative to pre-industrial conditions, as defined for emissions scenarios by the Intergovernmental Panel on Climate Change (IPCC). RCP 2.6 is Low, 4.5 is Medium, 6.0 is Medium–High, and 8.5 is High emissions.
§ This is the "no dispersal" projection result.

of a frequency and severity that intermittently regenerated rather than maintained large areas of old, fire tolerant forest." The intermittent regeneration likely followed LIDs which varied in intensity, but were at least partly intense enough to kill substantial woody plants. Large, infrequent disturbances included fires, insect outbreaks, diseases, droughts, and blowdowns (Foster et al. 1998).

Many historical LIDs in dry-forest landscapes occurred in periodic climatic episodes. Large fires were often during droughts, as in 1848 when 41 of 63 fire-history sites across southwestern dry forests recorded this fire year (Swetnam and Baisan 1996), and in 1910 when 1.2 million ha burned in the northern Rocky Mountains (Odion et al. 2014). About 10 bark beetles had large outbreaks in dry forests (Bentz et al. 2010) when tree defenses were weakened by drought or other events, weather favored beetle reproduction, and mass attack could overcome tree resistance (Bentz et al. 2010, Negrón and Fettig 2014). An example is the 200,000- to 300,000-ha 1895–1909 mountain pine beetle (MPB; *Dendroctonus ponderosae*) outbreak in the Black Hills, South Dakota (Graham et al. 2016). Historical droughts, such as the A.D. 1574–1594 drought in the Southwest, also likely led to extensive tree mortality in dry forests (Swetnam and Betancourt 1998, Williams et al. 2013).

Large disturbances were likely infrequent in historical fire regimes and in other disturbance regimes. Modern fire regimes globally nearly all have log-normal fire-size distributions in which large fires are exponentially less frequent than small fires (Hantson et al. 2016). Historical fire-

Table 2. Examples of probabilistic studies and ancillary supporting sources that showed evidence of historical mixed-severity fire regimes, with substantial area of high-severity fire, that fostered heterogeneous historical dry-forest landscapes in the western United States.

Data source	Author(s)	Location(s)
Probabilistic		
Early aerial photographs	Hessburg et al. (2007)	WA, OR
Forest Inventory and Analysis data	Odion et al. (2014)	W USA
Early forest-reserve reports	Baker et al. (2007), Baker (2012, 2014), Williams and Baker (2014)	AZ, CA, OR, Rocky Mountains
Reconstructions–General Land Office surveys	Williams and Baker (2012*a*, *b*)	AZ, CO, OR
Reconstructions–Tree-rings at landscape scale	Sherriff et al. (2014)	CO
Ancillary supporting sources		
Early historical accounts	Baker (2012, 2014)	CA, OR
Early photographs	Baker (2009)	Rocky Mountains
Reconstructions–Paleo-charcoal	Compilation in Baker (2015)	W USA

Note: AZ, Arizona; CA, California; CO, Colorado; OR, Oregon; WA, Washington; W USA, western USA.

size and patch-size distributions in dry forests also had inverse-J shapes suggesting log-normal distributions (Williams and Baker 2012*a*, Baker 2017*a*). While rare, LIDs could be concentrated in episodes across large land areas, as were severe fires in the late 1800s in the southern Rocky Mountains (Veblen et al. 2000, Schoennagel et al. 2011, Baker 2017*b*), and large MPB outbreaks across the western United States and Canada (Jarvis and Kulakowski 2015).

The severely disturbed extent of LIDs had historical rotations (the expected time to affect the area of a landscape once) of one or more centuries. High-severity fires that killed >70% of basal area in dry forests historically had rotations of about 2–8 centuries (Baker 2015); moderate- to high-severity fires that killed 20% or more of basal area had rotations of 235–319 yr (Odion et al. 2014). Tree age distributions and early observations suggest large insect outbreaks and droughts were also infrequent events in dry forests (Blackman 1931, Swetnam and Betancourt 1998). The historical rotation for outbreaks of MPB, the main outbreak beetle in the western United States (Meddens et al. 2012), might be somewhat longer in ponderosa pine than lodgepole pine (*Pinus contorta*) forests, since ponderosa pine forests are more heterogeneous (Chapman et al. 2012). Jarvis and Kulakowski (2015) reconstructed MPB outbreaks in lodgepole pine at 10 sites in 200,000 ha of western Colorado and found four episodes from 1742 to 1910 that affected 0.5, 0.7, 0.5, and 0.4 of the 10 sites, a rotation of about 80 yr (168 yr/2.1). The rotation for drought-caused mortality in dry forests is unknown as there are no historical reconstructions. Pan-continental droughts that affected several U.S. regions occurred historically in ~12% of the last 1000 yr, but megadroughts of a decade or more, mainly in the Southwest and Central Plains, were rare in the last 500 yr (Cook et al. 2014). The 1574–1594 event, mentioned earlier, is the only historical one known to have caused extensive mortality in dry forests.

In the case of fires, the few percent that are large typically account for most of the total burned area (Strauss et al. 1989) and are often more intense (Swetnam and Baisan 1996). This importance of only a few percent of fires, the largest fires, to total burned area is evident in modern dry forests (Farris et al. 2010) and other forests (Baker 2009). Larger fires often have a mix of intensities and higher intensity, since fires become large because of rapid spread, driven by wind and drier fuels that allow more fuel consumption, increasing fire intensity (Alexander 1982). Large beetle outbreaks and lengthy droughts also appear to cause most tree mortality (Allen et al. 2010, Baker and Williams 2015, Graham et al. 2016), likely because resistance thresholds in trees are difficult to cross with smaller, less severe events (Romme et al. 1998).

Large, infrequent disturbances updated resistance, resilience, and legacies that facilitated recovery and bet-hedging

Large, infrequent disturbances with varying severities historically provided episodic adjustment across dry-forest landscapes, reducing area, density, and basal area of less disturbance-resistant trees, while increasing more disturbance-resistant trees, although current trees simply regenerate at times. Competition was lessened and the canopy was opened (Fig. 3a, b), often reducing vulnerability to subsequent disturbances for decades or longer (Parks et al. 2016).

Large, infrequent disturbances episodically tested and updated resistance, resilience, and bet-hedging across changing landscapes. Large, infrequent disturbances fostered diverse surviving tree species, sizes, and regeneration strategies that provided resistance and resilience to subsequent diverse disturbances (Table 3). This diverse stand and landscape structure and composition after LIDs could effectively provide stand- and landscape-level bet-hedging against an uncertain array of subsequent disturbances (Baker and Williams 2015). At the stand scale, bet-hedging was provided by combinations of large, old trees with thick bark that resisted mortality in fires and some beetle outbreaks (Graham et al. 2016, Welch et al. 2016), abundant small trees that resisted mortality in beetle outbreaks and droughts (Baker and Williams 2015), and diverse tree species so that some trees were not vulnerable to particular insects or diseases. At the landscape scale, areas of large trees, other fire-resistant trees, and low tree density provided landscape resistance to severe fires. Low-to-moderate fuel continuity allowed fires to spread, but with patchiness. Openings reduced ignitions, slowed disturbance spread, and reduced severity, while natural breaks could slow or terminate fires. Low–moderate contiguity of large trees and diverse patches may have reduced beetle spread and limited the size of patches of tree mortality (Graham et al. 2016). Young, recovering forests had high tree survival in beetle outbreaks (Graham et al. 2016) and droughts (Allen et al. 2010).

Natural recovery exemplifies resilience: "…the capacity of a system to absorb disturbance and reorganize while undergoing change so as to retain essentially the same function, structure, identity and feedbacks" (Walker et al. 2004:2). Large, infrequent disturbances in dry forests left behind complex effects from variable disturbance types and severities (Fig. 3), and these legacies (Foster et al.

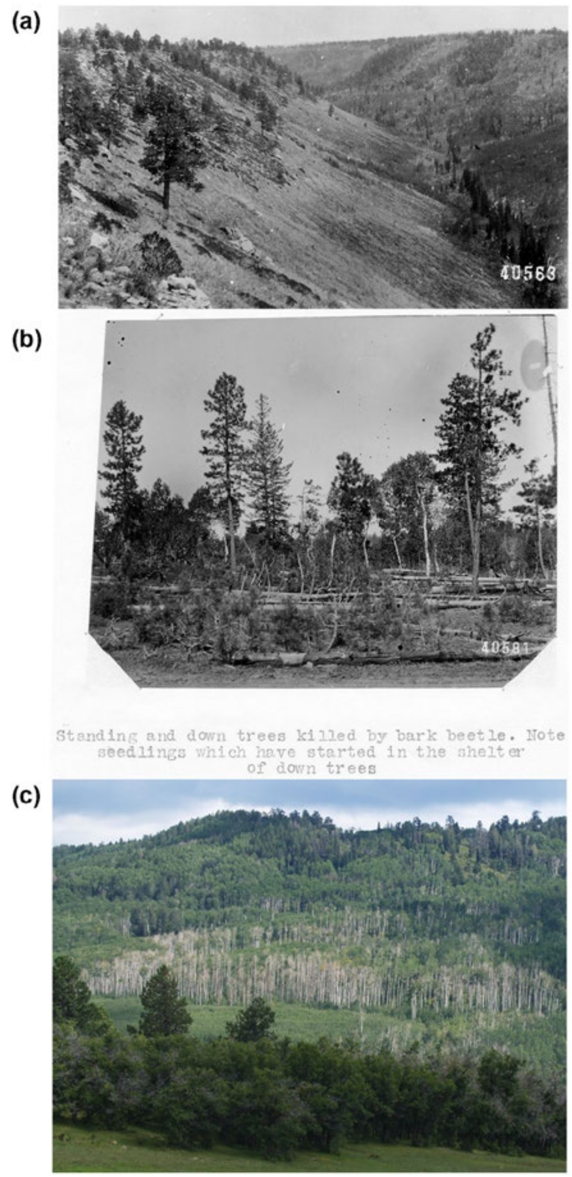

Fig. 3. Legacies after large, infrequent disturbances in dry forests: (a) a historical moderate- to high-severity fire in dry forests on the Uncompahgre Plateau, western Colorado, photograph in 1903 from Riley (1904); (b) a historical beetle outbreak in dry forests on the Uncompahgre Plateau, western Colorado, photograph in 1903 from Riley (1904); and (c) sudden aspen decline (SAD), a recent drought-linked disturbance, in southwestern Colorado, photograph by W. L. Baker, in 2006.

Table 3. Some historical structures (table entries), created by LIDs and environmental heterogeneity, that provided resistance and resilience at the stand and landscape scales to the three main types of LIDs in dry forests.

Property	Moderate- to high-severity fires	Beetle outbreaks	Droughts
Resistance–stand scale	Abundant large trees, some small trees	Abundant small trees, some large trees	Abundant small trees
	Fire-resistant trees	Diverse tree species	Diverse tree species
	Moderate fuel continuity (e.g., patches of rocks, low fuels)	Contiguous patches of small trees	Diverse topo-edaphic settings, some with more moisture
	Lower tree density/fuels, where this occurred, reducing fire severity	Lower tree density, where this occurred	Lower tree density, where this occurred
	Higher tree density/cover leading to shaded, moister fuels, where this occurred		
Resistance–landscape scale	Areas of large trees	Low–moderate contiguity of areas of large trees	Low–moderate contiguity of areas of large trees
	Areas of fire-resistant trees	Diverse patches dominated by different tree species	Diverse patches dominated by different tree species
	Areas of low tree density/fuels, where they occurred	Areas of low tree density, where they occurred	Areas of low tree density, where they occurred
	Limited areas of young, recovering forests	Large areas of young, recovering forests	Large areas of young, recovering forests
	Moderate fuel continuity	Discontinuous suitable host trees	
	Areas of higher tree density/ cover leading to shaded fuels		
	Openings that slowed fire spread (e.g., grasslands, wetlands)	Openings that broke up contiguous suitable host trees	
	Natural fire breaks (e.g., rock outcrops, streams, moist stands)	Natural openings with few or no host trees	
Resilience–stand scale	Resprouting trees and shrubs	Resprouting trees and shrubs	Resprouting trees and shrubs
	Surviving large seed trees, some patches of surviving small trees	Abundant small trees, some large surviving trees for seed	Abundant small trees, some large surviving trees for seed
		As much diversity in tree species as possible	As much diversity in tree species as possible
Resilience–landscape scale	Large seed trees, likely to survive, every 50–100 m, limited patches of small trees	Large areas with abundant small trees likely to survive, some patches of large trees	Large areas with abundant small trees likely to survive, some patches of large trees
	Diverse tree densities, basal areas, and tree species composition	Diverse tree densities, basal areas, and tree species composition	Diverse tree densities, basal areas, and tree species composition
	Most severely burned area within 100–200 m of an unburned edge	Patches with a diversity of dominant tree species	Patches with a diversity of dominant tree species

Note: LIDs, large, infrequent disturbances.

1998) or ecological memory (Johnstone et al. 2016) facilitated natural recovery. Resilience was enhanced by resprouting trees and shrubs, large old trees that provided post-disturbance seed, and variable tree densities and basal areas that provided diverse post-disturbance recovery (Table 3).

Highly variable historical tree regeneration, particularly in the Southwest

Successful ponderosa pine regeneration was limited by a required coincidence of favorable processes from seed formation to seedling survival (Pearson 1923, Feddema et al. 2013, Savage et al. 2013). However, land-survey records from 22,206 km of transects across 1.7 million ha of dry forests in the late 1800s showed that seedlings and/or saplings were present over 35–57% and dense over 20–30% of dry-forest area in Oregon, California, and part of northern Arizona (Baker and Williams 2015). Pulses of regeneration seen in some age structures were favored by canopy-reducing disturbances, particularly fire that created mineral seedbeds and reduced competition by grass, followed by fire-free periods or pluvials, that sustained regeneration (Dugan and Baker 2015). Moderate- to high-severity fires led to more regeneration than did low-severity fires (Wu 1999, Ehle and Baker 2003, Schoennagel et al. 2011, Baker and Williams 2015).

About 14% of dry-forest area, mostly in the Southwest, had sufficiently frequent low-severity fire (Baker 2017*a*) and drier climate to potentially limit regeneration to exceptional pluvials and fire-free periods (Covington and Moore 1994, Savage et al. 1996, 2013). Land-survey records document that seedlings and/or saplings were present over only 4–13% of two large landscapes in Arizona and one in Colorado (Baker and Williams 2015). However, forest age structure in two cases showed more continuous regeneration not limited to wet or fire-free periods, with broad peaks evident in one case (Mast et al. 1999). Broad episodes bring into question whether regeneration was rare and confined to unusual climatic episodes (Savage and Mast 2005).

Contrasting regeneration findings in the Southwest are also documented in early forest-reserve reports. Leiberg et al. (1904:28) said of the 329,000-ha San Francisco Peaks forest-reserve area on the western part of the Mogollon Plateau in northern Arizona:

> Reproduction of the yellow pine is, generally, extremely deficient as regards seedling and young sapling growth, except in an area lying east of Stoneman Lake and south of Morman Lake. Apparently there has been an almost complete cessation of reproduction over very large areas during the past twenty or twenty-five years, and there is no evidence that previous to that time it was at any period very exuberant.

What happened to favor regeneration near the lakes is unexplained, but a nearby landscape also had abundant regeneration. Stabler (1906:7) said of the eastern extension of the Mogollon Plateau onto Black Mesa and into the White Mountains:

> The reproduction of the yellow pine portion of the commercial forest type is wonderfully good. This in spite of the fact that the pine bunchgrass is as a rule very thick and vigorous and but little of it kept down by grazing. The fact that the grass is not grazed makes the numerous ground fires more serious than they otherwise would have been, but in spite of these fires. . .the reproduction is good and occurs in all ages.

A compelling explanation is lacking for contrasts in historical regeneration over large land areas.

Historically slow and incomplete natural recovery after LIDs in dry forests

Severely disturbed dry forests historically regenerated variably, but often slowly, and could remain unforested or sparsely forested for ≥100 yr (Table 4). Post-fire regeneration was at times very dense over extensive area in the Southwest (Fig. 4a, b). High-severity fires could be followed by extended tree regeneration lasting 20–60 yr, which could also be lagged by 15–20 yr and even have >50-yr lags with little or no tree regeneration (Table 4). Openings (grasslands, shrublands) created or maintained by high-severity fires could persist for 130–150 yr or more (Tables 4, 5) and be quite large. For example, in the Sierra, Show (1924:83) reported:

> Perhaps the most striking characteristic of the timber region of northern California. . .is the very large area occupied by brushfields. The brushfields, for the most part, are the results of fires which have destroyed the timber and allowed the brush to occupy the ground; in round numbers 1,500,000 acres [607,000 ha] are now in this condition. Of this million and a half acres probably 75% is restocking naturally, scattered individuals and groups of trees having survived the fires of the past, and can be depended on to take care of themselves. . ..

Forests often, but not always, recovered after intense fires, particularly if surviving seed trees were nearby; if so, trees regenerated and tree density and basal area increased, and forests often became denser (Fig. 4c). Probabilistic studies found dense middle-aged forests and created or maintained grasslands and shrublands in all dry-forest landscapes (Table 2). However, many pathways of forest recovery likely occurred (Kashian et al. 2007). In dry forests, open forest patches and some dense forest patches may have simply persisted and grown older, and some dense forest patches may have been thinned by competition or disturbances (Oliver 1995, Zhang et al. 2013) until a mature forest re-established (Moir and Dieterich 1988).

Including the lag before tree regeneration, recovery of a mature forest after high-severity fire historically required >100 yr (Table 6). Old growth could be reached within 150–200 yr (Mehl 1992, Hamilton 1993), but 150–300 yr for

Table 4. Historical lags in tree regeneration and the length of successful episodes of natural tree regeneration after high-severity fires in dry forests, based on tree-ring reconstructions and early observations.

Topic/Author(s)	Location	Observation
Huckaby et al. (2001)	Front Range, Colorado	Tree regeneration delayed on average by 18 yr after high-severity fires, ranging from 0 to 33 yr for 16 fires from A.D. 1531–1880
Boerker (1915:15)	Western Sierra, California	"Unlike the chaparral regions of southern California, this brush is only a temporary type and is, in most cases, the result of fire having destroyed the forest cover. . .In most cases, in from 5 to 10 years after the fire has consumed the timber, the brush takes possession of the land. . .after the brush has established itself, if seed trees are nearby, seedlings will get started and fight their way through the brush. It takes from 15 to 30 years for a seedling to get large enough to overtop the brush. . ."
Wu (1999)	San Juan Mts., Colorado	Tree regeneration concentrated within 20 yr after higher-severity fires
Baker (2017*b*)	Uncompahgre, Colorado	Tree regeneration sparse or lacking in a stand 24 yr after high-severity fire
Ehle and Baker (2003)	Front Range, Colorado	Tree regeneration concentrated within 20–25 yr after high-severity fires
Nagel and Taylor (2005:448)	Lake Tahoe Basin, California	"Tree regeneration into the chaparral stands was highest during the first two or three decades after the fire [6 fires in 1861–1882], but tree establishment continued for at least five decades after the last fire in all of the stands"
Lauvaux et al. (2016:82)	Southern Cascades, California	"Tree populations were multi-aged. Initial establishment [after 6 fires in 1864–1918] was slow and typically peaked five or more decades after the fire"
Duthie (1914:14)	Front Range, Colorado	Tree regeneration sparse or lacking for first 50 yr: "A careful reconnaissance of the region made in 1911 showed that there are over 10,000 acres of land from which all forest cover was consumed by these fires half a century ago, and upon which there has been practically no natural restocking"
Sherriff (2004)	Front Range, Colorado	Tree regeneration concentrated within 19–60 yr after high-severity fires
Huckaby et al. (2001:25)	Front Range, Colorado	". . .openings were created by a fire in 1851, and remained unforested 149 years later. . .the northern part of the area may have burned again in 1880, slowing tree regeneration"
Kaufmann et al. (2003:239)	Front Range, Colorado	". . .historical mixed severity fires and delays of regeneration into openings created by fire contributed to a very open, spatially complex and temporally dynamic landscape structure"
Pearson (1914:249)	Arizona and New Mexico	"A characteristic feature of the timbered mountains in Arizona and New Mexico at altitudes above 8000 feet is the occurrence of extensive burns. The original forests below 9500 feet were composed mainly of western yellow pine (*Pinus ponderosa*), Douglas fir. . ., limber pine. . ., Mexican white pine. . ., and white fir. . .The greater portions of the burns have grown up to quaking aspen. . ., but extensive areas are practically bare. Scattering trees of the original forest usually remain, and where this condition exists or where the burn is comparatively small conifers are generally restocking the land. . ."

conifers to regain dominance over aspen in mixed-conifer forests (Table 6). Historical high-severity fire rotations of 2–8 centuries (Baker 2015) would have often allowed full recovery to old growth before the next high-severity fire.

Fluctuating historical dry-forest landscapes of recovery had heterogeneous structure

Overall, historical dry-forest landscapes in the western United States fluctuated from infrequent large natural disturbances that included substantial severe fires, beetle outbreaks, and droughts that killed many trees, leaving a diversity of legacies, followed by 100–300 yr of natural recovery. Where tree density and basal area were reduced, vulnerability to droughts and beetle outbreaks often declined; where old trees persisted, vulnerability to severe fires was reduced. Slow, variable post-disturbance tree regeneration and growth made natural recovery after LIDs a dominant

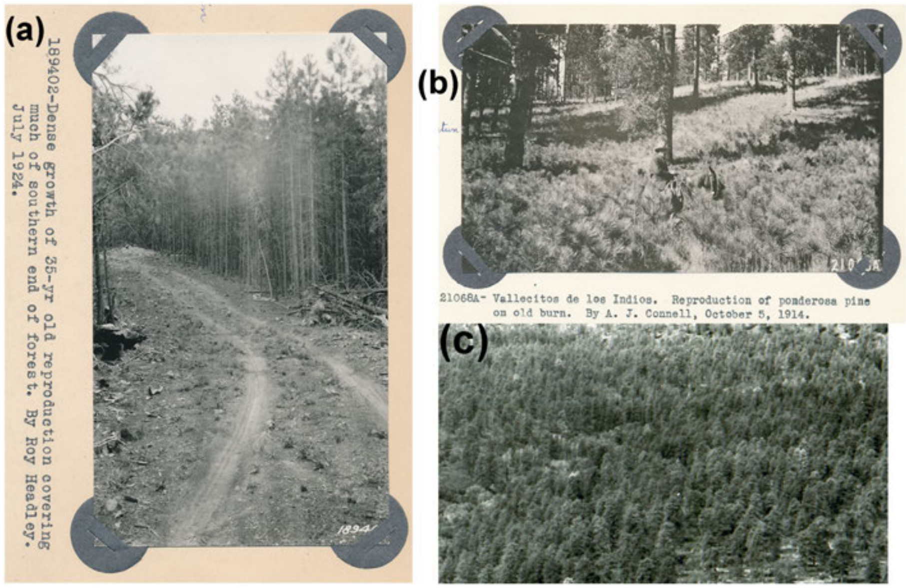

Fig. 4. Dense historical ponderosa pine regeneration after fire in the Southwest: (a) after likely large high-severity fire in the late 1800s in ponderosa pine forests, southern Coconino National Forest, Mogollon Plateau, Arizona, photograph taken in 1924 by Roy Headley, Historical Photo Collection, Region 3, U.S. Forest Service, Albuquerque, New Mexico; (b) after fire in the Jemez Mountains, New Mexico, photograph taken in 1914 by A. J. Connell, Historical Photo Collection, Region 3, U.S. Forest Service, Albuquerque, New Mexico; and (c) an example of a dense middle-aged historical forest, reproduced from a zoom of the right center of Fig. 2, an 1897 photograph by Whitman Cross.

ongoing process in most historical dry-forest landscapes. Episodes of LIDs across large areas meant that large land areas may have been synchronously recovering from natural disturbances. Infrequent disturbances and slow, variable natural recovery explain why historical dry-forest landscapes were spatially heterogeneous with a mix of old forests, middle-aged forests, recently disturbed forests, large and small openings with incipient or nearly completed regeneration, and more persistent openings, as documented by probabilistic landscape-scale studies (Table 2). This stand and landscape diversity conferred resistance, resilience, bet-hedging, and adaptation to diverse, unpredictable future disturbances, but substantial fluctuation still occurred.

Emerging Patterns of Transition to Committed Warming

Tree-mortality agents in dry forests over the last few decades

Increased tree mortality and regeneration decline or failure are expected during the transition. Increasing tree mortality is evident around the world (Allen et al. 2010). In dry forests, mortality is occurring from fires, beetle outbreaks, and directly from drought and temperature stress (Anderegg et al. 2013). Background rates of tree mortality (non-catastrophic, including all agents) increased significantly (3.3% per year, a doubling time of 22 yr), likely from warming, in the 15 old-forest plots most likely in dry forests, since they had short mean fire intervals (Van Mantgem et al. 2009: Table 1). In all plots censused from 1955 to 2007 across the western United States, 19% of trees died over the roughly 50-yr period (Van Mantgem et al. 2009), which is a 263-yr rotation (50/0.19). That would not lead to lasting loss of old forests, as 263 yr is ample time to regrow old trees, but if mortality doubled further, then it could become very limiting, and drought and heat stress could become the main cause of tree mortality (Allen et al. 2010).

Even with more severe (non-background) mortality from beetle outbreaks, droughts, and fires, there are survivors that play key stand-level roles

Table 5. Longer-term studies and observations of post-fire creation or maintenance of grasslands and shrublands after historical high-severity fires in dry-forest landscapes of the western United States.

Author(s)	Location	Years after fire	Observation
Guiterman (2016)	Jemez Mts., New Mexico	>115	Most of the area of 5 large patches (totaling 1142 ha) of mixed montane shrubland, dominated by Gambel oak (*Quercus gambelii*) that originated primarily in 1894–1900 remained largely unforested. Originating fires were likely mixed- to high-severity
Baker (2014)	Sierra Nevada, California	109–118	About 22% of montane chaparral, likely burned in high-severity fires in the late 1800s, did not become forested, and instead remained as montane chaparral, over periods of 109–118 yr
Nagel and Taylor (2005)	Northern Sierra, California	~120–140	About 38% of montane chaparral patches that originated after high-severity fires in 1861–1882 had not become forested by the 2000s
Lauvaux et al. (2016)	Southern Cascade Mts., California	~100–150	About 35% of montane chaparral patches that originated after high-severity fires in 1864–1918 had not become forested by the 2010s
Huckaby et al. (2001), Baker (2009:249)	Front Range, Colorado	~120–150	By about A.D. 2000 [120–50 yr after fires], some forests burned in high-severity fires in 1851 or 1880 had recovered to dense, middle-aged forests, but some openings were still unforested grasslands that were slowly reforesting
Baker (2017*b*)	Uncompahgre Plateau, Colorado	~130–150	About 40% of a large ponderosa pine and mixed-conifer landscape with evidence of high-severity fire in the late 1800s was nonforested (e.g., shrubs, small trees, grasslands); about half the nonforested area that was a mixture of grasslands, shrublands, recent burns, and areas with small trees was not forested by 2010, likely indicating at least century-scale stability after high-severity fires

in forest resilience (Table 3). In beetle outbreaks, most smaller trees survive as do some percentage of larger trees. A 1965–1978 MPB outbreak in the Colorado Front Range killed 25% of ponderosa pines of all sizes, especially 20–36 cm dbh, and reduced basal area by 38% (McCambridge et al. 1982). In British Columbia, a more severe 2005–2008 MPB outbreak, also with western pine beetle (*Dendroctonus brevicomis*), killed ~80% of trees, including 23–42% <15 cm dbh, 81% 15–30 cm, and 94% >30 cm over >175,000 ha, with little variation across a wide range of tree densities (Klenner and Arsenault 2009). In the Black Hills of South Dakota and Wyoming, an MPB outbreak over ~157,000 ha in 2004–2014 mostly killed ponderosas 23–43 cm dbh (Graham et al. 2016). Stands with <21 m^2/ha of basal area were little affected, but mortality increased up to 28–34 m^2/ha, where 74% of trees were killed, but 60% for >34 m^2/ha (Graham et al. 2016). Many trees survived, with means of 141 trees/ha of tree density and 11.7 m^2/ha of basal area. After beetle outbreaks, there were surviving trees of all sizes, especially small trees, as well as patches of surviving trees (Six et al. 2014). Dry forests were substantially renewed, and yet able to persist.

Mortality from droughts in dry forests has not been isolated, as beetles often ultimately kill many drought-affected trees. However, similar mortality patterns were evident with trees of all sizes killed and the highest percent mortality in larger trees (Ganey and Vojta 2011). Droughts put tall, old conifers especially at risk of replacement by shorter trees and shrubs (Bennett et al. 2015, McDowell and Allen 2015, McDowell et al. 2015), because taller trees are more physically vulnerable to failure to conduct water. A surprising 70% of a global sample of trees, in both dry and wet environments, operates with low physiological safety margins for escaping mortality from drought (Choat et al. 2012). Mortality consistent with these drought vulnerabilities is already occurring (Bennett et al. 2015). In contrast, larger trees generally better survive fires, because of thicker bark, elevated branches, and other adaptations (Baker 2009).

Table 6. Longer-term studies and observations of post-fire recovery to forest after historical high-severity fires in dry-forest landscapes of the western United States.

Author(s)	Location	Years after fire	Observations of post-fire succession in dry forests
MacKenzie et al. (2004)	Western Montana	60–100	Tree density/basal area approached pre-fire level within about 60–100 yr, basal- area increase slowed ~100 yr after high-severity fire
Baker (2014)	Sierra Nevada, California	109–118	About 78% of chaparral, likely burned in high-severity fires in the late 1800s, became forested over periods of 109–118 yr
Smith and Smith (2005), Baker (2017*b*)	Uncompahgre Plateau, Colorado	100–137	Conifers can begin to overtop aspen within about 100 yr, with mixed conifer–aspen stands at about 137 yr after high-severity fires
Nagel and Taylor (2005)	Northern Sierra, California	~120–140	About 62% of montane chaparral patches that originated after high-severity fires in 1861–1882 had become forested by the 2000s
Lauvaux et al. (2016)	Southern Cascade Mts., California	~100–150	About 65% of montane chaparral patches that originated after high-severity fires in 1864–1918 had become forested by the 2010s
Huckaby et al. (2001), Baker (2009:249)	Front Range, Colorado	~120–150	By about A.D. 2000 [120–150 yr after fires], some forests burned in high-severity fires in 1851 or 1880 had recovered to dense, middle-aged forests, but some openings were still reforesting
Baker (2017*b*)	Uncompahgre Plateau, Colorado	~130–150	About 40% of a large ponderosa pine and mixed-conifer landscape was nonforested (e.g., shrubs, small trees, grasslands) in the late 1800s; about half that area had become forested by 2010, likely indicating natural recovery after high-severity fires
Leiberg (1902:74)	Western Sierra, California	150	"The yellow pine on these tracts is mostly old growth; that is, the greater percentage of suitable size for mill timber is over 150 years of age"
Wu (1999:134)	San Juan Mts., Colorado	~150–200	"Even-aged stands still maintain their structure, such as a prominent post-fire cohort of aspen or ponderosa pine, 150 years after their last lethal fires, which occurred in the period from 1850 to 1880. . .therefore, this study estimates that all-age structure requires at least two hundred years to develop"
Kercher and Axelrod (1984)	Western Sierra, California	~250	Simulation suggested that about 250 yr would be required for Sierran mixed-conifer forests to recover and stabilize after severe disturbance
Baker (1925:89)	Central Rocky Mountains	≥250	"On the assumption that conifers found in the aspen zone will bear seed at 80 years, most areas ought to be well seeded in with reproduction in three tree generations or about 250 years in the Douglas fir-white fir zone. . .certain areas in the lower zones may require more than 250 years. . ."
Duthie (1914:14)	Front Range, Colorado	200–300	"It is estimated that two or three centuries would elapse before these burns would again be fully reforested if natural regeneration were depended upon to produce a satisfactory forest cover" (describing recovery after high-severity fires that occurred a half century earlier)
Zier and Baker (2006:261)	San Juan Mts., Colorado	Long periods	Over about a century, 40% of mixed-conifer forests visible in 25 scenes in early photographs showed increased conifers, while in 60%, there was no change in proportions of aspen and conifers, suggesting that ". . .long periods of time may be needed for conversion from aspen to conifers, if it occurs at all"

Given these vulnerabilities and documented mortality effects, what were recent sources of mortality; were severe fires, beetle outbreaks, or droughts the largest cause of non-background tree mortality over the last few decades? Details of analysis are in Appendix S1. Most important from this analysis is that insects-disease, on average, overall led to 2.1 times as much mortality area as did moderate- to high-severity fires, 1.7 times in ponderosa, and 2.6 times in dry mixed conifer (Table 7). Estimated rotations were 565 yr for moderate- to high-severity fires and 221 yr for insects-disease in dry mixed-conifer forests (Table 7). Rotations were 408 yr for moderate- to high-severity fires and 247 yr for insects-disease in ponderosa pine. These are similar to 2003–2012 mortality rotations of 500 yr for fire and 286 yr for beetles across all forests of the western United States from the inverse of annual mortality of 0.20% for fire and 0.35% for beetles, and the ratio of insects-disease to moderate- to high-severity fire of 1.75 is also similar (Berner et al. 2017). Under a hypothetical California-type drought scenario moving across dry forests (Appendix S2), an affected area of 5.5 million ha every six years would have a mortality area of 1.3 million ha (24%); if half from direct drought mortality, this would be a drought mortality rotation of 191 yr (6 yr/(0.65 million ha/20.7 million ha)). If so, droughts and insects-disease would likely account for about 3–4 times as much mortality area as severe fires.

Recent sizes and rates of beetle outbreaks, droughts, and moderate- to high-severity fire in dry forests are probably not yet outside the historical range of variability (Table 8), although evidence about historical variability is limited and some individual events have been exceptional locally (e.g., 2012–2016 California drought). Large beetle outbreaks have individually affected up to about 175,000 ha in dry forests, approaching the same scale as the 200,000- to 300,000-ha outbreak in the Black Hills in 1895–1909. The estimated recent beetle mortality rotation of 241 yr would not preclude full recovery of old-growth forests during or after the transition. The historical beetle-outbreak mortality rotation is too poorly known to be certain that this recent rate is or is not similar. Available evidence is insufficient to be able to assess historical vs. recent drought impacts on dry forests, but drought rates themselves are in general not outside historical variability in the western United States (Wuebbles et al. 2017; Appendix S3). However, if the frequency distribution of droughts does not change, $\sim 1°C$ elevated temperature alone will cause an increase in drought events sufficient to kill ponderosa pine seedlings by about 1.8 events by A.D. 2100 under RCP 2.6 (Adams et al. 2017). Larger recent moderate- to high-severity fires have individually affected about 30,000–60,000 ha, except for the 128,000-ha Rodeo–Chediski fire in Arizona (Table 8). Historical fire-size evidence is limited, in general, but the area burned at moderate to high severity on the Uncompahgre Plateau, Colorado, likely in 1879, was at the scale of about

Table 7. Affected area and estimated mortality area in dry forests across the western United States from 1999 to 2012 ($n = 14$ yr) from moderate- to high-severity fire and insects-disease.

Area and measure	Ponderosa pine	Dry mixed conifer	Total
Affected area			
Fire area (ha)	518,580	415,971	934,551
Insects-disease area (ha)	2,319,651	2,874,101	5,193,752
Fire rotation (yr)	265	367	311
Insect rotation (yr)	59	53	56
Ratio: insects-disease/fire area	4.5	6.9	5.6
Mortality area from multiplying affected fire area by 0.65 and affected insects-disease area by 0.24			
Fire area (ha)	337,077	270,381	607,458
Insects-disease area (ha)	556,716	689,784	1,246,500
Fire rotation (yr)	408	565	478
Insect rotation (yr)	247	221	233
Ratio: insects-disease/fire area	1.7	2.6	2.1
Fire area in 40 yr (% of total), if no change	9.8	7.1	8.4
Insects-disease area in 40 yr (% of total), if no change	16.2	18.1	17.2
Fire area in 40 yr (% of total), if projected	15.4	10.5	12.8
Total analysis area (ha)	9,825,679	10,910,705	20,736,384

Notes: Data on affected areas and total analysis areas are from Baker and Williams (2015). See Appendix S1 for an explanation of estimation of mortality area.

Table 8. Comparative sizes, durations, and rotations of recent large infrequent disturbances in dry forests and the expected mortality area during the 40-yr transition.

Attribute	Insects-diseases	Droughts	Moderate- to high-severity fires
Example events among the largest (ha) events since 1984 in dry forests†	∼ 157,000 ha SD/WY‡ >175,000 ha BC§	∼5, 500,000 ha CA¶ ∼700,000 ha U.S.#	30,146-ha 2012 Whitewater Baldy, NM 34,432-ha 2002 Hayman, CO 36,611-ha 2012 Ash Creek, MT 50,287-ha 2013 Rim, CA 56,174-ha 2011 Wallow, AZ 127,667-ha 2002 Rodeo–Chediski, AZ‖
Duration of these example events (yr)	4–14	5	1
Estimated recent mortality rotation (yr) across total dry-forest area††	233	191	478
Estimated historical mortality rotation (yr) across total dry-forest area for reference	>333‡‡	Unknown	362–491§§
Expected mortality area (% of total dry-forest area) in transition if no change in rotation¶¶	17.2	20.9	8.4
Projected mortality area (% of total area) in transition if climate change shortens rotation¶¶	Not available	Not available	12.8

Note: Province and state abbreviations: AZ, Arizona; BC, British Columbia; CA, California; CO, Colorado; NM, New Mexico; SD, South Dakota; WY, Wyoming.

† These are affected areas, in ha, not mortality areas.

‡ Graham et al. (2016).

§ Klenner and Arsenault (2009).

¶ From Tree Mortality Task Force (2017) and Potter (2017).

\# From Worrall et al. (2013) for roughly the area of aspen decline affecting dry mixed-conifer forests.

‖ From MTBS data (www.mtbs.gov); the area is the sum of the moderate- and high-severity classes in the MTBS pdf map of each fire.

†† From the text, in the case of drought, and from Table 7, in the case of fire and insects-diseases; rotation is the time, in years, it is expected to take for these disturbances to affect land area equal to whole landscapes.

‡‡ The original rotation estimate of >80 yr for affected area is given in the text. The rotation for mortality area can be estimated by dividing by 0.24, which is the estimate from Hicke et al. (2016) used in Table 7.

§§ The original rotation estimate from Odion et al. (2014) was 235–319 yr for affected area, and the rotation for mortality area can be estimated by dividing by 0.65, as explained in the text and used in Table 7. After division, the original 235–319 yr range becomes 362–491 yr.

¶¶ From Table 7.

75,000–90,000 ha (Baker 2017*b*), thus similar to larger recent fires. The geometric mean higher-severity patch size was 47% lower in the recent than the historical period across 624,156 ha of dry forests in the Colorado Front Range (Williams and Baker 2012*a*). Moderate- to high-severity fire in dry forests was not operating from 1984 to 2012 at rates that exceeded historical rates, and the fraction of fires that burned at high severity had not increased (Baker 2015). The recent fire-mortality rotation for moderate- to high-severity fires of 478 yr is within, but toward the long end of the estimated historical rotation of 362–491 yr (Table 8).

Assuming no increase, in ponderosa pine, a resulting fire-mortality rotation of 408 yr would lead to expected mortality area of 9.8% over the 40-yr transition (Table 7). In dry mixed conifer, a fire-mortality rotation of 565 yr would lead to mortality area of 7.1%. In ponderosa pine, an insects-disease mortality rotation of 247 yr would lead to mortality area of 16.2% over the 40-yr transition (Table 7). Similarly, in dry mixed conifer, an insects-disease mortality rotation of 221 yr would lead to mortality area of 18.1% (Table 7). Under a California-type drought scenario, a 191-yr mortality rotation would lead to a mortality area of 20.9%. Overall, if no change in rates over the 40-yr transition, actual mortality area from fire and insects-disease would total ∼26% of dry-forest area, 1/3 from moderate- to high-severity fires and 2/3 from insects-disease, a mortality rotation of 154 yr, which could still leave substantial area of old forests by the end of the transition. However, if a California-drought scenario ensued, an added 21% in 40 yr could lead to a total mortality area of ~47%, a rotation of ~85 yr, which could leave much less old forest, since it is particularly vulnerable to droughts and beetles.

Projecting possible increases in these disturbances during the 40-yr transition period is only roughly possible, and only for fire (Table 7; Appendix S3). There are no specific projections for only 1.5–2.0°C of warming on drought, fire, and insects. The U.S. Global Change Research Program reported low-to-medium confidence in a current anthropogenic climate-change effect on fire in the western United States (Wuebbles et al. 2017). Nonetheless, to estimate an upper bound on possible increases in moderate- to high-severity fire in dry forests, I used the midpoint of the low range of projected increases in area burned by A.D. 2046–2065 across 23 analysis areas under moderate emissions (RCP 4.5), which is 1.57 in ponderosa pine and 1.48 in dry mixed conifer (Baker 2015). These were the most recent area-burned projections, which are needed to estimate future mortality area. Using these, the percentage of mortality area from fire would increase from 9.8% to 15.4% in ponderosa pine and from 7.1% to 10.5% in dry mixed-conifer forests (Table 7). If combined with the hypothetical California-drought scenario, the total could reach about 51%.

Recent and projected tree regeneration in dry forests

Is there evidence of tree-regeneration decline in dry forests that could make the forest loss from tree mortality more permanent? Current rates and patterns of tree regeneration in all dry forests are relevant, but the only systematic monitoring is by the Forest Inventory and Analysis Program (FIA). Since 1995, FIA data are remeasured at 5- to 10-yr intervals on plots each representing about 2429 ha (Bechtold and Patterson 2005). Forest Inventory and Analysis data were used to analyze recent recruitment of juvenile vs. adult trees relative to climate in the western United States (Bell et al. 2014, Dobrowski et al. 2015). Ponderosa pine and Douglas-fir seedlings were much less likely to be present than were adults (28,177 plots), particularly along the warmer western and southern range margins of ponderosa pine (Bell et al. 2014). Similarly, for most conifers (13 species in 33,665 plots) in dry forests, juveniles occupied moister sites than did adults (Dobrowski et al. 2015). A caveat is that historical variability in tree regeneration was naturally high, as reviewed earlier, leaving in question whether these short periods of observation represent lasting trends.

These studies provide context, but tree regeneration after severe disturbances in dry forests is most relevant to the transition, since disturbances leave forests most dependent on regeneration. A focus has been on regeneration after high-severity fires; 24 studies, all I found, showed tree regeneration after these fires was almost universally heterogeneous (Table 9). Within the first 30 yr, substantial area lacked any conifer regeneration, while other area had adequate or dense regeneration (Table 9). Where studied, regeneration density was nearly always lowest in high-severity areas, relative to low- or moderate-severity areas. Ponderosa regeneration was commonly highest adjacent to the unburned margin of the fire and declined into the fire to low levels within 100–200 m, often attributed to seed-dispersal limitations, the hotter environment of open areas, or competition with shrubs or deciduous trees. Studies that analyzed topographic effects found regeneration especially deficient at low elevations and on south-facing slopes. Regeneration after high-severity fires in Colorado was concentrated in only three years with unusually high growing season precipitation over a 24-yr period, based on precisely dated seedlings (Rother and Veblen 2017). Less concentrated years of regeneration were evident in young adult trees, less precisely datable, after older New Mexico fires (Savage et al. 2013). Although regeneration was still sparse and favored near unburned margins at 28 and 45 yr post-fire, it was extrapolated to extend within ~50 yr across the 28-yr-old high-severity burn (Haire and McGarigal 2010). Substantial declines in post-fire tree regeneration occurred from warmer and drier conditions since 2000, suggesting possible declines with warming (Stevens-Rumann et al. 2018).

Tree regeneration after recent high-severity fires was often considered unnatural or deficient, but historical evidence now does not support this. Dense regeneration was earlier considered hyperdense and outside the natural range of variability (Savage and Mast 2005). Since then, we have found (1) dense regeneration occurred

Table 9. Studies of tree regeneration up to 64 yr after high-severity fires in dry forests of the western United States arranged by the number of years since fire.

Author(s)	Location	Years after fire	Post-fire seedling/sapling density (trees/ha)
Bonnet et al. (2005)	South Dakota Black Hills	2	>700 ha^{-1} in burn (19 transects in 1 fire) within 12 m of unburned edge, declining inward, still some at 120 m, 180 m; positive effect, scorched needles on burned mineral soil; negative, high understory cover
Keyser et al. (2008)	South Dakota Black Hills	2–5	By year 5 (36 sites in 1 fire), >1000 ha^{-1} in unburned, low and moderate severity; little in high severity
Meigs et al. (2009)	Oregon Eastern Cascades	4–5	Range 0–62,134 conifers/ha (64 plots in 4 fires); no difference among unburned, low, and moderate severity. In ponderosa forests, no ponderosa regeneration in high-severity fires and in mixed conifer limited conifer regeneration in high-severity fires
Ouzts et al. (2015)	Northern Arizona–New Mexico	7–10	Range 0–1433 conifers/ha (46 plots in 8 fires); 2 fires had 0 conifers/ha 7–10 yr after fire, 5 fires had <50 conifers/ha, 1 fire had 1500 conifers/ha; litter cover positively associated with seedlings
Crotteau et al. (2013)	California Southern Cascades	9–10	Mean 2235 conifers/ha in unburned (60 units in 1 fire), 2252 conifers/ha in low-severity, 7868 conifers/ha in moderate-severity, and 733 conifers/ha in high-severity fire; *Abies concolor* dominated regeneration over pines in more severe fire areas
Dodson and Root (2013)	Oregon Eastern Cascades	10	Mean 362 conifers/ha (18 plots in 1 fire); Range 0–1807 with 0 in 5 of the 7 plots <1000 m elevation
Collins and Roller (2013:1807)	Northern California Sierra	2–11	Omitting plots with post-fire management (leaving 21 patches in 5 fires), "there was no pine regeneration in over 90% of sampled patches". No significant effect from distance to unburned forest. Negative effect from shrubs, low seed production or soil moisture
Welch et al. (2016: Fig. 5)	California Sierra, Klamath, Southern Cascades	5–11	Mean about 500 trees/ha in yellow pine (246 plots in 12 fires), about 2000 trees/ha in dry mixed conifer (489 plots in 10 fires), interpolated from a bar graph. Overall across all vegetation types, not just yellow pine and dry mixed conifer, 54% of plots had 0–1 conifers, in interiors of severe burns, in dry areas, where more shrubs
Hanson (2018)	California Sierra Nevada/San Bernardino Mts.	1–12	Mean 3803 conifers/ha at ≤50 m into fire (20 plots in 7 fires), 1850 ha^{-1} at 51–150 m into fire (15 plots in 7 fires), 798 ha^{-1} at 151–300 m into fire (22 plots in 7 fires), and 336 ha^{-1} at >300 m into fire (25 plots in 7 fires). More within 50 m, but no significant difference among other distances. Percent shrub cover not correlated with density of conifer regeneration
Kemp et al. (2016)	Idaho–Montana Northern Rocky Mountains	5–13	Mean 7047–8153 conifers/ha (182 sites in 21 fires); Range 0–127,500 conifers/ha, but 5% of 182 sites had 0 conifers within 500 m; seedling presence probable if within 95 m of live seed source, especially if high basal area; fire severity little effect as most burn area was within 95 m of live trees
Owen et al. (2017)	Northern Arizona	12–13	Mean 84.1 conifers/ha in edge plots (6 plots in 2 fires), 41.4 conifers/ha in interior plots having no surviving trees within 200 m (6 plots in 2 fires); Range 13.0–153.8 conifers/ha in edge plots, 12.0–124.0 conifers/ha in interior plots. Regeneration significantly lower in interiors. Some long-distance dispersal (>300 m) found
Rother and Veblen (2016)	Colorado Front Range	8–15	Mean 37–1424 conifers/ha (302 plots in 6 fires), nearly all lower than pre-fire density, and 59% of plots had 0 conifers in 100 m^2 plot, with 83% of plots having <370 conifers/ha. Few seedlings in hot, dry lower elevations or on south-facing slopes, more seedlings within 50 m of live seed source, also in more southerly locations with summer rainfall
Ziegler et al. (2017)	South Dakota, Northern Colorado	11–15	Mean 43.0 trees/ha (18 plots in 3 fires)
Foxx (1996)	Northern New Mexico	0–16	Two sites in 1 fire had no seedlings in year 1, 0 and 210 trees/ha in year 8, and 218 and 318 trees/ha in year 16
Haffey (2014)	Arizona–New Mexico	6–16?	Only 24% of plots (179 plots in 9 fires) had ponderosa pine regeneration; within 150 m of a seed source, 38% of plots had tree regeneration; no regeneration beyond 250 m from a seed source. Nearly half of ponderosa pine seedlings were near a nurse structure, most often a log or large branch
Roccaforte et al. (2012)	Arizona	1–18	Range 0–11,234 conifers/ha (399 plots at 14 sites in 11 fires); 8 sites had 0 conifers/ha 1–12 yr after fire, 3 sites had 37–74 conifers/ha, 2 sites had 297–336 conifers/ha, and 1 site had 11,234 conifers/ha. Deciduous regeneration was dominant at all but 2 sites

(Table 9. *Continued*)

Author(s)	Location	Years after fire	Post-fire seedling/sapling density (trees/ha)
Chambers et al. (2016)	Colorado Front Range	11–18	Mean 225 trees/ha (305 plots in 5 fires) across unburned, low, and moderate severity. Mean tree density lowest in high severity (118 trees/ha) and in only 25% of plots, whereas 60% of low- to moderate-severity plots had regeneration. Regeneration greatest at high elevations and adjacent to unburned, declining to 10 conifers/ha at 200 m
Shatford et al. (2007)	Southern Oregon–Northern California	9–19	Mean 1694 trees/ha (24 plots in 8 fires); Range 83–8188 trees/ha. Plots showed a wide range from immediate and rapid regeneration to slow and constant to chronically limited. No significant effect of distance from seed source on seedling density; up to 84–1100 trees/ha >300 m from a seed source. Positive effect of shrub and hardwood cover
Guiterman et al. (2015)	Northern New Mexico	20	Mean 11 conifers/ha (10 plots in 1 fire); conifers present in 4 of 10 plots; maximum distance from a ponderosa seedling to unburned edge was 77 m
Rother and Veblen (2017)	Colorado Front Range	8–23	Ponderosa pine and Douglas-fir regeneration was concentrated in years with especially high growing season precipitation (413 dated seedlings at 10 sites in 5 fires); for all sites combined, three years (1995, 1998, and 2009) in twenty-four (1988–2011) accounted for most of the post-fire regeneration. Regeneration lags after the 5 fires were 0–4 yr
Passovoy and Fulé (2006)	Northern Arizona	3–27	Range 0–1052 conifers/ha (210 plots in 7 fires). Four of seven fires in years 4–8 had <50 conifers/ha and one had 26 conifers/ha at year 27, the other two fires had 170 conifers/ha and 1052 conifers/ha in years 4 and 9, respectively
Haire and McGarigal (2010)	Northern Arizona–New Mexico	28, 45	Little within years 1–8 (68 plots in 1 fire) or 1–15 (79 plots in 1 fire); ~8000, 2000 trees/ha near low-severity edge; most within 200 m of low-severity edge, but some to 304 m, 410 m; could reach all of fire area within ~50 yr
Savage and Mast (2005)	Northern Arizona–New Mexico	25–54	Regeneration began within 1–2 yr at 7 sites, within 6–10 yr at 3 sites (300 plots in 10 fires); 5 sites <200 trees/ha, 5 sites >400 trees/ha
Savage et al. (2013)	New Mexico	47–64	Regeneration did not begin for 3–20 yr (5 fires); Range (from 150 plots in 5 fires) per fire: 96–443 adult conifers/ha (≥1.4 m height and >6 cm dbh), 94–1629 seedlings and sapling conifers/ha for a total of 201–2112 trees/ha

Note: ? indicates that the Years after fire entry is uncertain.

historically over 20–30% of dry-forest areas in Oregon, California, and part of northern Arizona (Baker and Williams 2015); (2) dense younger established forests were historically common in nearly all dry-forest landscapes, suggesting past regeneration had been successful and dense (Williams and Baker 2012*b*); and (3) very dense post-fire trees are shown here to have covered large area on the southern Mogollon Plateau in northern Arizona (Fig. 4a) and occurred in the understory of burned forest in the Jemez Mountains, New Mexico (Fig. 4b). Dense and even very dense regeneration, in general and after high-severity fires, was within the historical range of variability in dry forests.

Some also considered poor regeneration after high-severity fires to indicate potentially unnatural type conversion of forests to shrublands or grasslands (Savage and Mast 2005, Haffey 2014), possible indicators of emerging tipping points (Reyer et al. 2015). However, of 24 studies, 21 (88%) covered only up to 27 yr after high-severity fires (Table 9). In general, 27 yr is insufficient, as historical tree regeneration after high-severity fires in dry forests could extend over periods of up to 60 yr (Table 4). Some large areas could even lack regeneration for ≥50 yr (Table 4) in part because of few climatically favorable periods for tree regeneration (Savage et al. 1996, Rother and Veblen 2017). A way to offset insufficient post-fire records is to extrapolate spatially (Haire and McGarigal 2010), but this has not generally been done (Table 9). Evidence is insufficient to conclude that post-fire tree regeneration is outside historical variation.

Historical tree regeneration after high-severity fires in dry forests failed or was slow at times, creating forest openings (Tables 4, 5), but recent studies often did not show modern failure was outside historical variability (Lauvaux et al. 2016). Opening creation by high-severity fire is likely operating at or below historical levels, since high-severity fires are at or below historical rates in dry forests (Baker 2015). Some openings have declined (Coop and Givnish 2007); thus, creation of new openings by high-severity fires is likely restorative (Baker 2017*b*, Boisramé et al. 2017). Openings also enhance resistance to fire spread (Boisramé et al. 2017, Owen et al. 2017) and increase the heterogeneity of landscape structure (Kaufmann et al. 2003), enhancing resistance and resilience (Table 3); thus, added openings in the transition are generally beneficial.

In contrast, tree regeneration after beetle outbreaks and droughts is not currently thought to be declining, because advance regeneration continues. In the multi-decadal period that background tree mortality increased in dry forests as temperatures rose, tree recruitment was unchanged (Van Mantgem et al. 2009). In beetle outbreaks, (1) 75% of trees <20 cm dbh survived (McCambridge et al. 1982), (2) 77% of trees <7.5 cm dbh and 58% of trees 7.5–15 cm dbh survived a severe outbreak (Klenner and Arsenault 2009), and (3) >95% of trees survived in stands with <18 m^2/ha of basal area, about 170 trees/ha in trees up to 37 cm dbh (Graham et al. 2016).

Future regeneration of dry-forest trees in general, not just after disturbances, was projected. Dobrowski et al. (2015) modeled the recruitment niche of 10 dry-forest trees relative to minimum temperature, evapotranspiration, and climatic water deficit. They then projected recruitment prevalence across the West through A.D. 2100 under RCP 8.5 (high emissions) and found recruitment declines of only about 10% or less (estimated from graphs) by A.D. 2060, at the end of the transition. Petrie et al. (2017) modeled climatic effects on stages in ponderosa regeneration (Feddema et al. 2013, Savage et al. 2013) and then projected future conditions with a water-balance model under RCP 4.5 and 8.5. Regeneration potential would be increased by +50% $\pm$ 106%, at 47 sites across the West by A.D. 2020–2059, from more flowering, seed production, and germination, especially in Arizona, Colorado, and New Mexico. After A.D. 2060, at the end of the transition, tree regeneration would decline, due to lower seedling production and survival, especially in the Pacific Northwest (−67%), but less so in the Intermountain region (−29%).

In summary, background rates of tree mortality are increasing in dry forests, and major recent droughts and beetle outbreaks have killed many trees. Recent droughts and beetle outbreaks together account for perhaps 3–4 times as much tree mortality as do moderate- to high-severity fires. Together, natural disturbances could cause tree mortality over 26–51% of dry forests in the transition. Tree regeneration is not apparently outside historical variability and is projected to only slightly decline or even increase. Some opening creation from tree mortality followed by tree-regeneration failure could actually restore grasslands and other openings. Current dry-forest area is not all at risk, as 1/2 to 3/4 could escape substantial mortality under committed warming, and the remainder could have more resistant and resilient forests that persist more than expected.

Transitioning Dry-Forest Landscapes

Large, infrequent disturbances that will enact tree mortality during the transition are capable of rapidly affecting millions of hectares and are generally beyond control. The spatial extent (25.5 million ha) of dry-forest landscapes and associated human communities and infrastructure provides large inertia for preparations. Our ability to control LIDs by manipulating forest structure is limited, and structurally ideal or restored landscapes may help, but a broader tie-in strategy, with a refocus on bet-hedging to enhance resilience to natural-process management may be more feasible and effective.

Limited ability to directly prevent LIDs or reduce their impacts on dry forests in the transition

Our ability to directly prevent LIDs or reduce their impacts is limited. Graham et al. (2016) reviewed the long history of failed attempts at controlling bark beetles through direct suppression or indirect manipulation of forest structure. At best, evidence suggests thinning, the most common manipulation, might modify the extent and pattern of tree mortality over limited area. Fettig et al. (2014) found thinning treatments to reduce tree mortality from MPB were costly and did not work during outbreaks without added direct control; thinning worked in some cases in ponderosa pine forests but had no significant effect in others. Six et al. (2014) also found thinning could possibly work at times, but failures occurred during outbreaks, and unthinned stands may actually have more survivors. Droughts are not directly controllable. Some drought treatments aim to protect particular trees by reducing competition (McDowell and Allen 2015), but this will likely ultimately fail under hotter droughts (Bennett et al. 2015). Fuel treatments to reduce fire spread and severity have also not been very effective: "Mechanical fuels treatments on U.S. federal lands over the last 15 yr (2001–2015) totaled almost 7 million ha. . .but the annual area burned has continued to set records" (Schoennagel et al. 2017:4586). Schoennagel et al. (2017) explained that treatments can reduce fire severity and increase low-severity fire in some dry forests, but the probability of having an effect is low, as only about 1% of treatments actually experience wildfire each year. Thinning treatments have been ineffective for LIDs in dry forests, in general, and are best as short-term, small-area holding actions (Six et al. 2014).

Can ideal or restored landscapes discourage LIDs from crossing tipping points?

Evidence that ideal or restored landscapes can discourage tipping points is also limited. To maintain MPBs in an endemic condition, discouraging an outbreak, Graham et al. (2016:157) suggested, based on high tree survival in an outbreak: ". . .heterogeneous landscapes composed of stands with heterogeneous structures and containing densities in the neighborhood of 80 feet^2 [18.3 m^2/ha] of basal area are resistant to MPB infestations. . ." However, they said forests in the late 1800s were dominantly in that condition when the largest known MPB outbreak in ponderosa pine forests occurred, the 200,000- to 300,000-ha 1895–1909 outbreak in the Black Hills. Thus, ideal landscapes might only be resistant to some beetle outbreaks. Lundquist and Reich (2014:472) said: "Existing models show that diverse composition and configuration is the best and possibly only long-term, large-scale approach to bark beetle management. . ." For droughts, ideal stands and landscapes have not emerged, and there is little historical evidence. For wildfires, low-density stands with large, old ponderosa pines and few understory trees and shrubs are most resistant and resilient to subsequent wildfires (Allen et al. 2002). However, probabilistic studies (Table 2) have shown this structure was a significant, but not dominant component of most historical dry-forest landscapes, which had more heterogeneous stands across heterogeneous landscapes (Table 2). Thus, historical and ideal landscapes appear congruent, and achievable through restoration, for droughts and beetle outbreaks, and at least partly for fires.

Idealized and historical stand and landscape structures are unlikely to prevent LIDs from causing substantial tree mortality, some tree-regeneration failures, and some opening creation, as these were natural components of historical processes of disturbance and recovery in dry forests. Large, infrequent disturbances occurred in historical dry-forest landscapes and led to substantial landscape change and large fluctuations. Dry-forest landscapes appear to have been capable of general recovery after LIDs (Table 6), but some nonforest, created by disturbance, persisted for 100–150 yr or more (Table 5). Whether tipping points were crossed or this simply represents slow natural recovery is uncertain, but in either case dry-forest landscapes were dynamic and subject to large fluctuations that created and renewed resistance and resilience features that fostered bet-hedging (Table 3).

Natural fluctuation means that restoration and management in dry forests are less a matter of restoring and managing forest structures (Table 3) and more a matter of restoring and managing natural disturbance and recovery processes. Most structures are inherently ephemeral, persisting for only years or decades, and are quickly recreated by disturbances, and thus do not warrant intentional restoration. Widespread micro-management of fuel loads and forest structures after LIDs, based on fears of hypothetical mass fires (Stephens et al. 2018), is likely a waste of resources, because extensive structure management to reduce severe fires has been ineffective (Schoennagel et al. 2017). However, old trees and their associated stand- and landscape structures could persist for centuries, are not recreated by disturbances, and have been lost to excessive logging. Structure restoration and management make sense for these long-persisting structures not created quickly by disturbances, but process management, and associated facilitative structures (e.g., bet-hedging) now make sense for most landscape restoration and management.

A tie-in strategy using bet-hedging and process management of disturbances in the transition

Given substantial uncertainty and limited ability to control LIDs, a broad tie-in strategy, using actions beneficial for people and nature no matter what occurs, could likely facilitate more forest persistence in the transition. Suggested actions include (1) refocusing intentional ecological restoration on bet-hedging using historically congruent structures that provide resistance and resilience to diverse future disturbances (Baker and Williams 2015), (2) expanding development of fire-safe landscapes for people and infrastructure (Schoennagel et al. 2017), (3) expanding managed fire, and (4) accepting that LIDs will beneficially revise resistance, resilience, and genetic adaptation (Six et al. 2014). Restoring forest structure is costly, and resistance structures may fail, favoring structures that facilitate more process-based restoration (Millar et al. 2007).

Droughts and beetle outbreaks are likely to be 3–4 times as important as fires during the transition, which means that abundant small trees and high tree species diversity are now the more important resistance and resilience structures for transitioning dry forests (Table 3). Most large restoration programs (Reynolds et al. 2013, Addington et al. 2018) are likely to be ineffective, as they are focused on structures resistant to fire, when it is more likely that drought and beetles will determine the structures that persist in the transition. These programs to thin forests to resist damage by moderate- to high-severity fires have unfortunately reduced the small trees and diverse tree species that most provide resilience to droughts and beetle outbreaks. These programs could be quickly modified to instead retain small and diverse trees. In forests already deficient in small and diverse trees, if only one prescribed fire occurs before managed wildfire for resource benefit ensues, that last fire will likely stimulate some tree regeneration to repopulate small trees. If low-severity fires are generally managed to mimic historical spatial and temporal variability, opportunities will likely occur for diverse trees to repopulate (Baker 2017*a*).

The unpredictability of future disturbances suggests hedging bets (Millar et al. 2007, Baker and Williams 2015) in stand-level restoration by maintaining large and small trees and available tree species diversity. After restoration, most stands, even open low-density stands, can have numerical dominance by small trees of all available species, but also sufficient replacement larger trees of all available species. Early land surveys across 1.7 million ha of dry forests showed small trees (typically <40 cm dbh) were, on average, 62% of total trees (Baker and Williams 2015). Given loss of large trees to logging, retaining all large trees, and mid-sized trees that are their future replacements, is sensible. After disturbances, successful tree regeneration is favored by large surviving trees that provide seed within about 100–200 m (Table 9). Larger trees may later be lost to hotter droughts and beetle outbreaks. However, if there were 20–50 larger (>40 cm dbh) trees per ha, and >5% survived, that could provide needed surviving large trees. Bet-hedging in restoration leaves abundant trees of all species and sizes with small trees dominant.

At the landscape scale, diverse historical forest structures could reduce the spread and effects of natural disturbances (Table 3) and bet-hedging at this key scale of LIDs is very important now. For fires, areas of large fire-resistant trees, openings, and naturally moist areas or shaded fuels provide resistance and favor survivors that aid post-fire resilience. For beetle outbreaks and droughts, diverse tree species and smaller trees provide the most important resistance and resilience. Recovering younger to middle-aged forests were common historically, based on studies in Table 2, and naturally conferred resistance and resilience to beetles and droughts. Kautz et al. (2017:534) found that "...more than 60% of global forests are in various stages of recovery from a past disturbance at any given time." Protecting young, naturally recovering forests is thus feasible, congruent with historical forests, and a key landscape part of a process-restoration approach (Baker 2017*b*). Young forests can survive beetle outbreaks and possibly droughts at much higher rates than older forests (Graham et al. 2016). To maximize bet-hedging, mixtures of diverse resistance and resilience structures across landscapes, with much more focus on beetle outbreaks and droughts, in addition to fire, are now more congruent with expected LIDs.

It would benefit both people and nature to rapidly increase protection of infrastructure, homes, and communities from increased wildfires, and this would also enable more managed use of natural disturbances. With $\sim$7 million ha of fuel-reduction treatments, but fires still burning homes (Schoennagel et al. 2017), we need to prevent the expansion of developments into fire-prone settings and finish full fire protection around all homes, infrastructure, and communities. Effective ways to reduce vulnerability and live with wildfire have been articulated (Cohen 2000, Baker 2009, Calkin et al. 2014, Moritz et al. 2014, Smith et al. 2016, Schoennagel et al. 2017). Tools include fire-safe construction, zoning, building codes, incentives, easements, growth boundaries, insurance policies, and other means (Kennedy 2006, Baker 2009, Schoennagel et al. 2017). Homeowners can use fire-safe construction focused on the home-ignition zone (Cohen 2000). Possibly most effective is for communities and developments to designate growth boundaries that enclose a wide margin of open, fire-resistant land uses that can serve as an effective fire break (e.g., ball fields, wetlands, irrigated agricultural fields), whether they are already in place or require construction. This alone would definitively stop expansion into fire-prone vegetation, protect key concentrations of people and infrastructure from fire, and make it more feasible to manage wildfires for resource benefit on adjoining public lands (Baker 2009).

Among LIDs, using more managed fire for resource benefit would be effective wherever it is safe and feasible, especially in the early part of the transition. Moderate- to high-severity wildfire has the longest recent rotation (Table 8) and is the only LID that can be directed. Prescribed fires are typically not sufficiently intense for effective restoration (Van Wagtendonk and Lutz 2007, Baker 2014), but prescribed burning once across landscapes and near homes and infrastructure is best before initiating managed fire (Baker 2017*a*). Managed wildfires can accomplish more renewal and enhancement of resistance and resilience, and also help prepare communities for future LIDs (Schoennagel et al. 2017). Expanding managed fire is scientifically supported (North et al. 2015, Schoennagel et al. 2017), and solutions to institutional barriers are identified (Stephens et al. 2016). Managed fires early in the transition are especially important to reduce tree density and basal area, which can lower vulnerability to droughts and beetle outbreaks more likely with higher temperatures later in the transition. Early managed fires could also stimulate tree regeneration, when it is favored (Petrie et al. 2017). Recovering small trees and entire stands recovering after fires provide resilience to droughts and beetles and foster asynchrony in tree populations that can slow disturbance spread (Millar et al. 2007, Seidl et al. 2016). If openings or low-density patches are created by early disturbances, those could also reduce later vulnerability. Openings are less likely to ignite (Baker 2009), may slow fire, and could hinder beetle spread.

Acceptance of the benefits of LIDs and protection of the post-LID environment are sensible, since we cannot prevent LIDs in the transition. For example, bark-beetle outbreaks may naturally thin and diversify forest structures (Oliver 1995, Graham et al. 2016), updating resistance and resilience, while increasing biodiversity and furthering genetic adaptation to emerging climates and LIDs (Six et al. 2014, Beudert et al. 2015). Large, infrequent disturbances also provide selection against individual trees not resistant to the LID or post-LID environment (Six et al. 2014). Survivors and post-disturbance regeneration can revise tree adaptations to both emerging climate and patterns of LIDs. Rapid evolutionary response to extreme climatic events is possible, even in long-lived trees (Grant et al. 2017). For example, MPB outbreaks favor survival of slower-growing ponderosa pines, even though faster-growing trees may outcompete them at other times (De la Mata et al. 2017). Also, since post-LID tree regeneration is favored within 100–200 m of surviving trees (Table 9), and LIDs can leave isolated patches of surviving trees that, by chance, have different gene frequencies, the opportunity for locally adapted genetic change is high. As Howe (1976:263) said: "Prevention of major conflagrations... would eliminate the ingredients for drift, i.e., the replacement of large, continuous populations by tiny islands of isolated interbreeders from which most ensuing regeneration would emanate..." To preserve genetic adaptation of trees to emerging climate and LIDs, it is important to not prevent LIDS, not plant trees, and not log post-disturbance survivors (Lindenmayer et al. 2008). Genetic adaptation to committed warming could enhance possibilities for more dry-forest persistence in the transition and during the extended period of adjustment after the initial transition to committed warming.

Conclusions

Limiting warming, as with the Paris Agreement, should enable more persistence of current dry forests in the transition to committed warming than projected by models. Here, I reviewed evidence that (1) LIDs historically produced diverse forest stands and landscapes that naturally provided resistance and resilience to subsequent disturbances; (2) LIDs cannot be generally prevented through direct control or indirect manipulation of forest structure; (3) fires, droughts, and beetle outbreaks are not yet having effects in dry-forest landscapes that appear outside historical variability; (4) in the last few decades, droughts and beetle outbreaks have caused roughly 3–4 times as much tree mortality as fires; (5) primary opportunities to enhance forest persistence are from expanded bet-hedging at stand and landscape scales focused on resistance and resilience to droughts and beetle outbreaks, and facilitating adaptation as disturbances occur; and (6) 1/2 to 3/4 of dry-forest area could possibly escape most mortality during the transition.

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Supporting Information

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RESEARCH ARTICLE

Restoring and managing low-severity fire in dry-forest landscapes of the western USA

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Abstract

Low-severity fires that killed few canopy trees played a significant historical role in dry forests of the western USA and warrant restoration and management, but historical rates of burning remain uncertain. Past reconstructions focused on on dating fire years, not measuring historical rates of burning. Past statistics, including mean composite fire interval (mean CFI) and individual-tree fire interval (mean ITFI) have biases and inaccuracies if used as estimators of rates. In this study, I used regression, with a calibration dataset of 96 cases, to test whether these statistics could accurately predict two equivalent historical rates, population mean fire interval (PMFI) and fire rotation (FR). The best model, using Weibull mean ITFI, had low prediction error and $R^2_{adj} = 0.972$. I used this model to predict historical PMFI/FR at 252 sites spanning dry forests. Historical PMFI/FR for a pool of 342 calibration and predicted sites had a mean of 39 years and median of 30 years. Short (< 25 years) mean PMFI/FRs were in Arizona and New Mexico and scattered in other states. Long (> 55 years) mean PMFI/FRs were mainly from northern New Mexico to South Dakota. Mountain sites often had a large range in PMFI/FR. Nearly all 342 estimates are for old forests with a history of primarily low-severity fire, found across only about 34% of historical dry-forest area. Frequent fire (PMFI/FR < 25 years) was found across only about 14% of historical dry-forest area, with 86% having multidecadal rates of low-severity fire. Historical fuels (e.g., understory shrubs and small trees) could fully recover between multidecadal fires, allowing some denser forests and some ecosystem processes and wildlife habitat to be less limited by fire. Lower historical rates mean less restoration treatment is needed before beginning managed fire for resource benefits, where feasible. Mimicking patterns of variability in historical low-severity fire regimes would likely benefit biological diversity and ecosystem functioning.

Introduction

Low-severity wildfires significantly shaped dry forests in the western USA, but historical rates (e.g., mean interval, area burned) of these fires remain uncertain in a time of altered and further changing fire regimes. Low-severity fires periodically burned the understory of historical dry forests, changing fuel loads, composition, diversity, and ecosystem processes without killing most canopy trees [1–2]. Dry forests in the western USA cover 25.5 million ha and include dry pine forests, dominated by ponderosa pine (*Pinus ponderosa*) or other dry pines, and dry mixed-conifer forests that also have firs (*Abies concolor*, *A. grandis*, *Pseudotsuga menziesii)* and other trees [3]. Past reconstructions of low-severity fire in dry forests, using tree-rings, focused on long records of dated fire years in small plots, and most were not intended to accurately estimate key rate parameters of low-severity fire [1–2] needed to restore and manage low-severity fire across large landscapes. These small-plot reconstructions have known inaccuracies and biases if inappropriately used for this purpose [1, 4–11]. Fortunately, new landscape-scale and small-plot reconstruction methods [1, 11] overcome many known inaccuracies and biases in estimating historical low-severity fire rates, but limited new estimates are available.

This situation leaves a weak current basis for restoring and managing low-severity fire, using historical rates as a guide, across dry-forest landscapes. Here I: (1) develop regressions for estimating mean historical rates of low-severity fire from past reconstructions using a calibration dataset, and then (2) apply these regressions to estimate mean historical rates of low-severity fire for a large dataset of past reconstructions across the western USA, and (3) assess the applicability of these new estimates across dry-forest landscapes. These new estimates are directly usable in restoring and managing low-severity fire in the parts of dry forests of the western USA where low-severity fire was historically predominant, and provide a West-wide perspective on variability in historical mean rates of low-severity fire in these parts of dry forests. As discussed later, variability around mean rates is also an essential attribute of a low-severity fire regime.

Estimated mean historical rates of low-severity fire need to be fairly accurate, for restoring and managing low-severity fire, because key effects of fires on biological diversity, ecosystem functioning, and post-fire recovery operate significantly differently across a narrow range of mean rates. For example, understory fuels in dry forests, reduced by a single fire, often recover to pre-fire levels in about 7–25 years [12–14]. If mean fire intervals for low-severity fires were 10–15 years, understory fuels would often have been kept at relatively low levels, but if mean intervals were 25 years or more, then understory fuels would more often have been fully recovered and generally higher. Fires that are too frequent can reduce the ecological roles of the forest floor in replenishing soil nutrients and organic matter, enhancing absorption of water and nutrients, and providing habitat for microbial communities, potentially reducing long-term forest productivity [15]. Habitat for wildlife that use snags or down wood could be adversely affected by fire that is too frequent [15], which can also reduce understory plant species richness, possibly due to depletion of soil nitrogen [16]. Native shrubs, historically abundant in some dry forests, may also be reduced by fire at intervals less than about 20–30 years [17]. However, fire-stimulated shrubs in the understory of dry forests may also decline if low-severity fire rates are too low [18]. Insufficient low-severity fire can allow tree density or other understory shrubs to increase, reducing nutrient cycling and understory diversity, and increasing fire severity [16, 19].

Maintenance of tree populations in dry forests also depends on the balance between tree natality and mortality, a balance strongly shaped by rates and patterns of fires. Fire intervals for successful tree regeneration were likely long relative to historical mean intervals, as fires at short intervals can kill most small trees [6]. Patchy surface fires could alone allow survival of small trees in unburned areas [20]. Also, seedlings regenerating in openings may produce limited fuels, enhancing fire patchiness that favors seedling survival [21]. Where fire kills overstory trees, a resulting mineral seedbed and reduced competition with grass can enhance tree regeneration, if other factors (e.g., seed production) co-occur [22]. A fire-quiescent period is also needed [23]. Long intervals may occur over large areas in wet periods, or stochastically from variability in fire. In contrast, mortality of larger trees from single low-severity fires can reach 7–8%; if repeated every 10 years, larger trees could be reduced by half in a century, but, assuming the same 7–8% rate repeated every 50 years, larger trees would be halved in 500 years [17]. Thus, tree populations, both young and old trees, are sensitive to rates and patterns of low-severity fire.

Rates and patterns of low-severity fire also affect how resistant and resilient dry forests are to future fire, drought, and beetle outbreaks [24]. Open, low-density forests relatively free of shrubs and small trees can be produced by repeated low-severity fires, and may be more resistant to subsequent higher-intensity fires than are denser forests, with more shrubs and small trees [25]. Forests subject to repeated low-severity fires could even be self-limiting, if the rate of fires is high, possibly promoting continuing low-severity fire rather than higher-severity fires [26]. However, if a deficiency in tree regeneration occurs because of too-frequent fires, dry forests would be vulnerable to subsequent regeneration lags or failures after droughts and beetle outbreaks that are a higher current risk than are severe fires [24]. Too little low-severity fire could increase fire severity, but too much could reduce higher-severity fires that enhance spatial heterogeneity, a key source of forest resilience to future disturbances [3].

Research has enhanced understanding of the importance of rates and patterns of low-severity fire to biological diversity, ecosystem functioning, and sustainability of dry forests, but estimated historical rates and patterns of low-severity fire remain uncertain. Newer methods for accurately reconstructing rates of historical low-severity fire promise to eventually resolve uncertainty, but improved estimates, the focus here, might be possible from past research.

Measures and estimators of mean rates of low-severity fires

Terms and measures

A *low-severity fire* in this study is a fire that burns in the understory of a forest, and is often defined as causing mortality or topkill of no more than about 20% of stand basal area [27–28]. These fires are not usually burning in the canopy independently, instead torching upwards from surface fuels into single or small groups of trees. These fires could also be called low-moderate severity to reflect some canopy mortality, but the extent of canopy mortality from these fires is poorly known [17].

Several measures of mean rates of fire also need explanation. At a point in a landscape, the average interval between fires is the point *mean fire interval* (point MFI). The average MFI across multiple points in a landscape provides a sample estimate of the *population mean fire interval* (PMFI) for a particular landscape, which is the grand mean fire interval across the landscape [6]. Fire-interval data at points have interval distributions that often are skewed, not normally distributed. Alternative measures of central tendency, such as the median, can characterize these distributions. These distributions often can also be fit by the flexible two- or three-parameter Weibull distribution, which has a shape parameter that describes the form of the distribution (e.g., lognormal), a scale parameter that represents the 63rd percentile of the distribution, and a shift parameter to set the location of the distribution [29]. The mean and median of the fitted Weibull distribution, which can offset unusual values in actual data [29], are useful alternative measures of central tendency. Descriptors of variation (e.g., standard deviation) are relevant for all measures. The *fire rotation* (FR) is the expected time for fire to burn an area equal to the area of a landscape of interest [17]. The FR for a landscape is equivalent to the PMFI, which was shown analytically [6] and through simulation [7–8]. Fire-interval data at points can be used to estimate the PMFI, or area-burned data across a landscape can be used to estimate the FR. PMFI estimates at points and FR estimates across areas are the fundamental, equivalent estimators of mean rates of fire, as they show how often points experience fire and the equivalent time it takes for fire to burn across a landscape.

Estimators of the Population Mean Fire Interval (PMFI)

For reconstructions of mean low-severity fire rates in the pre-EuroAmerican period, which are predominantly derived using tree-ring and fire-scar methods, the actual intervals needed for estimating PMFI can be sampled and processed in several ways. Fires do not physically leave a scar on every tree that burns [30], and the scarring fraction (SF), the fraction of live trees that receive a scar from a fire, may be moderate or even low. The intervals derived from scarred trees are thus simply estimators of the actual fire intervals that occurred at a point.

The most widely used fire-interval estimator is the mean composite fire interval (mean CFI), often also called the mean fire-return interval (MFRI) or even, to confuse matters, the MFI itself, which is not the estimator but instead what is being estimated. This estimator seeks to offset the fact that SF is < 1.0 by compositing scar records across a set of nearby trees, which together are expected to contain a more complete record of fires that burned the point. To calculate mean CFI, the user creates a pooled "composite" list of fire years that burned any tree in a set of sample trees, then the estimated intervals are those between fire years in the composite list. However, this composite list of all fires may contain small spot fires that have little ecological effect, and users often also report estimates for larger fires that scarred more than 10%, 25%, or another percentage of scarred trees. Various measures of central tendency can be calculated, including the mean, median, and Weibull measures. I distinguish variants here using combined terms, such as mean CFI-all fires, mean CFI-10% scarred, or median CFI-25% scarred. Mean CFI-10% scarred, for example, is the mean composite fire interval for fires recorded on $\geq$ 10% of scarred trees.

Another commonly used estimator is the mean individual-tree fire interval (mean ITFI). This estimator is calculated in two steps. First, the intervals between fires on an individual scarred tree are used to estimate the MFI for that tree. Second, the grand mean of each tree's estimated MFI is calculated across a set of sample trees. In this case, restrictions (e.g., 25% scarred) are not used, but alternative measures of central tendency are, so there are fewer variants.

Finally, we developed an estimator, the mean all-tree fire interval (mean ATFI), which seeks to offset SF < 1.0 by using an estimated SF to predict the total number of scars that would have occurred if SF was 1.0 [7–8, 11]. This estimator has been shown to be the best available estimator of PMFI [11], but it is not used in this paper because few ATFI estimates are currently available.

Estimators of the Fire Rotation (FR)

Area-burned estimates for calculating FR can be derived from three main sources: (1) area burned in recent fires from agency polygon fire records or fire-atlas records or from remotely sensed data, (2) historical area burned from fire-year maps reconstructed from scarred-tree or plot locations, or (3) historical area burned reconstructed using a ratio method and scarred-trees or plot records, or comparable data in a table or graph.

Polygon fire records or fire-atlas records are available from public land-management agencies, and are most complete and accurate after about A.D. 1980. Early data are often from fire perimeters sketched on a map, but later data may have been from remotely-sensed data [31]. Small fires were not always mapped. Accuracy of boundaries of fires in fire-atlas data, relative to tree-ring reconstructions and remote-sensing data, was moderately high in one study, sufficiently accurate to use in some research [31]. In another study, tree-ring methods underestimated fire extent relative to fire-atlas maps, which also had some errors [32]. A larger study showed closer agreement between fire-atlas data and tree-ring reconstructions of fires [1].

Fire-year maps are typically reconstructed from tree-ring and fire-scar data collected at a grid of points or a set of random points. Fire scars near the points are dated, dates are displayed on a map or in GIS, and a fire perimeter is placed around the points common to a fire year [33–34]. The boundary is positioned using a set of fire-spread principles [35], Voronoi polygons centered on the points [1], convex hulls [32], fuzzy-set methods [36], inverse-distance weighting [23, 33], or indicator-kriging [33–34]. If grid points are close, unburned area may be most accurately mapped, but a larger grid spacing is often needed to allow sufficient area to be sampled, leading to less precision in boundaries and unburned areas [34]. Smaller fires also will be missed more often with larger grid spacing. Larger fires that contribute most to fire rotation are mapped the best. Fire rotation has been shown to be estimated within about 10% of the value obtained from fire-atlas data [1, 11].

A non-spatial ratio method estimates area burned within a study area as proportional to the percentage of sample trees scarred in a particular fire year or the percentage of plots in which a particular fire year is recorded on sample trees. The equation [37] is:

$$A_i = (AT^* NS_i)/(NST - NRE) \tag{1}$$

where A_i is area burned in year i, AT is the study area size, NS_i is the number of scarred trees or plots recording a fire in year i, NST is the total number of scarred trees or plots, and NRE is the number of scarred trees or plots eliminated by subsequent fires. This method is most accurate when the number of scarred trees or plots is large and these are well distributed across a sample area [1, 37]. However, scarred trees are often clustered [30], which could lead to ratio estimates that are biased and too short. Because the location of scarred trees or plots is not used, unburned area may also be underestimated. In a large modern corroboration study, the ratio method accurately estimated area burned of larger fires ($>$ 100 ha), that accounted for 97% of total area burned, and fire rotation from total plots was 89% of fire rotation from fire-atlas data [1].

FR can be calculated, using any of the three sources of data, by the equation [17]:

$$FR = (ObservationPeriod/FractionBurned) \tag{2}$$

where FR is fire rotation, in years, $ObservationPeriod$ is the period, in years, for which there are mapped or reconstructed records of fire, and $FractionBurned$ is the fraction of the study area estimated to have burned during the observation period, obtained by summing the areas of fires or the estimated fraction burned from ratio estimates.

Perspectives on estimating PMFI/FR and interpreting mean CFI

A central area of analysis and discussion by our research group has been about whether past mean CFI and ITFI estimates from small plots accurately estimate PMFI/FR. Other studies (e.g. [38]) were more focused on reconstructing a long history of dated fire years across a network of locations, not so much accurate rates of fire across landscapes. I continue the rate focus here. An earlier review suggested mean CFI is too short and mean ITFI is too long as an estimator of PMFI/FR [6]. This study suggested mean CFI was often too short from compositing across too much area or samples and mean ITFI was too long, as it does not offset unrecorded fires that occur because SF is $<$ 1.0 [6].

Reflecting a need for rate estimates, some studies mostly used mean CFI as comparable to, or effectively an estimator of FR [39–40]. Others also used historical median CFI as an estimator of historical FR [41]. Another compared estimated median CFI, ITFI, and FR, found median ITFI was closest to FR, and suggested median ITFI might be used to estimate FR in low-severity fire regimes [42]. In contrast, other studies suggested fire scars provide estimates of the PMFI/FR that are generally too long: ". . . our findings clearly demonstrate that analysis of fire scars will likely underestimate past fire occurrence" ([10]:1500). However, when compositing fire-scar records over larger areas and more trees, mean CFI declines toward 1.0, a fire every year [1, 43], an estimate of PMFI/FR that is nearly always too short. Given uncertainty about estimators of low-severity fire rates, some studies suggested that summary statistics, such as mean CFI or FR, should not even be used in restoring and managing low-severity fire (e.g. [44]).

Other studies suggested that multiple descriptors of fire regimes (i.e., including mean CFI) are desirable (e.g. [1]). Studies, that favored mean CFI and ITFI as one of multiple statistics, suggested they must be interpreted correctly. For example, regarding mean CFI-all fires, one study said it was not designed to estimate area burned, and if it does not, that is not a problem in mean CFI, but an error in interpreting it [1]. Other studies also suggested it is a problem if mean CFIs are interpreted as indicating how often the entire stand burned ". . . since fires are quite variable in burn patterns" ([2]:1091). Similarly, other studies suggested managers need to recognize that fires indicated by mean CFI burned in variable spatial and temporal patterns, including unburned areas [45]. A study in California said: ". . . the composite MFIs are not equivalent to average point fire intervals, population means [sic] fire intervals or natural fire rotation. They are an estimation of average intervals between fires of any size, or of an estimated size class, occurring anywhere within a study area" ([46]:52). That mean CFI declines with increasing sampling area is also interpreted by some not as a fundamental flaw [6], but instead as an added descriptor of a fire regime [47–49]. Complex power-function patterns across spatial scales, observed as mean CFI declines toward 1.0 with more samples, are thought in this study to elucidate cross-scale spatial properties of fire regimes. Thus, ". . . measures of fire frequency are area dependent, and . . . fire return intervals cannot be described by a single number independent of spatial scale" ([48]:820). However, scale-dependent values are only known for CFI measures, not other rate measures. In summary, there is now general agreement that mean CFI and its variants (e.g., median CFI) and ITFI are not intended to estimate the PMFI/FR. Mean CFI is accepted to not indicate area burned, the pattern of the fire, or PMFI/FR.

Accurate estimators of the PMFI/FR are still needed. Fortunately, recent modern calibrations have validated new methods for estimating PMFI and FR that do not need to use mean CFI or ITFI and have promising accuracy [1, 11]. However, it may be decades before better estimates from these new methods become sufficiently common to be able to guide restoration and management of low-severity fire. In the meantime, past mean CFI and ITFI plot estimates are abundant, and required large efforts to gather and process. Moreover, plot data on fire history likely will remain a fundamental sampling component of spatial fire histories, and could provide detail about spatial variability in FR and MFI across landscapes. Mean ITFI is less studied; it remains unclear how it might perform as an estimator of PMFI/FR, but it may suffer from the unrecorded fire problem, so that mean ITFI may be too long [6]. Now that there are more spatial estimates of FR, further analysis of the relationships of CFI, ITFI, and PMFI/FR is warranted, to see whether a variant of CFI or ITFI may estimate PMFI/FR.

Materials and methods

I assembled two datasets for analyzing the relationships of CFI, ITFI, and PMFI/FR in dry forests of the western USA (Fig 1) using an analysis of bias and inaccuracy followed by regression analysis. I also recorded and analyzed fire-history sampling measures (e.g., number of samples) and their effects on these relationships.

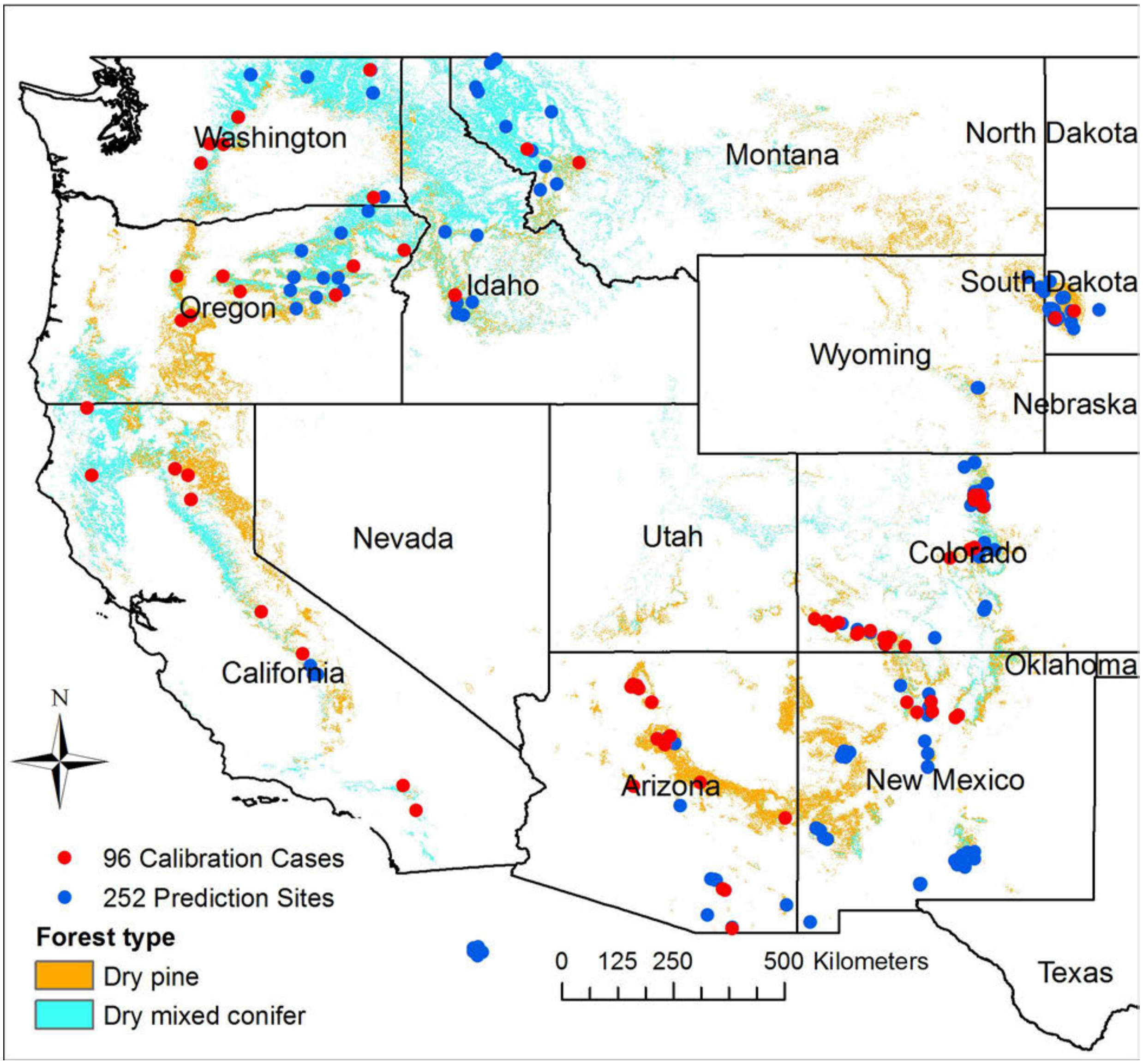

Fig 1. The 96 calibration cases and 252 prediction sites from the International Multiproxy Paleofire Database. Note that multiple plots were often done near one site, thus the number of dots is fewer.

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The 252-site fire-history dataset

To obtain a large sample of fire-history sites in dry forests to use to analyze methods and estimators in common use, I searched the International Multiproxy Paleofire Database (IMPD) (https://www.ncdc.noaa.gov/data-access/paleoclimatology-data/datasets/fire-history) for all fire-scar sites between 102˚ and 125˚ west longitude and 30˚ and 60˚ north latitude, finding 436 sites. I excluded 77 sites not clearly in dry pine or dry mixed-conifer forests. Some were also excluded because their FHX file (containing the fire-history data) in the IMPD was not usable (n = 26), the dataset was too small (n = 6) or calculations could not be completed (n = 12). I also removed 63 sites usable in a calibration dataset, described next, which left 252 sites. I left in 9 sites from Mexico and one from Canada that are nearby and relevant to the western USA.

I downloaded from the IMPD an FHX file containing fire-history records for each site and used the Fire History Analysis and Exploration System (FHAES; frames.nbii.gov) Version 2.0.2 [50] to calculate CFI and ITFI estimators. To reduce differences in the period of record, I restricted calculations for all sites to the period extending from the earliest fire to the latest fire within the A.D. 1600–1900 period. The purpose of restricting analysis to fire-to-fire periods is that scar-to-scar fire intervals are traditionally used. I did not want to introduce a possible confounding variable by using an arbitrary period. After restriction, I omitted sites with < 50 years of record, an arbitrary criterion aimed at minimizing short records.

For each case, I also recorded ancillary information, from the original publication reporting the study or from the FHX file, including the sample area, the number of sampled scarred trees, the total number of fire scars across all sampled trees, the analysis years used. and the types of targeted sampling used, including: (1) seeking the best information/longest record, (2) seeking multi-scarred trees, (3) seeking clusters of scarred trees, (4) seeking scars on dead wood, or (5) placing plots or selecting study areas in areas with many scarred trees or in old forests with long records of fire. I also recorded whether fire severity was studied, and I recorded the location of the samples found in the FHX file or publication.

The 96-case calibration and analysis dataset

To analyze the relationship of CFI and ITFI estimators and FR, I searched for and found 44 fire-history studies with 96 fire-history reconstructions and alternative calculations of fire rates in dry forests in which the study: (1) estimated CFIs and/or ITFIs and (2) in areas of at least 80 ha, also estimated FR or provided data sufficient to allow FR to be calculated from data in the paper or in an FHX file (S1 Table). The purpose of this dataset was to analyze whether CFI and ITFI estimators can predict FR. I included all sites from the IMPD, meeting the criteria defined earlier, for which sample area was given and was ≥ 80 ha, and for which there was a usable FHX file. Other sites > 80 ha were included that did not have an FHX file, but were documented in a publication. If area was reported as a range, I used the midpoint. The 80-ha minimum is an arbitrary limit to increase the area used for estimating FR. Analysis periods did not need to be pre-EuroAmerican or identical among sites, but had to have ≥ 50 years of record. If measures were not calculated in the study, I restricted analysis to scar-to-scar intervals, beginning with the first scar after ≥ 10 samples had accumulated, and ending with the last fire.

FR was calculated in the study, or by me if the study did not do this, using the previously-described area-burned estimates: (1) area burned from agency polygon fire records (n = 1) or fire-atlas records (n = 2), (2) estimates of area burned from fire-year maps reconstructed from scarred-tree or plot locations (n = 24), or (3) estimates of area burned from the ratio method and scarred-tree or plot locations (n = 63), or data in a table or graph (n = 6). For published studies, I recorded whether FR was estimated from total number of scarred trees/plots or recorders. In a few cases, this was uncertain and I recorded the most likely. A recorder is a tree scarred at least once, which increases the probability of recording fires [30]. If the study did not estimate FR, I used FHAES and Minitab to estimate FR from fire-history data in the IMPD for sites for which an FHX file was available and usable. I copied the summary table, provided in FHAES for each FHX file, into Minitab 17 [51] to do calculations. I made ratio estimates, and calculated them separately based on both total number of scarred trees and number of recorders. Sites were included more than once if different methods to calculate FR were provided in the study or could be calculated. As in the case of the 252-site dataset, I obtained and recorded ancillary information for each site.

The 342-site merged dataset

To allow calculation of histograms for particular attributes across all the sites, I merged sites in the two datasets. I removed post-EuroAmerican sites from the 96-case calibration dataset, then merged it with the 252-site prediction dataset, yielding a dataset of 342 sites (S2 Table). These include some alternative estimates from the same site or area by different studies or from using different methods, data sources, time periods or with different boundaries or other differences.

I did a rough analysis of whether sampled stands were old forests in the pre-EuroAmerican era. Old-growth dry forests are generally at least 150–200 years old, but also have attributes other than age [52], so here I call forests older than 150–200 years just "old forests." To roughly estimate the age of sampled forest stands, I used the beginning year of analysis for each stand, as defined in the study (first fire year if not). Stands with beginning years before A.D. 1700 were likely generally $\geq$ 200 years old in A.D. 1900, thus would have been old forests in the pre-EuroAmerican era. Although some could have been younger, if the oldest sample trees were not abundant, often the beginning year of analysis was defined by a minimum number of sample trees (e.g. [10]). Although imprecise, this should roughly estimate sampling in old forests. I also reviewed GLO-survey and aerial-photo reconstructions of fire severity to assess the percentage of historical landscapes with a history of predominantly low-severity fire. The GLO reconstructions use a calibrated and validated low-severity fire model [53]. The calibrated model predicts low-severity fire where historical tree density was $<$ 178 trees/ha, percentage of large trees was $>$ 29.2%, and percentage of small trees was $<$ 46.9% [53].

Can CFI and ITFI measures predict PMFI/FR?

The calibration dataset included 21 estimators of the rate of low-severity fires based on CFI, ITFI, and PMFI/FR and three sample-size variables. Sample-size variables included sample area (ha), total number of scarred trees, and scar density, expressed as total scarred trees per 100 ha (e.g. [54]). These variables are included because previous analyses found that CFI estimators were related to sample size [6]. The 21 estimators of the rate of low-severity fires included five measures of central tendency (mean, median, Weibull scale, Weibull mean, and Weibull median) for CFI-all fires, CFI-10% scarred, CFI-25% scarred, ITFI, plus the PMFI/FR based on recorders.

These 21 variables are used to individually predict PMFI/FR based on total scarred trees/plots, not based on recorders, for several reasons. Most of the best available estimates, from fire-year maps and ratio estimates using plots in a grid, are based on fires from total scarred trees in the plot. For ratio estimates from just scarred trees, recorders or all scarred trees each have strengths and limitations (S1 Text), summarized here. The use of all scarred trees is consistent with most plot-scale fire-year estimates. Recorders are two to three times less abundant than single-scarred trees, so area burned is inherently less detailed if only recorders are used, likely generally inflating area burned and shortening the estimated PMFI/FR. However, recorders do have a higher probability, than do unscarred trees, of recording a fire or of documenting it did not burn at a particular point [30]. Recorders are also multi-scarred trees, that inherently omit unscarred and single-scarred trees, that can indicate where fires did not burn, also inflating area burned and shortening PMFI/FR. PMFI/FR estimates from targeted trees (typically multi-scarred) were reduced to about 86–95% of estimates from equal-size probabilistic samples [55], supporting this expected effect. Also, about 1/3 of fires may be missed if only recorders are used [S1 Text]. More research is needed on using unscarred trees, single-scarred trees, recorders ($\geq$ 2 scars), or all scarred trees to estimate area burned, but all scarred trees likely provide the best estimates.

To understand the direction and magnitude of differences between the 21 estimators and the PMFI/FR, I calculated bias and inaccuracy for the 21 estimators relative to PMFI/FR-total scarred trees/plots for the calibration dataset. Bias is quantified by relative mean error (RME):

$$RME = \sum_{i=1}^{n} [(M_i - FR_i)/FR_i]/n \tag{3}$$

where M_i is value i of n total available estimates for CFI or ITFI estimator M of the 21 estimators and FR_i is the corresponding estimate of PMFI/FR-total scarred trees/plots [56]. RME measures relative bias as sample sizes differ. I also calculated the standard error of each mean and tested the null hypothesis that mean bias is zero using a one-sample t-test in Minitab 17 [51]. Inaccuracy or error was also calculated using a relative measure, relative mean absolute error (RMAE):

$$RMAE = \sum_{i=1}^{n} [|(M_i - FR_i)|/FR_i]/n \tag{4}$$

where symbols are as above. This quantifies the difference or error between the 21 estimators versus PMFI/FR-total scarred trees/plots as a percentage of this PMFI/FR estimate [56]. I also calculated the standard error of each mean and then tested the null hypothesis that mean inaccuracy is zero using a one-sample t-test in Minitab 17 [51].

Can bias and inaccuracy be overcome by adjusting estimators using regression models? Scatter plots showed that PMFI/FR-total scarred trees/plots versus CFI and ITFI estimators were generally linear (e.g., Fig 2A), thus I fit linear regression models, using the lm function in R version 3.2.3 [57], to predict PMFI/FR-total scarred trees/plots from each of the 21 estimators. Sample size differed among the regressions, because individual estimators were not available for all 96 cases. After initial fitting, for each measure I removed 1–2 outliers with the largest studentized residuals (i.e., > 3.0). After refitting, I examined a plot of residuals versus fitted and a normal probability plot to identify trends in residuals, which were lacking for all models.

To estimate prediction error, which is useful itself but also provides a model-selection criterion, I completed a 10-fold cross-validation using the cv.lm function in the DAAG package in R. The output is the mean square error (MSE) of predicted estimates, and its square root is the root mean square error (RMSE), a prediction analog of the standard error of the estimate in fitted regression equations. Prediction error from cross-validation is asymptotically equivalent to Akaike's information criterion (AIC), a commonly applied model-selection criterion [58], but low prediction error is most germane for this application.

Do sample-size variables improve these models? To test this, I redid the regressions with three sample-size predictors (sample area, total scarred trees, scarred trees/100 ha) in addition to each of the 21 estimators in the previous models. This time I used best-subset regression in Minitab 17 [51], and the best predictor models were chosen by the lowest Mallow's C_p statistic, where each included variable also had to be significantly ($\alpha = 0.05$) related to FR-total scarred trees/plots. I again removed 1–2 outliers based on studentized residuals and examined histograms of residuals and normal probability plots, but found no trends in residuals.

Results

Bias, inaccuracy, and regression models to estimate PMFI/FR

Bias was significantly different from 0.0 for all estimators except mean ITFI and inaccuracy was significantly different from 0.0 for all estimators (Table 1). Mean RMEs of -69% to -75% for CFI-all fires, -60% to -69% for CFI-10% scarred, and -38% to -49% for CFI-25% scarred estimators, combined with low standard errors, show that CFI measures all lead to estimates of PMFI/FR that are consistently too short (Table 1). Bias diminished from CFI-all to CFI-25%, but all estimators, except mean ITFI, were still biased. Inaccuracy for CFIs had similar

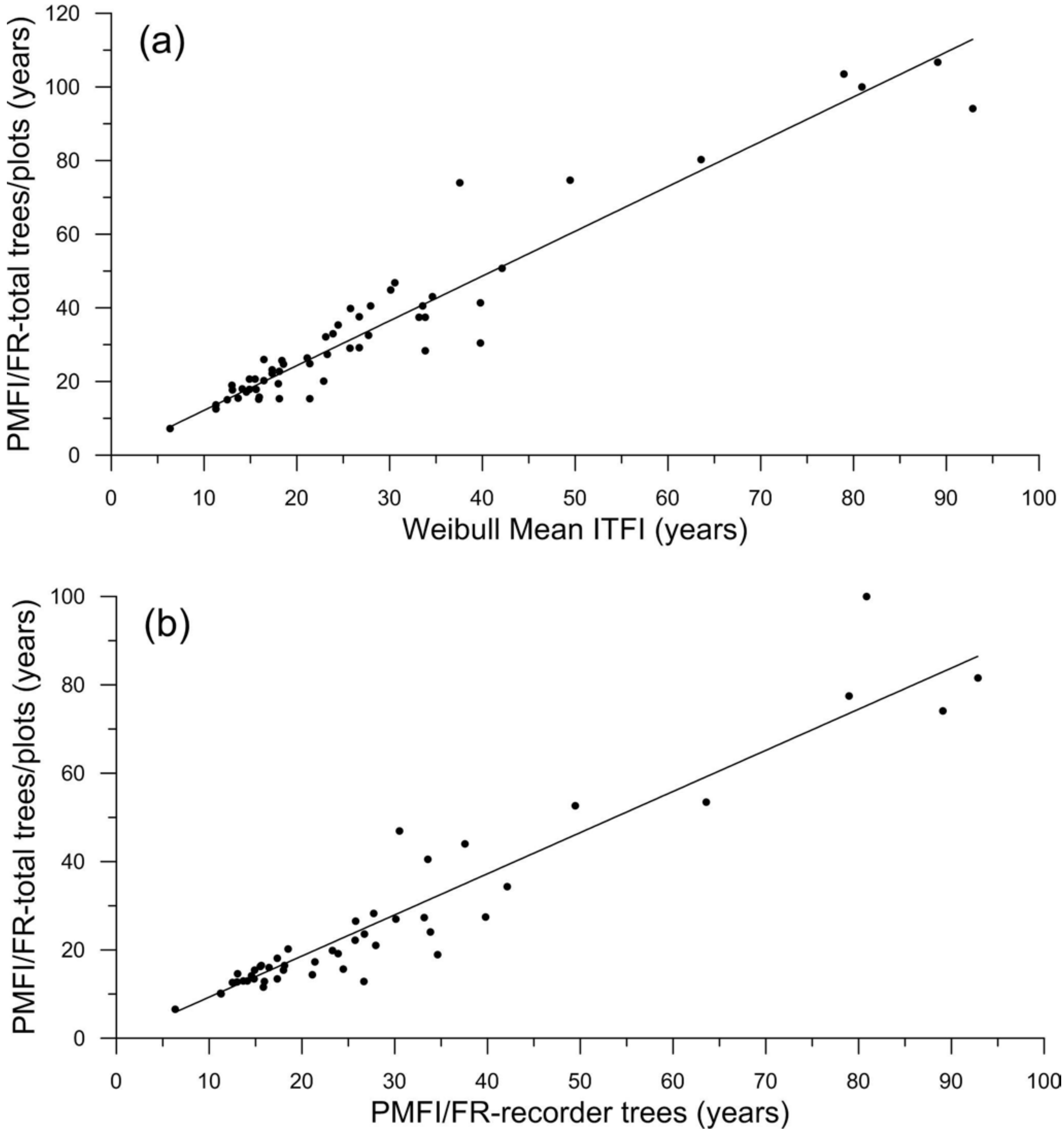

Fig 2. Scatterplots showing the linear relationships between: (a) Weibull mean ITFI and fire rotation-total trees/plots, and (b) Fire rotation-total trees/plots and fire rotation-recorder trees.

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magnitudes, patterns, and trends, with the best still having inaccuracies of 40–50%. ITFI estimators had lower bias and inaccuracy than CFI measures, with bias ranging from -3 to -30% and inaccuracy ranging from 16–33%. Only mean ITFI was unbiased, but still had 30% inaccuracy. FR-recorders also produced significantly biased and inaccurate estimates of FR, averaging 27% too low.

Table 1. Bias and inaccuracy in composite fire interval (CFI) and individual-tree fire interval (ITFI) estimates if used to estimate fire rotation-total trees/plots within the 96-case calibration dataset.

Measure	Test of bias					Test of inaccuracy			
	n	Mean RME (%)	s.e. of mean (%)	t	p	Mean RMAE (%)	s.e. of mean (%)	t	p
Mean CFI—all fires	84	-69.35	2.11	-32.79	<0.001	69.42	2.09	33.27	<0.001
Median CFI—all	76	-70.10	2.62	-26.77	<0.001	70.22	2.57	27.27	<0.001
Weibull Scale CFI—all	58	-69.25	2.28	-30.41	<0.001	69.25	2.28	30.41	<0.001
Weibull Mean CFI—all	58	-72.10	2.03	-35.47	<0.001	72.10	2.03	35.47	<0.001
Weibull Median CFI—all	58	-75.46	1.92	-39.33	<0.001	75.46	1.92	39.33	<0.001
Mean CFI—10% scarred	61	-63.29	2.08	-30.46	<0.001	63.39	2.03	31.27	<0.001
Median CFI—10%	62	-68.69	2.14	-32.11	<0.001	68.79	2.09	32.95	<0.001
Weibull Scale CFI—10%	57	-60.09	2.11	-28.48	<0.001	60.09	2.11	28.48	<0.001
Weibull Mean CFI—10%	57	-63.84	1.86	-34.29	<0.001	63.84	1.86	34.29	<0.001
Weibull Median CFI—10%	57	-68.03	1.85	-36.70	<0.001	68.03	1.85	36.70	<0.001
Mean CFI—25% scarred	71	-42.12	2.28	-18.43	<0.001	43.19	1.98	21.82	<0.001
Median CFI—25%	65	-48.88	2.66	-18.35	<0.001	50.77	2.04	24.91	<0.001
Weibull Scale CFI—25%	56	-38.41	2.64	-14.54	<0.001	40.24	2.09	19.30	<0.001
Weibull Mean CFI—25%	56	-44.66	2.35	-19.00	<0.001	45.92	1.86	24.74	<0.001
Weibull Median CFI—25%	56	-49.11	2.24	-21.90	<0.001	49.88	1.91	26.14	<0.001
Mean ITFI	67	-2.71	6.76	- 0.40	0.689	30.10	5.66	5.32	<0.001
Median ITFI	66	-29.71	2.75	-10.79	<0.001	33.43	1.99	16.78	<0.001
Weibull Scale ITFI	56	-8.50	2.53	-3.35	0.001	16.34	1.69	9.65	<0.001
Weibull Mean ITFI	56	-16.64	2.31	-7.22	<0.001	21.19	1.48	14.33	<0.001
Weibull Median ITFI	56	-28.25	2.10	13.43	<0.001	29.68	1.71	17.37	<0.001
FR–recorders	52	-26.79	2.01	-13.31	<0.001	26.79	2.01	13.33	<0.001

doi:10.1371/journal.pone.0172288.t001

Prediction error and fit show that the best regression models to predict PMFI/FR-total scarred trees/plots (Table 2) were from ITFI estimators, particularly Weibull mean ITFI (RMSE = 7.52, R^2_{adj} = 0.972), Weibull scale ITFI (RMSE = 8.04, R^2_{adj} = 0.970), and Weibull median ITFI (RMSE = 9.46, R^2_{adj} = 0.958), although the mean ITFI model was also good (RMSE = 10.30, R^2_{adj} = 0.944). Models based on CFI-25% scarred measures had moderately low prediction errors (RMSE from 11.0–13.7) and high R^2_{adj} values of 0.870–0.929. Models using CFI-10% had higher prediction errors and somewhat lower fit (Table 2). The poorest models were from CFI-all measures (Table 2). Weibull mean models consistently had lowest prediction errors and highest R^2_{adj} compared to models based on mean, median, Weibull scale, or Weibull median (Table 2).

Sample-size variables were not significant in most models (Table 3). The few models with significant sample-size variables had R^2_{adj} values generally improved only slightly, averaging higher by only 0.006–0.010 except for the model for mean CFI-all fires, which was 0.086 higher (Tables 2 and 3). Thus, simpler models in Table 2 should suffice for estimating PMFI and FR, except that the sample-size model may be worth using in the case of mean CFI-all fires (Table 3).

Using prediction error (RMSE) as the criterion, supplemented by fit (R^2_{adj}), the best model (Table 2) is based on Weibull mean ITFI, which had the lowest RMSE of 7.52 years and the highest R^2_{adj} of 0.972 (Table 2). The Weibull mean ITFI model was thus used for all PMFI/FR estimation for the 252-site dataset. Given its 7.52 year RMSE, 15–20 year bins are appropriate for reporting estimates, as about 68% of predictions are expected to be within the ± 1 RMSE of 7.52 years. Models other than the Weibull mean ITFI model (Tables 2 and 3) can also be used for deriving estimates from CFI and ITFI estimates, assuming prediction error and fit are acceptable.

Table 2. Linear regression models for estimating PMFI/FR-total scarred trees/plots, based on the 96-case calibration dataset. All slopes (β) were significant ($p < 0.001$) at $\alpha = 0.05$.

Estimator	β †	Outliers‡	*n*	R^2_{adj}	$RMSE_{§}$
Mean CFI—all fires	2.440	25, 89	82	0.721	18.14
Median CFI—all fires	2.450	25, 89	74	0.675	18.52
Weibull Scale CFI—all fires	2.655	25, 93	56	0.755	19.05
Weibull Mean CFI—all fires	2.915	25, 93	56	0.762	18.63
Weibull Median CFI—all fires	3.294	25, 93	56	0.730	20.12
Mean CFI—10% scarred	2.467	25, 89	59	0.837	15.65
Median CFI—10% scarred	2.783	25, 89	60	0.812	16.34
Weibull Scale CFI—10% scarred	2.423	25, 93	55	0.856	16.09
Weibull Mean CFI—10% scarred	2.666	25, 93	55	0.865	15.39
Weibull Median CFI—10% scarred	2.992	25, 93	55	0.826	17.66
Mean CFI—25% scarred	1.715	2, 89	69	0.923	11.00
Median CFI—25% scarred	1.834	26, 89	63	0.870	13.67
Weibull Scale CFI—25% scarred	1.597	2	55	0.925	11.96
Weibull Mean CFI—25% scarred	1.749	2	55	0.929	11.36
Weibull Median CFI—25% scarred	1.867	2	55	0.906	13.00
Mean ITFI	1.121	2, 70	65	0.944	10.30
Median ITFI	1.366	24, 26	64	0.896	12.57
Weibull Scale ITFI	1.108	2	55	0.970	8.04
Weibull Mean ITFI	1.216	2	55	0.972	7.52
Weibull Median ITFI	1.361	2	55	0.958	9.46
PMFI/FR-recorders	1.337	None	52	0.961	10.39

† All models have the form: PMFI/FR-total scarred trees/plots = β * predictor

‡ Numbers represent row numbers in the 96-case calibration dataset (S1 Table)

§ RMSE = root mean square error, the prediction error, in years, from the 10-fold cross validation

doi:10.1371/journal.pone.0172288.t002

Estimated historical PMFI/FRs across the 342-site dataset

Overall, estimated historical PMFI/FR across the 342 sites had a mean of about 39 years and a median of about 30 years (Table 4). Mean PMFI/FR did not differ significantly between dry pine forests and dry mixed-conifer forests (Table 4; t (181) = -0.34, p = 0.731). Maps and histograms show that shorter historical PMFI/FRs (< 25 years) were concentrated in Arizona and New Mexico, but also were scattered across parts of all other states, except for few in South Dakota, Wyoming, Colorado, and Mexico (Figs 3 and 4). Historically long PMFI/FR (> 55 years), in contrast, were common only in a band from northern New Mexico to western South

Table 3. Best linear regression models for estimating PMFI/FR-total scarred trees/plots, including estimators in Table 2 plus measures of sample size, based on the 96-case calibration dataset. Only cases where sample-size variables were significant are shown here, otherwise the best models are in Table 2.

Estimator	Best model	*n*	R^2_{adj}
Mean CFI—all fires	1.817 Mean CFI-all + 0.000896 Sample area (ha) + 0.927 Scarred Trees/100 ha	82	0.807
Mean CFI—10% scarred	2.347 Mean CFI-10% scarred + 0.0447 Scarred Trees	59	0.847
Mean ITFI	1.178 Mean ITFI—0.037 Scarred Trees	65	0.951
Median ITFI	1.260 Median ITFI + 0.360 Scarred Trees/100 ha	64	0.902
PMFI/FR-recorders	1.281 FR from recorders + 0.0702 Scarred Trees	52	0.966

doi:10.1371/journal.pone.0172288.t003

Table 4. Overall statistics for historical low-severity PMFI/FR in dry forests and by forest type, based on the merged 342-site dataset. Sample size was 342 overall, 223 in dry pine, 119 in dry mixed conifer.

Statistic	Overall (years)	Dry Pine (years)	Dry Mixed Conifer (years)
Mean	38.62	39.11	37.69
95% confidence interval for mean	35.13–42.10	35.40–42.83	30.42–44.97
Standard deviation	32.75	28.17	40.08
Minimum	7.20	7.20	10.21
1st quartile	19.55	18.80	21.24
Median	29.68	29.95	29.20
95% confidence interval for median	27.01–31.70	26.40–34.63	25.07–31.58
3rd quartile	46.11	50.49	37.62
Maximum	327.16	175.09	327.16

doi:10.1371/journal.pone.0172288.t004

Dakota, and were otherwise only scattered in a few locations in California, Oregon and Washington, with no occurrences in Idaho and Montana (Figs 3 and 4). Variability in historical PMFI/FRs was substantial but generally modest within a state, with coefficients of variation (CV) typically between about 30–60%, although California had a high CV and Arizona had a low CV (Table 5). Minima were typically 7–15 years except 20–30 years in South Dakota, Wyoming, and Mexico. Maxima were not very indicative, as a few long PMFI/FR were not uncommon (Fig 4). However, the 3rd quartile of about 93 years in Colorado, 56 years in Wyoming, and 50 years in South Dakota suggests that long historical PMFI/FRs were common in the southern Rocky Mountains and Black Hills (Table 5, Fig 3). At the state level, Colorado stands out in having the greatest variability and total range in historical PMFI/FRs (Fig 4I), and Arizona stands out as having the lowest variability and total range (Fig 4A).

Another pattern is that in the most mountainous areas with the steepest environmental gradients and topographic diversity, the full range (all four classes) in historical PMFI/FRs often was found in a small area (Fig 3). This high diversity occurred in northeastern Washington, the central Sierra Nevada, northern New Mexico, southwestern Colorado, north-central Colorado, and in western South Dakota, but not in Arizona, Idaho, Montana, or Oregon (Fig 3). However, even in these areas, with the exception of Arizona, some diversity in historical PMFI/FR was found over relatively short distances (Fig 3), suggesting the importance of local factors in addition to the large trends evident across the western USA.

Most studies of low-severity fire in dry western forests were conducted in forest stands that were mostly old forests in the pre-EuroAmerican era (Fig 5). Stands with beginning analysis years before A.D. 1750 were likely generally $\geq$ 150 years old, and stands with beginning analysis years before A.D. 1700 were likely generally $\geq$ 200 years old, in A.D. 1900, thus meeting the age criterion for old-growth forests (Fig 5). A history of predominantly low-severity fire in the century before the late-1800s was found across about 34%, on average (ranging from 2.5–62.4%), of eleven dry-forest landscapes across the western USA (Table 6). Thus, estimated historical PMFI/FRs apply primarily to old forests, which were likely concentrated historically in the 34% of overall dry-forest landscapes with a history of predominantly low-severity fire.

Discussion

Limitations of CFIs and ITFIs if used to estimate PMFI/FR

Researchers in the past commonly sampled fire scars and trees to generally increase the length of the fire-history record, minimize physical damage to trees, and maximize efficiency [55]. Unfortunately, these methods also produced CFI and ITFI estimates that are biased and

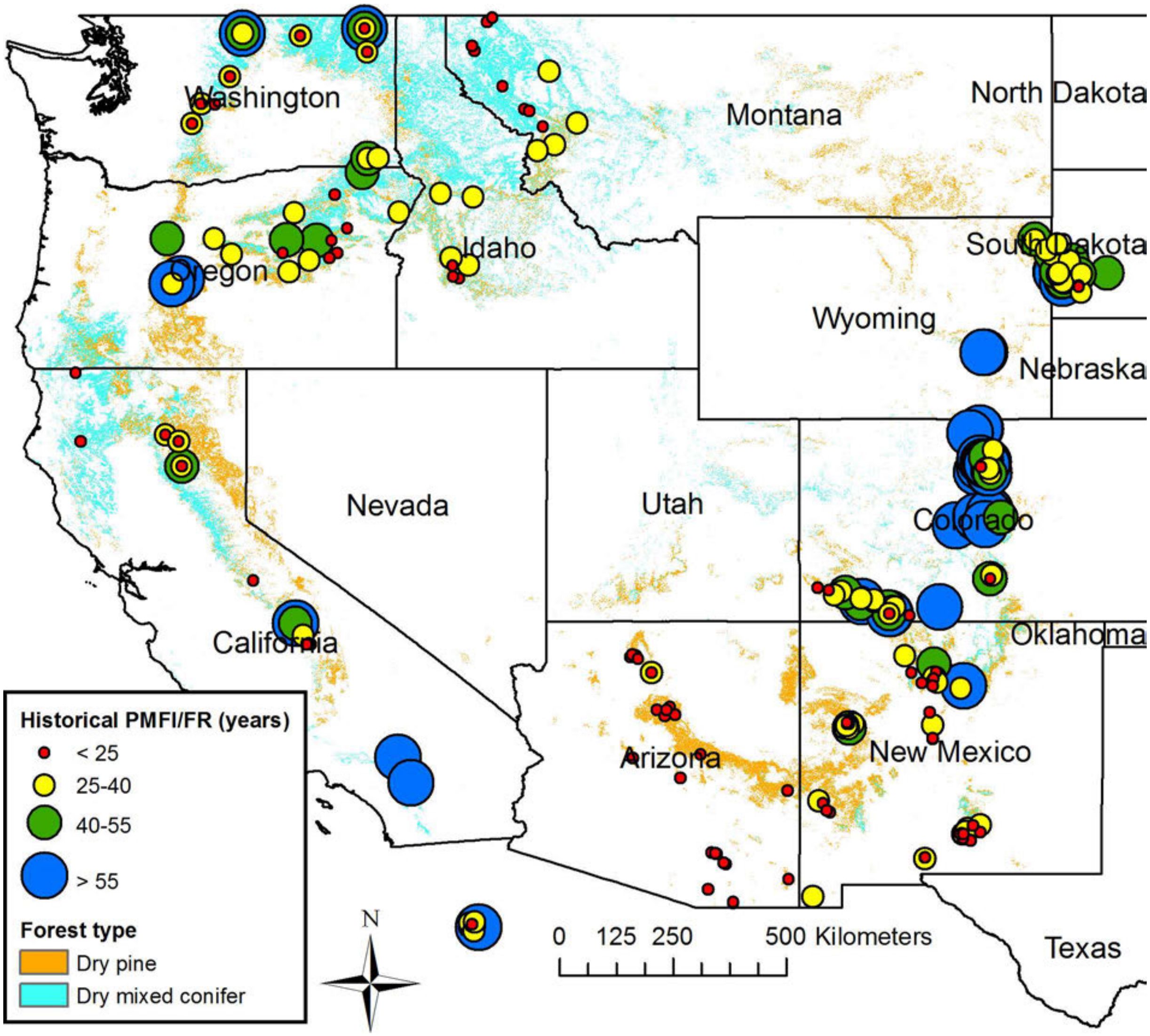

Fig 3. Estimated historical low-severity population mean fire interval/fire rotation (PMFI/FR) for the combined set (n = 342) of calibration cases and prediction sites in dry forests of the western USA.

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inaccurate if used to estimate the PMFI/FR (Table 1), as also found from modern calibration [11], and this is now accepted to be an inappropriate use. However, it is possible to estimate PMFI/FR accurately from past CFI and ITFI estimates using linear regression (Table 2).

Further discussion of the limitations of past measures as estimators of the PMFI/FR is thus generally moot, but for those interested, I include further analysis in S1 Text and a summary here. The main factors unique to underestimation of PMFI/FR by CFI measures likely include: (1) overcompensation—sampling and compositing across too large an area, (2) loss of long real fire intervals to the compositing process, and (3) restriction rules that do not omit enough small fires. ITFI measures do not use compositing and have lower bias and inaccuracy, but still are biased and inaccurate (Table 1). Both CFI and ITFI measures must be missing longer intervals from a sampling bias, because their estimates are low relative to PMFI/FR. Major factors likely are targeting trees and sampling areas with the most scars, excluding trees with no or one scar, and censoring intervals at the beginning and end of a tree's record (S1 Text).

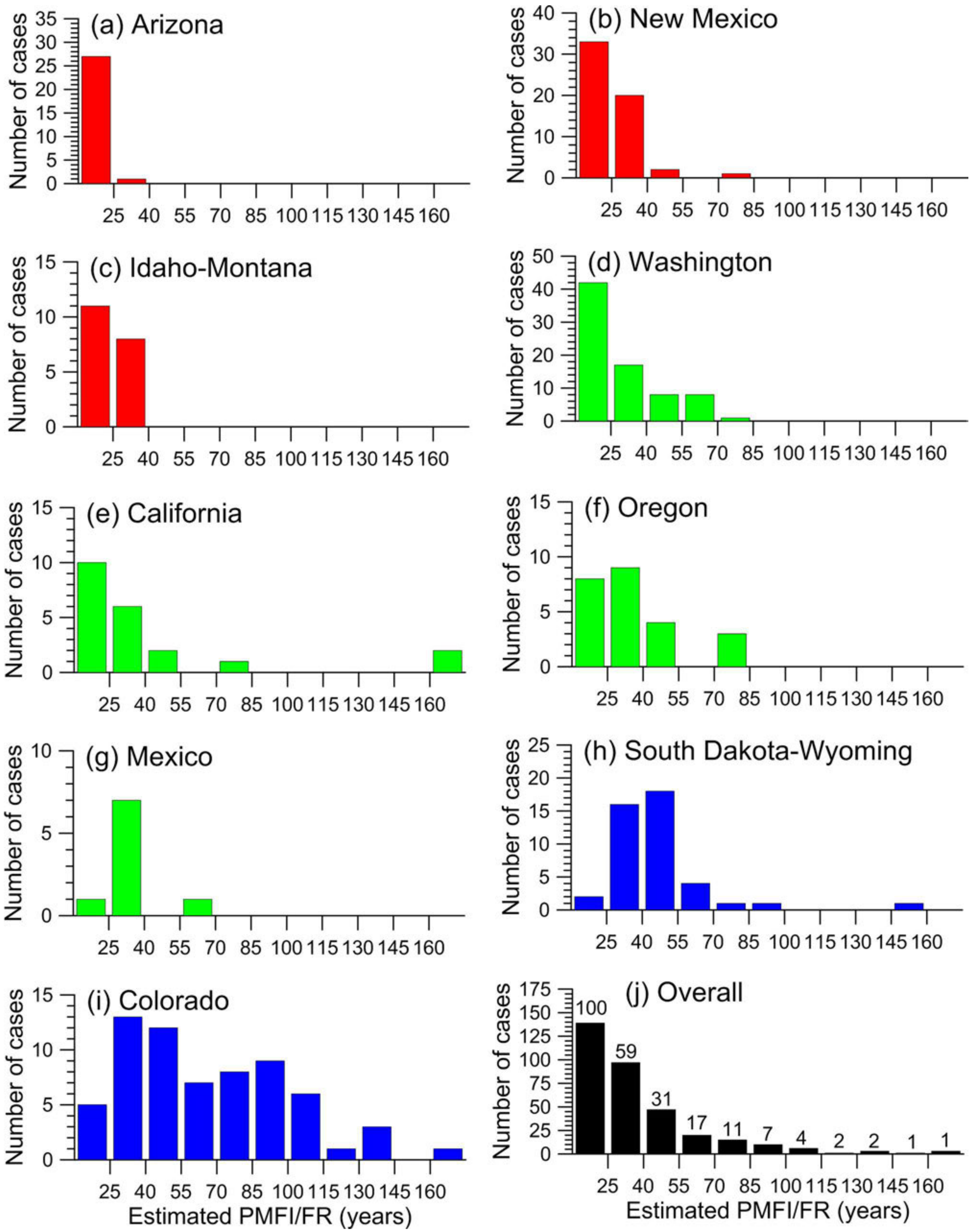
(a) Arizona
(b) New Mexico
(c) Idaho-Montana
(d) Washington
(e) California
(f) Oregon
(g) Mexico
(h) South Dakota-Wyoming
(i) Colorado
(j) Overall
Number of cases
25 40 55 70 85 100 115 130 145 160
Estimated PMFI/FR (years)
100
59
31
17
11
7
4
2
2
1
1

Fig 4. Histograms showing the variability in historical PMFI/FRs (342 sites). These are shown among: (a-i) the eleven western states and (j) overall. In (j) the numbers above the bars indicate the percentage of the distribution that exceeds the lower limit of each bin. For example, 59% of the distribution had historical PMFI/FR > 25 years. Idaho and Montana were combined, as were South Dakota and Wyoming, because of insufficient samples and similarity of histograms within these pairs of adjoining states. Colors indicate similar histograms, with the shortest historical PMFI/FRs predominating in Arizona, New Mexico, and Idaho-Montana, intermediate in Washington, California, Oregon, and Mexico, and the longest in South Dakota-Wyoming and Colorado.

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Targeting can also reduce estimated PMFI/FR itself (S1 Text), and needs to be avoided in new landscape methods.

Inference space for the PMFI/FR estimates

Studies of new probabilistic landscape methods for reconstructing PMFI/FR encourage ". . . clearly defining the inference space, not extrapolating to unrepresentative areas . . ." ([55]:1030), and this is also important for estimates of PMFI/FR from regression. The dataset of 342 sites spans dry forests in the western USA (Fig 1). The set of published studies corresponding to this dataset (S2 Table) includes many of the studies of low-severity fire in dry forests in the western USA, but other studies exist. This dataset and these other studies likely are not a probabilistic sample of historical dry forests, however, as many studies targeted old forests or forests with concentrations of fire scars (S1 Text) and occurred in forests that likely were old in the pre-EuroAmerican era (Fig 5). Old trees were historically dominant in some dry-forest landscapes and old trees were not uncommon in many forests, but young to middle-aged forests historically dominated most dry-forest landscapes [24, 62, 63]. Based on the GLO reconstructions and early aerial photographs (Table 6), the PMFI/FR estimates here apply most clearly to no more than about 34% of dry-forest landscapes, particularly in old forests. That leaves about 66% of dry-forest landscapes without PMFI/FR estimates. It is possible that some estimates do apply to parts of these other forests, possibly representing the low-severity parts of mixed-severity fire regimes on sites that had not recently burned at high severity. However, it is impossible to determine this from data in FHX files or for the 74% of studies that did not reconstruct fire severity (S1 Text).

Several studies, that targeted old forests to obtain long fire records, indicated that younger forests had few fire scars and, because these studies were focused on long and complete records of fire years, they avoided sampling younger stands. In El Malpais, New Mexico: "The most abundant, best preserved fire-scarred samples were found at sites on the northwestern and western peripheries of the malpais . . . We found no fire-scarred samples on the kipukas in the northern and eastern portions of the malpais, and found few samples in the southern portions. These areas contained ponderosa forests that appeared younger than elsewhere, perhaps due to more recent, intense stand-replacing fires . . ." ([64]:136). Sampling was concentrated in

Table 5. Statistics for historical low-severity PMFI/FR in dry forests by state, based on the merged 342-site dataset. Sample sizes were 28 in AZ, 21 in CA, 65 in CO, 7 in ID, 12 in MT, 56 in NM, 24 in OR, 40 in SD, 76 in WA, 3 in WY, 9 in MX and 1 in BC.

Statistic	AZ (yrs)	CA (yrs)	CO (yrs)	ID (yrs)	MT (yrs)	NM (yrs)	OR (yrs)	SD (yrs)	WA (yrs)	WY (yrs)	MX (yrs)	BC (yrs)
Mean	15.48	54.21	65.70	25.96	21.81	24.59	36.41	46.19	30.60	47.18	35.04	40.49
s.d.	4.26	83.01	35.32	8.28	6.77	11.24	19.09	22.23	16.09	14.92	13.06	-
CV	27.52	153.13	53.76	31.90	31.04	45.71	52.43	48.13	52.58	31.62	37.27	-
Minimum	7.20	8.56	15.20	16.88	13.25	10.21	15.30	21.18	11.00	29.95	23.15	40.49
1st quartile	12.51	18.68	35.05	17.00	14.88	16.25	24.12	35.20	19.73	29.95	28.62	-
Median	15.22	27.20	60.45	27.07	22.47	22.28	29.66	41.84	23.89	55.79	32.28	40.49
3rd quartile	17.98	40.77	92.67	32.95	26.95	30.62	42.33	49.97	38.30	55.79	35.88	-
Maximum	25.70	327.16	175.09	37.37	32.83	74.70	83.25	158.70	81.93	55.79	68.08	40.49

doi:10.1371/journal.pone.0172288.t005

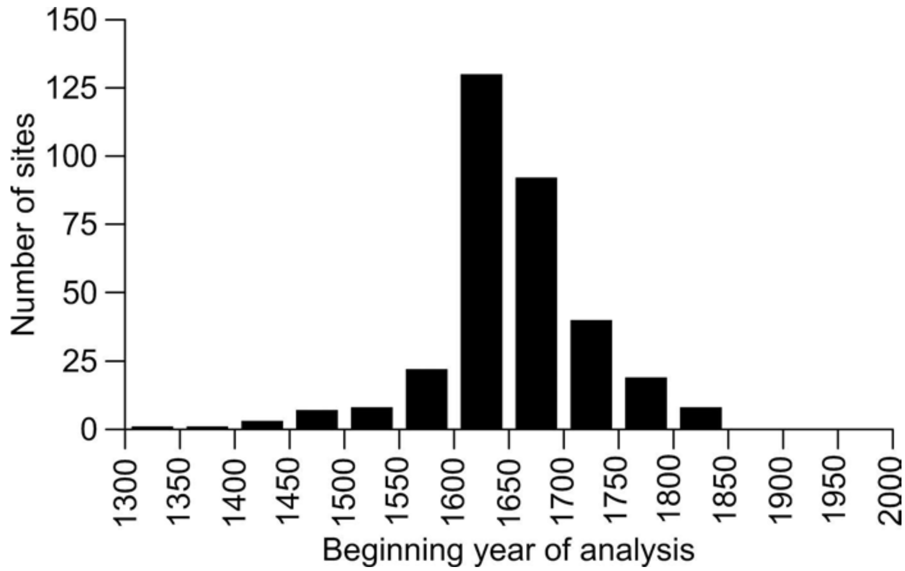

Fig 5. Beginning year of analysis for the 331 sites with available data in the 342-site merged dataset.

doi:10.1371/journal.pone.0172288.g005

areas with abundant fire-scars, but later this targeting was forgotten, and these areas were portrayed as representing the whole El Malpais landscape: "These increased fuel loadings in malpais forests have essentially changed the trajectory of fire behavior to one that now favors the occurrence of high-intensity, stand-replacing fires in contrast to the low-intensity, stand-maintenance fires that occurred prior to Euro-American settlement . . ." ([64]:234). Similarly, no fire scars were found in 5 of 12 transect locations in mixed-conifer forests in northern New Mexico [65]. The study sampled scars on relatively flat ridges nearby, where scars were abundant, and composite fire intervals from these sites were assumed to apply to the whole mixed-

Table 6. Area and percentage of 11 dry-forest landscapes in the western USA that meet the low-severity model, based on GLO surveys and early aerial photographs.

Source/Author(s)	Study area		Fits low-severity model	
	Location	**Area (ha)**	**%**	**Area (ha)**
General Land Office Surveys				
Williams and Baker [53]	Mogollon Plateau, AZ	405,214	62.4	252,854
	Black Mesa, AZ	151,080	12.0	18,130
	Front Range, CO	65,525	2.5	1,638
	Blue Mountains, OR	304,709	40.3	122,798
Baker [59]	North-E Cascades, OR	146,555	32.5	47,630
	Central-E Cascades, OR	147,502	10.4	15,340
	South-E Cascades, OR	104,160	29.4	30,623
Williams and Baker [60]	Coconino Plateau, AZ	41,214	58.8	24,234
Baker [61]	N. Sierra, CA	115,766	12.6	14,587
	S. Sierra, CA	187,085	26.4	49,390
Early Aerial Photographs				
Hessburg et al. 2007	E. WA & E. OR	112,115	21.6	24,200
Symopsis				
Total		1,780,925		601,424
Mean percentage			33.8	

doi:10.1371/journal.pone.0172288.t006

conifer forest [65]. This was also the pattern in northern Colorado: "Most of the 67 fire-scarred trees that were sampled were found on ridges or in open areas (Fig 1). It was uncommon to find scarred trees in dense stands" ([66]:138).

These observations suggest low-severity fire was likely less frequent or even rare in younger and denser historical dry forests, that likely were common in the 66% of dry forests lacking a history of exclusive low-severity fire (Table 6). However, specific studies of rates of low-severity fire are lacking for stands $\leq$ 150–200 years old in the pre-EuroAmerican era (Fig 5), that are the predominant forests today. Because they are not in the inference space for past fire-history studies in dry forests, it is not valid to infer that today's young to middle-aged forests would have been subject to low-severity fires at the historical mean rates in the 342-site dataset or in other comparable published fire-histories for dry forests.

Historical dry forests not predominantly frequent-fire forests

Dry pine and dry mixed-conifer forests have been described as frequent-fire forests, an attribute still supported for only about 14% of overall dry-forest area, with multidecadal low-severity fire likely historically over about 86% of overall dry-forest area in the western USA. Only about 41% of the old, dry forests, which were likely concentrated in about 34% of western USA dry forests (41% of 34% = 14% of overall dry forest), had frequent fire, with a historical PMFI/FR < 25 years (Fig 4J). Old forests with frequent fire were historically concentrated in Arizona and found at scattered sites across the West (Fig 3), particularly in New Mexico, Washington, Idaho, Montana, and California (Fig 4A–4E). In contrast, about 59% of cases in old forests and thus about 20% of dry forests in the western USA (59% of 34% = 20% of overall dry forest) had a historical mean PMFI/FR $\geq$ 25 years (Fig 4J). Low-severity fire was likely even less frequent in the remaining overall 66% of dry-forest landscapes lacking a history of exclusive low-severity fire (Table 6). Altogether roughly 14% of dry forests in the western USA historically had frequent (PMFI/FR < 25 years) low-severity fire and 86% of dry forests in the western USA historically instead had multidecadal low-severity fire.

Even in the 34% of dry-forest landscapes with an exclusive history of low-severity fire, the overall mean PMFI/FR was 39 years, half the cases had PMFI/FR > 30 years, and a quarter of cases had PMFI/FR > 46 years (Table 4). These old forests are better described overall as having diverse rates of low-severity fire, spanning the range from frequent to multidecadal. This diversity in rates varied on two scales, first across large regions from predominantly multidecadal (median > 40 years), in Colorado, South Dakota, and Wyoming, to predominantly frequent in Arizona, New Mexico, and Idaho-Montana, with other states having broader mixtures, ranging from frequent to multidecadal (Figs 3 and 4). Second, individual smaller areas often contained a diversity of rates over short distances, particularly in mountain ranges, often spanning or nearly spanning a broad range from frequent to multidecadal (Fig 3).

Estimated historical PMFI/FR mean rates are relevant, because many ecological processes and structures change across a narrow range in rates. In the roughly 86% forests with PMFI/FR $\geq$ 25 years, fuels that required about 7–25 years to build back up after a low-severity fire [12–14]. would, on average, have been fully recovered for an extended period before the next fire. Shrubs would likely have been able to fully recover and dominate for substantial periods. Small trees that rely on seed (e.g., ponderosa pine) would also have been able to regenerate and become common in forest understories, as documented in several historical dry forests [24]. The role of the forest floor in replenishing soil nutrients and organic matter, enhancing absorption of water, and fostering microbial communities [15] would not have been limited by too-frequent fires. Greater opportunities for trees to regenerate and less mortality from low-severity fire also help to explain dense areas of dry forests that occurred historically across substantial parts of many dry-forest landscapes (e.g. [53, 59]). Natural fuels, less limited by low-severity fire, would have favored higher-severity fires via ladder fuels. Adverse effects on habitat for wildlife that use snags or coarse down wood [15] would be less because of less low-severity fire, and fires of higher intensity would likely increase snags and coarse dead wood.

In contrast, in the roughly 14% of historical dry forests with historical PMFI/FR $<$ 25 years, levels of fuels, including shrubs and small trees, would have been more consistently kept low (Fig 3). Frequent low-severity fires would likely have fostered a diversity of grasses and forbs, but would have limited shrubs and small trees. In these settings, lower-density forests would have been favored and higher-severity fires would have been discouraged, at least by fuel conditions [19, 53]. Potential adverse ecosystem and wildlife effects of frequent low-severity fire [15] would remain a natural historical characteristic of these primarily southwestern frequent-fire forests (Fig 3). However, high local and regional diversity in rates (Fig 3) meant that a diversity of processes, rates, and structures occurred across even the old-forest part of many dry-forest landscapes, within both small areas and across the western USA.

Limitations and error in calibration and prediction PMFI/FRs

The calibration cases (S1 Table) are from larger land areas and include estimates of PMFI/FR that are directly usable as a guide for restoration and management in old, dry forests. The appropriate estimate in S1 Table is FR-YrsTot, which was directly estimated in the study in many cases. Where a direct estimate was not made, I estimated PMFI/FR-YrsTot from PMFI/FR-YrsRec using the equation in Table 2 and Fig 2B.

The 252 prediction cases (part of S2 Table) are from single-plot samples in smaller plot areas, and likely have more error. The estimated prediction error for PMFI/FR in a small plot was a 7.52 year RMSE, which suggests bins about 15-years wide, as in Fig 3, would likely contain about 68% of observations. Bins about 30-years wide would contain about 95% of observations. Smaller plots used at the 252 sites also may not individually provide an adequate sample of a forest area. In an accuracy study, estimates from small plots required averaging across 5–6 plots representing 600–1000 ha to achieve mean relative errors $<$ 30% in estimating PMFI/FR [11]. The estimated PMFI/FRs from the available set of small plots cannot be pooled to decrease this error, as they are not necessarily samples from one population. The problem for small plots is inherent stochastic variability in realized fire intervals, even from a fixed fire regime in a particular land area [67], and errors in the sample and estimators. Thus, the PMFI/FR estimates are a significant improvement over using CFI and ITFI, but greater accuracy can be expected from larger studies in the calibration dataset and also from future landscape-scale reconstructions.

Most of the 342 estimates are likely low estimates for two reasons. Targeting multi-scarred trees reduces CFI and ITFI estimates, but also reduces estimated PMFI/FR by not sampling trees with one scar or no scar that can indicate areas that did not burn in a particular fire (S1 Text). Thus, the area burned by each fire may be inflated and the PMFI/FR too short. Because 94% of 250 cases with evidence did target multi-scarred trees (Table A in S1 Text), this affects almost all estimates of PMFI/FR. Targeted sampling of individual trees led to PMFI/FR estimates reduced to about 86–95% of estimates from equal-sized probabilistic samples [55]. This would mean that PMFI/FR estimates here need to be multiplied by 1.05–1.18. Also, both calibration and prediction PMFI/FR estimates are low estimates in many cases because PMFI/FR could not be estimated separately for low-severity fires in the 74% of cases where fire-severity was not studied. Even where fire severity was studied, the study did not report separate rates, instead only rates for fire severities combined (S1 Text). Because estimates are for old forests with a history of low-severity fire, the higher-severity component was likely not large, but could affect longer estimates (Table B in S1 Text). Combining these two factors likely would increase estimated PMFI/FR, but more research is needed to narrow and validate the needed corrections before they are applied. In contrast, FR estimates in the calibration are from all trees, not recorders, and regression equations applied to the predicted dataset are from all trees. Estimates from recorders would be lower, but I explained earlier why the truth is likely closer to FR-all trees. Further research is warranted, and could possibly resolve all remaining uncertainties, leading to improved equations and estimates.

PMFI/FRs as a guide to restore and manage low-severity fires

In spite of these limitations, these new PMFI/FR estimates are the best available and usable estimates of historical mean rates of low-severity fire to use as a guide in restoring and managing low-severity fire in dry forests of the western USA. Past CFI and ITFI estimates were not intended to estimate PMFI/FR and would be misapplied, with adverse impacts on biological diversity and ecosystem functioning, if used directly for this purpose, as is shown by their biases, inaccuracies, and needed adjustments using regressions (Tables 1, 2).

Estimated historical PMFI/FRs specify how long, on average, it took to burn across a land area (the FR), and how long the intervals were, on average, between fires at points in the land area (the PMFI). They can be estimated at multiple scales, from small plots to large land areas, although with greater accuracy over larger land areas. Congruent estimates of modern and historical low-severity PMFI/FR can be made, and directly compared. Modern estimates can be made using digital fire maps (e.g., Monitoring Trends in Burn Severity at: http://www.mtbs.gov) or other sources. All that is needed is to add up the areas of fires that burned in a particular landscape of interest at low severity over a particular period, calculate the area of the landscape, and use Eq 2. Temporal and spatial variability in PMFI/FR can be estimated as well, using subareas or sub-periods (e.g. [23]). Comparison of modern and historical rates of low-severity fire facilitates monitoring the effectiveness of restoration and management programs, and analysis of trends in rates of modern relative to historical fires [3].

Fire-size distributions are also important, but those from small plots have inherently limited value. At this point, distributions of annual area burned, which approximate fire sizes, can be shown for some larger study areas in dry forests (Fig 6). I compiled data for these histograms from graphs or tables in the sources. Note that this is area burned at all severities, not just low severity, and is not restricted to old-forest parts of landscapes. Several graphs show that the most fire years were in the smallest size class, with decreasing abundance in larger size classes. Historical fire sizes could reach at times into the 5,000–11,000 ha size classes, at least in three study areas (Fig 6C, 6F and 6G). In many study areas, the maximum area burned reached the size of the study area (Fig 6B, 6C, 6D, 6E and 6F), suggesting fires could have been larger.

In an Arizona dry-forest landscape, 5.1% of total fires, that were the largest fires, contributed 97% of total burned area [1]. This pattern, common in forests [17], also suggests that in dry forests most of the burned area is from infrequent large fires, with frequent small fires not adding much to total burned area. This pattern of variable fire sizes and infrequent large fires is important to mimic, as it fosters diverse times since fire, at any instant, across a landscape, which allows species with different responses to fire to all remain viable across landscapes [16].

Low-severity fires can kill up to about 20% of basal area [27–28], and it is usually expected that this mortality is from torching or passive crown fires that kill individual trees or small groups of trees. However, little is known about the size and distribution of patches of mortality in low-severity fire regimes. Only about 23% of reconstructions of low-severity fires analyzed fire severity and even these provided little information about this topic (S1 Text), as it is difficult to reconstruct the size of mortality patches. Early historical observations provide some

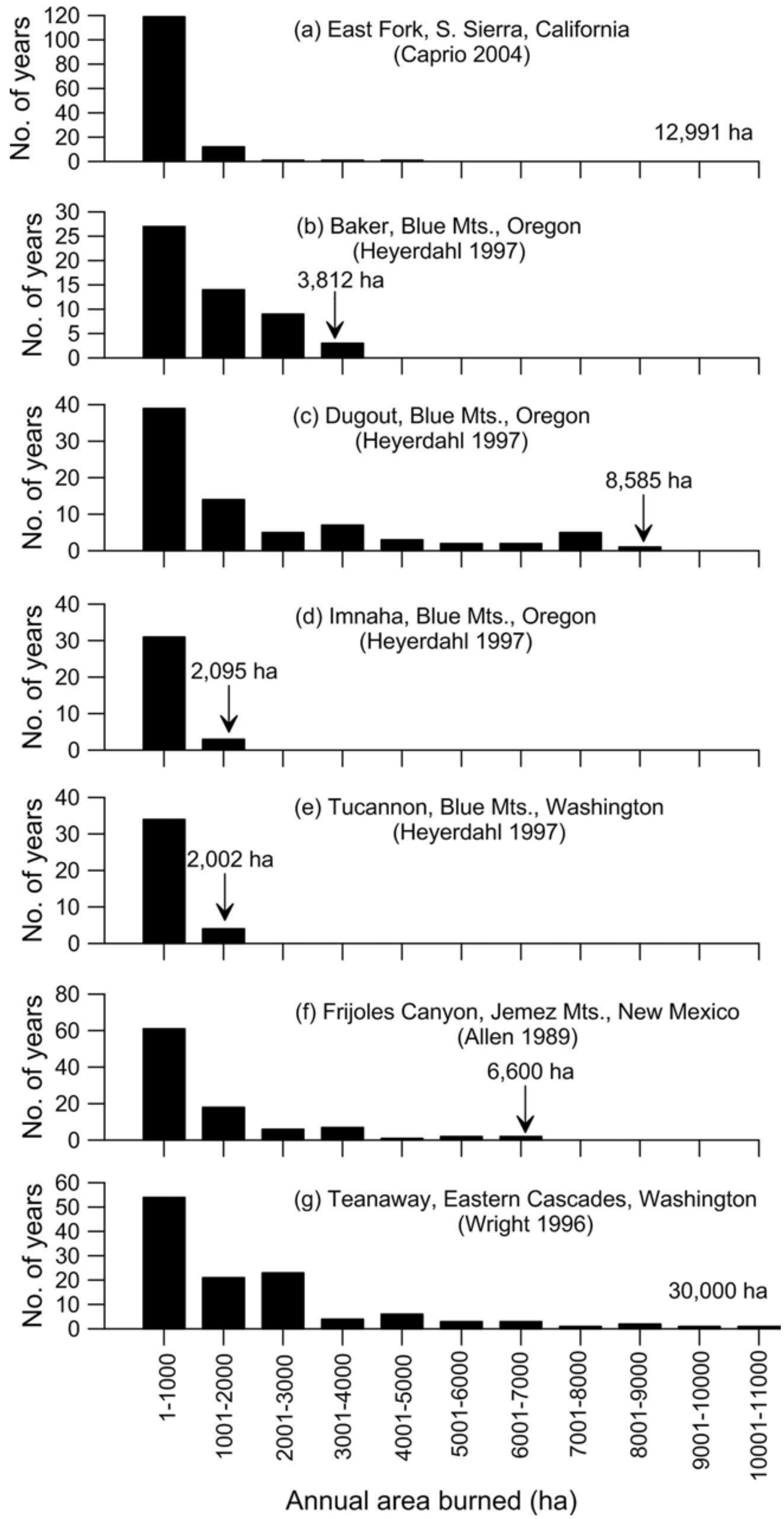

Fig 6. Size distribution for historical annual area burned in seven large study areas in dry forests of the western USA. Study area size is given above arrows or at the right of the x-axis.

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evidence. For example, generally low-severity fires in Sierran mixed conifer forests were observed to also have about 15% high-severity fire in small patches [20]. Small high-severity patches from low-severity fires in these forests were described [68] as mostly < 2–4 ha ([61]: Table A1 Q18-Q25). More research is needed on early historical observations of canopy mortality from low-severity fires, but some is historically congruent and expected from low-severity fires in modern forests.

Unburned areas within the perimeters of fires are also important for biological diversity and natural recovery, as these areas serve as refugia for less fire-tolerant plants or those that regenerate by seed, facilitating survival in these areas and natural recovery within the burned area [69]. This study found 35% average unburned area inside 154 modern fire perimeters in Yosemite National Park, California, which included substantial dry-forest area. Unfortunately, little is known about the extent of unburned area in historical low-severity fires in dry forests. It is known that prescribed burning that fully blackens burn units can reduce spatial heterogeneity in fire that promotes coexistence of multiple species [16]. Also, as reviewed in the introduction, unburned areas historically were locations where tree regeneration to replace tree mortality could survive. Thus, including unburned area within burn units, rather than blackening the whole unit, is ecologically important to restore and maintain tree populations and biological diversity.

The extent of needed burning to restore and manage old dry forests and the rest of dry forests is lower than previously thought. Earlier estimates were largely based on the assumption that reported mean CFI estimates represent PMFI/FR, which they do not (Tables 1, 2) and apply to all dry forests, which they also do not. Estimated historical rates of low-severity fire in dry forests in the U.S. Landfire program, for example, typically incorrectly use reported mean CFI estimates as though they represent PMFI/FR, although some actual PMFI/FR estimates are also used. These are applied to all dry forests, not just old forests. Both misapplications likely have adverse effects on biological diversity and ecosystem functioning. Prescribed burning in U.S. national forests, national parks, and on other public lands, where Landfire or other estimates from mean CFIs have been used as a guide, is likely too much by 1.6–3.3 times, depending on the CFI measure used (See *ß* inTable 2) in the roughly 14% of dry-forest area that was historically old forest with frequent fire (PMFI/FR < 25 years). Mean rates are likely too high by > 1.6–3.3 times in the 86% of the area of dry-forest landscapes that historically had multidecadal low-severity fire.

A need for less low-severity fire in restoration and management of dry forests is good news, because costs of prescribed burning and other restoration treatments are high, effects on invasive species, ecosystem processes, and biological diversity are a concern [15, 70], and the feasibility of restoring and managing low-severity fire is higher with longer rates. Longer rates also mean that completed treatments may have already been sufficient in many old-forest areas, and further management of low-severity fire can be redirected to using managed fire for resource benefit [71]. Where initial treatment is incomplete, one prescribed fire should suffice before a managed-fire program can begin. At that point, managers can monitor low-severity fire using historical mean PMFI/FR rates, fire-size distributions, and other attributes (e.g., unburned area) as a guide.

In locations where managed fire for resource benefit is infeasible, and an ongoing prescribed-burning program must be used, burning at rates longer than the mean PMFI/FRs reported here and using a diversity of rates and patterns of prescribed fires would be congruent with the findings. First, substantially lower rates (longer PMFI/FR) are warranted, if forests are not old forests, because estimated rates here apply mostly to old forests and the prevailing younger forests today likely burned historically at longer PMFI/FR. Second, the rates reported here are likely somewhat too short, as explained in "Limitations and errors. . ." Finally, lower rates would likely reduce the spread of invasive species and adverse effects on ecosystem processes and biological diversity. Also, historical rates varied substantially within small areas, particularly where there was topographic diversity, but also because of natural variability over time. It makes sense to similarly vary prescribed burning rates within local areas, leaving some areas unburned for longer periods. An approximation of the percentage of western USA old-forest parts of landscapes that experienced longer historical rates of fire is in Fig 4J. More local data can be derived from S2 Table, which lists PMFI/FR by state.

Data presented here can generally be used, with other evidence and tools, to create more comprehensive and spatially informative local understanding about mean historical PMFI/FR to guide local restoration and management of low-severity fire in old-forest parts of landscapes. Data in S2 Table have latitude and longitude and other ancillary information, and can be downloaded (S2 Dataset) and used directly or be read into a GIS program, where topography, land ownership and other information can be added for context. As new data are added to the IMPD, an FHX file for each new site can be downloaded and read into FHAES. Weibull mean ITFI can be calculated, which can then be used (Table 2) to estimate historical PMFI/FR, if not already provided in the study. Geographical coordinates, usually in the FHX file, allow new data to be added to the database (S2 Dataset) for use in GIS. Estimates of historical mean CFI and ITFI are available in the published scientific literature for other sites, not in the IMPD, which can also be used to estimate historical PMFI/FR using the equations in Tables 2 and 3, then added to the dataset (S2 Dataset) and input into GIS for local analysis. Of course, these estimates usually apply to only old-forest parts of historical landscapes.

Dry-forest landscapes until recently were thought to have historically been primarily old-growth forests, with a history of frequent low-severity fire, across their extent (e.g. [72]), but this has been refuted by GLO reconstructions and early aerial photographs (Table 6), paleo-ecological evidence [24], and early forest-reserve reports and other evidence [63, 73]. Even in Arizona, which had abundant old forests with frequent fire (Fig 3), denser forests and high-severity fire were extensive at certain times and in certain places, as on Black Mesa and parts of the Mogollon Plateau [60, 73]. It is sensible to restore low-severity fire to its former dominance in the parts of dry-forest landscapes with a history of primarily low-severity fire, historically averaging about 34% of western dry-forest landscapes (Table 6). Estimated mean PMFI/FRs here provide a guide for restoration and management of low-severity fire in extant old-forest parts of landscapes. For most dry-forests today, which are not old, using frequent fire (PMFI/FR $<$ 25 years) in restoration is not supported, and fuels do not need to be substantially reduced, because historical PMFI/FRs naturally allowed historical shrubs and small trees to fully recover after fires. Restoration of low-severity fire is still needed. The most appropriate approach, given likely long but uncertain mean rates of historical low-severity fire, is for most dry forests today to receive at most one prescribed fire, followed by managed fire for resource benefit, with the goal of mimicking mean historical PMFI/FRs and variability in fire (fire-size distributions, unburned area) as forests reach old age.

Supporting information

S1 Dataset.
(XLS)

S2 Dataset.
(XLS)

S1 Dataset metadata.
(PDF)

S2 Dataset metadata.
(PDF)

S1 Table. Authors, locations, and values for CFI and ITFI estimators and PMFI/FR in the 96-case calibration dataset.
(PDF)

S2 Table. Authors, sites, the Weibull mean ITFI estimate, and the calibrated or predicted PMFI/FR for the merged 342-site dataset.
(PDF)

S1 Text. Why CFIs and ITFIs underestimate PMFI/FR.
(PDF)

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Investigation: WLB.

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Bet-hedging dry-forest resilience to climate-change threats in the western USA based on historical forest structure

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Dry forests are particularly subject to wildfires, insect outbreaks, and droughts that likely will increase with climate change. Efforts to increase resilience of dry forests often focus on removing most small trees to reduce wildfire risk. However, small trees often survive other disturbances and could provide broader forest resilience, but small trees are thought to have been historically rare. We used direct records by land surveyors in the late-1800s along 22,206 km of survey lines in 1.7 million ha of dry forests in the western USA to test this idea. These systematic surveys (45,171 trees) of historical forests reveal that small trees dominated (52–92% of total trees) dry forests. Historical forests also included diverse tree sizes and species, which together provided resilience to several types of disturbances. Current risk to dry forests from insect outbreaks is 5.6 times the risk of higher-severity wildfires, with small trees increasing forest resilience to insect outbreaks. Removal of most small trees to reduce wildfire risk may compromise the bet-hedging resilience, provided by small trees and diverse tree sizes and species, against a broad array of unpredictable future disturbances.

Keywords: dry forests, wildfires, insect outbreaks, droughts, climate change, resilience, land surveys, bet-hedge

INTRODUCTION

Dry forests globally may be particularly vulnerable to climatic change, because their setting is prone to wildfires, insect outbreaks, and droughts; these disturbances may increase, and post-disturbance tree recruitment is often poor. Recruitment limitation in forests is a widespread concern (Clark et al., 1999), particularly where moisture is limiting, as in *Pinus* forests in drier parts of precipitation gradients (Dorman et al., 2013). For example, dry forests of the western USA (Figure S1), which include montane ponderosa pine (*Pinus ponderosa*) forests and dry mixed-conifer forests also with firs (*Abies* spp.) and Douglas-fir (*Pseudotsuga*), can have poor tree recruitment that limits their recovery after fires, insect outbreaks, and droughts. Tree recruitment in dry *P. ponderosa* forests of the western USA over the last century has been poor, concentrated in episodic pluvials (Savage et al., 1996), and spatially variable (Stein, 1988; Roccaforte et al., 2012). Mortality of *P. ponderosa* at their ecotone with lower-elevation woodlands during a 1950s drought (Allen and Breshears, 1998) also indicates vulnerability. Rising temperatures and drought could further reduce tree recruitment in dry forests (Anderson-Teixeira et al., 2013). Climate envelopes of seedlings vs. established trees of *P. ponderosa* suggest general recruitment failure is underway, possibly a precursor to broader range contraction (Bell et al., 2014).

In contrast, paleoecological research shows that dry forests of the western USA persisted for thousands of years in the face of wildfires, insect outbreaks, and droughts (Jenkins et al., 2011), suggesting recruitment was not generally deficient and historical forests were resilient. However, this persistence appears incongruent with the hypothesis that these dry forests historically had low abundance of seedlings, saplings and small trees (Covington and Moore, 1994; Allen et al., 2002). This hypothesis is based in part on tree-ring reconstructions, which show that large trees were historically dominant in most sampled stands (Williams and Baker, 2012a). However, small trees could have been common, but missed in tree-ring reconstructions because small trees had high mortality rates and may decompose by the time of reconstruction (Allen et al., 2002). Also, tree-ring reconstructions are not located systematically across landscapes and plot-level size-class distributions are often averaged, masking variability (Williams and Baker, 2013). Nonetheless, frequent surface fires were thought to have limited small trees, and some early accounts do suggest low abundance of tree recruitment (Leiberg et al., 1904; Covington and Moore, 1994; Allen et al., 2002). Today, large trees are likely less abundant and small trees more abundant than historically (Covington and Moore, 1994), but our focus is only on historical abundance of small trees, not current abundance. The common hypothesis is that low-severity fires historically limited small trees, so they were a low percentage of total trees and were found across a low percentage of land area.

We use a previously untapped historical source, the General Land Office (GLO) land surveys, which provide spatially extensive direct empirical data on historical tree recruitment (seedlings/saplings, small trees). We use seven study areas that span dry forests of the western USA (Figure S1) to test the hypothesis that dry forests historically had little tree recruitment. We formalize this for the two data sources from the GLO surveys and two components of recruitment abundance: H_1: Small trees were <20% of total trees, and H_2: Seedlings and saplings (trees < 10 cm diameter) were present on <20% of forest area.

Past specific estimates of percentages were lacking; we used test values that conservatively represent the hypotheses. Small trees are ≥10 cm dbh, with an upper size limit of 30–50 cm, defined for each study area (Williams and Baker, 2012a). We measured and compared recent risks of higher-severity wildfires and insect outbreaks in dry forests, separated into ponderosa pine forests and dry mixed-conifer forests, across the western USA using government data. We reviewed the role of tree recruitment in recovery after these disturbances. We suggest a strategy to maintain the resilience of dry forests to future disturbances, based on our findings.

MATERIALS AND METHODS

Data from the public land survey system, conducted by the U.S. General Land Office, have been widely used in the USA to reconstruct historical vegetation (Schulte and Mladenoff, 2001). Surveys in the study areas were generally done in the late-1800s before widespread expansion of EuroAmerican land uses. The system consists of 9.6 × 9.6 km townships containing thirty-six 1.6 × 1.6 km sections. Surveyors marked quarter corners at the 0.8 km mark and section corners at the 1.6 km mark along section lines. Surveyors were required to record azimuth, distance, species, and diameter of two bearing trees at quarter corners and four trees at section corners. Here we used surveyors' direct estimates of tree diameters. In an accuracy study, we found surveyors estimated diameters with sufficient accuracy to place trees in 10-cm diameter bins (Williams and Baker, 2010). After applying an empirical correction, diameter distributions from bearing trees were 87–88% similar to distributions from plot data (Williams and Baker, 2011), thus are quite accurate. Bearing trees are a statistically valid sample, as they have low bias and error (Williams and Baker, 2010).

We also used section-line data recorded by surveyors. Surveyors in forests were required to record, in order of abundance, the dominant overstory trees and understory plants, often including small trees (seedlings and saplings) and shrubs (Williams and Baker, 2012a). Surveyors also often recorded qualitative estimates of understory tree density. Not all surveyors followed the instructions, thus we limited analysis to the set of surveyors who did so for at least one section-line. The section-line data represent a statistically valid line-intercept estimate of cover (Butler and McDonald, 1983).

To provide data to test hypothesis H_1, we totaled small and large trees in each of the seven study areas and for the composite (**Table 1**, **Figure 1**). Small trees were defined as ≥10 cm but ≤40 cm, except ≤30 cm in the Colorado Front Range, where tree growth is slower (Williams and Baker, 2012a) and ≤50 cm in the western Sierra, where tree growth is faster (Baker, 2014).

Table 1 | Study areas, corresponding number of trees and section-line length in forested area, and the percentage of forest section line-length with seedlings and saplings.

Hypotheses and variables	Front range, Colorado[a]	Coconino Plateau, Arizona	Mogollon Plateau, Arizona	Black Mesa, Arizona	Blue Mts., Oregon	Eastern Cascades, Oregon	Western Sierra, California	Total or mean
Dry-forest study area (ha)	65,525	41,214	405,214	151,080	304,709	398,346	329,943	1,696,031[b]
H_1: SMALL TREES WERE < 20% OF TOTAL TREES								
Number of trees	1055	1643	10,848	2741	7496	11,856	9532	45,171[b]
Small-tree diameters used (cm)	≤30	≤40	≤40	≤40	≤40	≤40	≤50	≤30 to 50
Small trees (% of total trees)	91.8	69.5	51.8	81.1	62.0	62.4	60.9	61.6[c]
Chi-square test result[d]	$X^2 = 3404$	$X^2 = 2517$	$X^2 = 6859$	$X^2 = 6403$	$X^2 = 8267$	$X^2 = 13,326$	$X^2 = 9976$	$X^2 = 48,772$
	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$
H_2: SEEDLINGS AND SAPLINGS WERE PRESENT ON < 20% OF FOREST AREA								
Section-line length (km)	4004	413	4230	1441	5878	3873	2367	22,206
Seedlings/Saplings present (%)	3.8	43.4	13.3	8.0	34.6	57.4	54.9	29.6
Chi-square test result[f]	$X^2 = 657$	$X^2 = 140$	$X^2 = 119$	$X^2 = 150$	$X^2 = 780$	$X^2 = 3385$	$X^2 = 1780$	$X^2 = 1238$
	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$
Seedlings/Saplings dense (%)	0.2	28.8	1.9	-	22.4	30.3	20.0	14.3
Seedlings/sapling pines[e]	0.9	1.4	9.8	7.9	32.7	51.0	42.3	24.8
Seedlings/Sapling firs[e]	0.5	0.0	0.0	0.0	27.1	27.8	39.7	16.4
Seedling/Sapling oaks[e]	0.5	43.3	8.8	7.1	0.0	0.2	42.4	7.6
Seedling/Sapling other trees[e]	2.5	0.4	1.2	2.0	0.3	2.6	25.1	4.0

[a] *Study areas include the Colorado Front Range (Williams and Baker, 2012a), Coconino Plateau, Arizona (Williams and Baker, 2013), Mogollon Plateau and Black Mesa, Arizona and Blue Mountains, Oregon (Williams and Baker, 2012a), Eastern Cascades of Oregon (Baker, 2012), and western Sierra Nevada, California (Baker, 2014).*

[b] *Total.*

[c] *Percentage for the composite across the seven study areas.*

[d] *Degrees of freedom = 1 and N = the number of trees, for all chi-square tests.*

[e] *Seedling/Sapling pines, firs, oaks, and other trees may be overlapping, as a line can have, for example, both pines and firs.*

[f] *Degrees of freedom = 1 and N = the number of 1-km line-lengths, for all chi-square tests.*

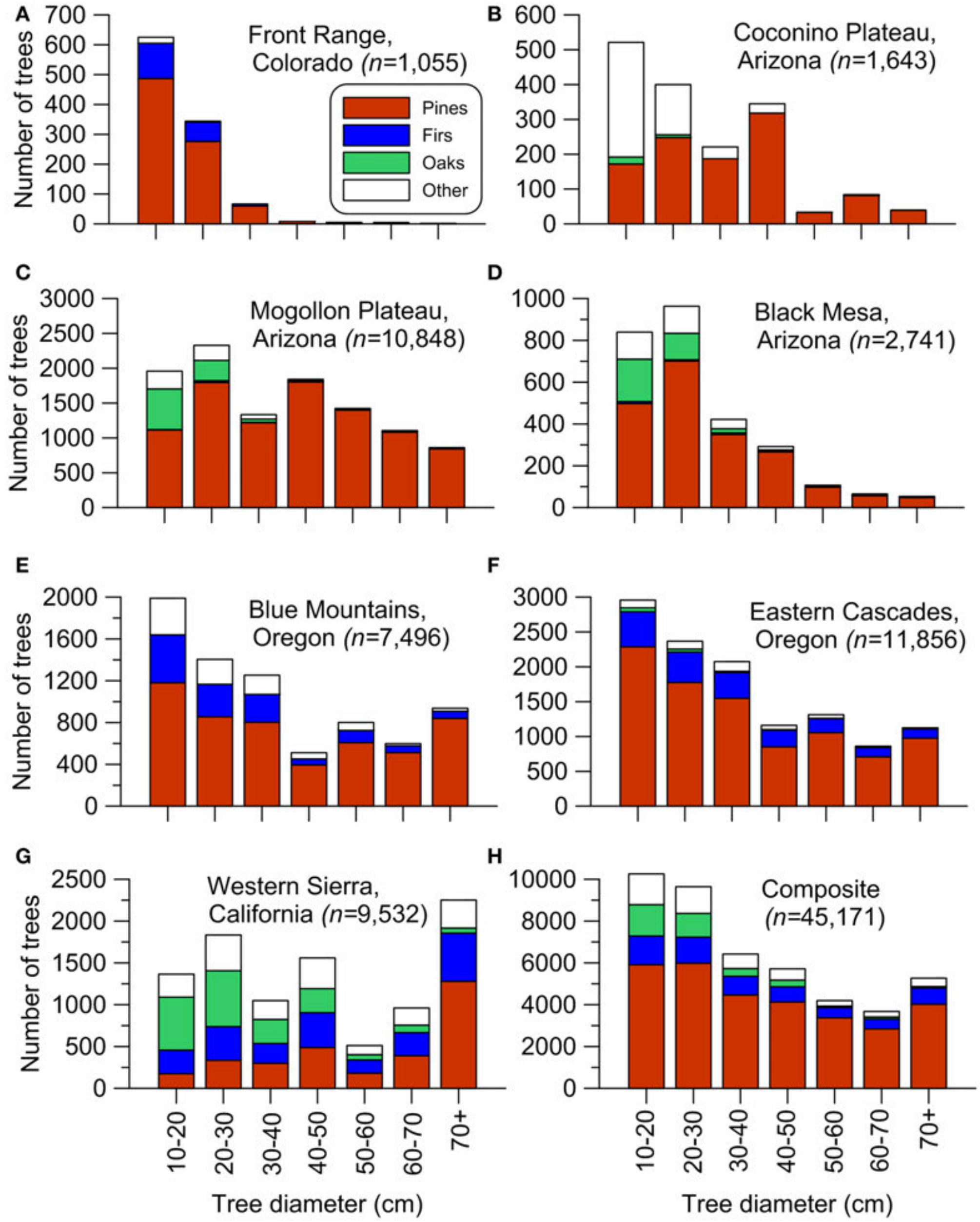

FIGURE 1 | Historical tree size-class distributions for the seven study areas and a composite across all the study areas: (A) Front Range, Colorado, (B) Coconino Plateau, Arizona, (C) Mogollon Plateau, Arizona, (D) Black Mesa, Arizona, (E) Blue Mountains, Oregon, (F) Eastern Cascades, Oregon, (G) Western Sierra, California, (H) The composite of all areas. Distributions use 10-cm bins compatible with the accuracy of diameters measured by the surveyors (Williams and Baker, 2011). Other trees, not found in every area, include *Pinus edulis* and *Juniperus* spp., *Calocedrus decurrens*, *Populus tremuloides*, and *Larix occidentalis*. As in **Table 1**, small trees were defined as trees $\geq$10 cm but $\leq$40 cm diameter, except $\leq$30 cm in Colorado **(A)** and $\leq$50 cm in California **(G)**.

These diameters generally represent trees that are less than about 140 years old (Bright, 1912; Baker, 2012, 2014; Williams and Baker, 2013). Trees this size today are often thought to have widely established after EuroAmerican settlement because of logging, livestock grazing, and fire exclusion (Covington and Moore, 1994; Allen et al., 2002; Franklin and Johnson, 2012), and thus may be removed in restoration treatments. To test H_1, we used a chi-square goodness-of-fit test of a null hypothesis that small trees were 0.2 of total trees and large trees were 0.8 of total trees. If this null was rejected, we rejected H_1 if small trees were <0.2 of total trees. To control error rates, we Bonferroni-corrected $\alpha = 0.05$, for 8 planned tests, one per study area and one for the composite (**Table 1**, **Figure 1**), to $\alpha = 0.00625$.

To provide data to test H_2, we totaled 1-km section lines for which surveyors recorded understory trees in each of the study areas and for the composite. Similarly, to test H_2, we used a chi-squared goodness-of-fit test of a null hypothesis that the area with seedlings/saplings was 0.2 of the total forested area and the area without seedlings/saplings was 0.8 of the total forested area. If this null was rejected, we then rejected H_2 if seedlings/saplings were found across <0.2 of total forest area. We also Bonferroni-corrected an initial $\alpha = 0.05$ for 8 planned tests.

We used maps of ponderosa pine and dry mixed-conifer forests from Landfire Biophysical Settings (www.landfire.gov). Wildfire area and severity were from raster maps of actual burned area, not fire perimeters, from the Monitoring Trends in Burn Severity (MTBS) program (http://www.mtbs.gov). Insect-caused mortality was from the US Forest Service Forest Health Technology Enterprise Team (http://foresthealth.fs.usda.gov/portal/Flex/IDS). Insect outbreaks were detected using annual aerial surveys. To limit analysis to dry western forests, aerial survey polygons and wildfires were both clipped by the maps of ponderosa pine and dry mixed conifer. The annual sample area varied, but averaged about 9.8 million ha of ponderosa pine and 10.9 million ha of dry mixed-conifer forests (Table S1), about 80% of the 25.8 million ha area of western dry forests.

Comparison of wildfire and insect outbreaks was done for each year both datasets were available. We compared moderate- and high-severity wildfire area, which are the severities with substantial tree mortality, with areas where tree mortality from insects was also substantial, as it was visually detected from aerial surveys. We calculated the rate of wildfire using the fire rotation, which is the number of survey years divided by the fraction of the survey area impacted by fire in those years. The rate of insect outbreaks was determined similarly. Some outbreak areas appeared to overlap in subsequent years and potentially be cumulative. We performed a union and spatial dissolve in GIS to derive a conservative estimate of total area impacted by insect outbreaks over the analysis period. Additional details are in Supplementary Methods.

RESULTS

SMALL TREES HISTORICALLY ABUNDANT AND DOMINANT

Hypothesis H_1 is rejected across all seven study areas and the composite (**Table 1**). Small trees generally dominated historical dry forests, ranging from 51.8 to 91.8% of total trees across the seven study areas and equaling 61.6% of trees in the overall composite (**Table 1**, **Figure 1**). Small trees can be suppressed older individuals, but were predominantly <140 years old (Bright, 1912; Williams and Baker, 2012a). Small trees were somewhat diverse, with pines most abundant, but also firs, oaks and other conifers and hardwoods (**Figure 1**). Hypothesis H_2 is rejected for study areas in California and Oregon, but not in Arizona and Colorado (**Table 1**).

HIGHER RECENT THREAT FROM INSECT OUTBREAKS THAN FROM WILDFIRE

Data from government agencies show that insect outbreaks were recently a more significant threat to dry forests than were moderate- to high-severity wildfires; similar data are not available for droughts. It is conservatively estimated (i.e., consolidating all areas of spatial overlap) that insect outbreaks caused substantial detectable tree mortality in 5,193,752 ha of western dry forests over the 1999–2012 period for which spatial data were available, which is 5.6 times the 934,551 ha impacted by moderate- to high-severity wildfires (Table S1). Mean ratios of insect to fire impact were 4.5 in ponderosa pine and 6.9 in dry mixed-conifer forests (Table S1). At the rates during 1999–2012, it would require 311 years for moderate- to high-severity wildfires to burn once across an area equal to the area of western dry forests, but only 56 years for insect outbreaks to impact this area (Table S1). Rotations for fire varied from 265 years in ponderosa pine to 367 years in dry mixed-conifer forests, and for insects from 53 years in dry mixed-conifer to 59 years in ponderosa pine forests (Table S1).

DISCUSSION

NATURAL DISTURBANCES FOSTERED HISTORICALLY ABUNDANT SMALL TREES AND DIVERSE TREE SIZES

Historical dominance of small trees in dry forests (**Figure 1**) does not support the hypothesis that surface fires generally kept small trees rare. Small trees had successfully recruited and were dominant in all dry-forest areas (**Figure 1**). These small, established trees are given more weight, than smaller, more ephemeral seedlings/saplings, for which evidence is more mixed. Seedlings/saplings were abundant in the majority of areas, except two southwestern landscapes (Black Mesa, Mogollon Plateau) and the Colorado Front Range (**Table 1**). Early scientific sources corroborate limited seedlings/saplings in these areas (Leiberg et al., 1904; Williams and Baker, 2012b). Early foresters emphasized preserving advanced recruitment during logging (Pearson, 1923). Thus, recent high-severity fires do not have unprecedented poor recruitment (Savage and Mast, 2005). Seedling/sapling populations in these landscapes must have fluctuated, since small trees had been able to recruit and dominate all dry forests (**Figure 1**). Particular sequences of fires, droughts, and other disturbances may explain fluctuating seedling/sapling populations (Dugan and Baker, in press), and reinforce the historical role of advanced recruitment.

Dominance of small trees, and even ephemeral seedling/sapling populations in most areas, indicates more imperfect limitation of tree recruitment by historical low-severity fires than previously thought. Other disturbances, including droughts, insect outbreaks, and more severe fires likely killed canopy trees and increased tree recruitment, particularly if followed by pluvials (Savage et al., 1996; Dugan and Baker, in press). The Colorado Front Range and Black Mesa (Williams and Baker, 2012a) had the greatest dominance of small trees (**Figures 1A,D**), and our reconstructions showed these areas had more higher-severity fires (Williams and Baker, 2012a,b). Historical abundance of small trees and importance of higher-severity fires in structuring tree populations across dry-forest landscapes are supported by an independent dataset of tree ages (Odion et al., 2014). Higher-severity fires likely interacted with other disturbances to produce diverse tree sizes that were together more resilient to disturbance than would have been the case if only low-severity fires had occurred and large trees had dominated. Historical dominance by small trees and diverse trees sizes are consistent with long-term persistence and resilience of dry forests after disturbances (Jenkins et al., 2011).

ABUNDANT SMALL TREES AND DIVERSE TREE SIZES CONFER RESILIENCE IN MODERN FORESTS

Modern observations also document key, but contrasting roles for advance recruitment and surviving larger trees in forest resilience after fires, insect outbreaks, and droughts. Higher-severity fires may be followed by variable recruitment, including poor recruitment, lags in recruitment, or abundant recruitment in some areas (Roccaforte et al., 2012), with large, surviving trees and proximity to them important (Bonnet et al., 2005; Haire and McGarigal, 2010).

About a dozen bark-beetles, that kill trees over large areas of dry forests in the western USA, are the major outbreak insects (Bentz et al., 2010; Weed et al., 2013). In this case, larger trees are differentially susceptible, which often leaves smaller surviving trees as the key source of post-outbreak recruitment. Vulnerability of larger trees to bark beetles is related to greater food resources (Raffa et al., 2008). In a 1970s outbreak of mountain pine beetle (*Dendroctonus ponderosae*) in ponderosa pine in Colorado, tree survival was substantially higher for trees <20 cm diameter (McCambridge et al., 1982). Similarly, western pine beetles (*Dendroctonous brevicomi*s) kill relatively few trees <40 cm (Miller and Keen, 1960). However, *Ips* in Arizona preferentially kill smaller trees (Negrón et al., 2009). Nonetheless, advance recruitment generally dominates post-outbreak recruitment. After spruce beetle (DeRose and Long, 2010) and mountain pine beetle outbreaks (Astrup et al., 2008), small trees present before outbreaks dominated post-outbreak recruitment. Since these small trees were more diverse than pre-outbreak canopy trees, post-outbreak forests may have greater resilience to future outbreaks (Diskin et al., 2011; Kayes and Tinker, 2012).

Drought often also differentially kills the largest, oldest trees, with less mortality in small and mid-sized trees (Allen et al., 2010), thus also leaving advance recruitment. Drought effects on tree mortality can be widespread and affect forests for centuries (Allen et al., 2010). Drought also influences the occurrence of wildfires, insect outbreaks, and regional tree mortality (Allen et al., 2010), thus it is difficult to parse the impacts of drought alone.

The upshot is that both small trees and surviving larger trees and a diversity of tree species provide resilience to disturbances. Surviving larger trees are particularly important after higher-severity fires and abundant small trees are particularly important after insect outbreaks and droughts.

RESTORING AND MAINTAINING THE BET-HEDGING RESILIENCE OF HISTORICAL FORESTS

Current restoration strategies that seek to increase forest resilience focus predominately on impacts from severe wildfires, but bark-beetle outbreaks and other insects affected 5.6 times the area of western dry forests impacted by moderate- to high-severity fires over the most recent 14-year period (1999–2012). Current rates of moderate- and high-severity fire, with a combined rotation of 311 years (Table S1), would likely not prevent recovery of old-growth forests in the interlude between fires, but rates of insect outbreaks, with a rotation of 56 years (Table S1), could prevent recovery of most older dry forests. Previous research, using the same data sources, in a more limited and lower-elevation area in the southwestern United States, found that beetle-outbreaks affected 2.5–4 times as much area as moderate- to severe wildfires (Williams et al., 2010). Both wildfires (Dennison et al., 2014) and beetle-outbreaks (Bentz et al., 2010; Weed et al., 2013) are increasing in parts of the western United States. Future outcomes are uncertain and complex, however, as beetle-outbreaks can affect wildfire probability (Simard et al., 2011), and as tree mortality occurs, both beetle outbreaks and wildfires could become self-limited (Williams et al., 2010).

Ecological restoration of public dry forests in the western USA is increasingly a goal, because these forests were altered by unsustainable logging, livestock grazing, and fire exclusion that allowed abundant small trees to recruit (Covington and Moore, 1994). Retaining older trees, while removing most small trees up to ages or sizes of trees recruited since EuroAmerican settlement (Figure S2A), is thus often a restoration focus (Covington and Moore, 1994; Allen et al., 2002; Abella et al., 2006; Franklin and Johnson, 2012). Typical upper tree age and size limits are 120–150 years old or 30–50 cm diameter (Abella et al., 2006; Franklin and Johnson, 2012).

We show here, however, that these small trees were the tree sizes historically dominant in these forests (**Figure 1**, **Table 1**), thus removing most small trees so they are no longer dominant is not ecological restoration. There are also efforts underway to increase resilience of forests to droughts by removing most small trees and lowering stand density. However, stand density does not appear to play a major role in level of tree mortality from drought (Ganey and Vojta, 2011). Thus, strategies to reduce most small trees are neither restorative nor very effective.

We suggest diverse historical tree sizes and abundant and dominant small trees long provided bet hedging in dry-forest landscapes subject to unpredictable disturbances. These forests can be more effectively restored and their resiliency to future disturbances increased by maintaining or restoring the historical abundance, dominance, and diversity of small trees, while also restoring large trees depleted by logging (Figure S2B). This can be achieved with historically congruent diversities of forest structures across landscapes, based on GLO and other spatial reconstructions. This bet-hedging landscape approach to ecological restoration is consistent with long-term persistence of historical forests, the high current threat from insects, and would likely confer more resilience to disturbances, that may all increase in the future, than would just retaining larger or older trees across large areas.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: http://www.frontiersin.org/journal/10.3389/fevo.2014.00088/abstract

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S1 Text. Why CFIs and ITFIs underestimate PMFI/FR

Since both mean composite fire interval (CFI) and mean individual tree fire interval (ITFI) typically underestimate the population mean fire interval or fire rotation (PMFI/FR), it is logical to infer that both methods are missing longer intervals because of biased estimates (Table 1 main text). Compositing alone could explain nearly the whole bias in CFI measures, as explained below, but ITFI measures are not composites and are still biased, although less so. The most likely explanation for bias in ITFI measures, and also a contributor to bias in CFI measures, as estimators of PMFI/FR, is targeted sampling. These potential sources of bias are reviewed in detail here.

Compositing overcompensates, destroys long fire intervals, and restriction rules do not remedy this

Scarring fraction, compositing, and widespread over-compensation

The purpose of compositing is to compensate for the incomplete scar record on individual trees, since trees can often resist scarring even if burned (Baker and Dugan 2013). Scarring fraction (SF) is the fraction of burned live trees that survive a fire but receive a scar. Studies of SF are few (e.g., Collins and Stephens 2007, Stephens et al. 2010). A study of 16 fires in ponderosa pine forests in northern Arizona found a mean SF of 0.375, ranging from 0.121 to 0.728 across 52 plot samples (Baker and Dugan 2013).

Given a particular SF, how many trees must be sampled or composited to compensate for SF < 1.0 (Baker and Dugan 2013)? The minimum is to have a sample size that has a high probability of recording each fire on at least one scarred tree. The probability, P, of at least one tree scarring in a sample of n live trees, for a scarring fraction, SF, is given by:

$$P = 1.0 - (1.0 - \mathrm{SF})^{n} \qquad (1)$$

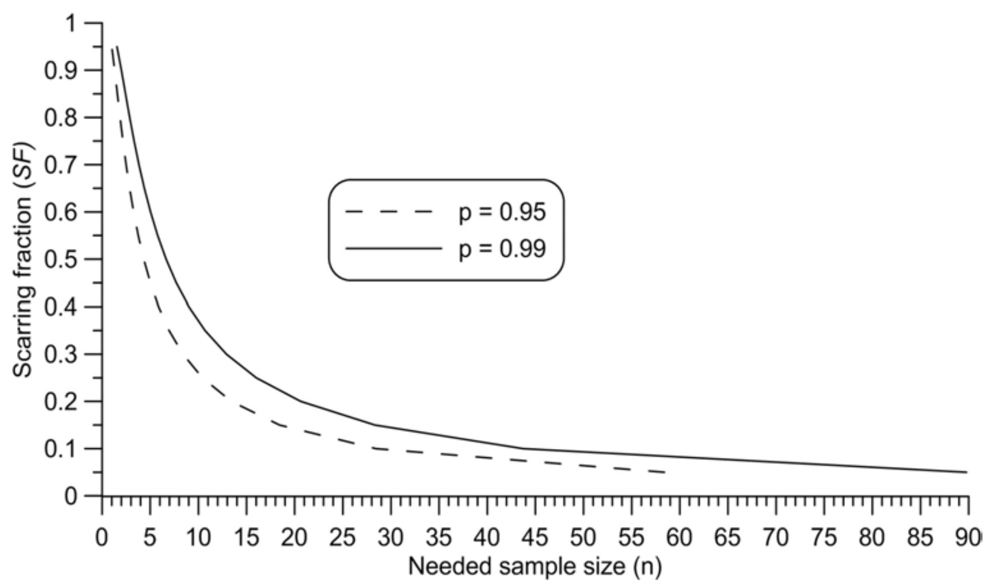
and the corresponding estimate of n, for a particular SF, is given by:

$$n = \log(1.0 - P) / \log(1.0 - \mathrm{SF}) \qquad (2)$$

The necessary sample sizes to achieve a probability ≥ 0.95 or 0.99 of detecting a fire are modest, typically < 20 trees, whether scarred or not, to detect fires with SF < 0.25 (Figure A).

However, this equation does not adjust for scar healing. Scars can, but do not always, heal from the sides and disappear under new bark unless subsequent fires occur (Baker and Dugan 2013). However, Fiegener (2002) examined over 8,000 stumps and snags in a Sierran mixed-conifer forest and found only 2% with scars. An empirical study of scar healing after fires in northern Arizona ponderosa pine forests

Figure A. Scarring fraction and its effect on the sample size needed to have either a 0.95 or 0.99 probability of scarring at least one tree.



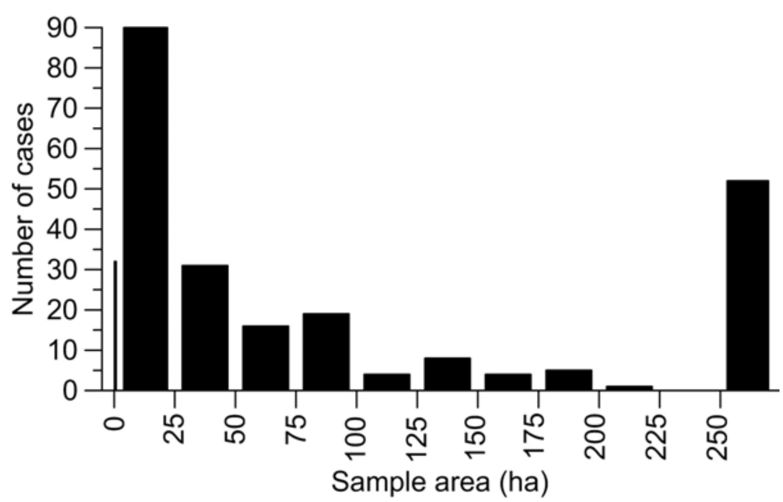
showed that larger initial scars have longer expected healing times and subsequent fires increased healing times of all scars (Baker and Dugan 2013). Healing rates from this study can be used to estimate the needed sample size to find at least one unhealed scar for a fire after 100 years, or another time since fire, using an equation developed from forests differing in time since fire:

$$\text{Effective mean SF} = \text{Initial SF} \times \exp(-0.0125 \times \text{Time since fire}) \qquad (3)$$

For example, if expected mean SF is 0.400, then six sample trees would likely ($P = 0.95$) contain at least one scar (using Eq. 2 or Figure A) from a fire burned recently. In contrast, the effective mean SF if that same fire had burned 100 years ago, and scars had healed since then, would be 0.115, from Eq. 3, thus requiring a sample of about 26 trees each > 100 years old (Eq. 1, Figure A). Similarly, a lower SF of 0.200 would require 51 trees > 100 years old for a fire 100 years ago. These calculations suggest that sufficient trees to detect fires in historical landscapes could likely be obtained from unlogged areas that are on the order of about 1 ha in area or even less.

In contrast, most compositing is from areas far too large for the area and number of sample trees usually needed to compensate for SF < 1.0. In the merged dataset of 342 sites, only 262 reported area sampled. Of those, only 32 (12.2%) were from areas < 1.0 ha (Figure B). One concern is whether SF rates estimated in this study are higher than they would have been in historical forests, because fire exclusion increased fuel loads in modern forests, likely increasing SF. Some effect is likely, but the effect would not change the general pattern of widespread overcompensation. First, if a preceding fire occurred within 30 years, then SF was reduced from a mean of 0.393 to 0.324, only an 18% reduction, in the Baker and Dugan (2013) study. This would have a minor effect of increasing the number of needed sample trees from 26 trees to 31 trees, having almost no effect on the widespread pattern of over-compensation evident in Figure B. Second, even in the extreme case of a 0.05 mean SF in historical forests, assuming a historical fire rotation of < 10 years (Stephens et al. 2010), only 208 trees, whether scarred or not, would be needed after 100 years to achieve a probability ≥0.95 of detecting a fire. This could be obtained in most historical dry forests in < 2-3 ha. Even at this extreme level of SF, only 18.7% of the 262 sites were from areas < 3 ha, thus 81.3% of studies were over-compensating.

Figure B. Number of cases (n = 262 total) by sample area (ha). Many cases did not report area. The first bin is from 0-1 ha, the remainder are 25-ha wide, and the last bin is for areas > 250 ha.



This over-compensation particularly biases CFI estimates toward values that are too short, since mean CFI declines as sample area or number of sample trees increases (Arno and Petersen 1983, Baker and Ehle 2001, Everett 2003, Kou and Baker 2006a, b). ITFI and FR estimates, in contrast, do not systematically decline with larger samples, and may even become more precise. Compositing records across an area or number of trees that is too large could explain why CFI estimates are too short relative to FRs, but cannot explain why ITFI estimates, which do not use compositing, are also too short.

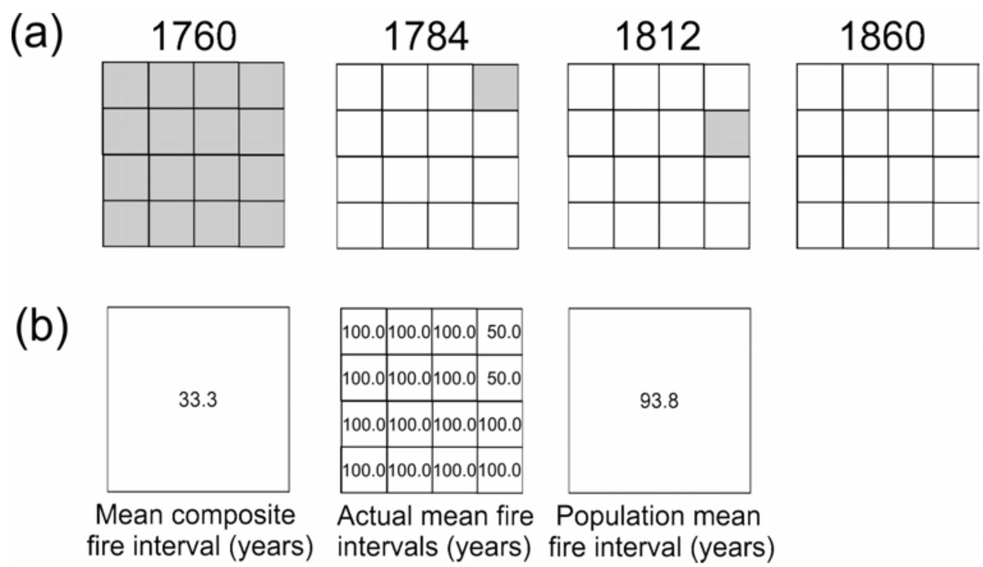
Compositing not only over-compensates, but also destroys long fire intervals that are real

Compositing is a processing step, separate from finding and collecting an adequate sample. Several methods can be used to process sample data, including calculating mean ITFI (Dugan and Baker 2014), or estimating FR, thus the compositing step is not essential. How does the compositing step contribute error if used to estimate PMFI/FR? Most fires are small and only a few are large (Baker and Ehle 2001). When a composite list is created, and intervals are calculated among fires in the list, each small fire year counts the same as a large fire year. Even though some compositing might offset incomplete evidence, at the same time it destroys other evidence. Longer fire-free intervals that are real occur in unburned parts of landscapes adjacent to where small fires occurred, and some long intervals that are false because scarring is incomplete also occur. However, all these long intervals, whether real or false, are erased across the whole sample area when a composite list is created, rather than disappearing only in the area where a small fire occurred (Figure C). Since longer intervals, some of which are real, are all lost to compositing, this in part explains why mean CFI underestimates PMFI/FR.

CFI restriction rules are ad hoc, inconsistent, and likely insufficient in excluding small fires

Some suggest that there is only a problem with mean CFI and its use if it is presented without omitting spot fires:
"...this becomes a problem only if the fire chronology is presented with all fires, even the smaller spot fires, and is interpreted by the reader as if the chronology indicates how often the entire stand burned" (Stephens et al. 2003 p. 1091).

Figure C. How compositing destroys long fire intervals that are real: (a) a landscape that burned (shaded area) in each of three fire years between 1760-1860, and (b) the actual mean fire intervals (center panel), the mean composite fire interval for the landscape (left panel) calculated from a composite list of fires (1760, 1784, 1812, ending in 1860), thus three intervals in 100 years = 33.3 years, and the PMFI (right panel), which is the grand mean of the actual mean fire intervals (1500 years/16 = 93.8 years). Note that the reason that the mean CFI underestimates the PMFI is that compositing treats the 1784 and 1812 fire years as though they burned the same land area as the 1760 fire year, which also eliminates the 100.0-year fire intervals that occurred over most of the landscape.



Restriction rules are traditionally applied to filter out fires, like spot fires, that are small, using the number of fire scars or the percentage of total scarred trees that record a fire year (e.g., 10%, 25%). However, no way is known to objectively identify a spot fire or other small fire that should be omitted, since fire-size distributions are typically nearly linear on a log-log plot and have no natural breaks (Kou and Baker 2006b). Also, distributions vary in slope among forest types and environments (Kou and Baker 2006b), so imposing a particular filter (e.g., 25%) has varying effects. This means that restriction rules are ad hoc and inherently inconsistent in their effects.

Moreover, 10% and 25% filters that are typically applied, may be insufficient to limit the fires that should be included in a composite list, if the goal is that mean intervals between fires in the list estimate the PMFI/FR. In a spatial reconstruction of fire sizes in dry forests, Farris et al. (2010) found that 414 total fires occurred in their study area from 1937-2000, but only 21 fires (5.1% of total fires) accounted for 97% of total burned area. This suggests a restriction rule would have to exclude 95% of fires to limit a composite to the fire years that account for most of the total burned area, which would more likely accurately estimate PMFI/FR. Together, the ad hoc, inconsistent, and insufficient extent of traditional restriction rules in part explain why mean CFI underestimates PMFI/FR.

Censoring incomplete fire intervals leaves out long intervals in both CFI and ITFI estimates

Fire-history data contain incomplete intervals at the beginning and end of a period of record unless those periods begin and end with fires (Polakow and Dunne 1999). Incomplete intervals can be included or omitted ("censored") in analysis of fire-interval data (Polakow and Dunne 1999). Censoring (i.e., using only scar-to-scar intervals) biases both mean CFI and ITFI by omitting incomplete intervals at the beginning or end of a tree's record. Incomplete fire intervals occur on most trees, but longer intervals have more chance, than shorter intervals, of appearing as incomplete intervals, indicated by no scars or one scar on a tree (Kou and Baker 2006a). Simulation has shown that in a landscape subject to low-severity fires at modest intervals (e.g., 50 years), actual intervals at some locations may be several times longer (Kou and Baker 2006a) and even up to an order of magnitude longer than the mean interval (Parsons et al. 2007). These are real intervals that occur by chance, not an artifact of incomplete scarring. Censoring is biased against these expected long fire intervals and leads to estimates of PMFI/FR that are too short and have reduced variability, since longer intervals are omitted (Kou and Baker 2006a). These effects from censoring were also found in two studies in Mediterranean shrublands, in which censoring reduced the scale parameter (indicator of length of fire intervals) of a Weibull fire-interval distribution and also reduced estimated variability in fire intervals (Polakow and Dunne 1999, Moritz et al. 2009).

These censoring effects have ecological implications in dry forests subject to periodic fires, since most composited fire-scar records, which are traditionally censored, lack evidence of the long intervals needed for tree regeneration and survival of fire-intolerant species. We suggested that the interval before the first fire scar (origin-to-scar interval--OS) on individual trees may record the fire-free period needed for trees to successfully regenerate (Baker and Ehle 2001), since both wide-area and local processes producing long intervals should be recorded as OS intervals. Mean OS intervals are, in fact, usually much longer than mean scar-to-scar intervals in the same stands, and many are sufficiently long to allow tree regeneration (Baker and Ehle 2001). Mean OS intervals in ponderosa pine forests were 51 years in the Black Hills (Brown et al. 2008), 55.4 years in Rocky Mountain National Park (Baker and Ehle 2003), 81 years across five studies (Baker and Ehle 2001), and 101.5 years in one case in northern Arizona (Van Horne and Fulé 2006). Arguments can be made for and against including the OS interval in CFI estimates (e.g., Baker and Ehle 2003, Van Horne and Fulé 2006, Stephens et al. 2010). However, long intervals that are real do occur and are directly censored by traditional use of only scar-to-scar intervals in CFI and ITFI estimates, contributing to underestimation of PMFI/FR by CFI measures.

Targeted sampling likely a significant source of underestimates of PMFI/FR by ITFI, as well as by CFI

<u>Why researchers target fire-history evidence and why it remains a concern for estimating PMFI/FR</u>

Researchers target fire-history evidence to increase the length of record and maximize the data obtained with minimal physical effort and damage to trees (Farris et al. 2013). If only 50 scarred trees can be sampled, more fire years per scarred tree and a longer mean length of record will nearly always be obtained from 50 trees selected by targeting than from a random sample.

Unfortunately, targeting fire-history evidence at the scale of individual trees, sampling areas, and landscapes produces biased estimates of fire history (Lorimer 1985, Johnson and Gutsell 1994, Baker and Ehle 2001). The consequences are generally that estimates of historical PMFI/FR are too short and fire-severity is underestimated. The magnitude of targeting and its effects is now better known. Targeting remains common in fire-history studies, as illustrated in Table A, which shows that targeting of individual trees, particularly multi-scarred trees and old trees, was widespread, almost universal for multi-scarred trees, and almost 1/3 of studies placed study plots where there were concentrations of scarred trees and old trees.

Table A. Percentage of 342 sites in which various types of targeting sampling were used.

Targeting type and measures	**Yes**	**No**	**No explanation**
1. Target trees to get best information or longest record of fires?			
Number of cases	114	68	160
Percentage of yes/no (%)	62.6	37.4	-
2. Target multi-scarred trees?			
Number of cases	235	15	92
Percentage of yes/no (%)	94.0	6.0	-
3. Target clusters of scarred trees?			
Number of cases	37	9	296
Percentage of yes/no (%)	80.4	19.6	
4. Target scars on dead wood?			
Number of cases	270	27	45
Percentage of yes/no (%)	90.9	9.1	
5. Target tree species thought to better record fire			
Number of cases	12	13	317
Percentage of yes/no (%)	48.0	52.0	
6. Target plot locations in old forests and concentrations of scars			
Number of cases	73	153	116
Percentage of yes/no (%)	32.3	67.7	
7. Target study areas in old forests and concentrations of scars			
Number of cases	18	168	156
Percentage of yes/no (%)	9.7	90.3	

Specific studies of some of these types of targeting are now available (Baker and Ehle 2003, Van Horne and Fulé 2006, Kou and Baker 2006a, Brown et al. 2008, Farris et al. 2010, 2013), but the most significant types are less studied. Studies whose findings supported targeted sampling (e.g., Van Horne and Fulé 2006) for some purposes did not study using targeted sampling for estimating PMFI/FR, the focus here, thus targeted sampling has not been supported for this purpose.

Targeting individual trees

Targeting individual trees typically includes a bias component and a non-random sampling component. The bias component is from omitting trees with no scars or one scar and preferentially or exclusively using trees with multiple scars. The non-random sampling component comes from purposely choosing particular multi-scarred trees rather than randomly sampling them.

Significant bias is likely from omission of trees with no or single scars, which are traditionally omitted because only scar-to-scar intervals provide estimates of complete fire intervals. However, no scars or single scars on a tree may be false, because fires do not scar every tree that burns, but no or single-scarred trees also include real but incomplete fire intervals. No or single scars that represent real, incomplete long fire intervals are more likely where fire intervals also are longer (Kou and Baker 2006a). More long intervals and more of the length of long fire intervals are inherently present on unscarred or single-scarred trees, assuming tree ages are similar to those of multiple-scarred trees. Since longer fire intervals are more likely to be omitted by individual-tree targeting, all types of estimators (i.e., CFI, ITFI) from multi-scarred trees are biased toward being too short (Kou and Baker 2006a). FR estimates are also biased toward being too short if only multi-scarred trees are sampled, because trees with no scar or one scar can indicate places where a fire did not burn, and these omissions inflate area burned for that fire year, and shorten the estimated FR.

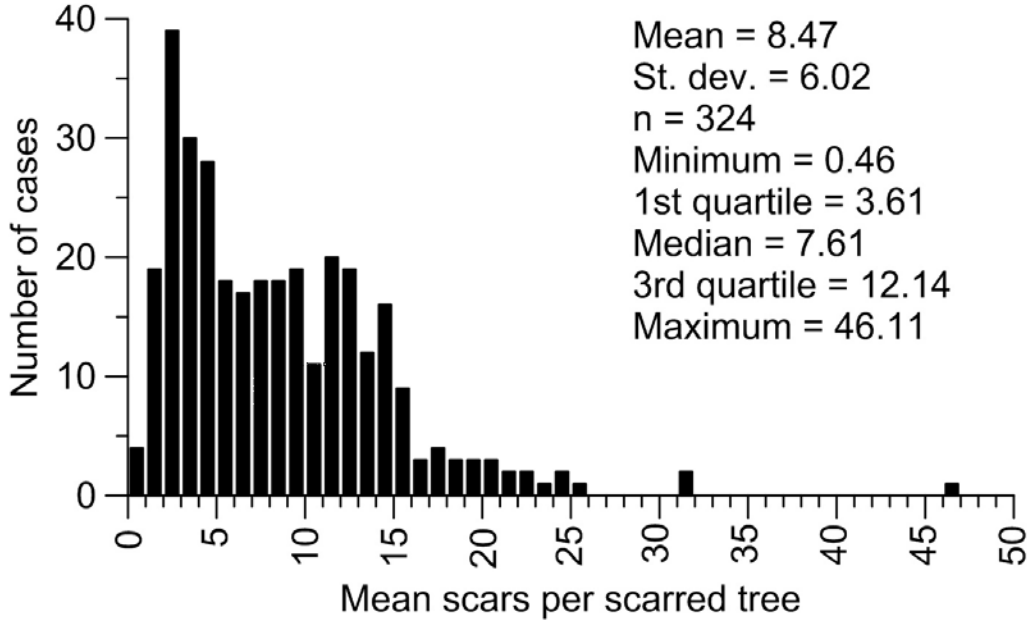
How does targeting trees with more than one scar (multi-scarred trees, recorder trees, and open-scarred trees) lead to CFI and ITFI values that underestimate PMFI/FR? Trees have visible, open scars because they are the trees that have had fires often enough to prevent healing. We found that the time for a fire scar to heal had a median of 38 years and was <100 years for 89% of scars (Baker and Dugan 2013). Longer fire intervals, that are real, have a high probability of not being selected by targeting trees with > 1 scar, because longer intervals often are expressed as no scars or one scar. Of course, long intervals can be an artifact of incomplete scarring, so that including all of them would lead to bias, but excluding all of them does too. Targeting trees with > 1 scar omits trees most likely to have long real fire intervals and selects trees with short fire intervals.

The substantial numerical dominance of unscarred and single-scarred trees in dry forests suggests omission of longer real fire intervals by individual-tree targeting of trees with > 1 scar could be among the most significant sources of bias in CFI estimates and possibly the main source of bias in ITFI estimates. In a sample of 906 pre-EuroAmerican trees we collected on 8 transects in northern Arizona, near Flagstaff and in Grand Canyon National Park, 779 trees had no scar (86%), 111 had one scar (12%), and only 16 trees had two or more scars (2%). In a mixed-conifer forest in the western Sierra, 98% of nearly 8,000 stumps and snags examined for scars did not have any scars, only 13 (0.2%) had one scar, and 48 (0.6%) had two or more scars (Fiegener 2002). Multi-scarred trees are rare in modern landscapes.

The magnitude of effects of omitting trees with no or one scar is unstudied, but within the set of multi-scarred trees with ≥2 scars, the effect of restricting fire history to increasing levels of multiple-scarring was studied (Fiegener 2002). To gauge how relevant this study is to multi-scarred sets of trees actually used in fire histories, I analyzed the number of scars per tree found by studies in the merged dataset, although data were available for only 324 cases. First, I calculated mean number of scarred trees, over each site's sample period, which is less than the total number of sample trees, since trees usually each cover only part of the sample period. Then, I calculated mean number of scars per scarred tree as total number of scars/mean number of scarred trees from the summary table for the FHX file in FHAES.

A histogram of mean scars per scarred tree had a mean of 8.47 scars/sample tree and a median of 7.61 scars/sample tree (Figure D). Fiegener (2002) found that restriction to ≥3 scars reduced ITFI from 17.4 years to 16.8 years (to 96.6%). This is above the minimum of 0.46 scars/tree in the distribution (Figure D). Restriction to ≥4 scars, just above the 1st quartile in the distribution, reduced ITFI to 15.8

Figure D. Histogram of the ratio of mean scars per sample tree in the 342-case merged dataset, and parameters of the distribution. Mean scars could not be calculated for 18 cases.



years (to 90.8%), restriction to ≥7 scars, just below the median, reduced ITFI to 14.6 years (to 83.9%), and restriction to ≥10 scars, below the 3rd quartile, reduced ITFI to 13.2 years (to 75.9%). These responses show that the more scars on a multi-scarred tree, the shorter is the mean ITFI estimate.

Roughly the median level of multi-scar targeting (≥7 scars), which reduced ITFI to 83.9%, or by 16.1%, closely matches the -16.64% bias in Weibull mean ITFI relative to PMFI/FR-total scarred trees/plots (Main text--Table 1). Other ITFI measures have biases of -2.71 to -29.71 (Main text-Table 1), so the close match with the Weibull Mean ITFI could possibly be a coincidence. Mean CFI-10% scarred also declined from 6.7 years to 5.7 years (85.1%) with restriction to ≥3 scars, but fluctuated or increased with higher levels of restriction (Fiegener 2002). Thus, the response of ITFI to targeting multi-scarred trees could explain much of why ITFI underestimates PMFI/FR, but the response of CFI estimators was inconsistent, suggesting it is possible too, but also may not be a main effect for CFI measures.

Mean CFI, ITFI, and FR estimates are further biased and shortened by non-random sampling of multi-scarred trees. Van Horne and Fulé (2006) found a statistical difference, using 95% confidence intervals, between mean CFI for an individual-tree targeted sample and a large census. Comparison of a random sample and a targeted sample, each of 40 trees, shows that mean CFI in the targeted sample was 79.1% (2.23/2.82) of the mean CFI in the random sample for all fires, 98.4% (3.00/3.05) for mean CFI-10%-scarred, and 86.9% (5.43/6.25) for mean CFI-25%-scarred. Farris et al. (2013) re-analyzed the Van Horne and Fulé (2006) dataset and added two other datasets, which together showed targeted samples had a mean CFI-all fires that was 78.9-112.5%, a mean CFI-10% scarred that was 93.5-131.4%, and a mean CFI-25% scarred that was 80.0-96.1%, of the corresponding mean CFI from a probabilistic sample. In Brown et al. (2008), a target-supplemented sample (their Figure 4d) had a mean CFI that was 88.9% (24/27) of that from a systematic plot sample (their Figure 4c). ITFI and FR estimates from recorders are similarly affected. Van Horne and Fulé (2006) found that mean ITFI in a targeted sample was 83.3% of mean ITFI in a random sample. Everett (2003) sampled fire-scarred trees using a grid at two sites and chose the closest fire-scarred tree, thus a probabilistic sample without targeting multi-scarred trees. No comparable estimate from non-random sampling and a targeted sample was made, but Everett's estimated ITFIs were in the 3rd and 4th quartiles of the distribution of estimated ITFIs in the 96-case calibration dataset, consistent with the possibility that ITFIs were long because of lack of targeting.

Farris et al. (2013) showed that individual-tree targeting and non-random sampling even led to ratio-based estimates of FR at three sites that were reduced to 85.5%, 88.3%, and 94.8% of FR estimates from equal-sized probabilistic samples. As suggested earlier, this may be because places with long fire intervals that are real are omitted. These omissions may be places that particular fires did not burn, thus fire size for those fire years is inflated, leading to FR estimates that are too short.

Another impact of individual-tree targeting is reduced completeness of the fire record and over-representation by small, low-severity fires. Fiegener (2002) found that targeting trees with ≥ 5 scars reduced detected fires from 76 to 68 (to about 89.5%), but reduced detection of larger fires to 77%, thus increasing the proportion of small fires in the sample. Baker and Ehle (2003) also found that targeting multi-scarred trees identified and emphasized more small, low-severity fires, including one-tree fires that are another central source of bias in CFI estimates (Baker and Ehle 2001). A non-targeted sample did as well or better at identifying large, low-severity and mixed-severity fires (Baker and Ehle 2003). Also, 18% of 60 total fires and 30% of the most ancient fires (pre-1700), including a significant high-severity fire, found in a non-targeted sample would have been missed if only trees with ≥4 scars were sampled (Baker and Ehle 2003). Targeting multi-scarred trees thus leads to an incomplete fire record, missing significant fires, and a bias toward small fires that produce CFI and ITFI estimates that are too short.

A related type of individual-tree targeting focuses only on "recorder" trees with at least one previous fire scar (thus ≥2 fire scars), which are thought to preferentially record fires, leading to a more complete fire record. To have increased the probability of receiving a subsequent scar, these trees had to have been effectively open, with a scar lacking bark, at the time of the next fire. Previously scarred trees do have a much higher probability of receiving a new scar than do unscarred trees (Baker and Dugan 2013). However, they are much less common than unscarred trees, and unscarred trees appear to typically be scarred at a sufficient rate in a fire to outnumber scars on recorder trees. For example, in a single fire, 73% of scarred trees were first scars and only 27% were recorders that had a previous scar (Stephens et al. 2010), suggesting previously scarred trees were poorer recorders of the fire, in terms of number of scars per unit area, even though scarred at a higher rate. In a larger Rocky Mountain National Park (RMNP) study (Baker and Ehle 2003), for 24 fires that showed up both as first scars and on recorders, 62% of the scars documenting these 24 fires were not on recorders, while 38% were on recorders, a significant difference ($\chi^2 = 4.76$, $p = 0.029$) and lower rate per unit area for recorders, just as in the Stephens et al. study. Moreover, we found 60 total fires, and 32% of these fires showed up only as first scars while 28% of the 60 fires showed up only on recorders, suggesting neither source alone provides a complete fire history. However, there was not much difference in the ability of more numerous unscarred and less numerous recorder trees to record complete histories of fire. Moreover, recorders have the same additional biases, as estimators of PMFI/FR, as do other multi-scarred trees, as reviewed above.

Targeting open-scarred trees often aims at trees with a cat-face or deep semicircular wound, which typically also means they are multi-scarred trees and qualify as recorders. In a study of a single fire in a California Sequoia grove, 68% of open-scarred trees were scarred in a 1797 fire, but only 20% of intact trees were scarred (Kilgore and Taylor 1979). Across many fires, in a California study, a significantly greater mean fraction (0.22) of oaks with open scars at the time of a fire had scars from the fire than did intact trees (0.09), but twice as many intact trees on the sites had scars since there were 5.5 times as many intact trees as trees with open scars (McClaran 1988). This pattern is similar to that of recorders. Mean CFI did not differ between open-scarred and intact trees at one site, but open-scarred trees had 27% fewer fire dates (McClaran 1988). Thus, targeting open-scarred trees thought to be better recorders of fires also leads to omission of fires and the other biases of multi-scarred trees.

Species targeting focuses on particular tree species thought to be better recorders of fire. For example, one might obtain fire scars from ponderosa pine trees on the edge of piñon-juniper woodlands, because the ponderosa are thought to have a better record, from a higher SF (e.g., Miller and Rose 1999). However, fires that burned the ponderosa likely did not penetrate into the woodlands much, if at all (Huffman et al. 2008), thus the apparent difference in SF may reflect real differences in burning rates. To avoid a targeting effect from assuming that the tree species with more scars has a more complete record, data can be acquired from piñon-juniper woodlands and adjusted for their lower scarring fraction. This is what the ATFI method allows, a separate SF for differing trees on the same site (Kou and Baker 2006a).

Individual-tree targeting of older trees for sampling occurs because older trees have a potentially longer record (Farris et al. 2013). This type of targeting may also occur if trees with multiple scars are targeted, since trees generally must get older before they have multiple scars. By definition, individual trees with long fire-scar records have a history of only low-severity fires at that tree, thus a targeted sample of only old trees is certain to indicate a long history of low-severity fire. When fire is moderate- to high-severity, evidence of fire severity on surviving older trees underestimates fire severity in the stand (Hessburg et al. 2007). A targeted sample of old trees in a landscape with trees of other ages thus provides strongly biased evidence about the fire severities that affected the stand.

Targeting sampling areas in landscapes

Targeting particular landscapes or parts of landscapes also leads to bias, generally toward CFI and ITFI estimates that are too short relative to PMFI/FR, since the methods of individual-tree targeting are also used at the landscape scale. Researchers seeking to reconstruct pre-EuroAmerican fire regimes may select parts of landscapes with concentrations of multi-scarred trees, recorders, open-scarred trees or catfaces, and old trees or old-growth forests. In almost 1/3 of the cases where targeting or lack of it was reported, researchers located plots specifically in these areas and in about 10% of cases researchers chose study areas with these concentrations (Table A). These plot locations and study areas may contain long fire records and many fire scars, and are attractive to researchers seeking long fire records (Farris et al. 2013). However, these parts of landscapes also are forests that had a predominance of low-severity fire and little to no mixed- or high-severity fire for hundreds of years, as most trees would otherwise be younger. As explained in the main text, researchers may target areas with abundant fire scars and omit or reduce sampling in areas that lack or have few scars, then also may inappropriately assume that fire history in areas with abundant fire scars also applies to areas with few or no fire scars.

In contrast, probabilistic sampling areas, particularly if appropriately small (e.g., 1 ha) may commonly lack scarred trees or have few. Heyerdahl (1997) sampled using plots located in a grid, thus without targeting sampling areas in landscapes, and found that scarred trees were lacking in more than half the plots at three study sites. These areas could in part have had few scars because of a low scarring fraction, but could also have been areas that really did not burn for a long period. If the latter, then omitting these long intervals, that are real, would bias results toward underestimating fire severity and bias estimated rates of low-severity fire toward shorter intervals. This kind of targeting is not clearly rejected by supporters of targeting (Farris et al. 2013 p. 1030), although they encourage "...clearly defining the inference space, not extrapolating to unrepresentative areas..." This kind of targeting did clearly include extrapolating to unrepresentative areas in past fire histories that are the subject of this paper. I am not singling out particular authors, as most used sampling methods that were common practice at the time, largely aimed at finding and sampling the best evidence (Farris et al. 2013).

However, targeting of old forests, that inherently have a history of low-severity fire, likely explains the unexpected findings of landscape analyses of fire history that did not use targeting. When an objective, large sample (303,156 ha) of historical dry forests was studied in the Pacific Northwest using early aerial photography, middle-aged forests resulting from mixed- and high-severity fires were found to have dominated historical landscapes and old, park-like forests, exclusively with low-severity fire, were found to have been comparatively uncommon (Hessburg et al. 2007 p. 7): "Moreover, old, park-like or similar ponderosa pine stand structures did not dominate the landscapes, and this was particularly perplexing because this was to be the signature outcome of frequent low severity fires." Similarly, spatially extensive reconstruction across landscapes using the early land surveys, found evidence of abundant denser and younger forests from mixed- and high-severity fire across dry forests in northern Arizona, the Colorado Front Range, and the Blue Mountains in Oregon, where previous fire-history studies had found predominantly low-severity fire and open, low-density old forests (Williams and Baker 2012). Finally, stand age data from the Forest Inventory and Analysis Program also showed that young and middle-aged forests, not park-like old forests, were most common historically in ponderosa pine and mixed-conifer forests across relatively undisturbed parts of the western USA (Odion et al. 2014). These studies with probabilistic sampling at the landscape scale show that targeting parts of landscapes containing old forests with abundant fire scars led to very substantial over-estimation of the historical extent of low-density old forests that predominantly had low-severity fire.

Unstudied fire severity in dry forests also inflates low-severity fire rates

Estimates from CFIs, ITFIs, and FRs likely often included all fire severities, not just low severity, and the low-severity rates alone are thus likely longer. Fire severity has been relatively infrequently studied in dry forests. Baker and Ehle (2003) found that only about 25% of fire-scar studies also collected the age-structure data needed to determine whether higher-severity fires occurred historically. Of the 335 cases in the merged dataset with data, 254 (74.3%) did not study fire severity, 80 (23.4%) did study fire severity, and 8 (2.3%) did not explain whether they studied fire severity. Most studies that did analyze fire severity did not distinguish fire severities when they reported fire rates (e.g., Taylor and Skinner 1998). Where fire severity was studied, some mixed- or high-severity fires were nearly always found, but few studies estimated PMFI or FR for the higher-severity fires. Thus, most fire-history studies provide estimates of rates for all fires combined, including low, moderate- and high-severity fires.

The potential effect of combined fire severities on estimated rates for low-severity fire can be illustrated by subtracting, using partitioning (Baker 2009), reported rates of moderate- to high-severity fire from rates of low-severity fire. Odion et al. (2014) reported historical rates of combined moderate- to high-severity fire ranged from 115-128 years in the eastern Cascades of Oregon to 319 years on the Mogollon Plateau. I used the full range of 115-319 years to remove the moderate- to high-severity component, and found that 10-year combined PMFI/FRs would have a 10.3-11.0 low-severity component, but 50-year combined PMFI/FRs would have a 59.3-88.5-year low-severity PMFI/FR after removing the moderate- to high-severity component (Table B). I did not apply an adjustment, for this fire-severity issue, to estimated PMFI/FRs because the adjustments are imprecise and have a large range, and because sites where fire severity was unstudied did not necessarily have higher-severity fires. Nonetheless, this finding illustrates the limitation of unstudied fire severity, and shows that many estimates of low-severity PMFI/FR are likely low estimates.

Table B. Partitioning combined fire rotations (FR) into components for low- versus moderate and high-severity fire for three example levels of combined fire rotations.

	10-year combined-severity PMFI/FR	25-year combined-severity PMFI/FR	50-year combined-severity PMFI/FR
a. Combined annual probability of fire (1/FR)	0.10000	0.04000	0.02000
UPPER LIMIT OF LOW-SEVERITY RANGE			
b. Annual probability of fire for moderate-high component of 115 years†	0.00870	0.00870	0.00870
c. Annual probability of fire for low-severity component, from a - b.	0.09130	0.03130	0.01130
d. Net fire rotation for low-severity component, from 1 / c.	10.95 years	31.95 years	88.50 years
LOWER LIMIT OF LOW-SEVERITY RANGE			

e. Annual probability of fire for moderate-high component of 319 years†	0.00313	0.00313	0.00313
f. Annual probability of fire for low-severity component, from a - e.	0.09687	0.03687	0.01687
g. Net fire rotation for low-severity component, from 1 / g.	10.32 years	27.12 years	59.28 years
NET ESTIMATED LOW-SEVERITY RANGE	10.32-10.95 years	27.12-31.95 years	59.28-88.50 years

Notes

† The 115-319 year range for moderate- to high-severity fire rotation in dry forests is from Odion et al. (2014)

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Human-started wildfires expand the fire niche across the United States

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The economic and ecological costs of wildfire in the United States have risen substantially in recent decades. Although climate change has likely enabled a portion of the increase in wildfire activity, the direct role of people in increasing wildfire activity has been largely overlooked. We evaluate over 1.5 million government records of wildfires that had to be extinguished or managed by state or federal agencies from 1992 to 2012, and examined geographic and seasonal extents of human-ignited wildfires relative to lightning-ignited wildfires. Humans have vastly expanded the spatial and seasonal "fire niche" in the coterminous United States, accounting for 84% of all wildfires and 44% of total area burned. During the 21-y time period, the human-caused fire season was three times longer than the lightning-caused fire season and added an average of 40,000 wildfires per year across the United States. Human-started wildfires disproportionally occurred where fuel moisture was higher than lightning-started fires, thereby helping expand the geographic and seasonal niche of wildfire. Human-started wildfires were dominant (>80% of ignitions) in over 5.1 million km^2, the vast majority of the United States, whereas lightning-started fires were dominant in only 0.7 million km^2, primarily in sparsely populated areas of the mountainous western United States. Ignitions caused by human activities are a substantial driver of overall fire risk to ecosystems and economies. Actions to raise awareness and increase management in regions prone to human-started wildfires should be a focus of United States policy to reduce fire risk and associated hazards.

anthropogenic wildfires | fire starts | ignitions | modern fire regimes | wildfire causes

The United States has experienced some of the largest wildfire years this decade, with over 36,000 km^2 burned in 2006, 2007, 2012, and 2015 (1). There is national and global concern over how fire regimes have changed in the past few decades and how they will change in the future (2–4). In the western United States, there is strong evidence that regional warming and drying, including that directly attributed to anthropogenic climate change, are linked to increased fire frequency and size and longer fire seasons (5–9). However, the role that humans play in starting these fires and the direct role of human-ignitions on recent increases in wildfire activity have been overlooked in public and scientific discourse because of the difficulty in ascribing a cause, either human- or lightning-started (10). Humans primarily alter fire regimes in three ways: changing the distribution and density of ignitions, shifting the seasonality of burning, or altering available fuels (2, 3). Geographic variability in regional and continental-scale fire activity in the United States is strongly tied to proxies for these human-caused changes, including population and road density, and different land-use and development patterns (10–15). Although changing climate and fuels also influence fire regimes across the United States (10, 16, 17), there can be no fire without an ignition source. Here, we explore the role that human-started wildfires play in modern United States fire regimes.

Ignitions are often presumed to be saturated (18, 19), and therefore have limited ability to predict fire activity. However, several studies suggest that humans play an important role in redistributing ignitions (20–22), particularly where lightning rarely occurs or where lightning is not concurrent with dry conditions (23). The human–fire connection in the modern era appears strongest at intermediate levels of development, as fires become less likely in the landscape beyond a certain population density, level of urbanization, or dependence on fossil fuels (11, 13, 24). Overall, humans expand the spatial and temporal "fire niche" by introducing ignitions into landscapes when fuels are sufficiently dry enough to ignite and carry fire, but when lightning is rare. Human ignitions are therefore a critical force acting to expand how the fire niche is realized across United States ecoregions.

National-scale analysis of human alteration of the fire niche is critical given that the annual expense of fighting wildfires has exceeded $2 billion in recent years, and the accrued direct and indirect impacts of wildfire on infrastructure and communities could be 30 times that amount (25). Policies that govern wildfire management and response are also directed at the national level, demanding analysis at a national scale (10, 22, 26). Although recent human influence on fire regimes has been studied at local (13) to regional scales (14), human influence nationally remains poorly understood (10). National policies can strongly influence fire regimes (27) and, with sufficient information on human ignitions, policy directives could target human behavior in ways that remediate increasing trends in wildfire risk.

Here, we ask how human ignitions have altered the spatial extents, seasonality, and temporal trends in wildfire across the coterminous United States. We analyze over 1.5 million records of both human- and lightning-started fires in the United States from

Significance

Fighting wildfires in the United States costs billions of dollars annually. Public dialog and ongoing research have focused on increasing wildfire risk because of climate warming, overlooking the direct role that people play in igniting wildfires and increasing fire activity. Our analysis of two decades of government agency wildfire records highlights the fundamental role of human ignitions. Human-started wildfires accounted for 84% of all wildfires, tripled the length of the fire season, dominated an area seven times greater than that affected by lightning fires, and were responsible for nearly half of all area burned. National and regional policy efforts to mitigate wildfire-related hazards would benefit from focusing on reducing the human expansion of the fire niche.

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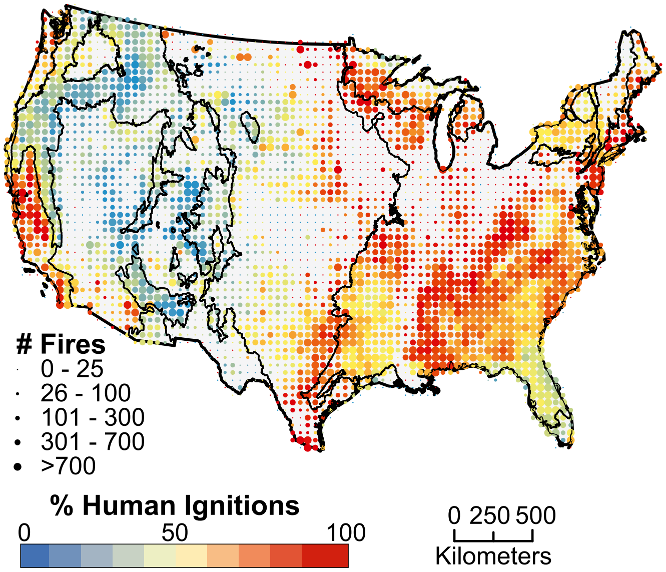

Fig. 1. The total number of wildfires (dot size) and the proportion started by humans (dot color: red indicating greater number of human started fires) within each 50 km × 50-km grid cell across the coterminous United States from 1992 to 2012. Black lines are ecoregion boundaries, as defined in the text.

1992 to 2012 (28). All of these wildfires necessitated an agency response to manage or suppress them, and therefore posed a threat to ecosystems or infrastructure; this record does not include intentionally set prescribed burns or managed agricultural fires. To our knowledge, this is the most comprehensive assessment of the role of human-started wildfires across the United States over the past two decades. We compare: (*i*) the spatial extents of human- vs. lightning-started wildfires, (*ii*) the seasonality of human vs. lightning wildfires, (*iii*) the climate niche for human- vs. lightning-started wildfires, and (*iv*) 21-y trends in large human vs. lightning wildfires. Our analysis documents the pronounced expansion of wildfire extent, seasonality of wildfires, and increasing numbers of large wildfires through time as a result of human-related ignitions across the United States.

Human-Related Ignitions Vastly Expanded the Extent of Wildfire

Human-started wildfires represented 84% of the 1.5 million wildfires included in this analysis (n = 245,446 lightning-started fires; n = 1,272,076 human-started wildfires). The eastern United States and western coastal areas were dominated by human-started wildfires, whereas lightning-started fires dominated the mountainous regions of the western United States (Fig. 1, Table 1 and Table S1). Here we define a fire regime as dominated by either human or lighting ignitions when one cause accounts for more than 80% of the number of fires in a given 50 × 50-km grid cell. Based on this definition, 5.1 million km^2, or 60% of the total land area of the coterminous United States, was dominated by human-started wildfires, whereas only 0.7 million km^2, or 8% of the area, was dominated by lightning-started fires. In addition to expanding the numbers of fires, humans also expanded the total area burned. Human-started wildfires burned a total of 160,274 km^2, or ~44% of the total area burned from 1992 to 2012 (Table 1).

Human-Related Ignitions More Than Tripled the Length of the Wildfire Season

Human ignitions dramatically expanded the wildfire season in the United States, particularly during spring. The length of the human-started wildfire season [defined as the interquartile range (IQR) of human-ignited fires] was 154 d, more than triple that of the lightning wildfire season (IQR = 46 d) (Fig. 2 and Table 1). This national-scale expansion is driven by earlier (spring) human-started fires in eastern ecoregions coupled with later (late summer or fall) human-started fires in western ecoregions (Table S2). The median discovery date for human-started fires was over 2-mo (May 20th) earlier than lightning-started fires (July 25th). Summed across the 21-y record, the most common day for human-started fires by far was July 4th, US Independence Day, with 7,762 fires starting that day over the course of the record (Fig. 2), whereas, the most common day for lightning-started fires was July 22nd. Of all lightning-ignited fires, 78% occurred in the summer (June–August), 9% in the spring (March–May), and 12% in the fall (September–November). In contrast, human-ignited wildfires were more evenly distributed throughout the year, with 24% in summer, 38% in spring, 19% in fall, and 19% in winter. This pronounced expansion of the wildfire season was also evident spatially (Fig. 3), with human-ignited wildfires occurring predominantly in spring in the eastern United States and in the fall and winter in Texas and the Gulf states. See Table S1 for state-level analysis. When lightning-started fires were rare (<5% and >95% quantile; i.e., before May 13th or after September 16th), humans ignited 842,289 wildfires, effectively increasing the number of wildfires 35-fold compared with the 24,081 lightning-ignited wildfires during these spring, fall, and winter seasons.

Table 1. The number of wildfires, total burned area (ha), and fire season length (IQR, in days), by ecoregion (ordered by percent human-caused fires) and within the coterminous United States from 1992 to 2012

	No. of fires			Area burned (ha)			Length (IQR, days)		
Ecoregion	Human	Light	Human caused (%)	Human	Light	Human caused (%)	Human	Light	Human expansion (%)
MC	87,274	2,855	97	2,143,282	253,210	89	85	45	189
NF	61,673	2,574	96	302,561	82,721	79	51	79	N/A
ETF	815,499	44,859	95	3,827,045	829,293	82	167	66	253
MWCF	14,586	925	94	19,251	27,291	41	67	52	129
GP	134,944	17,586	88	3,992,557	2,564,955	61	148	47	315
SSH	7,504	2,167	78	340,873	254,418	57	55	41	134
TWF	4,832	1,917	72	357,150	350,477	50	98	52	188
NAD	55,422	52,044	52	2,394,677	8,880,691	21	92	40	230
NFM	76,735	94,017	45	1,895,622	5,731,733	25	75	36	208
TS	13,607	26,502	34	754,393	1,152,064	40	85	39	218
CONUS	1,272,076	245,446	84	16,027,412	20,126,852	44	154	46	335

CONUS, Coterminous United States; ETF, Eastern Temperate Forests; GP, Great Plains; MC, Mediterranean California; MWCF, Marine West Coast Forests; NAD, North American Desert; NF, Northern Forests; NFM, Northwest Forested Mountains; SSH, Southern Semiarid Highlands; TWF, Tropical Wet Forests; TS, Temperate Sierras.

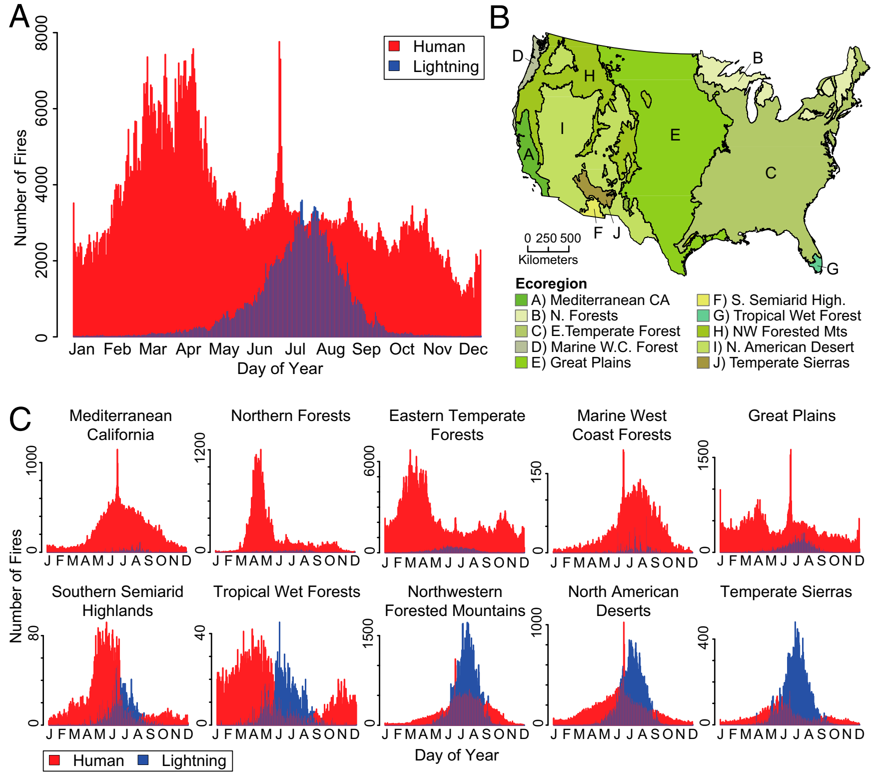

Fig. 2. Frequency distributions of human and lightning-caused wildfires by Julian day of year. (*A*) Frequency distribution of wildfires across the coterminous United States from 1992 to 2012 (n = 1.5 million); (*B*) map of United States ecoregions; (*C*) frequency distributions of wildfires by ecoregions, ordered by decreasing human dominance.

Human-Driven Expansion of the Fire Niche

Humans greatly expanded the natural fire niche (Fig. 4), which we calculated as the co-occurrence of the average monthly lightning density and 1,000-h dead fuel moisture. Regions and seasons of moderate to high lightning-started fire density (>0.4 fires per 1,000 km^2 per month) had a median lightning-strike density of 0.19 (IQR: 0.065–0.57) strikes per square kilometer per month and a median 1,000-h fuel moisture of 11.9% (IQR: 9.25–15.6%) (Fig. 4*A*). In contrast, regions and seasons of moderate to high human-started fire density (>0.4 fires per 1,000 km^2 per month) had a median lightning-strike density of only 0.11 (IQR: 0.025–0.39) strikes per square kilometer per month and a median 1,000-h fuel moisture of 17.8% (IQR: 15.95–19.25%) (Fig. 4*B*). The median fuel moisture and lightning conditions when human-started wildfires occurred were significantly different from those values for lightning-started fires (P < 0.0001). Areas and months of moderate to high human-caused fire density had approximately 40% fewer lightning strikes, and nearly 50% higher fuel moisture levels (based on median values) than for moderate to high lightning-caused fire density. Additional exploration of the fire niche for human-started and lightning-started fires relative to lightning density, fuel moisture, and net primary production (NPP), a proxy for fuels, is provided in Figs. S1 and S2.

Increasing Trends in Large Human-Started Wildfires

During the 21-y time period, there were significant increasing trends in large wildfires ignited by both lightning (n = 4,312; Theil-Sen estimated slope = 12.2; P = 0.001) and humans (n = 4,143; Theil-Sen estimated slope = 3.6; P = 0.004) (Fig. S3). There was a strong dichotomy in human vs. lightning trends seasonally (Fig. 5). Overall trends in lightning-caused fires were primarily driven by increasing numbers of large summer fires (Fig. 5*B*), whereas overall trends in human-caused fires were primarily driven by increasing numbers of large spring fires (Fig. 5*D*). Spatially, lightning-caused fires increased the most in the Northwest Forested Mountains ecoregion (Fig. S4*A*), whereas human-caused wildfires increased the most in the Great Plains ecoregion (Fig. S4*B*).

Discussion

Humans, the keystone fire species (29), play a primary role in spatially and temporally redistributing ignitions and resulting wildfires. We document that over 84% of the government-recorded

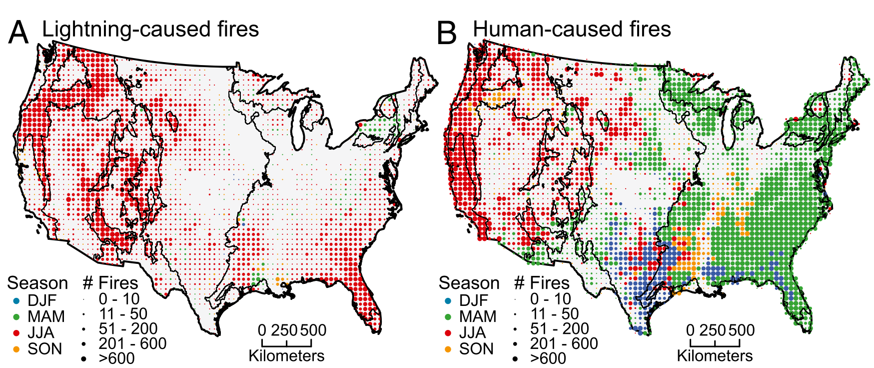

Fig. 3. Comparison of seasonality for (*A*) lightning- vs. (*B*) human-ignited wildfires. Human ignitions expand the seasonal fire niche considerably into spring and fall months. Colors show the season with the maximum ignitions caused by lightning and human within each 50 km × 50-km grid cell. Size of dot indicates the number of unique lightning and human fires between 1992 and 2012. Ecoregion boundaries are overlaid for visualization.

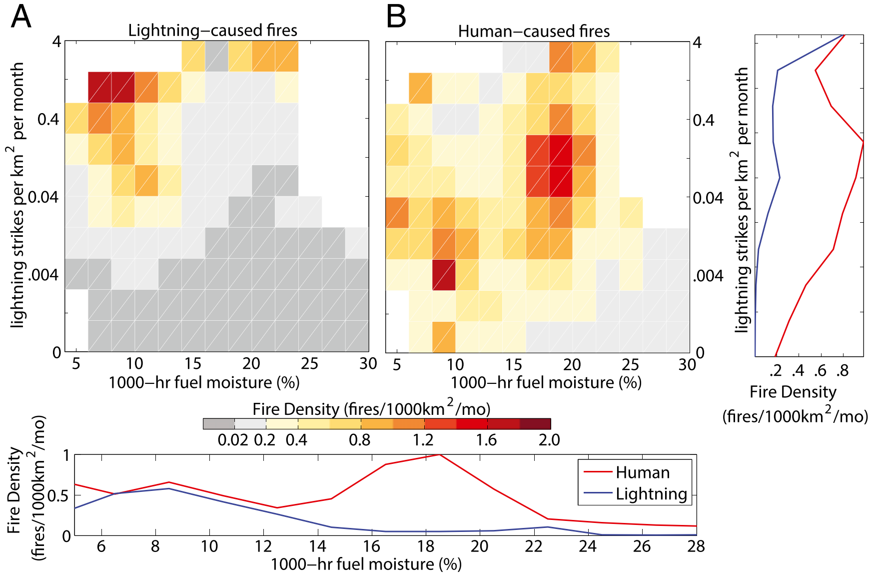

Fig. 4. Human vs. lightning fire niche relative to fuel moisture and lightning density, with greatest resulting wildfire density represented by dark red. (*A*) Lightning-started fires occur in areas with high lightning-strike density and dry fuels. (*B*) Human-started wildfires expand the fire niche to include areas with low lightning-strike density as well as areas with higher fuel moisture. Graphs on the bottom and far right show histograms of 1,000-h dead fuel moisture and lightning strikes, respectively, for human- and lightning-started fires.

wildfires were started by people from 1992 to 2012. Sixty percent of the total land area of the coterminous United States was dominated by human-started wildfires, whereas only 8% of the area was dominated by lightning fires. Humans tripled the length of the wildfire season, extending burning into the spring, fall, and winter months. During the spring, fall, and winter, people added more than 840,000 wildfires, a 35-fold increase over the number of lightning-started fires in those seasons. This expansion of the fire-niche was caused by human-related ignitions under higher fuel moisture conditions, compared with lightning-started fires. Moreover, during this 21-y record, large human-started wildfires increased significantly.

There was a strong national east–west dichotomy in the spatial distribution of human-started wildfires. Although human-started wildfires were pervasive across the United States (Fig. 1), the expansion of human-started wildfires relative to lightning-started fires was most dramatic in the eastern United States and central and southern California (Figs. 1 and 2*C*). Recent work for California confirms the important role of humans, with anthropogenic variables explaining half of the variability in fire probability over the past four decades (30). In contrast, lightning-started fires were found primarily in the intermountain west and Florida and occurred predominantly in the summer, reflecting national lightning strike patterns (31) (Fig. 2*C*). This finding supports other studies of human vs. lightning ignition sources that have found an important distinction between eastern and western United States fire patterns (10, 21) and drivers (32). Some explanations for this distinction include higher population and housing densities, lower proportions of public land, and more extensive land use and development in the eastern United States (33, 34), all of which could lead to more sources of anthropogenic ignitions. Synchrony between lightning activity and the seasonal nadir of fuel moisture in the western United States also likely contributes to these geographic differences. However, even with a projected increase in the number of lightning strikes as a result of anthropogenic climate change (50% by 2100) (35), humans would still remain the dominant ignition source across the majority of the United States land area. The majority of the wildfires requiring agency suppression in the east can be attributed to escaped fires from debris burning occurring in the spring months (or winter in Texas and the Gulf Coast) (Fig. 3). Between 1992 and 2012, wildfires caused by debris burning tended to be small (median

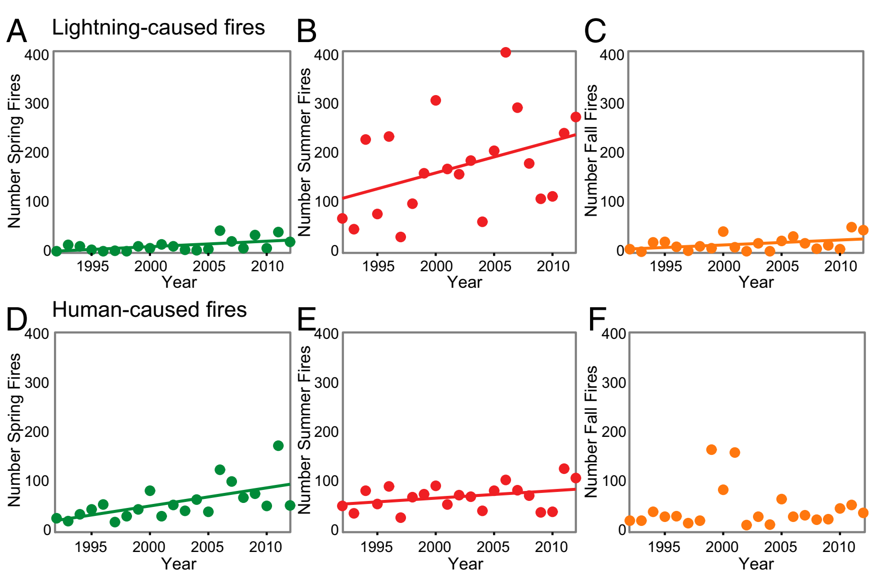

Fig. 5. Trends in the number of large wildfires verified by MTBS records from 1992 to 2012 for lightning-started fires (*A*–*C*) vs. human-started fires (*D*–*F*) in the spring (green: *A* and *D*), summer (red: *B* and *E*), and fall (orange: *C* and *F*). Where trend lines are shown, Theil-Sen estimated slopes are significantly different from zero ($P < 0.05$).

fire size 0.4 ha, IQR: 0.14–1.62 ha), but still an important source of risk to surrounding ecosystems. At finer scales, there are also notable patterns in human- vs. lightning-started wildfires (Fig. S5). Increased wildfires can follow road networks (36), the wildland–urban interface (13), and boundaries between agricultural and forested areas (37), highlighting just a few examples of how human activities and cultural drivers provide ignitions that substantially change the distribution of fire across the United States (38).

Our findings reinforce the strong imprint of people on fire regimes through changes in wildfire seasonality, which has been documented globally (39). In the past few decades, early onset of warmer and drier conditions has promoted greater fire activity across the western United States (6, 7, 40). However, our study highlights the equally important role of human ignitions in changing modern fire regimes by increasing the fire season length to encompass the entire year. The vast majority (78%) of lightning-started fires occurred during the summer months, whereas 76% of human-started fires occurred during the spring, fall, and winter months. Moreover, this trend varies substantially by ecoregion, reflecting again the principle dichotomy between the eastern and western United States (Fig. 3). Human-started fires extend the fire season earlier in the east, and later in the west (Fig. 3 and Table S2). Observations suggest that climate change has extended the duration of the fire weather season across most of the globe, including parts of the United States by a couple of weeks over the past three decades (5, 9), whereas we show that human ignitions in the United States increased the length of the fire season by more than three mo. There was also a notable mark of American culture on the distribution of wildfires, with the peak day of wildfires occurring on July 4th, concurrent with Independence Day fireworks displays (Fig. 2). Indeed, Americans start over twice as many wildfires on July 4th as any other summer day. A similar cultural mark has also been demonstrated globally with a marked decline in wildfires on Sunday compared with other weekdays (41).

Thus, at the national scale, human ignitions dramatically expand the spatial and seasonal niche of fire. The key components that define the fire niche are ignition sources, fuel mass, and desiccation. By exploring the fire niche along these axes, our results show that lightning fires are primarily constrained to areas with a lightning-strike density of greater than 100 strikes per grid cell per month (0.04 strikes/km^2 per month) and are concurrent with drier fuels ($<$ 15% fuel moisture) (Fig. 4). Human ignitions expand fires into regions with higher fuel moisture (Fig. 4) and higher NPP (Figs. S1 and S2), suggesting that humans create sufficient ignition pressure for wetter fuels to burn. As a consequence, human ignitions have expanded the fire niche into areas with historically low lightning-strike density, such as Mediterranean California, or low concurrence of lightning and dry conditions, such as Eastern Temperate Forests (Fig. 1).

Over the past two decades, there was a significant increase across the United States for both human- and lightning-caused large fires (Fig. S3). The significant increase in large lightning fires is driven primarily by fires in summer months (Fig. 5) in the Northwest Forested Mountains ecoregion of the western United States (Fig. S4). This finding is consistent with other studies that have demonstrated an increase in large fires across the western United States (6, 7, 40), likely as a consequence of changes in climate and fuels rather than ignitions. In contrast, the significant trend in human-caused fires is primarily driven by an increase in large fires during spring months (Fig. 5) in the Great Plains ecoregion of the United States (Fig. S4). This increasing trend suggests that earlier springs as a result of climate change (42, 43) may be interacting with human ignition sources to increase the risk of large fires in the central United States.

The strong year-to-year variability in human ignitions (Fig. S3 and S4) may reflect the degree to which human choices can affect fire regimes. However, interannual climate variability also influences fuel moisture, NPP, and short-term weather conditions that enable the spread of human-ignited wildfires (44). There was a significant temporal correlation between large human- and lightning-started fires (R = 0.75). This pattern has been observed previously in the western United States (23) and suggests that large-scale climate drivers affect the frequency of both human- and lightning-caused fires. It is unknown how human actions will be affected by hotter and drier conditions, potentially increasing or decreasing ignitions from land use, recreation, and other sources. Increased public awareness and focused policy and management, particularly in years with elevated fire risk associated with climatic anomalies, are needed to reduce the number of human-caused ignitions.

In conclusion, we demonstrate the remarkable influence that humans have on modern United States wildfire regimes through changes in the spatial and seasonal distribution of ignitions. Although considerable fire research in the United States has rightly focused on increased fire activity (e.g., larger fires and more area burned) because of climate change, we demonstrate that the expanded fire niche as a result of human-related ignitions is equally profound. Moreover, the convergence of warming trends and expanded ignition pressure from people is increasing the number of large human-caused wildfires (Fig. 5). Currently, humans are extending the fire niche into conditions that are less conducive to fire activity, including regions and seasons with wetter fuels and higher biomass (Figs. 3 and 4). Land-use practices, such as clearing and logging, may also be creating an abundance of drier fuels, potentially leading to larger fires even under historically wetter conditions. Additionally, projected climate warming is expected to lower fuel moisture and create more frequent weather conditions conducive to fire ignition and spread (45), and earlier springs attributed to climate change are leading to accelerated phenology (42). Although plant physiological responses to rising CO_2 may reduce some drought stress (46), climate change will likely lead to faster desiccation of fuels and increased risk in areas where human ignitions are prevalent.

Uncertainty remains regarding how anthropogenic climate change will alter wildfire activity geographically and seasonally (47, 48), particularly in areas where human-caused fires dominate. Moreover, the current wildland–urban interface, where houses intermingle with natural areas, constitutes 9% of the United States total land area (33) but is projected to double by 2030, predominantly in the intermountain West (49). This expected development expansion will increase not only ignition pressure, but also the vulnerability of new infrastructure. Human-driven expansion of the spatial and temporal distribution of ignitions makes national- and regional-scale policy interventions and increased public awareness critical for reducing national wildfire risk.

Materials and Methods

For this analysis, we used the publically available US Forest Service Fire Program Analysis-Fire Occurrence Database (FPA-FOD) (28). This comprehensive dataset includes United States federal, state, and local records of wildfires (both on public and private lands) that were suppressed from 1992 to 2012, a total of ∼1.6 million records. Previous studies have focused on the western United States (20), federal lands (22), or records from just one agency (21). Each entry includes at minimum the location, discovery date, and cause of the wildfire. We excluded 114,191 wildfires with an unknown cause and analyzed the spatial, seasonal, and temporal patterns of human- vs. lightning-started wildfires. In total, 1,517,522 wildfires were included in the analysis. Human-started wildfires were caused by a variety of sources, including the US Forest Service-designated categories of equipment use, smoking, campfire, railroad, arson, debris burning, children, fireworks, power line, structure, and miscellaneous fires (28). Spatially, we calculated the proportion of human- vs. lightning-caused wildfires within equal-area 50 × 50-km grid cells across the coterminous United States. This grid size corresponds roughly to the size of an average United States county. For each grid cell, we calculated the season (winter, spring, summer, or fall) when the majority of human-caused and lightning-caused wildfires were started. All spatial analyses were conducted in the Albers-Conical equal-area projection. To determine the seasonal distribution of wildfires, we plotted the distribution of human- and lightning-started fires by the day of year for the coterminous United States and for individual ecoregions. We used the level 1 ecological regions of North America, developed by the Commission for Environmental Cooperation (50). We calculated the length of the human- and lightning-caused fire seasons as the IQR of the Julian day of recorded fire ignition: that is, the difference between the first and third quartiles.

We determined how humans expanded the fire niche by comparing the lightning-strike density (i.e., natural ignition pressure) and fuel-moisture conditions under which actual human- and lightning-started fire events occurred. We obtained daily 1,000-h dead fuel moisture data from the surface meteorological data (51) on a 4-km grid from 1992 to 2012, and computed monthly averages across the 21-y study period. We obtained 4-km gridded monthly lightning-strike data from the Vaisala National Lightning Detection Network (https://www.ncdc.noaa.gov/data-access/severe-weather/lightning-products-and-services) and averaged the data over the 21-y study period. To account for fuel limitations, we also explored the fire niche as a function of fuel amount (approximated by NPP). We used MODIS mean annual NPP data (1-km resolution, from 2002 to 2015) (52) for this purpose. These three datasets were aggregated to the common 50×50-km grid cell. We calculated the number of human- and lightning-started fires by grid cell using the FPA-FOD dataset (28). We excluded any grid cells from subsequent analyses that did not report at least one lightning-caused or human-caused wildfire over the period of record. We tested whether fire niche expansion (as determined by fuel moisture and lightning-strike density) caused by human ignitions was significant based on Mann–Whitney tests between human- vs. lightning-started fires.

To assess trends in human- vs. lightning-caused wildfires through time, we used only large fires that were independently verified by the Monitoring Trends in Burn Severity (MTBS) project (53). We specifically focused on these large fires (>400 ha in the west, >200 ha in the east; $n =$ 8,455) for comparability with previous research, which has examined temporal trends in the western United States and the link to climate warming (6, 7, 40), but has not investigated the relative contribution of human-started fires at a national scale. In addition to overall temporal trends, we tested for significant trends by ignition source versus season (spring, summer, fall) and versus ecoregion based on the level I ecological regions of North America (50). We explored a similar analysis using all available FPA-FOD data, but changes in reporting frequency through time for some states precluded a robust temporal analysis. We tested for trends in wildfire numbers through time using the nonparametric Theil-Sen estimator (54) and tested for trend significance using nonparametric Mann–Kendall tests (55).

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Adapting to Climate Change on Western Public Lands: Addressing the Ecological Effects of Domestic, Wild, and Feral Ungulates

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Abstract Climate change affects public land ecosystems and services throughout the American West and these effects are projected to intensify. Even if greenhouse gas emissions are reduced, adaptation strategies for public lands are needed to reduce anthropogenic stressors of terrestrial and aquatic ecosystems and to help native species and ecosystems survive in an altered environment. Historical and contemporary livestock production—the most widespread and long-running commercial use of public lands—can alter vegetation, soils, hydrology, and wildlife species composition and abundances in ways that exacerbate the effects of climate change on these resources. Excess abundance of native ungulates (e.g., deer or elk) and feral horses and burros add to these impacts. Although many of these consequences have been studied for decades, the ongoing and impending effects of ungulates in a changing climate require new management strategies for limiting their threats to the long-term supply of ecosystem services on public lands. Removing or reducing livestock across large areas of public land would alleviate a widely recognized and long-term stressor and make these lands less susceptible to the effects of climate change. Where livestock use continues, or where significant densities of wild or feral ungulates occur, management should carefully document the ecological, social, and economic consequences (both costs and benefits) to better ensure management that minimizes ungulate impacts to plant and animal communities, soils, and water resources. Reestablishing apex predators in large, contiguous areas of public land may help mitigate any adverse ecological effects of wild ungulates.

Keywords Ungulates · Climate change · Ecosystems · Public lands · Biodiversity · Restoration

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Introduction

During the 20th century, the average global surface temperature increased at a rate greater than in any of the previous nine centuries; future increases in the United States (US) are likely to exceed the global average (IPCC 2007a; Karl and others 2009). In the western US, where most public lands are found, climate change is predicted to intensify even if greenhouse gas emissions are reduced dramatically (IPCC 2007b). Climate-related changes can not only affect public-land ecosystems directly, but may exacerbate the aggregate effects of non-climatic stressors, such as habitat modification and pollution caused by logging, mining, grazing, roads, water diversions, and recreation (Root and others 2003; CEQ 2010; Barnosky and others 2012).

One effective means of ameliorating the effects of climate change on ecosystems is to reduce environmental stressors under management control, such as land and water uses (Julius and others 2008; Heller and Zavaleta 2009; Prato 2011). Public lands in the American West provide important opportunities to implement such a strategy for three reasons: (1) despite a history of degradation, public lands still offer the best available opportunities for ecosystem restoration (CWWR 1996; FS and BLM 1997; Karr 2004); (2) two-thirds of the runoff in the West originates on public lands (Coggins and others 2007); and (3) ecosystem protection and restoration are consistent with laws governing public lands. To be effective, restoration measures should address management practices that prevent public lands from providing the full array of ecosystem services and/or are likely to accentuate the effects of climate change (Hunter and others 2010). Although federal land managers have recently begun considering how to adapt to and mitigate potential climate-related impacts (e.g., GAO 2007; Furniss and others 2009; CEQ 2010; Peterson and others 2011), they have not addressed the combined effects of climate change and ungulates (hooved mammals) on ecosystems.

Climate change and ungulates, singly and in concert, influence ecosystems at the most fundamental levels by affecting soils and hydrologic processes. These effects, in turn, influence many other ecosystem components and processes—nutrient and energy cycles; reproduction, survival, and abundance of terrestrial and aquatic species; and community structure and composition. Moreover, by altering so many factors crucial to ecosystem functioning, the combined effects of a changing climate and ungulate use can affect biodiversity at scales ranging from species to ecosystems (FS 2007) and limit the capability of large areas to supply ecosystem services (Christensen and others 1996; MEA 2005b).

In this paper, we explore the likely ecological consequences of climate change and ungulate use, individually and in combination, on public lands in the American West. Three general categories of large herbivores are considered: livestock (largely cattle [*Bos taurus*] and sheep [*Ovis aries*]), native ungulates (deer [*Odocoileus* spp.] and elk [*Cervus* spp.]), and feral ungulates (horses [*Equus caballus*] and burros [*E. asinus*]). Based on this assessment, we propose first-order recommendations to decrease these consequences by reducing ungulate effects that can be directly managed.

Climate Change in the Western US

Anticipated changes in atmospheric carbon dioxide (CO_2), temperature, and precipitation (IPCC 2007a) are likely to have major repercussions for upland plant communities in western ecosystems (e.g., Backlund and others 2008), eventually affecting the distribution of major vegetation types. Deserts in the southwestern US, for example, will expand to the north and east, and in elevation (Karl and others 2009). Studies in southeastern Arizona have already attributed dramatic shifts in species composition and plant and animal populations to climate-driven changes (Brown and others 1997). Thus, climate-induced changes are already accelerating the ongoing loss of biodiversity in the American West (Thomas and others 2004).

Future decreases in soil moisture and vegetative cover due to elevated temperatures will reduce soil stability (Karl and others 2009). Wind erosion is likely to increase dramatically in some ecosystems such as the Colorado Plateau (Munson and others 2011) because biological soil crusts—a complex mosaic of algae, lichens, mosses, microfungi, cyanobacteria, and other bacteria—may be less drought tolerant than many desert vascular plant species (Belnap and others 2006). Higher air temperatures may also lead to elevated surface-level concentrations of ozone (Karl and others 2009), which can reduce the capacity of vegetation to grow under elevated CO_2 levels and sequester carbon (Karnosky and others 2003).

Air temperature increases and altered precipitation regimes will affect wildfire behavior and interact with insect outbreaks (Joyce and others 2009). In recent decades, climate change appears to have increased the length of the fire season and the area annually burned in some western forest types (Westerling and others 2006; ITF 2011). Climate induced increases in wildfire occurrence may aggravate the expansion of cheatgrass (*Bromus tectorum*), an exotic annual that has invaded millions of hectares of sagebrush (*Artemisia* spp.) steppe, a widespread yet threatened ecosystem. In turn, elevated wildfire occurrence facilitates the conversion of sagebrush and other native shrub-perennial grass communities to those dominated by alien grasses (D'Antonio and Vitousek 1992; Brooks 2008), resulting in habitat loss for imperiled greater sage-grouse (*Centrocercus urophasianus*) and other sagebrush-dependent species (Welch 2005). The US Fish and Wildlife Service (FWS 2010) recently concluded climate change effects can exacerbate many of the multiple threats to sagebrush habitats, including wildfire, invasive plants, and heavy ungulate use. In addition, the combined effects of increased air temperatures, more frequent fires, and elevated CO_2 levels apparently provide some invasive species with a competitive advantage (Karl and others 2009).

By the mid-21st century, Bates and others (2008) indicate that warming in western mountains is very likely to cause large decreases in snowpack, earlier snowmelt, more winter rain events, increased peak winter flows and flooding, and reduced summer flows. Annual runoff is predicted to decrease by 10–30 % in mid-latitude western North America by 2050 (Milly and others 2005) and up to 40 % in Arizona (Milly and others 2008; ITF 2011). Drought periods are expected to become more frequent and longer throughout the West (Bates and others 2008). Summertime decreases in streamflow (Luce and Holden 2009) and increased water temperatures already have been documented for some western rivers (Kaushal and others 2010; Isaak and others 2012).

Snowmelt supplies about 60–80 % of the water in major western river basins (the Columbia, Missouri, and Colorado Rivers) and is the primary water supply for about 70 million people (Pederson and others 2011). Contemporary and future declines in snow accumulations and runoff (Mote and others 2005; Pederson and others 2011) are an important concern because current water supplies, particularly during low-flow periods, are already inadequate to satisfy demands over much of the western US (Piechota and others 2004; Bates and others 2008).

High water temperatures, acknowledged as one of the most prevalent water quality problems in the West, will likely be further elevated and may render one-third of the current coldwater fish habitat in the Pacific Northwest unsuitable by this century's end (Karl and others 2009). Resulting impacts on salmonids include increases in virulence of disease, loss of suitable habitat, and mortality as well as increased competition and predation by warmwater species (EPA 1999). Increased water temperatures and changes in snowmelt timing can also affect amphibians adversely (Field and others 2007). In sum, climate change will have increasingly significant effects on public-land terrestrial and aquatic ecosystems, including plant and animal communities, soils, hydrologic processes, and water quality.

Ungulate Effects and Climate Change Synergies

Climate change in the western US is expected to amplify "combinations of biotic and abiotic stresses that compromise the vigor of ecosystems—leading to increased extent and severity of disturbances" (Joyce and others 2008, p. 16). Of the various land management stressors affecting western public lands, ungulate use is the most widespread (Fig. 1). Domestic livestock annually utilize over 70 % of lands managed by the Bureau of Land Management (BLM) and US Forest Service (FS). Many public lands are also used by wild ungulates and/or feral horses and burros, which are at high densities in some areas. Because ungulate groups can have different effects, we discuss them individually.

Livestock

History and Current Status

Livestock were introduced to North America in the mid-sixteenth century, with a massive influx from the mid-1800s through early 1900s (Worster 1992). The deleterious effects of livestock—including herbivory of both herbaceous and woody plants and trampling of vegetation, soils, and streambanks—prompted federal regulation of grazing on western national forests beginning in the 1890s (Fleischner 2010). Later, the 1934 Taylor Grazing Act was enacted "to stop injury to the public grazing lands by preventing overgrazing and soil deterioration" on lands subsequently administered by the BLM.

Total livestock use of federal lands in eleven contiguous western states today is nearly 9 million animal unit months (AUMs, where one AUM represents forage use by a cow and calf pair, one horse, or five sheep for one month) (Fig. 2a). Permitted livestock use occurs on nearly one million square kilometers of public land annually, including 560,000 km^2 managed by the BLM, 370,000 km^2 by the FS, 6,000 km^2 by the National Park Service (NPS), and 3,000 km^2 by the US Fish and Wildlife Service (FWS).

Livestock use affects a far greater proportion of BLM and FS lands than do roads, timber harvest, and wildfires combined (Fig. 3). Yet attempts to mitigate the pervasive effects of livestock have been minor compared with those aimed at reducing threats to ecosystem diversity and productivity that these other land uses pose. For example, much effort is often directed at preventing and controlling wildfires since they can cause significant property damage and social impacts. On an annual basis, however, wildfires affect a much smaller portion of public land than livestock grazing (Fig. 3) and they can also result in ecosystem benefits (Rhodes and Baker 2008; Swanson and others 2011).

The site-specific impacts of livestock use vary as a function of many factors (e.g., livestock species and density, periods of rest or non-use, local plant communities, soil conditions). Nevertheless, extensive reviews of published research generally indicate that livestock have had numerous and widespread negative effects to western ecosystems (Love 1959; Blackburn 1984; Fleischner 1994; Belsky and others 1999; Kauffman and Pyke 2001; Asner

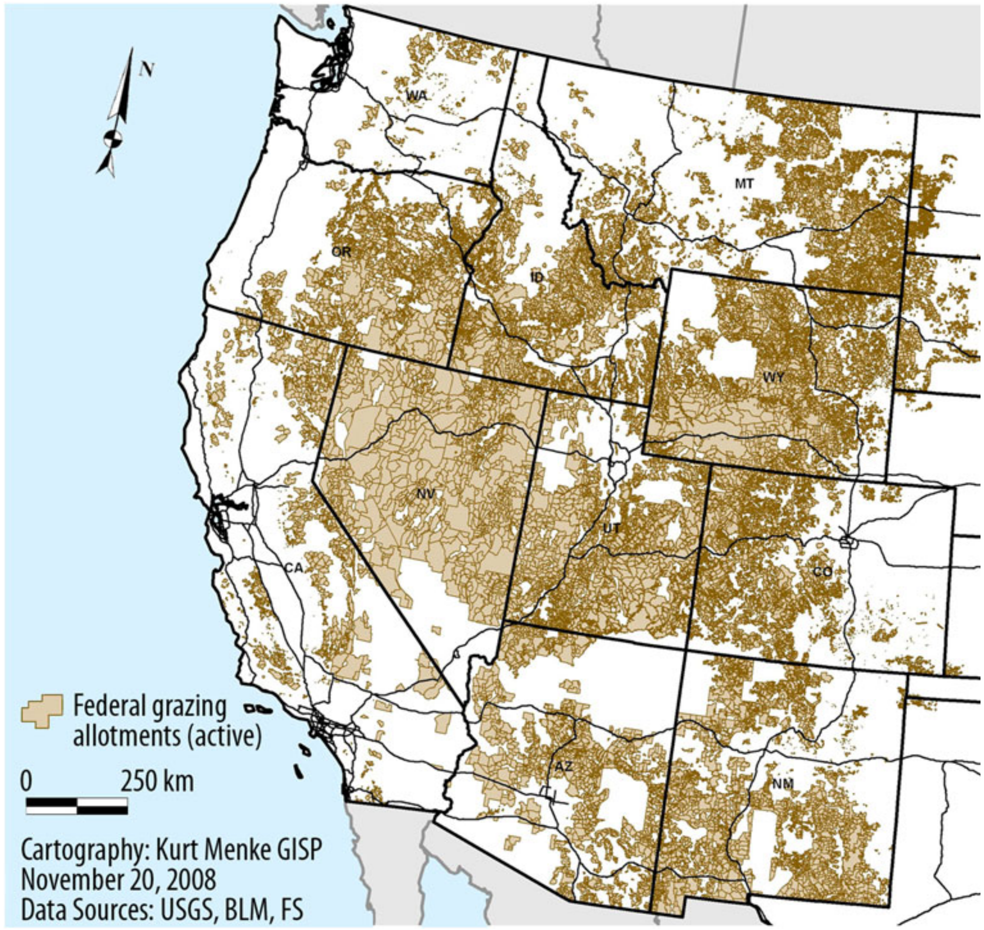

Fig. 1 Areas of public-lands livestock grazing managed by federal agencies in the western US (adapted from Salvo 2009)

and others 2004; Steinfeld and others 2006; Thornton and Herrero 2010). Moreover, public-land range conditions have generally worsened in recent decades (CWWR 1996, Donahue 2007), perhaps due to the reduced productivity of these lands caused by past grazing in conjunction with a changing climate (FWS 2010, p. 13,941, citing Knick and Hanser 2011).

Plant and Animal Communities

Livestock use effects, exacerbated by climate change, often have severe impacts on upland plant communities. For example, many former grasslands in the Southwest are now dominated by one or a few woody shrub species, such as creosote bush (*Larrea tridentata*) and mesquite (*Prosopis glandulosa*), with little herbaceous cover (Grover and Musick 1990; Asner and others 2004; but see Allington and Valone 2010). Other areas severely affected include the northern Great Basin and interior Columbia River Basin (Middleton and Thomas 1997). Livestock effects have also contributed to severe degradation of sagebrush-grass ecosystems (Connelly and others 2004; FWS 2010) and widespread desertification, particularly in the Southwest (Asner and others 2004; Karl and others 2009). Even absent desertification, light to moderate grazing intensities can promote woody species encroachment in semiarid and mesic environments (Asner and others 2004, p. 287). Nearly two decades ago, many public-land ecosystems, including native shrub steppe in Oregon and Washington, sagebrush steppe in the Intermountain West, and riparian plant communities, were considered threatened, endangered, or critically endangered (Noss and others 1995).

Simplified plant communities combine with loss of vegetation mosaics across landscapes to affect pollinators, birds, small mammals, amphibians, wild ungulates, and other native wildlife (Bock and others 1993; Fleischner 1994; Saab and others 1995; Ohmart 1996). Ohmart and Anderson (1986) suggested that livestock grazing may be the major factor negatively affecting wildlife in eleven western states. Such effects will compound the problems of adaptation of these ecosystems to the dynamics of climate change (Joyce and others 2008, 2009). Currently, the widespread and ongoing declines of many North American bird populations that use grassland and grass–shrub habitats affected by grazing are "on track to become a prominent wildlife conservation crisis of the 21st century" (Brennan and Kuvlesky 2005, p. 1).

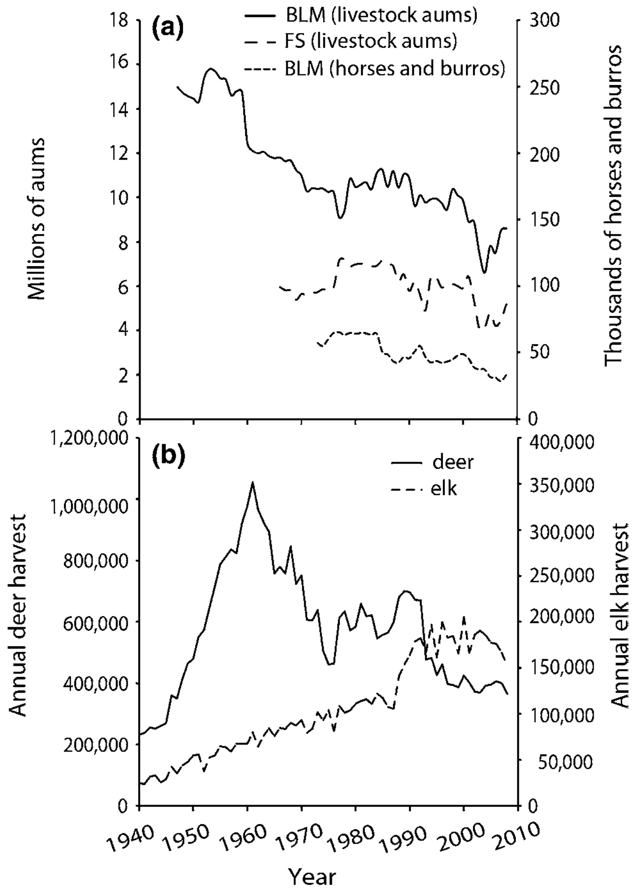

Fig. 2 **a** Bureau of Land Management (BLM) and US Forest Service (FS) grazing use in animal unit months (AUMs) and number of feral horses and burros on BLM lands, and **b** annual harvest of deer and elk by hunters, for eleven western states. *Data sources* **a** BLM grazing and number of horses and burros reported annually in Public Land Statistics; FS grazing reported annually in Grazing Statistical Summary; **b** deer and elk harvest records from individual state wildlife management agencies

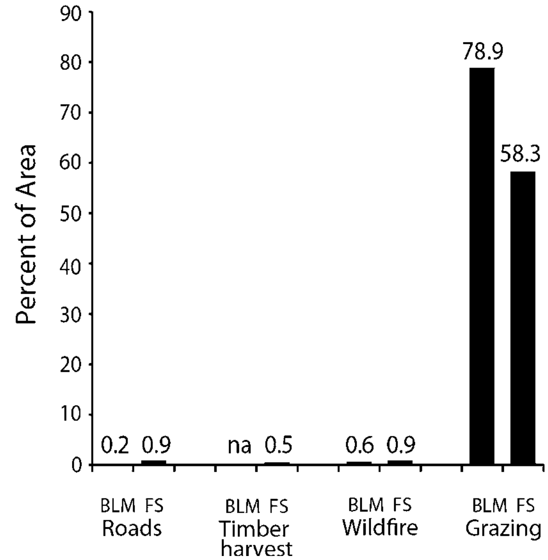

Fig. 3 Percent of Bureau of Land Management (BLM) and US Forest Service (FS) lands in eleven western states that are occupied by roads or are affected annually by timber harvest, wildfire, and grazing. *Data sources* Roads, BLM (2009) and FS, Washington Office; Timber harvest (2003–09), FS, Washington Office; Wildfire (2003–09), National Interagency Fire Center, Missoula, Montana; Grazing, BLM (2009) and GAO (2005). "na" = not available

Soils and Biological Soil Crusts

Livestock grazing and trampling can damage or eliminate biological soil crusts characteristic of many arid and semiarid regions (Belnap and Lange 2003; Asner and others 2004). These complex crusts are important for fertility, soil stability, and hydrology (Belnap and Lange 2003). In arid and semiarid regions they provide the major barrier against wind erosion and dust emission (Munson and others 2011). Currently, the majority of dust emissions in North America originate in the Great Basin, Colorado Plateau, and Mojave and Sonoran Deserts, areas that are predominantly public lands and have been grazed for nearly 150 years. Elevated sedimentation in western alpine lakes over this period has also been linked to increased aeolian deposition stemming from land uses, particularly those associated with livestock grazing (Neff and others 2008).

If livestock use on public lands continues at current levels, its interaction with anticipated changes in climate will likely worsen soil erosion, dust generation, and stream pollution. Soils whose moisture retention capacity has been reduced will undergo further drying by warming temperatures and/or drought and become even more susceptible to wind erosion (Sankey and others 2009). Increased aeolian deposition on snowpack will hasten runoff, accentuating climate-induced hydrological changes on many public lands (Neff and others 2008). Warmer temperatures will likely trigger increased fire occurrence, causing further reductions in cover and composition of biological soil crusts (Belnap and others 2006), as well as vascular plants (Munson and others 2011). In some forest types, where livestock grazing has contributed to altered fire regimes and forest structure (Belsky and Blumenthal 1997; Fleischner 2010), climate change will likely worsen these effects.

Water and Riparian Resources

Although riparian areas occupy only 1–2 % of the West's diverse landscapes, they are highly productive and ecologically valuable due to the vital terrestrial habitats they provide and their importance to aquatic ecosystems (Kauffman and others 2001; NRC 2002; Fleischner 2010). Healthy riparian plant communities provide important corridors for the movement of plant and animal species

(Peterson and others 2011). Such communities are also crucial for maintaining water quality, food webs, and channel morphology vital to high-quality habitats for fish and other aquatic organisms in the face of climate change. For example, well-vegetated streambanks not only shade streams but also help to maintain relatively narrow and stable channels, attributes essential for preventing increased stream temperatures that negatively affect salmonids and other aquatic organisms (Sedell and Beschta 1991; Kondolf and others 1996; Beschta 1997); maintaining cool stream temperatures is becoming even more important with climate change (Isaak and others 2012). Riparian vegetation is also crucial for providing seasonal fluxes of organic matter and invertebrates to streams (Baxter and others 2005). Nevertheless, in 1994 the BLM and FS reported that western riparian areas were in their worst condition in history, and livestock use—typically concentrated in these areas—was the chief cause (BLM and FS 1994).

Livestock grazing has numerous consequences for hydrologic processes and water resources. Livestock can have profound effects on soils, including their productivity, infiltration, and water storage, and these properties drive many other ecosystem changes. Soil compaction from livestock has been identified as an extensive problem on public lands (CWWR 1996; FS and BLM 1997). Such compaction is inevitable because the hoof of a 450-kg cow exerts more than five times the pressure of heavy earth-moving machinery (Cowley 2002). Soil compaction significantly reduces infiltration rates and the ability of soils to store water, both of which affect runoff processes (Branson and others 1981; Blackburn 1984). Compaction of wet meadow soils by livestock can significantly decrease soil water storage (Kauffman and others 2004), thus contributing to reduced summer base flows. Concomitantly, decreases in infiltration and soil water storage of compacted soils during periods of high-intensity rainfall contribute to increased surface runoff and soil erosion (Branson and others 1981). These fundamental alterations in hydrologic processes from livestock use are likely to be exacerbated by climate change.

The combined effects of elevated soil loss and compaction caused by grazing reduce soil productivity, further compromising the capability of grazed areas to support native plant communities (CWWR 1996; FS and BLM 1997). Erosion triggered by livestock use continues to represent a major source of sediment, nutrients, and pathogens in western streams (WSWC 1989; EPA 2009). Conversely, the absence of grazing results in increased litter accumulation, which can reduce runoff and erosion and retard desertification (Asner and others 2004).

Historical and contemporary effects of livestock grazing and trampling along stream channels can destabilize streambanks, thus contributing to widened and/or incised channels (NRC 2002). Accelerated streambank erosion and channel incision are pervasive on western public lands used by livestock (Fig. 4). Stream incision contributes to desiccation of floodplains and wet meadows, loss of floodwater detention storage, and reductions in baseflow (Ponce and Lindquist 1990; Trimble and Mendel 1995). Grazing and trampling of riparian plant communities also contribute to elevated water temperatures—directly, by reducing stream shading and, indirectly, by damaging streambanks and increasing channel widths (NRC 2002). Livestock use of riparian plant communities can also decrease the availability of food and construction materials for keystone species such as beaver (*Castor canadensis*).

Livestock effects and climate change can interact in various ways with often negative consequences for aquatic species and their habitats. In the eleven ecoregions encompassing western public lands (excluding coastal regions and Alaska), about 175 taxa of freshwater fish are considered imperiled (threatened, endangered, vulnerable, possibly extinct, or extinct) due to habitat-related causes (Jelks and others 2008, p. 377; GS and AFS 2011). Increased sedimentation and warmer stream temperatures associated with livestock grazing have contributed significantly to the long-term decline in abundance and distribution and loss of native salmonids, which are imperiled throughout the West (Rhodes and others 1994; Jelks and others 2008).

Water developments and diversions for livestock are common on public lands (Connelly and others 2004). For example, approximately 3,700 km of pipeline and 2,300 water developments were installed on just 17 % of the BLM's land base from 1961 to 1999 in support of livestock operations (Rich and others 2005). Such developments can reduce streamflows thus contributing to warmer stream temperatures and reduced fish habitat, both serious problems for native coldwater fish (Platts 1991; Richter and others 1997). Reduced flows and higher temperatures are also risk factors for many terrestrial and aquatic vertebrates (Wilcove and others 1998). Water developments can also create mosquito (e.g., *Culex tarsalis*) breeding habitat, potentially facilitating the spread of West Nile virus, which poses a significant threat to sage grouse (FWS 2010). Such developments also tend to concentrate livestock and other ungulate use, thus locally intensifying grazing and trampling impacts.

Greenhouse Gas Emissions and Energy Balances

Livestock production impacts energy and carbon cycles and globally contributes an estimated 18 % to the total anthropogenic greenhouse gas (GHG) emissions (Steinfeld and others 2006). How public-land livestock contribute to

Fig. 4 Examples of long-term grazing impacts from livestock, unless otherwise noted: **a** bare soil, loss of understory vegetation, and lack of aspen recruitment (i.e., growth of seedlings/sprouts into tall saplings and trees) (Bureau of Land Management, Idaho), **b** bare soil, lack of ground cover, lack of aspen recruitment and channel incision (US Forest Service, Idaho), **c** conversion of a perennial stream to an intermittent stream due to grazing of riparian vegetation and subsequent channel incision; channel continues to erode during runoff events (Bureau of Land Management, Utah), **d** incised and widening stream due to loss of streamside vegetation and bank collapse from trampling (Bureau of Land Management, Wyoming), **e** incised and widening stream due to loss of streamside vegetation and bank collapse from trampling (US Forest Service, Oregon), and **f** actively eroding streambank from the loss of streamside vegetation due to several decades of excessive herbivory by elk and, more recently, bison (National Park Service, Wyoming). *Photographs* **a** J Carter, **b** G Wuerthner, **c** and **d** J Carter, **e** and **f** R Beschta

these effects has received little study. Nevertheless, livestock grazing and trampling can reduce the capacity of rangeland vegetation and soils to sequester carbon and contribute to the loss of above- and below-ground carbon pools (e.g., Lal 2001b; Bowker and others 2012). Lal (2001a) indicated that heavy grazing over the long-term may have adverse impacts on soil organic carbon content, especially for soils of low inherent fertility. Although Gill (2007) found that grazing over 100 years or longer in subalpine areas on the Wasatch Plateau in central

Utah had no significant impacts on total soil carbon, results of the study suggest that "if temperatures warm and summer precipitation increases as is anticipated, [soils in grazed areas] may become net sources of CO_2 to the atmosphere" (Gill 2007, p. 88). Furthermore, limited soil aeration in soils compacted by livestock can stimulate production of methane, and emissions of nitrous oxide under shrub canopies may be twice the levels in nearby grasslands (Asner and others 2004). Both of these are potent GHGs.

Reduced plant and litter cover from livestock use can increase the albedo (reflectance) of land surfaces, thereby altering radiation energy balances (Balling and others 1998). In addition, widespread airborne dust generated by livestock is likely to increase with the drying effects of climate change. Air-borne dust influences atmospheric radiation balances as well as accelerating melt rates when deposited on seasonal snowpacks and glaciers (Neff and others 2008).

Other Livestock Effects

Livestock urine and feces add nitrogen to soils, which may favor nonnative species (BLM 2005), and can lead to loss of both organic and inorganic nitrogen in increased runoff (Asner and others 2004). Organic nitrogen is also lost via increased trace-gas flux and vegetation removal by grazers (Asner and others 2004). Reduced soil nitrogen is problematic in western landscapes because nitrogen is an important limiting nutrient in most arid-land soils (Fleischner 2010).

Managing livestock on public lands also involves extensive fence systems. Between 1962 and 1997, over 51,000 km of fence were constructed on BLM lands with resident sage-grouse populations (FWS 2010). Such fences can significantly impact this wildlife species. For example, 146 sage-grouse died in less than three years from collisions with fences along a 7.6-km BLM range fence in Wyoming (FWS 2010). Fences can also restrict the movements of wild ungulates and increase the risk of injury and death by entanglement or impalement (Harrington and Conover 2006; FWS 2010). Fences and roads for livestock access can fragment and isolate segments of natural ecological mosaics thus influencing the capability of wildlife to adapt to a changing climate.

Some have posited that managed cattle grazing might play a role in maintaining ecosystem structure in shortgrass steppe ecosystems of the US, if it can mimic grazing by native bison (*Bison bison*) (Milchunas and others 1998). But most public lands lie to the west of the Great Plains, where bison distribution and effects were limited or nonexistent; livestock use (particularly cattle) on these lands exert disturbances without evolutionary parallel (Milchunas and Lauenroth 1993; MEA 2005a).

Feral Horses and Burros

Feral horses and burros occupy large areas of public land in the western US. For example, feral horses are found in ten western states and feral burros occur in five of these states, largely in the Mojave and Sonoran Deserts and the Great Basin (Abella 2008; FWS 2010). About half of these horses and burros are in Nevada (Coggins and others 2007), of which 90 % are on BLM lands. Horse numbers peaked at perhaps two million in the early 1900s, but had plummeted to about 17,000 by 1971, when protective legislation (Wild, Free-Ranging Horses and Burros Act [WFRHBA]) was passed (Coggins and others 2007). Protection resulted in increased populations and today some 40,000 feral horses and burros utilize ~ 130,000 km^2 of BLM and FS lands (DOI-OIG 2010; Gorte and others 2010). Currently, feral horse numbers are doubling every four years (DOI-OIG 2010); burro populations can also increase rapidly (Abella 2008). Unlike wild ungulates, feral equines cannot be hunted and, unlike livestock, they are not regulated by permit. Nor are their numbers controlled effectively by existing predators. Accordingly, the BLM periodically removes animals from herd areas; the NPS also has undertaken burro control efforts (Abella 2008).

In sage grouse habitat, high numbers of feral horses reduce vegetative cover and plant diversity, fragment shrub canopies, alter soil characteristics, and increase the abundance of invasive species, thus reducing the quality and quantity of habitat (Beever and others 2003; FWS 2010). Horses can crop plants close to the ground, impeding the recovery of affected vegetation. Feral burros also have had a substantial impact on Sonoran Desert vegetation, reducing the density and canopy cover of nearly all species (Hanley and Brady 1977). Although burro impacts in the Mojave Desert may not be as clear, perennial grasses and other preferred forage species likely require protection from grazing in burro-inhabited areas if revegetation efforts are to be successful (Abella 2008).

Wild Ungulates

Extensive harvesting of wild (native) ungulates, such as elk and deer, and the decimation of large predator populations (e.g., gray wolf [*Canis lupus*], grizzly bear [*Ursus arctos*], and cougar [*Puma concolor*]) was common during early EuroAmerican settlement of the western US. With continued predator control in the early 1900s and increased protection of game species by state agencies, however, wild ungulate populations began to increase in many areas. Although only 70,000 elk inhabited the western US in the early 1900s (Graves and Nelson 1919), annual harvest data indicate that elk abundance has increased greatly since the about the 1940s (Fig. 2b), due in part to the loss of apex predators (Allen 1974; Mackie and others 1998). Today, approximately one million elk (Karnopp 2008) and unknown numbers of deer inhabit the western US where they often share public lands with livestock.

Because wild ungulates typically occur more diffusely across a landscape than livestock, their presence might be expected to cause minimal long-term impacts to vegetation. Where wild ungulates are concentrated, however, their browsing can have substantial impacts. For example, sagebrush vigor can be reduced resulting in decreased cover or mortality (FWS 2010). Heavy browsing effects have also been documented on other palatable woody shrubs, as well as deciduous trees such as aspen (*Populus tremuloides*), cottonwood (*Populus* spp.), and maple (*Acer* sp.) (Beschta and Ripple 2009).

Predator control practices that intensified following the introduction of domestic livestock in the western US resulted in the extirpation of apex predators or reduced their numbers below ecologically effective densities (Soulé and others 2003, 2005), causing important cascading effects in western ecosystems (Beschta and Ripple 2009). Following removal of large predators on the Kaibab Plateau in the early 20th century, for example, an irruption of mule deer (*O. hemionus*) led to extensive over-browsing of aspen, other deciduous woody plants, and conifers; deterioration of range conditions; and the eventual crash of the deer population (Binkley and others 2006). In the absence of apex predators, wild ungulate populations can significantly limit recruitment of woody browse species, contribute to shifts in abundance and distribution of many wildlife species (Berger and others 2001; Weisberg and Coughenour 2003), and can alter streambanks and riparian communities that strongly influence channel morphology and aquatic conditions (Beschta and Ripple 2012). Numerous studies support the conclusion that disruptions of trophic cascades due to the decline of apex predators constitute a threat to biodiversity for which the best management solution is likely the restoration of effective predation regimes (Estes and others 2011).

Ungulate Herbivory and Disturbance Regimes

Across the western US, ecosystems evolved with and were sustained by local and regional disturbances, such as fluctuating weather patterns, fire, disease, insect infestation, herbivory by wild ungulates and other organisms, and hunting by apex predators. Chronic disturbances with relatively transient effects, such as frequent, low-severity fires and seasonal moisture regime fluctuations, helped maintain native plant community composition and structure. Relatively abrupt, or acute, natural disturbances, such as insect outbreaks or severe fires were also important for the maintenance of ecosystems and native species diversity (Beschta and others 2004; Swanson and others 2011). Livestock use and/or an overabundance of feral or wild ungulates can, however, greatly alter ecosystem response to disturbance and can degrade affected systems. For example, high levels of herbivory over a period of years, by either domestic or wild ungulates, can effectively prevent aspen sprouts from growing into tall saplings or trees as well as reduce the diversity of understory species (Shepperd and others 2001; Dwire and others 2007; Beschta and Ripple 2009).

Natural floods provide another illustration of how ungulates can alter the ecological role of disturbances. High flows are normally important for maintaining riparian plant communities through the deposition of nutrients, organic matter, and sediment on streambanks and floodplains, and for enhancing habitat diversity of aquatic and riparian ecosystems (CWWR 1996). Ungulate effects on the structure and composition of riparian plant communities (e.g., Platts 1991; Chadde and Kay 1996), however, can drastically alter the outcome of these hydrologic disturbances by diminishing streambank stability and severing linkages between high flows and the maintenance of streamside plant communities. As a result, accelerated erosion of streambanks and floodplains, channel incision, and the occurrence of high instream sediment loads may become increasingly common during periods of high flows (Trimble and Mendel 1995). Similar effects have been found in systems where large predators have been displaced or extirpated (Beschta and Ripple 2012). In general, high levels of ungulate use can essentially uncouple typical ecosystem responses to chronic or acute disturbances, thus greatly limiting the capacity of these systems to provide a full array of ecosystem services during a changing climate.

The combined effects of ungulates (domestic, wild, and feral) and a changing climate present a pervasive set of stressors on public lands, which are significantly different from those encountered during the evolutionary history of the region's native species. The intersection of these stressors is setting the stage for fundamental and unprecedented changes to forest, arid, and semi-arid landscapes in the western US (Table 1) and increasing the likelihood of alternative states. Thus, public-land management needs to focus on restoring and maintaining structure, function, and integrity of ecosystems to improve their resilience to climate change (Rieman and Isaak 2010).

Federal Law and Policy

Federal laws guide the use and management of public-land resources. Some laws are specific to a given agency (e.g., the BLM's Taylor Grazing Act of 1934 and the FS's

Table 1 Generalized climate change effects, heavy ungulate use effects, and their combined effects as stressors to terrestrial and aquatic ecosystems in the western United States

Climate change effects	Ungulate use effects	Combined effects
Increased drought frequency and duration	Altered upland plant and animal communities	Reduced habitat and food-web support; loss of mesic and hydric plants, reduced biodiversity
Increased air temperatures, decreased snowpack accumulation, earlier snowmelt	Compacted soils, decreased infiltration, increased surface runoff	Reduced soil moisture for plants, reduced productivity, reductions in summer low flows, degraded aquatic habitat
Increased variability in timing and magnitude of precipitation events	Decreased biotic crusts and litter cover, increased surface erosion	Accelerated soil and nutrient loss, increased sedimentation
Warmer and drier in the summer	Reduced riparian vegetation, loss of shade, increased stream width	Increased stream temperatures, increased stress on cold-water fish and aquatic organisms
Increased variability in runoff	Reduced root strength of riparian plants, trampled streambanks, streambank erosion	Accelerated streambank erosion and increased sedimentation, degraded water quality and aquatic habitats
Increased variability in runoff	Incised stream channels	Degraded aquatic habitats, hydrologically disconnected floodplains, reduced low flows

National Forest Management Act [NFMA] of 1976), whereas others cross agency boundaries (e.g., Endangered Species Act [ESA] of 1973; Clean Water Act [CWA] of 1972). A common mission of federal land management agencies is “to sustain the health, diversity, and productivity of public lands” (GAO 2007, p. 12). Further, each of these agencies has ample authority and responsibility to adjust management to respond to climate change (GAO 2007) and other stressors.

The FS and BLM are directed to maintain and improve the condition of the public rangelands so that they become as productive as feasible for all rangeland values. As defined, “range condition” encompasses factors such as soil quality, forage values, wildlife habitat, watershed and plant communities, and the present state of vegetation of a range site in relation to the potential plant community for that site (Public Rangelands Improvement Act of 1978). BLM lands and national forests must be managed for sustained yield of a wide array of multiple uses, values, and ecosystem services, including wildlife and fish, watershed, recreation, timber, and range. Relevant statutes call for management that meets societal needs, without impairing the productivity of the land or the quality of the environment, and which considers the “relative values” of the various resources, not necessarily the combination of uses that will give the greatest economic return or the greatest unit output (Multiple-Use Sustained-Yield Act of 1960; Federal Land Policy and Management Act of 1976 [FLPMA]).

FLPMA directs the BLM to “take any action necessary to prevent unnecessary or undue degradation” of the public lands. Under NFMA, FS management must provide for diversity of plant and animal communities based on the suitability and capability of the specific land area. FLMPA also authorizes both agencies to “cancel, suspend, or modify” grazing permits and to determine that “grazing uses should be discontinued (either temporarily or permanently) on certain lands.” FLPMA explicitly recognizes the BLM’s authority (with congressional oversight) to “totally eliminate” grazing from large areas (> 405 km^2) of public lands. These authorities are reinforced by law providing that grazing permits are not property rights (*Public Lands Council v. Babbitt* 2000).

While federal agencies have primary authority to manage federal public lands and thus wildlife *habitats* on these lands, states retain primary management authority over resident *wildlife*, unless preempted, as by the WFRHBA or ESA (*Kleppe v. New Mexico* 1976). Under WFRHBA, wild, free-roaming horses and burros (i.e., feral) by law have been declared “wildlife” and an integral part of the natural system of the public lands where they are to be managed in a manner that is designed to achieve and maintain a thriving natural ecological balance.

Restoring Ungulate-Altered Ecosystems

Because livestock use is so widespread on public lands in the American West, management actions directed at ecological restoration (e.g., livestock removal, substantial reductions in numbers or length of season, extended or regular periods of rest) need to be accomplished at landscape scales. Such approaches, often referred to as passive restoration, are generally the most ecologically effective and economically efficient for recovering altered ecosystems because they address the root causes of degradation and allow natural recovery processes to operate (Kauffman and others 1997; Rieman and Isaak 2010). Furthermore, reducing the impact of current stressors is a “no regrets” adaptation strategy that could be taken now to help enhance

Fig. 5 Examples of riparian and stream recovery in the western United States after the removal of livestock grazing: Hart Mountain National Antelope Refuge, Oregon, in **a** October 1989 and **b** September 2010 after 18 years of livestock removal; Strawberry River, Utah, in **c** August 2002 after 13 years of livestock removal and **d** July 2003 illustrating improved streambank protection and riparian productivity as beaver reoccupy this river system; and San Pedro River, Arizona in **e** June 1987 and **f** June 1991 after 4 years of livestock removal. *Photographs* **a** Fish and Wildlife Service, Hart Mountain National Antelope Refuge, **b** J Rhodes, **c** and **d** US Forest Service, Uintah National Forest, **e** and **f** Bureau of Land Management, San Pedro Riparian National Conservation Area

ecosystem resilience to climate change (Joyce and others 2008). This strategy is especially relevant to western ecosystems because removing or significantly reducing the cause of degradation (e.g., excessive ungulate use) is likely to be considerably more effective over the long term, in both costs and approach, than active treatments aimed at specific ecosystem components (e.g., controlling invasive plants) (BLM 2005). Furthermore, the possibility that passive restoration measures may not accomplish all ecological goals is an insufficient reason for *not* removing or reducing stressors at landscape scales.

For many areas of the American West, particularly riparian areas and other areas of high biodiversity, significantly reducing or eliminating ungulate stressors should, over time, result in the recovery of self-sustaining and ecologically robust ecosystems (Kauffman and others 1997; Floyd and others 2003; Allington and Valone 2010; Fig. 5). Indeed, various studies and reviews have concluded that the most effective way to restore riparian areas and aquatic systems is to exclude livestock either temporarily (with subsequent changed management) or long-term (e.g., Platts 1991;BLM and FS 1994; Dobkin and others

1998; NRC 2002; Seavy and others 2009: Fleischner 2010). Recovering channel form and riparian soils and vegetation by reducing ungulate impacts is also a viable management tool for increasing summer baseflows (Ponce and Lindquist 1990; Rhodes and others 1994).

In severely degraded areas, initiating recovery may require active measures in addition to the removal/reduction of stressors. For example, where native seed banks have been depleted, reestablishing missing species may require planting seeds or propagules from adjacent areas or refugia (e.g., Welch 2005). While active restoration approaches in herbivory-degraded landscapes may have some utility, such projects are often small in scope, expensive, and unlikely to be self-sustaining; some can cause unanticipated negative effects (Kauffman and others 1997). Furthermore, if ungulate grazing effects continue, any benefits from active restoration are likely to be transient and limited. Therefore, addressing the underlying causes of degradation should be the first priority for effectively restoring altered public-land ecosystems.

The ecological effectiveness and low cost of wide-scale reduction in ungulate use for restoring public-land ecosystems, coupled with the scarcity of restoration resources, provide a forceful case for minimizing ungulate impacts. Other conservation measures are unlikely to make as great a contribution to ameliorating landscape-scale effects from climate change or to do so at such a low fiscal cost. As Isaak and others (2012, p. 514) noted with regard to the impacts of climate change on widely-imperiled salmonids: "…conservation projects are likely to greatly exceed available resources, so strategic prioritization schemes are essential."

Although restoration of desertified lands was once thought unlikely, recovery in the form of significant increases in perennial grass cover has recently been reported at several such sites around the world where livestock have been absent for more than 20 years (Floyd and others 2003; Allington and Valone 2010; Peters and others 2011). At a desertified site in Arizona that had been ungrazed for 39 years, infiltration rates were significantly (24 %) higher (compared to grazed areas) and nutrient levels were elevated in the bare ground, inter-shrub areas (Allington and Valone 2010). The change in vegetative structure also affected other taxa (e.g., increased small mammal diversity) where grazing had been excluded (Valone and others 2002). The notion that regime shifts caused by grazing are irreversible (e.g., Bestelmeyer and others 2004) may be due to the relative paucity of large-scale, ungulate-degraded systems where grazing has been halted for sufficiently long periods for recovery to occur.

Removing domestic livestock from large areas of public lands, or otherwise significantly reducing their impacts, is consistent with six of the seven approaches recommended for ecosystem adaptation to climate change (Julius and others 2008, pp. 1-3). Specifically, removing livestock would (1) protect key ecosystem features (e.g., soil properties, riparian areas); (2) reduce anthropogenic stressors; (3) ensure representation (i.e., protect a variety of forms of a species or ecosystem); (4) ensure replication (i.e., protect more than one example of each ecosystem or population); (5) help restore ecosystems; and (6) protect refugia (i.e., areas that can serve as sources of "seed" for recovery or as destinations for climate-sensitive migrants). Although improved livestock management practices are being adopted on some public lands, such efforts have not been widely implemented. Public land managers have rarely used their authority to implement landscape-scale rest from livestock use, lowered frequency of use, or multi-stakeholder planning for innovative grazing systems to reduce impacts.

While our findings are largely focused on adaptation strategies for western landscapes, reducing ungulate impacts and restoring degraded plant and soil systems may also assist in mitigating any ongoing or future changes in regional energy and carbon cycles that contribute to global climate change. Simply removing livestock can increase soil carbon sequestration since grasslands with the greatest potential for increasing soil carbon storage are those that have been depleted in the past by poor management (Wu and others 2008, citing Jones and Donnelly 2004). Riparian area restoration can also enhance carbon sequestration (Flynn and others 2009).

Socioeconomic Considerations

A comprehensive assessment of the socioeconomic effects of changes in ungulate management on public lands is beyond the scope of this paper. However, herein we identify a few of the *general* costs and benefits associated with implementing our recommendations (see next section), particularly with regard to domestic livestock grazing. The socioeconomic effects of altering ungulate management on public lands will ultimately depend on the type, magnitude, and location of changes undertaken by federal and state agencies.

Ranching is a contemporary and historically significant aspect of the rural West's social fabric. Yet, ranchers' stated preferences in response to grazing policy changes are as diverse as the ranchers themselves, and include intensifying, extensifying, diversifying, or selling their operations (Genter and Tanaka 2002). Surveys indicate that most ranchers are motivated more by amenity and lifestyle attributes than by profits (Torell and others 2001, Genter and Tanaka 2002). Indeed, economic returns from ranching are lower than any other investments with similar risk

(Torrell and others 2001) and public-land grazing's contributions to income and jobs in the West are relatively small fractions of the region's totals (BLM and FS 1994; Power 1996).

If livestock grazing on public lands were discontinued or curtailed significantly, some operations would see reduced incomes and ranch values, some rural communities would experience negative economic impacts, and the social fabric of those communities could be altered (Genter and Tanaka 2002). But for most rural economies, and the West in general, the economic impacts of managing public lands to emphasize environmental amenities would be relatively minor to modestly positive (Mathews and others 2002). Other economic effects could include savings to the US Treasury because federal grazing fees on BLM and FS lands cover only about one-sixth of the agencies' administration costs (Vincent 2012). Most significantly, improved ecosystem function would lead to enhanced ecosystem services, with broad economic benefits. Various studies have documented that the economic values of other public-land resources (e.g., water, timber, recreation, and wilderness) are many times larger than that of grazing (Haynes and others 1997; Laitos and Carr 1999; Patterson and Coelho 2009).

Facilitating adaptation to climate change will require changes in the management of public-land ecosystems impacted by ungulates. *How* ungulate management policy changes should be accomplished is a matter for the agencies, the public, and others. The recommendations and conclusions presented in the following section are based solely on ecological considerations and the federal agencies' legal authority and obligations.

Recommendations

We propose that large areas of BLM and FS lands should become free of use by livestock and feral ungulates (Table 2) to help initiate and speed the recovery of affected ecosystems as well as provide benchmarks or controls for assessing the effects of "grazing versus no-grazing" at significant spatial scales under a changing climate. Further, large areas of livestock exclusion allow for understanding potential recovery foregone in areas where livestock grazing is continued (Bock and others 1993).

While lowering grazing pressure rather than discontinuing use might be effective in some circumstances, public land managers need to rigorously assess whether such use is compatible with the maintenance or recovery of ecosystem attributes such as soils, watershed hydrology, and native plant and animal communities. In such cases, the contemporary status of at least some of the key attributes and their rates of change should be carefully monitored to ascertain whether continued use is consistent with ecological recovery, particularly as the climate shifts (e.g., Karr and Rossano 2001, Karr 2004; LaPaix and others 2009). To the extent possible, assessments of recovering areas should be compared to similar measurements in reference areas (i.e., areas exhibiting high ecological integrity) or areas where ungulate impacts had earlier been removed or minimized (Angermeier and Karr 1994; Dobkin and others 1998). Such comparisons are crucial if scientists and managers are to confirm whether managed systems are attaining restoration goals and to determine needs for intervention, such as reintroducing previously extirpated species. Unfortunately, testing for impacts of livestock use at landscape scales is hampered by the lack of large, ungrazed areas in the western US (e.g., Floyd and others 2003; FWS 2010).

Table 2 Priority areas for permanently removing livestock and feral ungulates from Bureau of Land Management and US Forest Service lands to reduce or eliminate their detrimental ecological effects

Watersheds and other large areas that contain a variety of ecotypes to ensure that major ecological and societal benefits of more resilient and healthy ecosystems on public lands will occur in the face of climate change
Areas where ungulate effects extend beyond the immediate site (e.g., wetlands and riparian areas impact many wildlife species and ecosystem services with cascading implications beyond the area grazed)
Localized areas that are easily damaged by ungulates, either inherently (e.g., biological crusts or erodible soils) or as the result of a temporary condition (e.g., recent fire or flood disturbances, or degraded from previous management and thus fragile during a recovery period).
Rare ecosystem types (e.g., perched wetlands) or locations with imperiled species (e.g., aspen stands and understory plant communities, endemic species with limited range), including fish and wildlife species adversely affected by grazing and at-risk and/or listed under the ESA
Non-use areas (i.e., ungrazed by livestock) or exclosures embedded within larger areas where livestock grazing continues. Such non-use areas should be located in representative ecotypes so that actual rates of recovery (in the absence of grazing impacts) can be assessed relative to resource trend and condition data in adjacent areas that continue to be grazed
Areas where the combined effects of livestock, wild ungulates, and feral ungulates are causing significant ecological impacts

Shifting the burden of proof for continuing, rather than significantly reducing or eliminating ungulate grazing is warranted due to the extensive body of evidence on ecosystem impacts caused by ungulates (i.e., consumers) and the added ecosystem stress caused by climate change. As Estes and others (2011, p. 306) recommended: "[T]he burden of proof [should] be shifted to show, for any ecosystem, that consumers do (or did) not exert strong cascading effects" (see also Henjum and others 1994; Kondolf 1994; Rhodes and others 1994). Current livestock or feral ungulate use should continue only where stocking rates, frequency, and timing can be demonstrated, in comparison with landscape-scale reference areas, exclosures, or other appropriate non-use areas, to be compatible with maintaining or recovering key ecological functions and native species complexes. Furthermore, such use should be allowed only when monitoring is adequate to determine the effects of continued grazing in comparison to areas without grazing.

Where wild native ungulates, such as elk or deer, have degraded plant communities through excessive herbivory (e.g., long-term suppression of woody browse species [Weisberg and Coughenour 2003; Beschta and Ripple 2009; Ripple and others 2010]), state wildlife agencies and federal land managers need to cooperate in controlling or reducing those impacts. A potentially important tool for restoring ecosystems degraded by excessive ungulate herbivory is reintroduction or recolonization of apex predators. In areas of public land that are sufficiently large and contain suitable habitat, allowing apex predators to become established at ecologically effective densities (Soulé and others 2003, 2005) could help regulate the behavior and density of wild ungulate populations, aiding the recovery of degraded ecosystems (Miller and others 2001; Ripple and others 2010; Estes and others 2011). Ending government predator control programs and reintroducing predators will have fewer conflicts with livestock grazing where the latter has been discontinued in large, contiguous public-land areas. However, the extent to which large predators might also help control populations of feral horses and burros is not known.

Additionally, we recommend removing livestock and feral ungulates from national parks, monuments, wilderness areas, and wildlife refuges wherever possible and managing wild ungulates to minimize their potential to adversely affect soil, water, vegetation, and wildlife populations or impair ecological processes. Where key large predators are absent or unable to attain ecologically functional densities, federal agencies should coordinate with state wildlife agencies in managing wild ungulate populations to prevent excessive effects of these large herbivores on native plant and animal communities.

Conclusions

Average global temperatures are increasing and precipitation regimes changing at greater rates than at any time in recent centuries. Contemporary trends are expected to continue and intensify for decades, even if comprehensive mitigations regarding climate change are implemented immediately. The inevitability of these trends requires adaptation to climate change as a central planning goal on federal lands.

Historical and on-going ungulate use has affected soils, vegetation, wildlife, and water resources on vast expanses of public forests, shrublands, and grasslands across the American West in ways that are likely to accentuate any climate impacts on these resources. Although the effects of ungulate use vary across landscapes, this variability is more a matter of degree than type.

If effective adaptations to the adverse effects of climate change are to be accomplished on western public lands, large-scale reductions or cessation of ecosystem stressors associated with ungulate use are crucial. Federal and state land management agencies should seek and make wide use of opportunities to reduce significant ungulate impacts in order to facilitate ecosystem recovery and improve resiliency. Such actions represent the most effective and extensive means for helping maintain or improve the ecological integrity of western landscapes and for the continued provision of valuable ecosystem services during a changing climate.

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Abstract

Surface fire intensity (kilowatts per metre) and crown fire initiation were predicted using Rothermel's 1972 and Van Wagner's 1977 fire models with fuel data from 47 upland subalpine conifer stands varying in age from 22-258 yr and 35 yr of daily weather data (fuel moisture and wind speeds). Rothermel's intensity model was divided into a fuel component variable and weather component variable, which were then used to examine the relative roles of fuel and weather on surface fire intensity (kilowatts per metre). Similar variables were defined in the crown fire initiation model of Van Wagner. Both surface fire intensity and crown fire initiation were strongly related to the weather components and weakly related to the fuel components, due to much greater variability in weather than fuel, and stronger relationship to the fire behavior mechanisms for weather than for fuel. Fire intensity was correlated to annual area burned; large area burned years had higher fire intensity predictions than smaller area burned years. The reason for this difference was attributed directly to the weather variable frequency distribution, which was shifted towards more extreme values in years in which large areas burned. During extreme weather conditions, the relative importance of fuels diminishes since all stands achieve the threshold required to permit crown fire development. This is important since most of the area burned in subalpine forests has historically occurred during very extreme weather (i.e., drought coupled to high winds). The fire behavior relationships predicted in the models support the concept that forest fire behavior is determined primarily by weather variation among years rather than fuel variation associated with stand age.

S2 Table. Authors, sites, the Weibull mean ITFI estimate, and the calibrated or predicted PMFI/FR for the merged 342-site dataset.

State/Author(s)†	Sites	State	Weibull Mean ITFI (years)	Calibrated or predicted	Calibrated/ Predicted PMFI/FR (years)
ARIZONA					
Dieterich and Hibbert (1990)	Battle Flat	AZ	6.31	Calibrated	7.20
Kaib and Swetnam, no publ.	Mt. Ord	AZ	8.72	Predicted	10.60
Dieterich (1980)	Chimney Springs	AZ	8.80	Predicted	10.70
Swetnam and Baisan (1996)	Walnut Canyon	AZ	8.81	Predicted	10.71
Swetnam et al. (2001)	Palisades	AZ	9.21	Predicted	11.20
Farris et al. (2013)	Centennial Forest	AZ	-	Predicted	12.03
Fulé et al. (2003a)	Galahad Point	AZ	11.26	Calibrated	12.50
Seklecki et al. (1996)	Rustler Park	AZ	10.32	Predicted	12.55
Farris et al. (2013)	Mica Mountain	AZ	-	Predicted	12.57
Baisan et al. (1998)	Rose Canyon Lower	AZ	10.80	Predicted	13.13
Danzer (1998)	Sawmill Canyon	AZ	10.92	Predicted	13.28
Fulé et al. (2003b)	Fire Point	AZ	11.25	Calibrated	13.60
Baisan and Swetnam (1990)	Mica Mountain	AZ	12.48	Calibrated	15.00
Baisan et al. (1998)	Mount Lemmon	AZ	12.36	Predicted	15.03
Fulé et al. (2003b)	Powell Plateau	AZ	13.68	Calibrated	15.40
Baisan et al. (1998)	Rose Canyon Upper	AZ	12.74	Predicted	15.49
Swetnam and Baisan (1996)	Josephine Saddle	AZ	12.90	Predicted	15.69
Baisan et al. (1998)	Rose Canyon East	AZ	13.40	Predicted	16.29
Fulé et al. (2003b)	Swamp Ridge	AZ	14.56	Calibrated	17.10
Danzer (1998)	Pat Scott Peak	AZ	13.06	Calibrated	17.70
Fulé et al. (2003b)	Grandview	AZ	14.89	Calibrated	17.90
Fulé et al. (2003b)	Rainbow Plateau	AZ	14.10	Calibrated	18.00
Fulé et al. (1997)	Camp Navajo	AZ	13.02	Calibrated	19.00
Huffman et al. (2015)	Mogollon Rim	AZ	-	Predicted	19.25
Heinlein et al. (2005)	San Francisco Peaks West	AZ	15.50	Calibrated	20.60
Dieterick (1983)	Thomas Creek	AZ	17.31	Calibrated	22.10
Heinlein et al. (2005)	San Francisco Peaks East	AZ	17.32	Calibrated	23.20
Fulé et al. (2003b; Dugan and Baker (2014)	Grandview	AZ	18.40	Calibrated	25.70
CALIFORNIA					
Caprio and Swetnam (1995)	Ash Peak Ridge	CA	7.04	Predicted	8.56

Taylor and Skinner (1998)	Thompson Ridge: 1850-1904	CA	-	Calibrated	12.30
Scholl and Taylor (2010)	Tuolomne River	CA	-	Calibrated	13.00
Beaty and Taylor (2001)	South-facing	CA	-	Calibrated	17.40
Caprio and Swetnam (1995)	Bobcat Point Pine	CA	15.09	Predicted	18.35
Taylor and Skinner (1998)	Thompson Ridge: 1626-1849	CA	-	Calibrated	19.00
Taylor and Skinner (2003)	Hayfork: 1628-1849	CA	22.86	Calibrated	20.00
Bekker and Taylor (2001)	White fir-Jeffrey pine	CA	-	Calibrated	21.50
Caprio and Swetnam (1995)	High Sierra Ridge Pine	CA	18.75	Predicted	22.80
Taylor (2000)	Prospect Peak: Jeffrey Pine	CA	-	Calibrated	24.50
Beaty and Taylor (2001)	Northern headwaters	CA	-	Calibrated	27.20
Beaty and Taylor (2001)	Combined study areas	CA	-	Calibrated	28.20
Taylor (2000)	Prospect Peak: Jeffrey Pine-White fir	CA	-	Calibrated	31.30
Bekker and Taylor (2001)	White fir-Sugar pine	CA	-	Calibrated	33.70
Beaty and Taylor (2001)	Southern headwaters	CA	-	Calibrated	37.20
Swetnam et al., no publication	Buck Rock Flat	CA	32.11	Predicted	39.05
Beaty and Taylor (2001)	North-facing	CA	-	Calibrated	42.50
Fiegener (2002)	Teakettle	CA	-	Predicted	49.87
Fiegener (2002)	Teakettle	CA	-	Predicted	75.31
Everett (2003)	Black Mountain	CA	-	Predicted	269.41
Everett (2003)	Big Pine Flat	CA	-	Predicted	327.16
COLORADO					
Grissino-Mayer et al. (2004)	Plateau	CO	15.86	Calibrated	15.20
Grissino-Mayer et al. (2004)	Five Pine Canyon	CO	15.96	Calibrated	15.80
Veblen et al. (2000)	BM34	CO	14.31	Predicted	17.40
Brown and Wu (2005)	Archuleta Mesa Plot A05	CO	15.19	Predicted	18.47
Grissino-Mayer et al. (2004)	Benson Creek	CO	14.86	Calibrated	20.70
Grissino-Mayer et al. (2004)	Hermosa Creek	CO	20.97	Predicted	25.50
Brown and Wu (2005)	Archuleta Mesa Plot A1	CO	21.49	Predicted	26.13
Grissino-Mayer et al. (2004)	Turkey Springs	CO	21.14	Calibrated	26.40
Veblen et al. (2000)	BM31	CO	23.68	Predicted	28.79
Grissino-Mayer et al. (2004)	Smoothing Iron	CO	25.74	Calibrated	29.00
Grissino-Mayer et al. (2004)	Taylor Creek	CO	26.72	Calibrated	29.20
Brown and Shepperd (2001)	Wet Mountains South	CO	25.85	Predicted	31.43
Brown and Wu (2005)	Archuleta Mesa	CO	23.13	Calibrated	32.10
Veblen et al. (2000)	BM14	CO	26.79	Predicted	32.58

Brown and Shepperd (2001)	M Kaufmanns Cabin	CO	26.80	Predicted	32.59
Bigio et al. (2010)	Vallecito Country Market	CO	27.75	Calibrated	32.60
Grissino-Mayer et al. (2004)	Monument	CO	33.16	Calibrated	37.50
Bigio (2013)	Marina Basin	CO	30.95	Predicted	37.64
Grissino-Mayer et al. (2004)	Burnette Canyon	CO	33.83	Predicted	41.14
Brown and Shepperd (2001)	Black Mountain	CO	35.28	Predicted	42.90
Brown and Wu (2005)	Archuleta Mesa Plot AA1	CO	36.00	Predicted	43.78
Veblen et al. (2000)	BM15	CO	30.12	Calibrated	44.92
Fulé et al. (2009)	Lower Middle Mountain	CO	30.53	Calibrated	46.80
Veblen et al. (2000)	BM28	CO	39.34	Predicted	47.84
Brown and Wu (2005)	Archuleta Mesa Plot B2	CO	40.02	Predicted	48.66
Veblen et al. (2000)	BM24	CO	40.26	Predicted	48.96
Brown and Wu (2005)	Archuleta Mesa Plot C2	CO	41.52	Predicted	50.49
Bigio et al. (2010)	Haflin Canyon	CO	42.11	Calibrated	50.80
Veblen et al. (2000)	BM11	CO	42.27	Predicted	51.40
Brown and Shepperd (2001)	Manitou Demo Plot	CO	42.34	Predicted	51.49
Veblen et al. (2000)	BM8	CO	47.23	Predicted	57.43
Veblen et al. (2000)	BM9	CO	48.61	Predicted	59.11
Brown and Shepperd (2001)	Mica Mine	CO	49.71	Predicted	60.45
Brown and Wu (2005)	Archuleta Mesa Plot AA15	CO	50.41	Predicted	61.30
Veblen et al. (2000)	BM22	CO	-	Calibrated	61.90
Brown and Shepperd (2001)	Parachute Hill	CO	51.75	Predicted	62.93
Veblen et al. (2000)	BM20	CO	53.57	Predicted	65.14
Veblen et al. (2000)	BM32	CO	58.07	Predicted	70.61
Brown et al. (2000)	Hot Creek	CO	58.44	Predicted	71.06
Brown and Wu (2005)	Archuleta Mesa Plot B3	CO	58.97	Predicted	71.71
Veblen et al. (2000)	BM13	CO	59.80	Predicted	72.72
Bigio (2013)	Steven’s Canyon	CO	37.56	Calibrated	74.00
Brown and Wu (2005)	Archuleta Mesa Plot C5	CO	64.07	Predicted	77.91
Veblen et al. (2000)	BM23	CO	63.59	Calibrated	80.30
Brown and Shepperd (2001)	Left Hand Canyon	CO	67.75	Predicted	82.38
Veblen et al. (2000)	BM5	CO	71.75	Predicted	87.25
Veblen et al. (2000)	BM12	CO	72.73	Predicted	88.44
Brown and Wu (2005)	Archuleta Mesa Plot A15	CO	73.80	Predicted	89.74
Donnegan et al. (2001)	BSA Shortcut	CO	75.54	Predicted	91.86
Brown and Shepperd (2001)	Cheesman Lake South	CO	76.87	Predicted	93.47

Donnegan et al. (2001)	Badger Mountain	CO	92.84	Calibrated	94.10
Brown and Shepperd (2001)	Washout Gulch Burn	CO	77.56	Predicted	94.31
Brown and Shepperd (2001)	Cheesman Lake North	CO	78.62	Predicted	95.60
Veblen et al. (2000)	BM6	CO	80.88	Calibrated	100.00
Veblen et al. (2000)	BM18	CO	78.98	Calibrated	103.50
Brown and Shepperd (2001)	Old Tree Cluster	CO	86.85	Predicted	105.61
Donnegan et al. (2001)	Salt Creek	CO	89.07	Calibrated	106.70
Brown and Shepperd (2001)	Lone Pine	CO	88.59	Predicted	107.73
Veblen et al. (2000)	BM39	CO	88.81	Predicted	107.99
Veblen et al. (2000)	BM10	CO	-	Calibrated	112.60
Veblen et al. (2000)	BM21	CO	99.22	Predicted	120.65
Brown and Shepperd (2001)	Lone Pine Upper	CO	107.72	Predicted	130.99
Donnegan et al. (2001)	China Wall	CO	-	Calibrated	138.30
Veblen et al. (2000)	BM19	CO	114.59	Predicted	139.34
Veblen et al. (2000)	BM17	CO	143.99	Predicted	175.09
IDAHO					
Heyerdahl et al. (2008)	Warm Springs Ridge	ID	13.88	Predicted	16.88
Heyerdahl et al. (2008)	Bannock Creek	ID	13.98	Predicted	17.00
Heyerdahl et al. (2008)	Wash Creek	ID	15.98	Predicted	19.43
Heyerdahl et al. (2008)	Keating Ridge	ID	22.26	Predicted	27.07
Heyerdahl et al. (2008)	Cove Mountain	ID	25.53	Predicted	31.04
Shapiro-Miller et al. (2007)	Powderhouse	ID	23.89	Calibrated	32.95
Heyerdahl et al. (2008)	Lowman RNA	ID	30.73	Predicted	37.37
MONTANA					
Heyerdahl et al. (2008)	Sophie Lake	MT	10.90	Predicted	13.25
Heyerdahl et al. (2008)	Sheldon Flats	MT	11.05	Predicted	13.44
Heyerdahl et al. (2008)	Butler Creek	MT	12.22	Predicted	14.86
Heyerdahl et al. (2008)	Blue Mountain	MT	12.28	Predicted	14.93
Heyerdahl et al. (2008)	McCormick Creek	MT	18.00	Calibrated	19.40
Heyerdahl et al. (2008)	McMillan Mountain	MT	17.71	Predicted	21.54
Heyerdahl et al. (2008)	Corona Road	MT	19.25	Predicted	23.41
Heyerdahl et al. (2008)	Hunter Point	MT	19.84	Predicted	24.13
Heyerdahl et al. (2008)	Sheafman Creek	MT	21.06	Predicted	25.61
Jones (2005)	Lubrecht	MT	23.26	Calibrated	27.40
Heyerdahl et al. (2008)	Crane Lookout	MT	25.47	Predicted	30.97
Heyerdahl et al. (2008)	Sawmill Creek RNA	MT	27.00	Predicted	32.83

NEW MEXICO					
Brown et al. (2001)	Pines at Sunspot	NM	8.40	Predicted	10.21
Swetnam and Dieterich (1985)	Langstroth Mesa	NM	8.62	Predicted	10.48
Kaye and Swetnam (1999)	Lower San Andreas	NM	9.74	Predicted	11.84
Morino (1996)	Upper Fillmore West	NM	10.55	Predicted	12.83
Grissino-Mayer & Swetnam (1997)	Cerro Bandera North	NM	10.71	Predicted	13.02
Swetnam and Dieterich (1985)	Gilita Ridge	NM	10.81	Predicted	13.14
Swetnam and Dieterich (1985)	McKenna Park	NM	11.05	Predicted	13.44
Kaye and Swetnam (1999)	Lower Pine Spring	NM	11.56	Predicted	14.06
Farris et al. (2013)	Monument Canyon	NM	-	Predicted	14.31
Brown et al. (2001)	James Ridge	NM	12.11	Predicted	14.73
Swetnam et al., no publication	Cerro Balitas	NM	12.32	Predicted	14.98
Morino (1996)	Upper Fillmore Side Cany. 1	NM	12.48	Predicted	15.18
Kaye and Swetnam (1999)	Upper San Andreas	NM	13.02	Predicted	15.83
Grissino-Mayer & Swetnam (1997)	Cerro Bandera East	NM	13.10	Predicted	15.93
Morino (1996)	Snag Saddle	NM	14.14	Predicted	17.19
Baisan and Swetnam (1997)	Capilla Peak Campground	NM	14.45	Predicted	17.57
Morino (1996)	Fillmore Side Canyon 2	NM	14.69	Predicted	17.86
Touchan et al. (1996)	Clear Creek Campground	NM	15.61	Calibrated	17.90
Grissino-Mayer & Swetnam (1997)	Candelaria	NM	14.97	Predicted	18.20
Grissino-Mayer & Swetnam (1997)	La Marchanita	NM	14.97	Predicted	18.20
Kaye and Swetnam (1999)	Cherry Canyon	NM	15.23	Predicted	18.52
Swetnam et al. (2001)	Black Mountain	NM	15.91	Predicted	19.35
Baisan and Swetnam (1997)	Canon de Turrieta	NM	16.02	Predicted	19.48
Morino (1996)	Rock House Spring	NM	16.10	Predicted	19.58
Brown et al. (2001)	Monument Canyon	NM	16.17	Predicted	19.66
Morino (1996)	Narrows	NM	16.42	Predicted	19.97
Kaye and Swetnam (1999)	Upper Pine Spring	NM	18.11	Predicted	22.02
Morino (1996)	Fillmore Side Canyon	NM	18.20	Predicted	22.13
Morino (1996)	Ledge Site	NM	18.44	Predicted	22.42
Brown et al. (2001)	Monument Canyon Upper	NM	18.66	Predicted	22.69
Touchan et al. (1996)	Pajarito Mountain Ridge	NM	19.04	Predicted	23.15
Baisan and Swetnam (1997)	La Luz Trail	NM	20.29	Predicted	24.67
Touch an et al. (1996)	Gallina Mesa	NM	18.54	Calibrated	24.70
Swetnam and Baisan (1996)	Ice Canyon	NM	21.58	Predicted	26.24
Swetnam (1990)	Bear Wallow	NM	21.74	Predicted	26.44

Swetnam and Baisan (1996)	Continental Divide Peak	NM	21.86	Predicted	26.58
Grissino-Mayer & Swetnam (1997)	Cerro Rendija	NM	22.19	Predicted	26.98
Grissino-Mayer & Swetnam (1997)	Mesita Blanca	NM	23.15	Predicted	28.15
Grissino-Mayer & Swetnam (1997)	Lost Woman	NM	23.50	Predicted	28.58
Swetnam et al., no publication	Laguna Garule	NM	23.79	Predicted	28.93
Grissino-Mayer & Swetnam (1997)	Hoya de Cibola Lava Flow	NM	24.03	Predicted	29.22
Swetnam and Baisan (1996)	El Calderon	NM	24.54	Predicted	29.84
Allen (1989)	Frijoles Canyon	NM	-	Predicted	30.88
Brown et al. (2001)	Delworth	NM	25.73	Predicted	31.29
Brown et al. (2001)	Fir Campground	NM	25.85	Predicted	31.43
Brown et al. (2001)	Peake Canyon	NM	28.63	Predicted	34.81
Touchan et al. (1996)	Camp May East	NM	28.91	Predicted	35.15
Brown et al. (2001)	Cosmic Ray Obs	NM	29.79	Predicted	36.22
Swetnam and Baisan (1996)	Continental Divide Saddle	NM	29.81	Predicted	36.25
Brown et al. (2001)	Sunspot	NM	29.89	Predicted	36.35
Baisan et al., no publication	Bonita Canyon	NM	30.70	Predicted	37.33
Touchan et al. (1996)	Canada Bonita South	NM	31.53	Predicted	38.34
Margolis and Balmat (2009)	Santa Fe Watershed Ponderosa Pine	NM	25.78	Calibrated	39.80
Touchan et al. (1996)	Cerro Pedernal	NM	33.56	Predicted	40.81
Grissino-Mayer & Swetnam (1997)	Hidden Kipuka	NM	38.90	Predicted	47.30
Margolis and Balmat (2009)	Santa Fe Watershed Dry Mixed Conifer	NM	49.46	Calibrated	74.70
OREGON					
Heyerdahl (1997), Heyerdahl et al. (2001)	Baker City	OR	18.11	Calibrated	15.30
Heyerdahl (1997), Heyerdahl et al. (2001)	Dugout	OR	21.39	Calibrated	15.30
Maruoka (1994)	Spring Mountain (12)	OR	16.40	Predicted	19.94
Heyerdahl (1997), Heyerdahl et al. (2001)	Baker City	OR	18.11	Calibrated	22.70
Maruoka (1994)	Seed Orchard (4)	OR	18.69	Predicted	22.73
Maruoka (1994)	Widow's Creek (1)	OR	19.72	Predicted	23.98
Maruoka (1994)	East Camp Creek (5)	OR	20.17	Predicted	24.53
Heyerdahl (1997), Heyerdahl et al. (2001)	Dugout	OR	21.39	Calibrated	24.80
Maruoka (1994)	Smoothing Iron Ridge (15)	OR	21.93	Predicted	26.67
Maruoka (1994)	Little Bear Burn (7)	OR	23.19	Predicted	28.20

Heyerdahl (1997), Heyerdahl et al. (2001)	Imnaha	OR	33.82	Calibrated	28.40
Maruoka (1994)	Five Mile Creek (6)	OR	24.11	Predicted	29.32
Maruoka (1994)	West Myrtle Creek (8)	OR	24.67	Predicted	30.00
Bork (1984)	Pringle Butte	OR	-	Calibrated	31.00
Heyerdahl, no publication	McKay Creek	OR	24.47	Calibrated	35.30
Heyerdahl (1997), Heyerdahl et al. (2001)	Imnaha	OR	33.82	Calibrated	37.50
Heyerdahl, no publication	Lytle Creek	OR	26.70	Calibrated	37.57
Maruoka (1994)	Raddue (2)	OR	33.27	Predicted	40.46
Heyerdahl, no publication	Green Ridge	OR	34.62	Calibrated	42.96
Maruoka (1994)	Troy (14)	OR	36.84	Predicted	44.80
Maruoka (1994)	Dixie Butte (3)	OR	43.64	Predicted	53.07
Bork (1984)	Lookout Mountain	OR	-	Calibrated	77.00
Bork (1984)	Cabin Lake	OR	-	Calibrated	79.00
Arabas et al. (2006)	Lava Cast Forest	OR	37.00	Calibrated	83.25
SOUTH DAKOTA					
Brown and Sieg (1999)	Pigtail Bridge	SD	17.42	Predicted	21.18
Brown and Sieg (1999)	Wind Cave North	SD	19.44	Predicted	23.64
Wienk et al. (2004)	Badger Game Prod. Area	SD	22.24	Predicted	27.04
Brown et al. (2008)	Mount Rushmore	SD	-	Calibrated	30.00
Brown (2003, 2006)	Black Hills Plot 105	SD	26.30	Predicted	31.98
Brown (2003, 2006)	Black Hills Plot 111	SD	26.85	Predicted	32.65
Brown and Sieg (1996)	Jewel Cave South	SD	27.20	Predicted	33.08
Brown (2003, 2006)	Black Hills Plot 204	SD	27.89	Predicted	33.91
Brown (2003)	Bear Lodge Central	SD	28.87	Predicted	35.11
Brown (2003, 2006)	Black Hills Plot 210	SD	28.95	Predicted	35.20
Brown (2003)	Reynold's Prairie	SD	28.95	Predicted	35.20
Brown (2003, 2006)	Black Hills Plot 213	SD	30.05	Predicted	36.54
Brown and Sieg (1996)	Jewel Cave North	SD	30.08	Predicted	36.58
Brown and Sieg (1999)	Gobbler Ridge	SD	31.02	Predicted	37.72
Brown (2003, 2006)	Black Hills Plot 207	SD	32.25	Predicted	39.22
Brown (2003)	Riflepit Gulch West	SD	32.72	Predicted	39.79
Brown (2003, 2006)	Black Hills Plot 202	SD	32.72	Predicted	39.79
Brown (2003, 2006)	Black Hills Plot 109	SD	33.02	Predicted	40.15
Brown and Sieg (1996)	Jewel Cave East	SD	33.58	Calibrated	40.52
Brown (2003, 2006)	Black Hills Plot 209	SD	34.04	Predicted	41.39

Brown et al. (2000)	Upper Pine Mid-Basin	SD	34.78	Predicted	42.29
Brown (2003)	Black Hills Exp. Forest	SD	35.22	Predicted	42.83
Brown (2003, 2006)	Black Hills Plot 112	SD	36.02	Predicted	43.80
Brown and Sieg (1996)	Jewel Cave Central	SD	36.47	Predicted	44.35
Brown (2003)	Bear Lodge North	SD	37.75	Predicted	45.90
Brown (2003, 2006)	Black Hills Plot 205	SD	38.42	Predicted	46.72
Brown (2003, 2006)	Black Hills Plot 106	SD	38.84	Predicted	47.23
Brown (2003, 2006)	Black Hills Plot 208	SD	40.05	Predicted	48.70
Brown (2003, 2006)	Black Hills Plot 113	SD	40.11	Predicted	48.77
Brown (2003, 2006)	Black Hills Plot 114	SD	40.75	Predicted	49.55
Brown (2003, 2006)	Black Hills Plot 203	SD	41.21	Predicted	50.11
Brown (2003)	Riflepit Gulch East	SD	41.35	Predicted	50.28
Brown (2003)	Riflepit Gulch North	SD	42.56	Predicted	51.75
Brown (2003, 2006)	Black Hills Plot 101	SD	42.75	Predicted	51.98
Brown (2003, 2006)	Black Hills Plot 206	SD	44.86	Predicted	54.55
Brown (2003, 2006)	Black Hills Plot 201	SD	46.10	Predicted	56.06
Brown (2003, 2006)	Black Hills Plot 110	SD	46.16	Predicted	56.13
Brown (2003, 2006)	Black Hills Plot 108	SD	63.33	Predicted	77.01
Brown (2003, 2006)	Black Hills Plot 103	SD	36.95 §	Predicted	90.16
Brown (2003, 2006)	Black Hills Plot 104	SD	64.33 ¶	Predicted	158.70
WASHINGTON					
Everett et al. (2000)	Entiat Mud Creek overall	WA	-	Calibrated	11.00
Everett et al. (2000)	Nile Creek overall	WA	-	Calibrated	12.20
Kernan and Hessl (2010)	Entiat	WA	-	Calibrated	13.10
Everett et al. (2000)	Entiat Mud Creek 165	WA	11.92	Predicted	14.49
Everett et al. (2000)	Entiat Mud Creek 230	WA	11.98	Predicted	14.57
Everett et al. (2000)	Entiat Mud Creek 201	WA	12.41	Predicted	15.09
Everett et al. (2000)	Entiat Mud Creek 205	WA	12.50	Predicted	15.20
Kernan and Hessl (2010)	Swauk	WA	-	Calibrated	15.80
Everett et al. (2000)	Entiat Mud Creek 199	WA	13.08	Predicted	15.91
Everett et al. (2000)	Entiat Mud Creek 202	WA	13.70	Predicted	16.66
Kernan and Hessl (2010)	Nile Creek	WA	-	Calibrated	17.00
Everett et al. (2000)	Nile Creek 10	WA	14.21	Predicted	17.28
Everett et al. (2000)	Entiat Mud Creek 196	WA	14.29	Predicted	17.38
Everett et al. (2000)	Entiat Mud Creek 207	WA	15.07	Predicted	18.33
Everett et al. (2000)	Entiat Mud Creek 203	WA	15.32	Predicted	18.63

Everett et al. (2000)	Nile Creek 5	WA	15.46	Predicted	18.80
Everett et al. (2000)	Quartzite 1	WA	15.54	Predicted	18.90
Everett et al. (2000)	Entiat Mud Creek 208	WA	16.02	Predicted	19.48
Everett et al. (2000)	Entiat Mud Creek 167	WA	16.17	Predicted	19.66
Everett et al. (2000)	Frosty 8	WA	16.38	Predicted	19.92
Wright (1996); Wright and Agee (2004)	Teanaway Demonstration Area	WA	16.43	Calibrated	20.20
Everett et al. (2000)	Quartzite 8	WA	16.73	Predicted	20.34
Everett et al. (2000)	Entiat Mud Creek 200	WA	16.84	Predicted	20.48
Everett et al. (2000)	Quartzite 6	WA	16.93	Predicted	20.59
Everett et al. (2000)	Nile Creek 3	WA	17.22	Predicted	20.94
Everett et al. (2000)	Quartzite 3	WA	17.40	Predicted	21.16
Everett et al. (2000)	Frosty 7	WA	17.47	Predicted	21.24
Everett et al. (2000)	Nile Creek 9	WA	17.66	Predicted	21.47
Everett et al. (2000)	Entiat Mud Creek 206	WA	17.78	Predicted	21.62
Everett et al. (2000)	Quartzite 2	WA	18.10	Predicted	22.01
Everett et al. (2000)	Nile Creek 4	WA	18.14	Predicted	22.06
Everett et al. (2000)	Frosty 4	WA	18.51	Predicted	22.51
Everett et al. (2000)	Frosty 3	WA	18.68	Predicted	22.71
Everett et al. (2000)	Frosty 2	WA	19.00	Predicted	23.10
Everett et al. (2000)	South Deep 1	WA	19.01	Predicted	23.12
Everett et al. (2000)	Quartzite 4	WA	19.17	Predicted	23.31
Everett et al. (2000)	Nile Creek 8	WA	19.36	Predicted	23.54
Everett et al. (2000)	Quartzite 5	WA	19.53	Predicted	23.75
Everett et al. (2000)	Entiat Mud Creek 204	WA	19.76	Predicted	24.03
Everett et al. (2000)	Nile Creek 2	WA	20.28	Predicted	24.66
Everett et al. (2000)	Nile Creek 6	WA	20.36	Predicted	24.76
Everett et al. (2000)	Frosty 1	WA	20.38	Predicted	24.78
Everett et al. (2000)	Nile Creek 1	WA	21.09	Predicted	25.65
Wright (1996); Wright and Agee (2004)	Teanaway Demonstration Area	WA	16.43	Calibrated	26.00
Everett et al. (2000)	Entiat Mud Creek 211	WA	21.71	Predicted	26.40
Everett et al. (2000)	Twenty Mile 3	WA	22.85	Predicted	27.79
Everett et al. (2000)	Frosty 6	WA	23.20	Predicted	28.21
Everett et al. (2000)	Quartzite 7	WA	24.28	Predicted	29.52
Everett et al. (2000)	Nile Creek 11	WA	24.65	Predicted	29.97

Heyerdahl (1997), Heyerdahl et al. (2001)	Tucannon	WA	39.80	Calibrated	30.50
Everett et al. (2000)	Twenty Mile 4	WA	25.60	Predicted	31.13
Everett et al. (2000)	Twenty Mile 1	WA	26.03	Predicted	31.65
Everett et al. (2000)	Frosty 5	WA	26.10	Predicted	31.74
Everett et al. (2000)	Twenty Mile 2	WA	26.68	Predicted	32.44
Everett et al. (2000)	Nile Creek 7	WA	29.34	Predicted	35.68
Everett et al. (2000)	Twenty Mile 6	WA	29.54	Predicted	35.92
Everett et al. (2000)	Twenty Mile 7	WA	31.24	Predicted	37.99
Everett et al. (2000)	South Deep 6	WA	31.58	Predicted	38.40
Everett et al. (2000)	Twenty Mile 8	WA	32.42	Predicted	39.42
Everett et al. (2000)	Twenty Mile 13	WA	33.75	Predicted	41.04
Heyerdahl (1997), Heyerdahl et al. (2001)	Tucannon	WA	39.80	Calibrated	41.40
Kernan and Hessl (2010)	South Deep	WA	-	Calibrated	45.30
Everett et al. (2000)	Twenty Mile 9	WA	40.17	Predicted	48.85
Everett et al. (2000)	Twenty Mile 12	WA	40.20	Predicted	48.88
Everett et al. (2000)	South Deep 7	WA	41.91	Predicted	50.96
Everett et al. (2000)	South Deep 3	WA	43.33	Predicted	52.69
Everett et al. (2000)	South Deep 5	WA	44.13	Predicted	53.66
Everett et al. (2000)	South Deep 9	WA	47.74	Predicted	58.05
Everett et al. (2000)	Twenty Mile 10	WA	48.96	Predicted	59.54
Everett et al. (2000)	South Deep 11a	WA	51.65	Predicted	62.81
Everett et al. (2000)	South Deep 11b	WA	51.65	Predicted	62.81
Everett et al. (2000)	South Deep 12	WA	51.65	Predicted	62.81
Everett et al. (2000)	South Deep 14	WA	51.82	Predicted	63.01
Everett et al. (2000)	South Deep 4	WA	53.37	Predicted	64.90
Everett et al. (2000)	South Deep 10	WA	55.09	Predicted	66.99
Everett et al. (2000)	Twenty Mile 11	WA	67.38	Predicted	81.93
WYOMING					
Brown (2003)	Cold Springs Creek	WY	24.63	Predicted	29.95
Brown et al. (2000)	Ashenfelder Lower	WY	45.88	Predicted	55.79
Brown et al. (2000)	Ashenfelder Upper	WY	45.88	Predicted	55.79
MEXICO					
Skinner et al. (2008)	PINO (San Pedro Martir)	MX	19.04	Predicted	23.15
Skinner et al. (2008)	BLAN (San Pedro Martir)	MX	22.18	Predicted	26.97
Skinner et al. (2008)	PYRA (San Pedro Martir)	MX	24.89	Predicted	30.27

Skinner et al. (2008)	WEST (San Pedro Martir)	MX	25.02	Predicted	30.42
Skinner et al. (2008)	TASA (San Pedro Martir)	MX	26.55	Predicted	32.28
Skinner et al. (2008)	VALL (San Pedro Martir)	MX	26.67	Predicted	32.43
Skinner et al. (2008)	PUER (San Pedro Martir)	MX	28.07	Predicted	34.13
Skinner et al. (2008)	CORO (San Pedro Martir)	MX	30.94	Predicted	37.62
Skinner et al. (2008)	AZUL (San Pedro Martir)	MX	55.99	Predicted	68.08
CANADA-BRITISH COLUMBIA					
Heyerdahl et al. (2012)	Middle Stein River Valley	BC	27.93	Calibrated	40.49

Notes
† Observations are in increasing order of calibrated/predicted PMFI/FR within each state
‡ Missing observations in this column occur because some calibration cases did not have an FHX file and did not report this statistic in the publication.
§ Mean ITFI could not be estimated, but mean CFI-all could be, is reported here, and was used to estimate PMFI/FR
¶ Mean ITFI could not be estimated, but mean CFI-10% could be, is reported here, and was used to estimate PMFI/FR

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Do Bark Beetle Outbreaks Increase Wildfire Risks in the Central U.S. Rocky Mountains? Implications from Recent Research

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CONSERVATION ISSUES

Do Bark Beetle Outbreaks Increase Wildfire Risks in the Central U.S. Rocky Mountains? Implications from Recent Research

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ABSTRACT: Appropriate response to recent, widespread bark beetle (*Dendroctonus* spp.) outbreaks in the western United States has been the subject of much debate in scientific and policy circles. Among the proposed responses have been landscape-level mechanical treatments to prevent the further spread of outbreaks and to reduce the fire risk that is believed to be associated with insect-killed trees. We review the literature on the efficacy of silvicutural practices to control outbreaks and on fire risk following bark beetle outbreaks in several forest types. While research is ongoing and important questions remain unresolved, to date most available evidence indicates that bark beetle outbreaks do not substantially increase the risk of active crown fire in lodgepole pine (*Pinus contorta*) and spruce (*Picea engelmannii*)-fir (*Abies* spp.) forests under most conditions. Instead, active crown fires in these forest types are primarily contingent on dry conditions rather than variations in stand structure, such as those brought about by outbreaks. Preemptive thinning may reduce susceptibility to small outbreaks but is unlikely to reduce susceptibility to large, landscape-scale epidemics. Once beetle populations reach widespread epidemic levels, silvicultural strategies aimed at stopping them are not likely to reduce forest susceptibility to outbreaks. Furthermore, such silvicultural treatments could have substantial, unintended short- and long-term ecological costs associated with road access and an overall degradation of natural areas.

Index terms: bark beetles, *Dendroctonus*, forest health, forest management, wildfire

INTRODUCTION

Forests in the western United States are being affected by the largest outbreaks of bark beetles in at least a century, which has caused concern about forest health and wildfire risk and led to proposals for tree removal in natural areas such as roadless forests. Such proposals stem in part from the rationale that bark beetle outbreaks increase wildfire risks due to increased dead fuels and that widespread treatment in beetle-affected forests is needed to lower such risks. Here, we review available peer-reviewed literature to determine if: (1) bark beetle outbreaks are associated with a higher incidence of wildfires in forest types in the central Rockies; and (2) if silvicultural treatments are effective at lowering beetle-associated tree mortality before, during, and after outbreaks. We briefly review the impacts that additional logging roads associated with broad-scale tree removal may have on the ecology of roadless natural areas. Our results may have broader policy implications in western forests as concerns over insect outbreaks have led to proposals to reduce environmental protections in favor of widespread thinning and post-disturbance tree removal.

INTERACTIONS AMONG FOREST INSECTS AND FIRES

We examined the long-standing belief that insect outbreaks lead to increased risk of fire (USDA Forest Service 2011). A large body of literature indicates that the occurrence of large, severe fires in subalpine, lodgepole pine (*Pinus contorta*), and spruce (*Picea engelmannii)*-fir (*Abies* spp.) forests is strongly contingent on climatic conditions, especially drought (e.g., Kipfmueller and Baker 2000; Romme et al. 2006; Sibold and Veblen 2006; Schoennagel et al. 2007; Jenkins et al. 2008; Simard et al. 2008, 2011).

The debate on how outbreaks affect fire risk and hazard is ongoing, but recent work emphasizes that the effect of outbreaks on subsequent fire risk is complex and is contingent on time since last outbreak and on biophysical setting. To date, the majority of studies have found no increase in fire occurrence, extent, or severity following outbreaks of spruce beetle (*Dendroctonus rufipennis*) and mountain pine beetle (*Dendroctonus ponderosae*) in Colorado, Wyoming, and other areas (Bebi et al. 2003; Kulakowski et al. 2003; Bigler et al. 2005; Kulakowski and Veblen 2007; Jenkins et al. 2008; Simard et al. 2008, 2011).

Theoretically, the effect of outbreaks on subsequent fires may vary with the time since the outbreak occurred (Romme et al. 2006). For example, it is reasonable to expect that foliar moisture in trees killed by beetles will decrease and canopy density will be reduced during and immediately after an outbreak. In subsequent years, canopy density may be further reduced as dead needles and small branches fall from killed trees reducing canopy bulk density, but increasing surface fire hazard (i.e., the type, volume, and arrangement of fuels that determines the ease of ignition and resistance to control regardless of the fuel type's weather-influenced moisture content). Although such a relationship may theoretically increase the risk of surface fires, studies on the influence of outbreaks on subsequent stand-replacing fires, over a range of years since outbreak, have found little or no increase in surface or canopy fire occurrence, extent, or severity (Bebi et al. 2003; Kulakowski et al. 2003; Bigler et al. 2005; Kulakowski and Veblen 2007; Jenkins et al. 2008; Simard et al. 2008, 2011) (Table 1).

Fire and Mountain Pine Beetle Outbreaks in Lodgepole Pine Forests

Although outbreaks of mountain pine beetle do alter fuel structure (Page and Jenkins 2007; Klutsch et al. 2009; Simard et al. 2011), the actual effects of these changes in fuels on subsequent fire risk (i.e., the chance that a fire might start based on all causative agents such as fuel hazard, ignition source, and weather) are complex, contradictory, and appear counterintuitive. For instance, lodgepole stands in which > 50 % of susceptible trees were killed by beetles in the 5 to 15 years preceding the 1988 Yellowstone fires had a higher incidence of crown fire than stands in which mortality was not as high (Turner et al. 1999). In contrast, stands with low to moderate beetle mortality (< 50% tree kill) had a lower incidence of high-severity crown fires. However, it is unclear whether these differences in fire behavior were primarily the result of the outbreak or of pre-outbreak stand structure (Simard et al. 2008), because beetle mortality occurred preferentially in older stands that were, in turn, inherently more likely to burn at high severity than younger stands because of differences in fuel structures even in the absence of beetle activity (Renkin and Despain 1992).

Other studies have found that beetle-kill may have decreased the hazard of high-severity crown fire by reducing the continuity of the canopy. For example, beetle-killed lodgepole pine stands, characterized by lower stand density, were affected by

Table 1. Forest types and relations between fire and insects in the Rocky Mountains.

Forest Type	Location	Insect-fire link	Citation
Lodgepole	Yellowstone	Beetle killed stands had significantly lower fire severity.	Omi 1997
Lodgepole	Yellowstone	Stands with higher mortally from bark beetles had higher incidence of crown fires. Stands with low to moderate beetle mortality had a lower incidence of crown fires.	Turner et al. 1999
Lodgepole	Yellowstone	Stands affected by outbreak in 1972-1975 were associated with a slightly higher probability of fire. Stands affected by outbreak in 1980-1983 were not more likely to burn.	Lynch et al. 2006
Lodgepole	Yellowstone	The probability of active crown fire in stands recently affected by beetles was significantly lower than in stands not affected by beetles.	Simard et al. 2011
Lodgepole and spruce	Colorado	Bark beetle outbreak did not affect the extent or severity of fire.	Kulakowski and Veblen 2007
Lodgepole and spruce	Intermountain west	Modeling study predicted a reduced risk of active crown fire 5 to 60 years after outbreaks.	Jenkins et al. 2008
Spruce	Colorado	Bark beetles caused no increase in the numbers of fires.	Bebi et al. 2003
Spruce	Colorado	Beetle-affected stands were not more susceptible to a low-severity fire.	Kulakowski et al. 2003
Spruce	Colorado	Previous bark beetle outbreaks had only a minor influence on fire severity.	Bigler et al. 2005
Spruce	Central Rocky Mountains	Modeling studies predicted reductions in the probability of active crown fire after bark beetle outbreaks.	DeRose and Long 2009

significantly lower fire severity compared to adjacent burned areas that had not been affected by beetles in the 3400-hectare Robinson Fire that burned in Yellowstone National Park in 1994 (Omi 1997). Lynch et al. (2006) also examined the influence of previous beetle activity on the 1988 Yellowstone fires by testing whether beetle-affected stands were more likely to have burned than those stands not affected by beetles. Stands affected by outbreak from 1972 to 1975 had a higher probability of burning, but the increase was relatively minor (about 11% greater compared to areas unaffected by beetles). In contrast, stands that were affected by outbreak from 1980 to 1983 were not more likely to burn in comparison to unaffected stands (Lynch et al. 2006).

It has been hypothesized that the risk of fire may increase only during and immediately after outbreaks of bark beetles when the dry red needles are still on the trees (Romme et al. 2006). However, Kulakowski and Veblen (2007) found that ongoing outbreaks of mountain pine beetle (and spruce beetle) did not affect the extent or severity of fire and suggested that changes in fuels brought about by outbreaks may be overridden by climatic conditions. Simard et al. (2011) examined fuel conditions for 35 years following outbreaks of mountain pine beetle in Yellowstone National Park. They documented reduced canopy moisture content after an outbreak, which was coupled with reduced canopy bulk density. In simulation models of fire behavior, under intermediate wind conditions (40 to 60 kilometers per hour), the probability of active crown fire in stands recently affected by beetles was significantly lower than in stands not affected by beetles (Simard et al. 2011). In addition, if winds were below 40 kph or above 60 kph, stand structure had little effect on fire behavior. Thus, although the canopy was drier immediately after an outbreak, no increase in fire risk was observed, likely because of the more important effect of reductions in canopy bulk density. Other modeling studies also have predicted a reduced risk of active crown fire 5 to 60 years after outbreaks, due to decreased canopy bulk density (Jenkins et al. 2008). In sum, outbreaks of bark beetles in lodgepole pine may have little or no effect on subsequent fires and may in some cases actually reduce the risk of fire.

Fire and Spruce Beetle in Subalpine Spruce-Fir Forests

There is increasing evidence that spruce beetle outbreaks have little or no affect on the occurrence or severity of fires in spruce-fir forests (Simard et al. 2008). It is well established that in this forest type, extensive fires are highly dependent on infrequent, severe droughts (e.g., Schoennagel et al. 2007). Under such extreme drought conditions, increased dead fuels from bark beetle outbreaks appear to play only a minor role, if any, in increasing fire risk. For instance, after a 1940s spruce beetle outbreak that resulted in dead-standing trees over thousands of hectares of subalpine forests in the White River National Forest of western Colorado, there was no increase in the numbers of fires compared to unaffected subalpine forests (Bebi et al. 2003). Beetle-affected stands were not more susceptible to a low-severity fire that spread through adjacent forests several years after the outbreak subsided (Kulakowski et al. 2003). During the extreme drought of 2002, large fires affected extensive areas of Colorado, including some spruce-fir stands that were previously affected by the 1940s outbreak of spruce beetle. Despite the expectation that these outbreaks would have led to an increased risk of severe fires, they had only a minor influence on fire severity (Bigler et al. 2005). Likewise, ongoing outbreaks of spruce beetle (and mountain pine beetle) had no detectable effect on the extent or severity of fires in 2002 (Kulakowski and Veblen 2007). These empirical findings are consistent with modeling studies that predict reductions in the probability of active crown fire for one to two decades after high-severity bark beetle outbreaks in pure stands of Engelmann spruce (*Picea engelmannii)* (DeRose and Long 2009). Other modeling studies have likewise predicted a reduced risk of active crown fire 5 to 60 years after outbreaks, due to decreased canopy bulk density (Jenkins et al. 2008).

The emerging view is that for lodgepole pine and spruce-fir forests: (1) the effect of bark beetle outbreaks on fuels is complex; and (2) weather and climate are more important in influencing fire risk and behavior the effects of insect outbreaks. When evaluating the influence of bark beetle outbreaks, it is important to recognize that outbreaks not only reduce foliar moisture content and increase the volume of dead wood, which can increase fire hazard, but that outbreaks also reduce canopy density, which can decrease fire risk (Simard et al. 2011). Therefore, when assessing the risk of wildfires following outbreaks, it is essential to recognize the relative importance of weather and climate to overall fire risk.

EFFICACY OF BARK BEETLE CONTAINMENT MEASURES

Prior to Outbreaks

The effectiveness of thinning to reduce forest susceptibility to bark beetles is believed to be related to tree vigor (Fettig et al. 2007); which may increase as moisture stress is decreased, and which in turn may make trees less susceptible to insect infestation. The premise is that if the trees are healthy and vigorous, they may be able to "pitch out" the attacking beetles, essentially flooding the entrance site with resin that can push out or drown the beetle (Figure 1).

Some studies have suggested that competition for light and water may reduce the vigor of surviving trees and increase susceptibility to bark beetle attacks (Fettig et al. 2007) and that thinning may, therefore, improve outbreak resistance. For instance, low-vigor ponderosa pine (*Pinus ponderosa)* in central Oregon was more often attacked by beetles than high-vigor trees during early stages of outbreaks (Larsson et al. 1983). Similarly, beetle activity has been associated with high tree densities in ponderosa pine and Douglas-fir *(Pseudotsuga* sp.) stands (Negrón et al. 2001; Negrón and Popp 2004). Ponderosa pine study plots in Colorado's Front Range infested by mountain pine beetle had significantly higher tree basal area and density (Negrón and Popp 2004). Douglas-fir beetles (*D. pseudotsugae*) more often attacked stands

Figure 1. Mountain pine beetle being pitched out. Photo taken by Whitney Cranshaw, Colorado State University, Bugwood.org.

containing a high percentage of basal area represented by high densities of Douglas-fir and slow growth during the five years prior to attack in Colorado's Front Range (Negrón et al. 2001).

Studies that have looked directly at thinning and its effects on tree vigor in Western forests have shown mixed results. While some studies have found that thinning reduces stand susceptibility in some circumstances (Fettig et al. 2007), other research has found bark beetles do not preferentially infest trees with declining growth. For example, Sánchez-Martínez and Wagner (2002) found that ponderosa pine forests of northern Arizona growing in dense stands were not more likely to be colonized by bark beetles.
Under some circumstances, thinning may alleviate tree stress at the stand level but is unlikely to be effective at mitigating susceptibility against extensive or severe outbreaks (Safranyik and Carroll 2006). Preisler and Mitchell (1993) found that thinned plots of lodgepole pine in Oregon were initially unattractive to mountain pine beetles; but when large numbers of attacks occurred, colonization rates were similar to those in unthinned plots. Similarly, Amman et al. (1988) studied the effects of spacing and diameter of trees and concluded that tree mortality was reduced as basal area was lowered. However, if the stand was in the path of an ongoing mountain pine beetle epidemic, spacing and density of trees had little effect (Amman et al. 1988).

While thinning has the potential to reduce tree stress, which can reduce susceptibility to insect attack, it also has the potential to bring about other conditions that can increase susceptibility. For example, thinning may injure surviving trees and their roots, which can provide entry points for pathogens and ultimately reduce tree resistance to other organisms (Hagle and Schmitz 1993; Paine and Baker 1993; Goyer et al. 1998). Although thinning can be effective in maintaining adequate growing space and resources, there is accumulating evidence to suggest that tree injury, soil compaction, and temporary stress due to changed environmental conditions caused by thinning may increase susceptibility of trees to bark beetles and pathogens (Hagle and Schmitz 1993).

From an adaptive management standpoint, it is most prudent to implement thinning in appropriate settings (e.g., already degraded areas in need of restoration) with sufficient controls that would lead to an improved understanding of the efficacy of these approaches, particularly under a range of climatic conditions. It is also important to consider how such strategies may alter normal stand structure. For example, thinning in Engelmann spruce forests is likely to create novel conditions that would be atypical for these ecosystems due to their naturally high tree densities (Daubenmire 1943). Further, thinning forest stands before epidemics is not likely to prevent major outbreaks, due to the inherent difficulties of manipulating stand structure over large enough areas and the overriding influence of climatic stress in driving outbreaks.

During Outbreaks

There is general agreement that silvicultural treatments cannot effectively stop outbreaks once a large-scale insect infestation has started. Citing multiple sources, Hughes and Drever (2001) found that most control efforts have had little effect on the final size of outbreaks. In another review, Romme et al. (2006) point out that once an extensive outbreak has started, timber management is unlikely to stop it. Control of such outbreaks is theoretically possible, but it would require treatment of almost all of the infected trees (Hughes and Drever 2001). Amman and Logan (1998) point to failed attempts to use direct control measures, such as pesticides and logging, after an infestation starts. They suggest that by the early 1970s, it was apparent that attempts to control the extensive mountain pine beetle outbreaks that were occurring in the northern Rockies, by directly killing the beetles, were not working.

If a bark beetle infestation is relatively restricted and concentrated in a limited area, it may be feasible to reduce the impact of that outbreak by removing infested trees from a forest stand, or by thinning a stand to reduce stress of trees competing for limited nutrients, sunlight, and moisture. However, specific climatic conditions are believed to be required for beetle populations to reach epidemic levels. As such, a small population of beetles is not sufficient for an outbreak to occur and would not necessarily lead to an outbreak. Conversely, under climatic conditions favorable for an outbreak, bark beetle outbreaks can erupt simultaneously in numerous, dispersed stands across the landscape. Thus, even if a growing population of beetles is successfully removed from one stand or the stand is thinned to increase vigor, under climatic conditions suitable for outbreaks, beetles from other stands are likely to spread over a landscape. Given that climate typically favors beetle populations and stresses trees over very large areas, successfully identifying and treating stands over a large enough region to have a significant impact on the overall infestation is impractical and costly.

Following Outbreaks

Post-disturbance harvest is common practice on forest lands and is designed to remove trees or other biomass in order to produce timber or other resources. This type of resource extraction has the potential to inadvertently lead to heightened insect activity (Nebeker 1989; Hughes and Drever 2001; Romme et al. 2006). In particular, snags and fallen logs contribute to the protection of soils and water quality and provide habitat for numerous cavity- and snag-dependent species (Romme et al. 2006), many of which prey on bark beetles and other economically destructive insects. Therefore, outbreaks could be prolonged because of a reduction in the beetle's natural enemies (Nebeker 1989), including both insects and bird species that feed on mountain pine beetles (Koplin and Baldwin 1970; Shook and Baldwin 1970; Otvos 1979). Furthermore, post-disturbance harvest can damage soil and roots by compacting them (Lindenmayer et al. 2008) leading to greater water stress in trees, which may reduce conifer regeneration by increasing sapling mortality (Donato et al. 2006) and, in general, may cause more damage to forests than that caused by natural disturbance events (DellaSala et al. 2006).

ROAD BUILDING FOR BARK BEETLE CONTROL

A broad scale program to treat forests that have been affected by bark beetle will require an extensive road system, which will likely have significant impacts to forest and aquatic ecosystems.

In general, the major physical results of roads on the terrestrial environment are increases in forest fragmentation and disruption of the movement of organisms and flow of ecological processes across the landscape (Lindenmayer and Fisher 2006). Aquatic systems have been impacted through the disruption of natural infiltration of water into the soil and increased runoff to streams (Forman and Alexander 1998). These effects have been particularly pronounced in mountainous regions, especially on high gradient streams and headwaters (Ziegler et al. 2001). Increased sediment input to streams can result in changes to channel morphology and channel substrate, as well as the creation of shallow pools (Beschta 1978). These changes to stream structure, an indirect effect of road construction, often adversely affect native fish habitat. Thus, any road network constructed to thin or harvest insect-infested stands will have to be carefully engineered to prevent increased sedimentation rates or alteration of hill slope processes (Beschta 1978). While proper engineering can help mitigate some negative effects, it does not mitigate the overall impact of roads on hydrologic processes, water flow, and fragmentation of wildlife habitat.

CONCLUSIONS AND RECOMMENDATIONS

Climate change and other factors are leading to unprecedented changes in western forest ecosystems (Logan et al. 2003; Carroll et al. 2004; Breshears et al. 2005; Bentz et al. 2009). One consequence of recent and predicted climate change is increased bark beetle activity leading to tree mortality over large areas (Logan and Powell 2001; Williams and Liebhold 2002; Carroll et al. 2004). Such ongoing outbreaks have led to widespread public concern about increased fire risk; however, outbreaks of mountain pine beetle and spruce beetle do not appear to substantially increase the risk of subsequent fire under most conditions. Instead, fire risk in spruce-fir and lodgepole pine is strongly tied to warm and dry conditions, such as those of recent decades. Insect containment measures have yielded mixed results and may pose significant risks to forested ecosystems. We recommend that priority be given to removing hazardous trees, which were killed by fire or insects and that might fall across roads or in campgrounds in areas of high human use to limit damages and potential loss of life. Moreover, in order to reduce existing and future risks of fire, it would be prudent to concentrate fuel reduction measures in the wildland-urban interface by creating defensible space, as the 40-meter zone around homes and structures has been shown to be critical to a home's ignitability (Cohen 1999). Thus, to be effective at reducing fire hazard to communities, tree-cutting can be directed at removing all flammable material (not just economically valuable timber) in the immediate vicinity of homes and settlements.

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Influence of Pre-Fire Tree Mortality on Fire Severity in Conifer Forests of the San Bernardino Mountains, California

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Abstract: High tree mortality due to drought and insects often is assumed to increase fire severity once ignition occurs. In 2002-2003, coniferous forests in the San Bernardino Mountains, California experienced a significant tree mortality event due to drought and an outbreak of western pine beetles (*Dendroctonus brevicomis*). In October 2003, fire burned approximately 5,860 ha of conifer forest types in many beetle- and drought-affected stands where most pre-fire dead trees had retained needles. We used pre- and post-fire GIS data to examine how fire severity was affected by pre-fire tree mortality, vegetation characteristics, and topography. We found no evidence that pre-fire tree mortality influenced fire severity. These results indicate that widespread removal of dead trees may not effectively reduce higher-severity fire in southern California's conifer forests. We found that sample locations dominated by the largest size class of trees (≥61 cm diameter at breast height (dbh)) burned at lower severities than locations dominated by trees 28-60 cm dbh. This result suggests that harvesting larger-sized trees for fire-severity reduction purposes is likely to be ineffective and possibly counter-productive.

INTRODUCTION

Tree mortality due to drought and insect attacks is common in western coniferous forests [1], but may be increasing in recent years in some areas [2, 3]. Bark beetles (Coleoptera: Scolytidae) are common native insects that kill firs and pines, and are capable of large-scale population increases following disturbances such as droughts [4, 5]. Dense forests are considered relatively more susceptible to insect mortality [6, 7], and recent studies have concentrated on how prescribed fire and thinning affect susceptibility of trees to insects [5, 8, 9]. However, few data are available on the influence of tree mortality on fire behavior.

Stands with high tree mortality due to drought and insects often are presumed to burn at higher severity during fires, increasing the mortality of dominant overstory vegetation in the stand [10, 11]. This assumption is based on expectations of greater dead fine and coarse fuel loads, including canopy fuels, resulting from pre-fire mortality [11]. The hypothesis that insect-caused tree mortality increases fire severity has relied upon two principal assumptions: (1) dead needles remaining on trees could increase the amount and vertical continuity of fine, dry fuels [11, 12]; and (2) tree mortality could open the canopy and intensify seasonal desiccation of understory fuels [12]. However, the few empirical studies testing this hypothesis have not found support for it. A widespread low-severity fire in subalpine forest in the White River National Forest, Colorado did not burn any stands affected by spruce beetle (*Dendroctonus rufipennis*) outbreaks that occurred several decades prior to the fire [13]. Furthermore, a regional analysis of 303 fires in the White River National Forest found that beetle-affected stands did not burn at higher severities than unaffected stands in fires occurring several decades after the outbreak [12].

The hypothesis that stands with recent high tree mortality due to drought or insects have an elevated probability of burning at higher severity when a fire occurs has never been empirically tested. We examined whether fire severity in two large fires that occurred in the midst of a tree mortality event was influenced by the number of trees killed by drought and insects. Specifically, we investigated whether pre-fire tree mortality increased fire severity in stands after ignition occurred. We did not examine the probability of fire igniting in a stand over broad spatial and temporal scales [e.g., 12, 14].

Beginning in rainfall year 1998-1999, southern California entered a period of major drought and higher temperatures. In 2000, the San Bernardino National Forest began to document unusually high mortality of incense-cedars (*Calocedrus decurrens*), and in 2001 slightly increased mortality was witnessed in ponderosa (*Pinus ponderosa*), Coulter (*P. coulteri*) and Jeffrey (*P. jeffreyi*) pines (L. Merrill, USDA Forest Service, unpublished data 2003).[1] In 2002, an outbreak of western pine beetles (*D. brevicomis*)

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resulted in what the USDA Forest Service identified as 'above background' mortality levels of ponderosa and Coulter pines, and many other conifer tree species were dying from drought alone (L. Merrill, USDA Forest Service, unpublished data 2003). In the first half of 2003, both western pine beetles and mountain pine beetles (*D. ponderosae*) were actively colonizing and killing thousands of conifer trees. By April 2003, the San Bernardino National Forest had mapped approximately 70,000 ha with elevated levels of conifer mortality (L. Merrill, USDA Forest Service, unpublished data 2003).

In late October 2003, one year after the beginning of the beetle population outbreak, two large human-ignited fires merged together in the San Bernardino Mountains and burned 5,863 ha of conifer and conifer-hardwood forest types, including stands with high levels of tree mortality due to drought and insects (Fig. **1**). The Old and Grand-Prix fires were driven by hot Santa Ana winds which typically sweep through southern California during the fall [15]. No widespread harvest of the beetle- and drought-killed trees had occurred at the time of the fires.

MATERIALS AND METHODOLOGY

We selected the San Bernardino Mountains study area because of the existence of Geographic Information System (GIS) layers depicting structural characteristics of vegetation, topography, pre-fire tree mortality immediately prior to the fires, and fire severity, allowing us to investigate the influence of numbers of recently dead trees on fire severity. We simultaneously investigated the effects of topography (slope and aspect), tree size, and canopy cover on fire severity in burned stands, because these factors also are known to influence fire behavior [16, 17].

Conifer forests in the San Bernardino Mountains consist of mixed-evergreen forests [18] below 1,500 m, and ponderosa pine, Jeffrey pine, Coulter pine, white fir (*Abies concolor*)–sugar pine (*P. lambertiana*), and bigcone Douglas-fir (*Pseudotsuga macrocarpa*) stands above 1,500 m [19, 20]. Various combinations of white fir, Jeffrey pine, ponderosa pine, Coulter pine, sugar pine, incense-cedar, and black oak *(Quercus kelloggii*) occur at higher elevations, and canyon live oak (*Q. chrysolepis*) and bigcone Douglas-fir

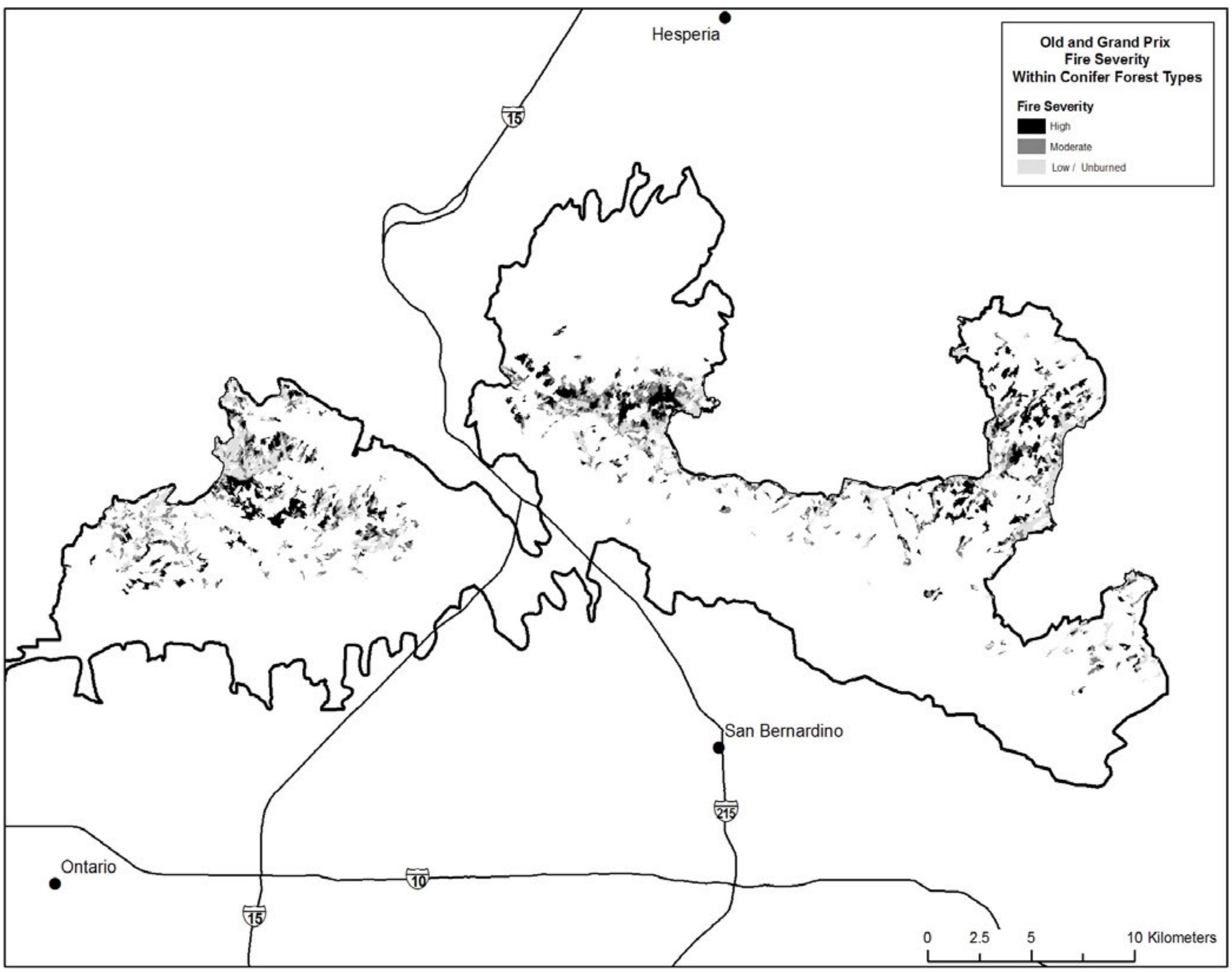

Fig. (1). Perimeters of the 2003 Old and Grand-Prix fires and RdNBR fire severity (low/unburned, moderate, moderate/high) within conifer forest types (Jeffrey Pine, Sierra Mixed Conifer, Montane Hardwood-Conifer). White areas within the fire perimeter are non-conifer vegetation.

dominate at lower elevations [21]. Historic fire return intervals in these forests were variable, with some forest types exhibiting relatively longer fire-free intervals associated with mixed-severity fire effects [20].

We acquired GIS data on vegetation type and structure [22] and pre-fire tree mortality [23] from the USDA Forest Service and GIS data on fire severity [24] and topography [25] from the US Geological Survey. The detailed methodology used by the agencies to create these GIS maps was explained in the metadata for the layers, and is summarized here. Our variables of interest were vegetation type, size of dominant trees, canopy cover, slope, aspect, number of dead trees per ha prior to the fire, and fire severity.

Vegetation type, size class of dominant trees, and canopy cover were derived from a map of existing vegetation from 2002-2003 (EVEG Tiles) [22]. The vegetation layer was generated using a combination of automated systematic procedures, remote-sensing classification, and photo editing and ground surveys to reduce bias while mapping large areas. Minimum mapping size for contrasting vegetation conditions based on cover type, vegetation type, tree cover, and diameter class was 1 ha and pixel size was 30 m.

Cover types were delineated using Landsat Thematic Mapper imagery into the following broad classes: (1) Conifer = >10% conifer cover as dominant type; (2) Mix = >10% tree cover and 20-90% hardwood cover; (3) Hardwood = >10% hardwood cover as dominant type; (4) Shrub = >10% shrub cover as dominant type; (5) Grass = >10% grass cover as dominant type; (6) Barren = <10% cover of any natural vegetation; (7) Agriculture; (8) Urban; and (9) Ice/snow. Attributes including tree cover from above and overstory tree diameter interpreted from aerial photography and satellite imagery were then mapped within the cover type classes and used to develop additional classifications. We used California Wildlife Habitat Relations (WHR) [26] to describe specific vegetation types, canopy cover, and tree size-class. "*WHR vegetation type,*" is derived primarily from CALVEG cover type and relative cover of conifer and hardwood trees for mixed vegetation types. For our study area, the *WHR vegetation types* consisted of Jeffrey Pine, Sierra Mixed Conifer, Montane Hardwood-Conifer, Eastside Pine, and Closed-cone Pine-Cypress. "*WHR density*" is a measure of tree density indexed by percent canopy cover and included: Sparse (10.0-24.9%), Poor (25.0-39.9%), Moderate (40.0-59.9%), and Dense (≥60%). "*WHR size*" identified size classes of overstory trees. *WHR size* included the following three classes: WHR size 3 = dominated by trees 15-27 cm dbh; WHR size 4 = dominated by trees 28-60 cm dbh; WHR size 5 = dominated by trees ≥61 cm dbh.

The GIS layer depicting tree mortality was created from annual aerial surveys conducted by the USDA Forest Service. Current-year tree mortality from 2001-2003 was sketch-mapped by an aerial observer who quantified the number of yellow to reddish brown trees. Polygons were categorized by mortality type (drought or insect kill) and number of trees affected per acre (we converted acres to hectares for this study). Generally, areas with <1 tree per acre of mortality were considered to have background levels of mortality and were not usually mapped during the flight. The resulting layer is a vector data set of polygons each associated with a level of tree mortality for that year. Each year's layer was non-cumulative with respect to numbers of dead trees; however, we used only the 2003 GIS map in our analyses because (1) prior to 2003, few polygons showed above-background levels of mortality within the fire perimeter and (2) we were interested only in very recent mortality since these trees were most likely to have retained dead needles to potentially contribute to fire severity. Therefore, the actual number of all dead trees in a given polygon was likely higher than reported herein.

The fire severity GIS data of the 2003 Old and Grand Prix fires were derived from Landsat Thematic Mapper data. Pre-fire and post-fire data were used to create a Relative delta Normalized Burn Ratio (RdNBR) image, which portrays fire severity to vegetation within a fire while accounting for variation in pre-fire live tree cover, as described in Miller and Thode [27]. Because we were interested in ascertaining whether pre-fire tree mortality influenced fire severity, we used a relative rather than absolute index. Absolute dNBR measures how much vegetation was killed by the fire, while RdNBR measures the amount of vegetation killed in relation to the amount of pre-fire vegetation [27]. Miller and Thode [27] found that RdNBR more accurately classified high-severity fire effects than dNBR in heterogeneous landscapes with variable amounts of pre-fire vegetation, such as our study area in the San Bernardino Mountains. Higher RdNBR values are correlated with more severe burning of vegetation. The RdNBR image was classified into 4 classes of fire severity based on cutoff thresholds informed by field data collected on understory, midstory, and overstory vegetation one year post-fire on several fires from 2001 through 2004 using Composite Burn Index (CBI) protocols [27]. We used CBI classifications because they provide information about fire effects on all vegetation strata from the forest floor to the upper canopy, and are a useful and easily understood measurement for managers.

The fire severity map identified 4 classes of fire severity. "*Unchanged*" included areas in which conditions one year after the fire were indistinguishable from pre-fire conditions. "*Low Severity*" represented areas of surface fire with little change in cover and little mortality of the dominant vegetation. "*Moderate Severity*" was between low and high and represented a mixture of effects on the dominant vegetation. "*High Severity*" represented areas where the dominant vegetation had high to complete mortality of canopy foliage due to the fire. We used this classification system to represent the severity of fire in the forest canopy in our analyses. For areas mapped as high severity using RdNBR, we categorized these as "moderate/high severity" because RdNBR measures fire-induced mortality of canopy foliage, rather than tree mortality. The RdNBR high-severity mapping category has a lower threshold of 80% canopy mortality, which equates to 65% tree mortality for trees >20 cm dbh [28]. Basal area mortality would likely be somewhat lower than 65%, since the larger trees that dominate in terms of basal area are less fire-susceptible than the abundant small trees that dominate in terms of tree density [29].

The GIS layers of vegetation type and structure, pre-fire tree mortality, and a Digital Elevation Model [25] were clipped to the Old and Grand Prix fire perimeters. We selected conifer and mixed hardwood-conifer type polygons from within the vegetation layer for analyses. We generated 500 randomly located points throughout the conifer and mixed hardwood-conifer forest types to create a table of sample stand locations. At each sample location we determined the values of the variables: (1) *slope* [%]; (2) *aspect* [degrees]; (3) *mortality* [drought and beetle killed only] expressed as the number of dead trees per ha from year 2003; (4) *WHR vegetation type*; (5) *WHR size*; (6) *WHR density*; and (7) *fire severity*.

We removed 31 locations from our sample due to small sample sizes within specific categories, including: (1) the 5 locations where *WHR size* = 3; (2) 8 locations of various *WHR vegetation types* that had <5 samples in categories; and (3) 18 locations in the Closed-Cone Pine-Cypress type. Final sample size was 469 random points in *WHR types* Jeffrey Pine, Sierra Mixed Conifer, and Montane Hardwood-Conifer (hereafter conifer forest), in *WHR size* classes 4 and 5. *WHR densities* were modified from categorical variables to the mean value of each category (17.5%, 32.5%, 50%, and 80%). Pre-fire tree mortality data was expressed in terms of total number of dead trees per ha in 2003 (immediately prior to the October fires).

We analyzed how fire severity was affected by pre-fire insect and drought mortality along with topography, tree size, and canopy cover variables using two model structures best suited to categorical response variables: binomial and rank-ordered logistic. For the binomial method, we created a generalized linear model (GLM) using a binomial error structure and a logit link function to examine the effects of explanatory variables on the probability that each randomly selected location experienced moderate/high severity fire. The binomial response variable was moderate/high severity burn = 1; and unchanged, low, or moderate severity burn = 0. For the rank-ordered method we performed ordered logistic regression (OLR) to fit an ordered logit model examining how explanatory variables affected the probability that each randomly selected location burned at low, moderate, or moderate/high severity. Our response variable, *fire severity*, was treated as ordinal under the assumption that the levels of *fire severity* have a natural ordering (low to moderate/high), but the distances between adjacent levels are unspecified. All analyses were performed using Stata 8.2 (Stata Corp. 2004, College Station, Texas 77845).

We generated binomial categorical variables for *aspect* (south, east, and west), *WHR type* (Sierra Mixed Conifer and Montane Hardwood-Conifer), and *WHR size* 5, conditioning the model on north-facing slopes of Jeffrey Pine dominated by trees 28-60 cm dbh (*WHR size* 4). *WHR density* was included to control for variation in stand density (canopy cover) within mortality polygons and across the landscape. *Slope*, *aspect*, *WHR size*, and *WHR vegetation type* variables were included because all of these factors can influence fire behavior [16, 17].

We used trend surface analysis to model broadscale spatial pattern in the burn-severity data as a control for spatial autocorrelation. This methodology has two primary aims [30, 31]: (1) to guard against false correlations between fire severity and explanatory variables, as may arise when an unmeasured environmental factor causes a common spatial structure in fire severity and in the measured explanatory variables; and (2) to determine if there is a substantial amount of broadscale spatially structured variation in the fire-severity data that is unexplained by the measured explanatory variables. We fitted a trend surface to fire severity by including variables for x and y spatial coordinates of each sample location, polynomial terms up to the third-degree, and interactions. Prior to analysis, x and y were centered on their respective means to reduce collinearity with higher-order terms [31] and standardized to unit variance. Nonsignificant ($P > 0.05$) trend surface terms were removed by stepwise selection.

RESULTS

Fire severity in the Old and Grand Prix fires was highly variable, as is typical of forest fires, leaving patches of unburned and lightly burned areas intermixed with moderate and moderate/high severity patches (Fig. **1**). Throughout conifer forest, the fires burned 1,882 ha (32%) at moderate/high severity; 2,010 ha (34%) at moderate severity; 1,385 ha (24%) at low severity; and 586 ha (10%) remained unchanged. The distribution of fire severity categories of our sample locations closely matched the distribution of fire severities in conifer forest throughout the study area (32% at moderate/high severity; 34% at moderate severity; 23% at low severity; and 12% remained unchanged). Tree mortality due to drought and beetle infestation prior to the fire ranged from an average of 0 to 21.83 dead trees per ha in each polygon. In smaller patches within a polygon the density of dead trees may have been much higher. Fifty percent of our sample locations had no pre-fire tree mortality above background level. Of the remaining 50% of our sample locations with above-background tree mortality levels, most observations were evenly distributed among four categories: (1) < 2.47; (2) 7.41-12.35; (3) 14.83; and (4) 19.77-22.24 dead trees per ha. The original data were reported in these categories and were expressed in terms of dead trees per acre. We converted acres to hectares to derive our dead tree density values.

The GLM indicated that pre-fire tree mortality due to drought and beetle infestation did not significantly affect the probability that a location within the fire burned at moderate/high severity ($P = 0.88$; Table **1**), while controlling for the effects of topography and vegetation characteristics. Burned locations in Montane Hardwood-Conifer vegetation were significantly more likely ($P = 0.04$) to burn at moderate/high severity than locations in Sierra Mixed Conifer or Jeffrey Pine vegetation. Western aspect decreased the probability of moderate/high severity fire ($P < 0.10$; Table **1**). The pseudo r^2 value was 0.067, indicating that 7% of the variation in probability of high-severity fire was explained by our model.

Similarly, the OLR indicated that pre-fire tree mortality did not increase the probability that a location within the fire area burned at higher severity ($P = 0.53$; Table **2**). Montane Hardwood-Conifer vegetation significantly increased the probability that a location burned at higher severity than Sierra Mixed Conifer or Jeffrey Pine vegetation ($P < 0.001$; Table **2**). Sample locations with western aspect and those dominated by trees $\geq$61 cm dbh were more likely ($P < 0.10$) to burn at lower severities relative to locations with north

Table 1. Table of Coefficients from a Binomial Generalized Linear Model (GLM) Examining Effects of Pre-fire Tree Mortality, Slope, Aspect, Vegetation Type, Tree Size Class, and Canopy Cover Class on Probability of Moderate/High Severity Fire in Conifer Types within the October 2003 Old and Grand Prix Fires in the San Bernardino National Forest, California (*n* = 469).

Variable	Coeff.	SE	z	P>|z|	95% CI	
Insect/drought mortality	0.005	0.035	0.15	0.882	-0.064	0.074
Slope	0.001	0.002	0.35	0.726	-0.002	0.003
East	-0.062	0.263	-0.24	0.812	-0.577	0.452
South	-0.293	0.345	-0.85	0.395	-0.970	0.383
West **	-0.585	0.300	-1.95	0.051	-1.173	0.003
WHR size 5	-0.361	0.228	-1.58	0.114	-0.808	0.087
WHR type MHC *	0.575	0.283	2.03	0.043	0.019	1.130
WHR type SMC	-0.637	0.698	-0.91	0.362	-2.004	0.731
WHR density	0.011	0.007	1.55	0.120	-0.003	0.024
Y-coordinate *	1.3E-04	4.2E-05	3.17	0.002	5.2E-05	2.2E-04
X-coordinate^2 *	2.5E-09	7.5E-10	3.24	0.001	9.7E-10	3.9E-09
Intercept	-2.247	0.701	-3.21	0.001	-3.620	-0.874

* = P < 0.05.
** = 0.05 < P < 0.10.

Table 2. Table of Coefficients from Ordered Logistic Regression (OLR) Examining Effects of Pre-fire Tree Mortality, Slope, Aspect, Vegetation Type, Tree Size Class, and Canopy Cover Class on Fire Severity (Low, Moderate, Moderate/High) in Conifer Types within the October 2003 Old and Grand Prix Fires in the San Bernardino National Forest, California (*n* = 415).

Variable	Coeff.	SE	z	P>|z|	95% CI	
Insect/drought mortality	0.020	0.032	0.63	0.532	-0.043	0.083
Slope	0.000	0.001	-0.16	0.874	-0.003	0.002
East	-0.191	0.240	-0.80	0.425	-0.662	0.279
South	-0.051	0.311	-0.16	0.870	-0.659	0.558
West **	-0.455	0.254	-1.79	0.073	-0.953	0.043
WHR size 5 **	-0.343	0.206	-1.67	0.095	-0.746	0.060
WHR type MHC *	0.915	0.244	3.75	<0.001	0.437	1.394
WHR type SMC	-0.302	0.577	-0.52	0.601	-1.433	0.829
WHR density	0.001	0.006	0.14	0.891	-0.010	0.012
XY *	5.8E-09	1.9E-09	3.03	0.002	2.1E-09	9.6E-09
Cutpoint 1	-0.634	0.521				
Cutpoint 2	1.109	0.524				

* = P < 0.05.
** = 0.05 < P < 0.10.

aspect or those dominated by trees 28-60 cm dbh (Table **2**). The pseudo r^2 value of 0.04 suggested that 4% of the variation in fire severity among locations was explained by our model.

DISCUSSION

We found that stands with recent high pre-fire tree mortality due to drought and insects did not burn at higher severity in coniferous forests of the San Bernardino Mountains, southern California, in the two fires we examined. Pollet and Omi [32] reported anecdotally that stands of lodgepole pine (*P. contorta*) that experienced an insect epidemic in the 1940s in Yellowstone National Park burned at lower severities compared to adjacent burned areas in the 1994 Robinson Fire. A widespread low-severity fire in subalpine forests in the White River National Forest, Colorado did not burn any beetle-affected stands [13]. Further, Bebi *et al.* [12] found that stands of Engelmann spruce (*Picea engelmannii*) and subalpine fir (*A. lasiocarpa*) in the White River National Forest influenced by a spruce beetle outbreak in the 1940s did not show higher susceptibility to 303 subsequent forest fires that burned after 1950. Our study area differed from these previous sites because most of the trees killed by insects and drought just prior to the fires in the San Bernardino Mountains were still standing and had retained needles. Despite differences in sites and forest types, previous studies and our results provide compelling evidence that when fire does occur, stands with considerable tree mortality due to drought and insects will not burn at higher severity than stands without significant tree mortality, either in the short or long term.

While pre-fire tree mortality had no effect on fire severity in burned stands, we found that sample locations dominated by the largest size class of trees ($\geq$61 cm dbh) burned at lower severities than locations dominated by trees 28-60 cm dbh (Table **2**). This result suggests that harvesting larger-sized trees for fire-severity reduction purposes is likely to be ineffective, and possibly counter-productive. These findings corroborate other recently published studies indicating that retention of the largest trees is likely to maintain normative fire behavior [33-35]. The smallest tree-size classes were not included in our analyses due to low sample sizes, so we could not determine the effects of still smaller tree-size classes on fire severity. An additional limitation on the potential effectiveness of fuel treatments to reduce fire severity in stands with high pre-fire mortality is the low likelihood that such stands will be affected by fire [14].

Weather conditions can supersede the influence of stand structure and fuels on fire behavior in mixed-severity fire regimes [36], which probably accounts for the low r^2 values of our models. We included topographical and stand structure variables, but we had no variables for wind speed, air temperature, and fuel and air moisture levels, for example. Odion and Hanson [36] analyzed the spatial patterns of fire severity for conifer forests in the three largest fires in the Sierra Nevada Mountains, California since 2000, and found that high-severity fire ranged from 10.9 to 28.9% of total area burned. Overall, we documented that 32% of conifer and mixed hardwood-conifer types burned at moderate/high severity in the 2003 Old and Grand Prix fires. The Old and Grand Prix fires may have had relatively high proportions of moderate/high severity due to the extreme fire weather resulting from Santa Ana winds, the lack of large-tree components due to past harvest, or some combination thereof.

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Chapter 4

c0020 # ~~Mammal Habitat Selection~~

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s0010 ## 4.1 INTRODUCTION

p0010 Mammals are ecologically and economically important members of the landscapes in which they live. Large herbivores like deer (*Odocoileus* spp.) and elk (*Cervus elaphus*), and predators like bears (*Ursus* spp.) and wolves (*Canis lupus*), are highly conspicuous and well-known "flagship" mammal species, whereas rodents, bats, and mustelids are cryptic but no less important in their ecosystems. Many species have developed broad ecological tolerance from exposure to environmental variation and natural disturbances over long time periods (Lawler, 2003). However, widespread hunting and excessive habitat fragmentation of landscapes by modern-day humans are qualitatively and quantitatively different from the natural disturbances to which these mammals were exposed in the past (Spies and Turner, 1999), and they have resulted in contraction of historical ranges and population declines. In North America alone notable population declines include elk, grizzly bears (*Ursus arctos*), gray wolves, Canadian lynx (*Lynx canadensis*), bighorn sheep (*Ovis canadensis*), beaver (*Castor canadensis*), the larger species of forest mustelids, and several herteromyid rodents.

p0015 Mixed- and high-severity wildfire is a natural disturbance in many vegetation systems of North America, the Mediterranean, Australia, and Africa (see Chapters 1, 2, 8~~, and 9~~). The effects of severe fire on organisms vary spatially and temporally, by habitat type, and by species, but how do these disturbances specifically impact mammals? As with any natural disturbance, some species are adversely affected ("fire-averse" species), others benefit ("fire-loving" or pyrophilous species), and still others have a neutral response to fires.

p0020 The dynamics of populations and communities of mammals after severe fire depend on factors such as the degree of ecological change, time since fire, size and spatial configuration of burned and unburned areas, extent of edge, isolation of habitat patches by urbanization and roads, and invasion of nonnative species (Smith, 2000; Shaffer and Laudenslayer, 2006; Arthur et al., 2012; Diffendorfer et al., 2012; Fontaine and Kennedy, 2012). In theory, mammalian populations should be stable and resilient across the landscape wherever prefire populations and critical habitats are not greatly reduced and/or fragmented by human activities, and where severe fires occur in a spatial and temporal pattern in which a species has evolved (Shaffer and Laudenslayer, 2006). The capability of fire-loving individuals to utilize severely burned areas or for fire-averse populations to recover after fire, however, can be compromised when prefire habitat fragmentation has resulted in small and/or isolated populations and where postfire management actions, such as logging of burned trees and use of herbicides and pesticides, adversely influence population dynamics and habitat use.

p0025 In this chapter I provide an overview of published studies about mammalian responses to mixed- and high-severity fires in forests, woodlands, shrublands, deserts, and grasslands around the world. I describe research on the effects of severe fire on four major taxonomic groups of mammals: bats, small mammals, carnivores, and ungulates. I emphasized peer-reviewed publications, particularly those with robust methodologies and analyses, because these are the accepted standard in science. I also used non-peer-reviewed data when necessary to supplement information from the peer-reviewed literature. I do not cite every published study but instead provide a balanced overview of severe-fire effects on these taxa. I encourage readers to investigate further the scientific literature on habitat use and population responses of mammals to severe fire because the state of the science is constantly evolving.

p0030 Few studies have documented direct effects of fire on wildlife (e.g., mortality from asphyxiation, heat stress, burning, or physiological stress; however, see Singer et al., 1989), wildlife biologists generally agree that direct mortality from fire is typically very low and does not significantly influence populations (Smith, 2000). Thus I focus here on the indirect responses ~~of~~ severe fire, such as postfire occupancy, abundance or density, survival, reproduction, and use of habitat (e.g., breeding, resting, foraging). I define "significant effects" according to the generally accepted scientific definition of statistical significance (i.e., at the 0.05 probability level). I exclude studies that simulated or modeled fires, choosing instead to focus on observations of real systems responding to severe wildfire.

p0035 Appendix 4.1 is a summary of published studies by mammalian taxa and directional response to severe wildfire (negative, neutral, positive) over three time periods after fire. I present results from studies comparing unburned habitats with high-severity burn from wildfire (rather than prescribed fire) and without the confounding effect of postfire logging. For small mammals, only species with enough detections to determine directional response were included in the appendix.

s0015 4.2 BATS

p0040 Bats perform unique and critical ecosystem services by consuming vast quantities of insects, thereby transferring nutrients, most notably nitrogen, from foraging to roosting areas via their feces (Gruver and Keinath, 2006). Bats are predators of adult mosquitoes and thus play an important role in controlling

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mosquito populations and reducing disease transmission (Reiskind and [illegible], 2009). Further, nectar-feeding bats are primary pollinators of many plants spe-cies throughout the world (Molina-Freaner and Eguiarte, 2003).

p0045 The current literature on the effects of fire on bats strongly suggests that mixed- and high-severity fires are explicitly beneficial. In a study comparing the relative activity of six phonic groups of mostly rare and sensitive bat species across unburned and moderate- and high-severity burned mixed-conifer stands 1 year after fire in the southern Sierra Nevada, bat activity in burned areas was equivalent to or greater than activity in unburned areas for all groups based on echolocation frequencies (Buchalski et al., 2013). Indeed, two of the phonic groups showed a positive response to high-severity fire but a neutral response to moderate-severity fire, demonstrating the importance of severity-specific responses. The positive response to mixed- and high-severity fire by bats mir-rors findings for a range of bird species (see Chapter 3) and provides evidence of a long evolutionary relationship between bats and severe fire.

p0050 Several studies have documented how roosting bats use basal hollows of large trees (Gellman and Zielinski, 1996; Zielinski and Gellman, 1999; Fellers and Pierson, 2002; Mazurek, 2004). (Figure 4.1) Basal hollows are cav-ities formed by repeated fire scarring and healing (Zielinski and Gellman, 1999). For bats that roost in basal hollows of large trees, high-severity fire may destroy or reduce the longevity of existing roost trees, but it also creates new roost trees. In addition, fire creates gaps in the canopy that increase the amount of solar radiation reaching the subcanopy where bats roost. These warmer temperatures may facilitate thermoregulation (Brigham et al., 1997; Boyles and Aubrey, 2006) and are particularly beneficial to [illegible]ductive females because increased temperatures are associated with increased fetal and neonate growth (Brigham et al., 1997; Johnson et al., 2009). Finally, high-severity fire c[illegible] a "pulse" of insect prey (e.g., aquatic insects (Malison and Baxter, 2010) ~~and~~ moths, beetles, and flies (Schwab, 2006)), as well as new natural edge habitat that provides novel foraging opportunities (Fellers and Pierson, 2002).

p0055 Comparisons of food web components between unburned watersheds [illegible] areas of low- and high-severity fires 5 years after fire in Douglas fir (*Pseudotsuga menziesii*) and ponderosa pine (*Pinus ponderosa*) forests in central Idaho showed high insect biomass in heavily burned areas and corre-spondingly high bat detection rates (Malison and Baxter, 2010). Notably, high-severity sites had almost five times more biomass of zoobenthic insects and more than three times the number of emerging adult aquatic insects than low-severity sites (and twice as many as unburned areas). The frequency of bat echolocation calls also was significantly greater at high-severity sites than at unburned sites, because aquatic insects emerging from streams into the ter-restrial enviro[illegible]t are an important food source for bats. In a review of the responses of [illegible] benthic macroinvertebrates to fire, Minshall (2003) con-cluded that "[r]esults for macroinvertebrates generally support the belief that fire and similar natural disturbance events are not detrimental to the sustained

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f0010 **FIGURE 4.1** Basal hollow in large trees are created by periodic fire scarring and healing, creating important roost sites for bats. A Townsend's big-eared bat (*Corynorhinus townsendii*) roost tree in a coast redwood (*Sequoia sempervirens*) in Grizzly Creek State Park, northern California. (Photo by M.J. Mazurek (2015)).

maintenance of diverse and productive aquatic ecosystems (i.e., those found in undisturbed forests)" (p. 159). While individual taxa respond differently to the physical changes in stream structure and short-term and long-term postfire changes in vegetation, Minshall noted that streams are inherently unstable and dynamic environments in which disturbance, including high-severity fire, is a regular occurrence, and many species are opportunistic and can shift food resources in response to fire.

p0060 In mid-elevation forests burned at mixed and high severity in western Montana, Schwab (2006) characterized roost sites and sampled potential prey sources for two forest-dwelling, insectivorous bat species, the little brown bat (*Myotis lucifugus*) and the long-eared myotis (*Myotis evotis*). These species roosted in larger-diameter snags (standing dead trees) in high-density stands of fire-killed trees. Proximity to perennial streams also was important in roost site selection for these two species in burned forests. Wildland fire apparently

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created an abundance of roosting sites and insect prey for bats. Although the abundance of Lepidoptera (moths) and Trichoptera (caddis flies) was similar in burned and unburned forests, the abundance of Diptera (flies) and Coleoptera (beetles) was significantly higher in burned forests. Overall, the median capture rate of all insects in the burn was 1.78 times higher than the median capture rate in unburned forests, ~~although~~ there was considerable variability in the composition and abundance of particular species. Eight of the 11 orders of insects were more abundant in burned sites. In addition, beetles, flies, and caddis flies were significantly more abundant in burned than unburned sites in the first year after fire, although they decreased significantly the second year after fire. Thus, retention of burned trees the first year is important for insectivorous bats. In fact, removing burned trees decreased mammalian (and avian) predation on the abundance of insects that occurred 1 year after fire. Snags in unburned forests can be recruited from existing green trees, but in severely burned forests postfire logging eliminates both existing and future snags for nearly a century because few trees are available for snag recruitment until large-diameter trees have regrown (Schwab, 2006).

p0065 As with many bird species, mixed- and high-severity fire in forest ecosystems likely enhances foraging opportunities for bats (Buchalski et al., 2013). Many insect species inhabiting coniferous forests are highly evolved to exploit severely burned forests and are aptly termed “pyrophilous.” Certain beetle species in particular are strongly attracted to highly burned forests. Saint-Germain et al. (2004) noted that, “[s]ome insect groups have adapted to recurrent forest fires by evolving sensory organs and life strategies that allow them to exploit these high quality habitats efficiently. Pyrophilous Buprestids of the genera *Oxypteris* and *Merimna* and the Cerambycid *Arhopalus tristis* (F.) have been shown to respond physiologically to smoke and/or heat generated by fire, and use them as signals leading toward the newly created habitat . . . Several other Coleoptera species uncommon in mature forests congregate in exceptionally high densities in burned stands” (p. 583).

p0070 In a study of fire-loving beetle communities in a large fire that burned boreal black spruce (*Picea mariana*) forest in Quebec, Canada, more than half of the 86 taxa captured were restricted to burned stands (Saint-Germain et al., 2004). Moreover, total captures and species richness were higher in burned stands, especially the oldest severely burned forests. Captures were significantly lower the second year after the fire for all burned stands, indicating that the utility of burned forests for these beetles is greatest in the first year following fire.

p0075 Insects utilizing dead trees occur at much lower abundances in low-severity sites, which by definition have far fewer fire-killed trees than high-severity sites. Malison and Baxter (2010) stated that, “[o]ur results suggest that high severity fires do not play the same ecological role as low severity fires and allowing high severity fires to burn (rather than suppressing them) in certain forest types could be important in maintaining ecosystem function” (p. 577). Similarly, in his severely burned study site, Schwab (2006) noted, “26% of all [insect] families captured were restricted to sites within the burn suggesting

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b0010 **BOX 4.1**

o0010 **(1)** Bats preferentially roost and forage in burned forests.
o0015 **(2)** High-severity fire creates a superabundance of native insect prey.
o0020 **(3)** Bats select denser stands of fire-killed trees for roosting in burned forests and
Au7 forage significantly more in forests burned by high-severity fire than in unburned and low-severity fire-affected forests.
o0025 **(4)** Large burned trees for roosting have significant positive benefits for bats.
o0030 **(5)** Postfire logging removes roost trees, reduces the abundance of prey, and reduces habitat suitability for bats.

a unique environment created only after fire." Thus, ecological changes caused by mixed- and high-severity fires cannot be mimicked by low-severity prescribed burns (also see Chapter 13 for similar discussion) (Box 4.1).

s0020 4.3 SMALL MAMMALS

p0110 Small mammals are critically important to ecosystems because they can influence vegetation structure and composition by dispersing seeds and ectomycorrhizal fungi and by aerating soils (Maser et al., 1978). They also provide an essential prey base for carnivores, and the distribution of small mammals can affect the use of space and the habitat selection of their predators (Carey et al., 1992; Ward et al., 1998). Small mammals have comparatively small home ranges and therefore are quite sensitive to habitat change, making them good biological indicators (Haim and Izhaki, 1994). Small mammal assemblages include rodents and insectivores of the families Soricidae (shrews), Talpidae (moles), Aplodontidae (mountain beavers), Sciuridae (squirrels, chipmunks, and marmots), Geomyidae (gophers), Heteromyidae (pocket mice and kangaroo rats), superfamily Muroidea (voles, mice, and woodrats), and Dipodidae (jumping mice). Larger-bodied small mammals include rodents in the Castoridae (beaver) and Erethizontidae (porcupine) families, as well as lagomorphs (pika, hares, and rabbits), and Australian and American marsupials (Marsupialia).

p0115 The occupation of severely burned areas by small mammals is related to regrowth of the vegetation structure with which various species are associated (Torre and Díaz, 2004; Lee and Tietje, 2005; Vamstad and Rotenberry, 2010; Diffendorfer et al., 2012; Kelly et al., 2012; Borchert and Borchert, 2013), as well as with seed and insect production and availability (Coppeto et al., 2006), and cavities created by woodpeckers in snags (Tarbill, 2010). I discuss fire effects on small mammals according to habitat type but give special attention to the deer mouse (*Peromyscus maniculatus*)—an exceptionally "fire-loving" species—in its own section. (Figure 4.2)

f0015 **FIGURE 4.2** Deer mice increase after severe fire in a variety of habitats. Deer mouse captured two years after forest dominated by Douglas-fir with some lodgepole pine, western larch, and ponderosa pine burned severely in the 2005 Tarkio Fire, Montana. (Photo by Rafal Zwolak (2005)).

s0025 Chaparral and Coastal Sage Scrub

p0120 The chaparral and coastal sage scrub vegetation types in central and southern California support an exceptionally rich diversity of rodents that are well-adapted to a regime of periodic, very-high-intensity fire (see Chapter 7). Many studies have examined small-mammal communities after both prescribed and wildfire in these vegetative types. During intense fires, some individuals among small, less vagile animals may suffer mortality, but many others survive in rock crevices, riparian areas, large downed logs, and underground burrows where temperatures remain cool and the air clean (Chew et al., 1959; Quinn, 1979; Lawrence, 1966; Wirtz, 1995; Smith, 2000). Following fire, small-mammal communities change over time (Diffendorfer et al., 2012; Arthur et al., 2012; Borchert and Borchert, 2013) and space (Schwilk and Keeley, 1998), depending on the vegetation associations of the various species. Species preferring open habitat, including pocket mice (*Chaetodipus* spp.), California voles (*Microtus californicus*), harvest mice (*Reithrodontomys megalotis*), and, especially, kangaroo rats (*Dipodomys* spp.) and deer mice can increase quite dramatically and quickly after severe shrubland fire. Over a period of several years, as shrubs resprout and grow denser and as different food sources become available, small-mammal species preferring a shrubby overstory, including woodrats (*Neotoma* spp.), California mice (*Peromyscus californicus*), brush mice (*Peromyscus boylii*), and cactus mice (*Peromyscus eremicus*), increase in number (Cook, 1959; Wirtz, 1977; Price and Waser, 1984; Brehme et al., 2011; Borchert and Borchert, 2013). Compared with unburned chaparral and grassland, severely burned chaparral had the highest rodent diversity 4 years after a high-intensity wildfire near Mount Laguna in San Diego County (Lillywhite, 1977). Published data are not currently available for lagomorphs in chaparral wildfires, but prescribed burning of chamise (*Adenostoma fasciculatum*) chaparral in northern California increased black-tailed jackrabbit (*Lepus californicus*) densities by 500-1000% the year following fire (Howard, 1995).

s0030 Forests

p0125 Forests offer important habitats for small mammals, especially shrews, mice, tree voles, and squirrels. Mixed- and high-severity fire in forested habitats can have pronounced effects on small-mammal populations by creating or transforming habitat structures such as live and dead trees, shrubs, and coarse woody debris. While some studies have shown that severely burned conifer forests in North America support fewer individuals of some rodents and insectivores immediately after fire compared with adjacent unburned sites (e.g., pinyon mice [*Peromyscus truei*; Borchert et al., 2014] and masked shrews (Sorex cinereus) and southern red-backed voles [*Myodes gapperi*; Zwolak and Forsman, 2007]), numbers begin to rebound several years after fire, often by individuals surviving in unburned refuges within the larger burn perimeter. Northern red-backed voles (*Myodes rutilus*), considered old-growth specialists, began repopulating an intense burn in boreal Alaska from surrounding unburned forest and started reproducing 3 years thereafter (West, 1982).

p0130 Unburned refuges and vegetation changes over time also mediate postfire mammal population dynamics in other forests types, notably *Eucalyptus* forests in Australia. Numbers of bush rats (*Rattus fuscipes*) and agile antechinus (*Antechinus agilis*) were reduced compared with populations in adjacent unburned forests 6 months after severe fire in a mountain ash (*Eucalyptus regnans*) forest, but the population in the burned area was composed of residual animals that had survived the fire rather than animals recolonizing from adjacent forests (Banks et al., 2011). Long-term studies are especially useful because responses relative to time since fire can be quantified. One study examined marsupial population dynamics over a 28-year period following severe wildfire in a southeastern Australia *Eucalyptus* forest reserve (Arthur et al., 2012). Bandicoots (*Isoodon obesulus* and *Perameles nasuta*) increased immediately following the fire, peaked 15 years later, and then declined, associated with an increase and decline of shrub cover. The potoroo (*Potorous tridactylus*) population was similar before and immediately after the fire but began to increase a decade later as tree cover increased. Wombats (*Vombatus ursinus*) exhibited a stable population trend for the first decade after the fire, then slowly declined along with a decline in ground litter cover. Finally, larger macropods (eastern gray kangaroo [*Macropus giganteus*], red-necked wallaby [*Macropus rufogriseus*], and swamp wallaby [*Wallabia biocolor*]) remained at high densities after the fire then declined a decade later as vegetation cover increased.

p0135 Rabbits and hares are associated with shrubs and small conifers that provide cover (Ream, 1981; Howard, 1995). Severe fire temporarily eliminates this

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habitat structure, but it quickly returns as the vegetation regrows, stimulated by intense fire. Snowshoe hares (*Lepus americanus*) in a boreal forest in Alberta, Canada, moved out of intensely burned sites to surrounding habitat immediately after fire but returned the second summer after the fire when shrubs resprouted, and the postfire population trajectory increased above prefire numbers (Keith and Surrendi, 1971).

p0140 Tree squirrels, including Douglas squirrels (*Tamiasciurus douglasii*) and northern flying squirrels (Glaucomys sabrinus), typically are associated with late-successional coniferous forests in California and the Pacific Northwest in the United States (Carey, 2000); thus they may be adversely affected by intense fire (Zwolak and Forsman, 2007), but few data currently are available to refute or support this hypothesis. Chipmunks and ground squirrels can occupy forests after severe fire where shrubs provide cover and food (Borchert et al., 2014). Townsend's chipmunks (*Neotamias townsendii*) were abundant in early seral forests with dense shrub cover (Campbell and Donato, 2014). Gray-collared chipmunks (*Tamias cinereicollis*) and least chipmunks (*Tamias minimus*) showed no significant response to wildfire in ponderosa pine forests of the southwestern United States (Converse et al., 2006), and the proportion and composition of two chipmunk species, *Tamias amoenus* and *Tamias ruficaudus*, did not differ between severely burned and unburned conifer forest in Montana (Zwolak and Forsman, 2007).

p0145 The increase in the availability, amount, and quality of forage for herbivorous small mammals is an important determinant of the post-severe-fire community. In plots recently burned by large, intense wildfires in a Mediterranean pine-oak woodland in Spain, the abundance of small mammals—mostly mice and shrews—was higher than expected based on vegetation characteristics alone (Torre and Díaz, 2004). The authors attributed small-mammal increases to large quantities of seeds and seedlings in burned sites.

s0035 Deserts

p0150 The role of severe fire and its effects on small mammals in desert grasslands is somewhat controversial (Killgore et al., 2009; Vamstad and Rotenberry, 2010). Most desert systems are not adapted to frequent fire because many species of long-lived perennial desert plants have low recruitment rates and long life spans and lack the ability to resprout. Fire size and frequency in some areas has increased recently because of the invasion of exotic grasses from livestock grazing (Brooks, 2000) and other causes (Burbidge and McKenzie, 1989). In general, most research shows a lack of significant long-term effects of intense fire on the abundance of desert small mammals, although fire can alter community composition. Similar to shrub types in southern California, rodents in the family Heteromyidae increased following a large, intense wildfire in a perennial grassland in southeastern Arizona, whereas species in the family Cricetidae declined immediately after fire, began increasing 4 years after fire, and returned to

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prefire levels by the sixth year (Bock et al., 2011). Rodent abundance and species richness were not different between burned and unburned plots after wildfires in Joshua tree woodlands of the Mojave Desert in the American Southwest (Vamstad and Rotenberry, 2010). Merriam's kangaroo rat (*Dipodomys merriami*) dominated the burned sites. As postfire vegetation changed from annuals to sub-shrubs and then to long-lived perennials, however, the composition of rodent species changed and the diversity of rodents increased over time.

p0155 Habitat type is important to fire effects in deserts. In Australia, wildfires in stony desert habitats with sparse grasses have less effect on habitat structure and small mammals than wildfires in sandy desert habitats with denser hummock grass spinifex (*Triodia* spp.) (Pastro et al., 2014). For example, an intense wildfire did not affect the total abundance and species richness of small mammals in the stony (gibber) desert in central Australia, although some species increased and others decreased immediately following fire (Letnic et al., 2013). By contrast, 9 months after intense wildfire in a spinifex grassland in the same region, small-mammal diversity declined compared with before the fire and with prescribed burned areas, although the abundance of animals captured was similar (Pastro et al., 2011). Data were unavailable from wildfires, but hare (*Lepus* spp.) abundance increased by 300% after prescribed burning in East African savanna grasslands (Ogen-Odoi and Dilworth, 1984).

s0040 Deer Mice

p0160 In North America, generalist deer mice are often the most abundant rodent after severe fire in a variety of vegetation types (Borchert et al., 2014). This species responds strongly and positively to high-intensity fire in both shrubland and conifer forests. Deer mice increased significantly over time in moderately and severely burned mixed-conifer forests in the San Bernardino Mountains of southern California over a 5-year period after fire (Borchert et al., 2014). During 2 years subsequent to intense fire, deer mice were invariably the most numerous species in burned study sites in a Douglas-fir-Western larch ~~(*Larix occidentalis* Nutt.)~~ forest in Montana (Zwolak and Forsman, 2008). Converse et al. (2006) attributed increased abundance of deer mice after wildfire in southwestern United States ponderosa pine forests to ~~increased~~ seed production or ~~greater~~ detectability of seeds after fire.

p0165 Dramatic increases in deer mice in severely burned conifer forests were not simply a result of colonization of the burn by animals from surrounding unburned forests. When population densities were low, the vast majority of individually ear-tagged deer mice were found in forest areas after severe fire, and mice appeared regularly in unburned forests only when population densities were high (Zwolak and Forsman, 2008). This finding indicated that severely burned forest was *preferred* deer mouse habitat and that the postfire population increase was intrinsic to the burn; thus the burn itself was a source habitat.

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b0015 **BOX 4.2**

o0035 **(1)** After intense wildfire, small-mammal communities are dynamic and associated with vegetation structure at different successional stages.

o0040 **(2)** Intense fire may increase the availability and abundance of seeds and seedlings for herbivorous small mammals.

o0045 **(3)** Unburned refuges and time since fire are important determinants of small-mammal communities following intense fire.

o0050 **(4)** The richness and abundance of small-mammal species is high following intense fire in chaparral and coastal sage scrub communities of southern California. Heteromyid rodents and deer mice often dominate severely burned shrublands, and heteromyids dominate postburn desert grasslands.

o0055 **(5)** Some small-mammal species decrease shortly after intense fire in North American conifer forests, but they can recover to prefire levels within 1 to several years after fire. Deer mice dramatically increase following intense fire.

p0170 Overall, these observations from small-mammal studies in mixed- and severely burned shrublands, forests, and grasslands underscore the important roles played by high-severity fire patches, unburned refuges within a fire area, and the time since fire in population dynamics after severe fire (Box 4.2).

s0045 4.4 CARNIVORES

p0205 Carnivores are critically important "top-down" regulators of ecosystem processes. Elimination of top carnivores unleashes a cascade of adverse effects, including relaxation of predation as a selective force on prey species, spread of disease, explosions of herbivore populations, and subsequent reproductive failure and local extinction of some plants, birds, herptiles, and rodents (Crooks and Soule, 1999; Terborgh et al., 2001). ~~Soulé~~ Large carnivores include ursids (bears), canids (wolves), and larger felids (puma, lions, and jaguars). Medium-sized carnivores, or "mesocarnivores," include canids (coyotes and foxes), Procyonidae (ringtails and raccoons), mustelids (wolverine, marten, fisher, weasels, mink, and badgers), Mephitidae (skunks), and smaller felids (lynx and bobcats). Currently published research on carnivores in mixed and severe wildfires is limited primarily to forested habitats.

s0050 Mesocarnivores and Large Cats

p0210 Many mesocarnivores are associated with forested habitats. Some are habitat generalists, whereas others are forest specialists, riparian associates, or semi-aquatic (Buskirk and Zielinski, 2003). Martens (*Martes* spp.) occur in dense coniferous or deciduous forests across the northern hemisphere. They also regularly use severely burned habitats. Some evidence suggests martens use burns

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only when postfire trees are not logged. For instance, stone marten (*Martes foina*) were not detected in an intensely burned but extensively postfire-logged Aleppo pine (*Pinus halepensis*) forest in Greece the second and third years after wildfire and logging (Birtsas et al., 2012). These martens were found only in Turkish red pine (*Pinus brutia*) forests burned by wildfire 9 years earlier and not in nearby unburned forests (Soyumert et al., 2010). In coniferous forests of the Alaskan taiga, resident and transient American martens (*Martes americanus*) were captured in a 6-year-old unlogged burn more often than in an island of unburned mature forest surrounded by the burn (Paragi et al., 1996). The authors did not quantify burn severity in their study area but described fire-affected sites as having portions of "severe" burn, and most of the vegetation was in early to mid-seral stages, with dead, fire-scarred trees still standing, consistent with mixed- and high-severity fire. There was no age difference between martens trapped live in the mature forests ~~versus~~ and those trapped in the burn, and marten foraging intensity was greatest in the recently burned area (Paragi et al., 1996). Conversely, martens avoided stands of boreal forests burned from 2 to 20 years prior (Gosse et al., 2005), but the study did not quantify or describe burn severity nor specify whether the burned forest was logged.

p0215 Larger cousins to the marten, fisher (*Martes pennanti* or *Pekania pennanti*) are rare mesocarnivores associated with dense, mature, boreal and mixed conifer-hardwood forests of North America (Powell and Zielinski, 1994). A recent study in the southern Sierra Nevada, however, used scat sampling to detect fisher habitat preferences and demonstrated that the species used denser, mature forests that had experienced moderate- and high-severity fire 10 and 12 years prior and that were not logged after fire (Hanson, 2013) (Figure 4.3). It is likely that both martens and fishers use severely burned forests for foraging rather than denning. These results provide intriguing evidence that even old-forest specialist species are adapted to and can exploit postfire conditions in regions where mixed- and high-severity fire is natural (see Chapter 3, Box 3.1: spotted owls).

p0220 Foxes apparently prefer severely burned forest areas over unburned areas, but they may be less tied to forest structure than martens and fishers and thus less sensitive to postfire logging. Red fox (*Vulpes vulpes*) in Turkish red pine forests were detected more often in the 9-year-old unlogged wildfire area (Soyumert et al., 2010); in postfire-logged Aleppo pine forests in Greece, red foxes were detected most often in severely burned areas, rather than moderately and unburned areas (Birtsas et al., 2012). In 3 of 4 years after intense wildfire in mixed-conifer forests of the San Bernardino Mountains in southern California, gray foxes (*Urocyon cinereoargenteus*) were detected more often in mixed-severity burned over unburned areas, and in two of the years no foxes at all were captured in the unburned area, but coyote (*Canis latrans*) were detected more often in unburned forests (Borchert, 2012). Both gray fox and coyote scats were more numerous in areas burned by intense wildfire than in unburned areas

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f0020 **FIGURE 4.3** Representative foraging ~~detection~~ location based upon global positioning system coordinates for a confirmed female Pacific fisher scat detection site several hundred meters into the interior of the largest high-severity fire patch (>5000 ha) in the McNally Fire of 2002, Sequoia National Forest, California. *(Photo by Chad Hanson (2014).)*

2 years after fire in interior chaparral, Madrean evergreen woodland, and ponderosa pine forest in Arizona (Cunningham et al., 2006).

p0225 Striped skunk (*Mephitis mephitis*), ringtail (*Bassariscus astutus*), and raccoon (*Procyon lotor*) were photocaptured only in mixed-conifer forests in southern California burned by high-intensity fire, but each were photographed only once (Borchert, 2012). Bobcat (*Lynx rufus*) were photocaptured in similar numbers in severely burned and unburned forest, but captures in the burned area decreased over time over the 4 years of the study. Finally, mountain lion (*Puma concolor*) were photocaptured more often in severely burned forest, but the overall sample was small (four lion in burned areas, one lion in unburned areas).

s0055 Bears

p0230 Although grizzly bears are flexible in the habitats they use, in British Columbia, Canada, radio-collared grizzly bears strongly selected open forest burned by wildfires 50-70 years earlier at high elevations because these sites supported prolific huckleberries (McLellan and Hovey, 2001). Wildfire also promotes the regeneration of whitebark pine (*Pinus albicaulis*) seeds, another important food source for bears (Kunkel, 2003). Wildfire is not equivalent to logging, as regenerating timber harvests were rarely used by bears in any season (McLellan and Hovey, 2001).

p0235 One study compared the demographics and physiology of black bears (*Ursus americanus*) occupying burns of two ages, 13 and 35 years old, in spruce

(*Picea* spp.) and aspen (*Populus tremuloides*) forests of the Kenai Peninsula of Alaska (Schwartz and Franzmann, 1991). The authors did not specify burn intensity, but they noted that 5% of the older burn was logged after fire for "improvement" of moose (*Alces alces*) habitat, and they pointed out that the more recent fire burned at a greater intensity than the older fire. The density of bears and the percentage of cubs born were similar between the two sites, but all age groups of bears were significantly larger in the recent burn area. Bears in the older burn area consumed more cranberries, whereas the number of moose calves consumed per bear was much larger in the recent burn area, likely explaining the larger size of the bears. Females in the recent burn area also produced litters at a younger age and had a shorter interval between weaning of yearlings than females in the older burn area. Moreover, cub survival was significantly higher in the recent burn area. The vigor of black bear populations was associated with moose abundance, which was significantly enhanced in the 13-year-old fire area.

p0240 Another study compared the demography of a population of black bears in interior chaparral, Madrean evergreen woodland, and ponderosa pine forest, burned by high-intensity wildfire for 3 years after fire using (1) the population in a nearby unburned site for 3 years and (2) results from earlier demographic research on the fire site from 20 years earlier, conducted over a 6-year period (Cunningham and Ballard, 2004). The sex ratio at the 3-year-old burned site was more skewed toward males than in either the unburned reference site or 20 years before the burn. The authors presumed that the fire had reduced the adult female population; however, it is also possible that the female population already had been reduced in the 20 years before the fire occurred, when the population was not monitored. Indeed, an alternative scenario could be that the population of both adult females and males had been declining at Four Peaks before fire, and the fire actually attracted males to the site, who have larger home ranges, thus skewing the sex ratio.

p0245 The above study reported complete reproductive failure in the 3 years after fire at the burned site compared with 36% of cubs surviving to 1 year of age on the unburned control site (Cunningham and Ballard, 2004). More cubs had survived to year 1 at the burned site 20 years before the fire. During the 1970s, however, complete reproductive failure also occurred in the absence of fire during 3 of the 6 years of study. Thus years of complete reproductive failure in that study area were not unusual. Overall, reproductive success was lowest in the burned forest compared with the same site 20 years before fire and an unburned reference site, suggesting the possibility of negative short-term effects of high-intensity fire on black bear reproduction. The mortality of adult bears from hunting, however, was 2.5 times higher in the fire area than in the unburned area (Cunningham et al., 2001), which would be expected to influence cub survival, potentially confounding results. The overall density of black bears in the fire area was higher than prefire densities in the area (Cunningham et al., 2001) (Box ~~4.3 and~~ 4.4).

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b0020 **BOX 4.3 Seed Dispersal by Carnivores**

p0250 Fleshy fruits are an important component of the diet of many carnivores, especially during certain seasons when other resources are scarce. Indeed, the germination of many seeds is facilitated by passage through the carnivore gut because it removes the fruit pericarp and scarifies the seed coat (Herrera, 1989). Carnivores are important dispersers of seeds because they have relatively large home ranges and long gut retention times, thus spreading the seeds far from the parent plant. This may be an important mechanism whereby early seral habitats are seeded. For example, in experimental and field tests in severely burned Aleppo pine forest in Spain, Rost et al. (2012) demonstrated that carnivores, including red fox, stone marten, and European badger (*Meles meles*), were important dispersers of Mediterranean hackberry (*Celtis australis*) seeds into the burned areas. These carnivores traveled long distances into the fire area, dispersing seeds more than 1 km from the parent plant. Moreover, seeds collected from scat (i.e., that had passed through the gut) in the burned study area had a significantly greater germination rate than unscarified seeds, both in the greenhouse and in the field.

b0025 **BOX 4.4**

o0060 **(1)** Grizzly bears use areas burned by intense wildfire because of increases in berry production, although results from studies of the effects of intense fire on black bear demographics are equivocal.

o0065 **(2)** Martens and fisher are mesocarnivores that are dense, mature forest specialists for denning and resting but use severely burned forests that were not logged after fire, most likely for foraging.

o0070 **(3)** Foxes regularly use severely burned forests (regardless of postfire logging for one Mediterranean species), but results from research on coyotes are equivocal.

o0075 **(4)** Carnivores are important dispersers of seeds deep into severely burned forest areas.

s0060 4.5 UNGULATES

p0280 As major herbivorous components of ecosystems, ungulates can act as keystone species with profound effects on vegetation development and productivity in forests, woodlands, and grassland ecosystems throughout the world (Hobbs, 1996; Wisdom et al., 2006). Hobbs (1996) stated, “ungulates are not merely outputs of ecosystems, they may also serve as important regulators of ecosystem processes at several scales of time and space” (p. 695). Ungulates, Hobbs further noted, are “important agents of environmental change, acting to create spatial heterogeneity, accelerate successional processes, and control the switching of

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ecosystems between alternative states." Ungulates regulate nitrogen cycling and influence plant size and morphology (Singer et al., 2003). Because grazing and browsing by ungulates affects the biomass, structure, and type of vegetation available to burn, these animals can actually regulate the dynamics of fire (Hobbs, 1996; Wisdom et al., 2006).

p0285 Episodic disturbance agents such as fire strongly interact with ungulate herbivory over space and time. For example, removal of fine fuels by ungulate grazers may reduce the frequency of ground fires but can increase crown fires by enhancing the development of ladder trees, especially when combined with a relatively long absence of fire (Hobbs, 1996). Further, postfire plant regeneration provides forage species that are highly palatable to ungulates, which attracts ungulates to burned areas, where they influence vegetation regrowth after fire (Canon et al., 1987; Wan et al, 2014). Moose rapidly immigrated to burned areas after a large wildfire in mixed coniferous-deciduous forests of northern Minnesota (Peek, 1974). In fact, fire size can moderate the adverse effects of ungulate herbivory on vegetation recovery. Compared with small fires, large fires "swamp" the effects of ungulate herbivory, for example, by providing sufficient new growth production to offset browsing, and enabling woody species such as aspen to grow to tree height (Biggs et al., 2010). In intensively burned ponderosa pine, mixed-conifer, and spruce-fir forests of northern New Mexico, elk selectively foraged on grasses over shrubs (Biggs et al., 2010). In 25 wildfires throughout five national forests in Utah, larger areas of aspen forest that burned with greater severity had the highest growth potential for aspen regeneration, and these high burn-severity conditions stimulated defensive chemicals in plants that lowered the levels of damage done by ungulate browsing (Wan et al., 2014). Wan et al. noted that "this effect may be particularly strong if amplified over large post-fire landscapes by saturating the browse capacity of the ungulate community." (See Box 4.5).

p0290 Positive effects of high-severity fire on ungulates likely are most pronounced in vegetation types that are most adapted to high-intensity fires, such as aspen forests and shrublands. Mountain or bighorn sheep selected intensely burned shrublands up to 15 years after fire in Montana (DeCesare and Pletscher, 2006) and in southern California mountains (Bleich et al., 2008). Wildfire increased the carrying capacity of southern California ~~mountain~~ sheep (*Ovis canadensis nelsoni*) in the San Gabriel Mountains, dramatically increasing the number of animals in this endangered population (Holl et al., 2004). A large natural fire on the eastern slopes of the Sierra Nevada mountains in California improved the winter range of Sierra bighorn sheep (*Ovis canadensis sierrae*) by increasing green forage availability, shifting diet composition to include more forbs, and possibly decreasing predation risk from mountain lions by increasing visibility (Greene et al., 2012). Overall, large, high-severity fire in bighorn sheep shrubland/forest habitats increases forage quality and availability as well as visual openness, which is critical because several populations are listed as endangered.

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p0295 Studies investigating the impact of fire on mule deer (*Odocoileus hemionus*), a common herbivore in the western United States, indicate that populations tend to increase after severe fire, especially in chaparral communities. In a review of the literature on ungulate responses to fire, Smith (2000) reported mule deer density in intensely burned chaparral was more than twice as high as that in mature chaparral in California, and it increased 400% the first year after high-intensity fire in chamise chaparral. Density then decreased each year afterward until preburn levels were reached 5-12 years later. Chamise chaparral burned by a large wildfire in California had more deer use per square mile than unburned chamise chaparral (Bendell, 1974). In northern coastal California, mule deer densities in chaparral burned by high-intensity wildfire the year before were four times greater than in unburned chaparral (Taber and Dasmann, 1957). Because the fire described in this study was relatively small, deer may have moved from one area to another rather than actually increasing the population via higher birth rates. Similarly, ~~black-tailed~~ deer in central coastal California strongly preferred burned habitat, with a 400% increase in the density of deer in prescribe-burned chaparral near oak woodlands, relative to preburn density, by the second growing season (Klinger et al., 1989). Here the increase in the use of burned chaparral was attributed to movements of deer from adjacent oak woodlands rather than an intrinsic increase in population size. Heavy use of prescribe-burned chamise chaparral by mule deer was reported in the San Jacinto Mountains of southern California (Roberts and Tiller, 1985).

p0300 Other studies documented postfire increases in the number of mule deer in conifer forests. Visual observations of 543 mule deer indicated a preference for burned over unburned Douglas fir/ninebark and burned ponderosa pine/bluebunch wheatgrass habitat types during winter and spring in the Selway-Bitterroot Wilderness of Idaho, although the authors did not specifically define the burn severity of sites used by deer (Keay and Peek, 1980). Two other studies that documented increases in mule deer in burned forests hypothesized that postfire logging removes protective cover, a critical habitat element for mule deer. Significantly more deer droppings were located in pinyon-juniper woodlands of Arizona burned by high-intensity fire 13 years earlier than in adjacent unburned areas (McCulloch, 1969). The author surmised that the standing forest of dead trees and fallen trunks provided some cover for deer from predators. Both mule deer and elk used intensely burned lodgepole pine ~~(*Pinus contorta*)~~ forests at two sites in Wyoming significantly more than paired clearcut sites of the same ages (9 and 5 years old) based on fecal pellet counts (Davis, 1977). Davis (1977, p. 787) stated: "[D]eer and elk use was greater in burned areas with standing dead timber than in clearcut areas without it. In the Sierra Madre study area, the burned and clearcut plots both had the same number of plant species present, and they both had standing dead timber. However, the burned plot with much more standing dead timber had more deer and elk use. Fire opened up the canopy allowing light to enter, stimulating growth of forage plants, while the dead trees left standing provided good protective cover" (see Figure 4.4).

f0025 **FIGURE 4.4** Mule deer respond positively to high-severity fire in forests. In this photo, mule deer forage on fresh vegetation growing in the first post-fire year following the Rim fire of 2013 on the Stanislaus National Forest, central Sierra Nevada. *(Photo by Chad Hanson (2014).)*

p0305 Available studies generally report increases in the reproductive rates and body condition of female mule deer in burned habitats. The reproductive rate was 1.32 fawns per doe in the first year after wildfire in northern coastal California, compared with 0.77 fawns per doe in unburned chaparral (Taber and Dasmann, 1957). After 3 years, the reproductive rate of deer at the burned site declined to that of deer in the unburned site. Chamise chaparral burned by a large wildfire produced heavier deer, and does had a higher frequency of ovulation, gave birth to more fawns, and wintered in better condition than does in dense, unburned chamise (Bendell, 1974). Another study, however, documented no difference in fawn-to-doe ratios between burned and unburned chaparral interspersed with oak woodlands in central California (Klinger et al., 1989).

p0310 Foraging studies indicate that mule deer populations in chaparral habitats burned by high-intensity fire often increase as a result of the increased availability of browse. *Ceanothus*—a high-quality food for ungulates (Hobbs, 1996)—is abundant after fire because it reproduces from seed that is scarified by burning (Smith, 2000). Thimbleberry (*Rubus parviflorus*) also generally increases after fire (Smith, 2000). Moreover, fire can increase the palatability of foliage for deer as well as the crude protein content (Smith, 2000). The improved quantity and quality of browse may be related to the fire-caused increase in available nutrients in the soil. As such, deer populations often benefit from the increased food production and nutritional value of their food in recently burned areas. Length and surface enlargement factor of papillae (the surface area within the intestine for absorbing nutrients) of necropsied mule deer were greater in those from high-intensity burned than unburned ponderosa pine habitat in the southern Black Hills of South Dakota (Zimmerman et al., 2006). These physiological factors indicate higher forage quality, such as greater concentration of volatile fatty acids. The authors concluded that fire was beneficial at the mucosal level for mule deer: the increase in forage quality from burning caused a rapid change in papillary morphology, allowing the deer to take up more nutrients.

p0315 Lichens in boreal habitats are preferred winter forage for caribou (*Rangifer tarandus*), yet large wildfires that depleted lichens had no effect on home-range size, range fidelity, or the survival and fecundity of woodland caribou (*Rangifer tarandus caribou*) in Alberta, Canada (Dalerum et al., 2007). Caribou avoided foraging in burned compared with unburned areas (Dalerum et al., 2007; Joly et al., 2010), although burn severity was not quantified, and some of the fires occurred 50 years before study. Lichens are significantly reduced by wildfire and take decades to recover to prefire abundance (Joly et al., 2010) (Box 4.5).

Au20 b0030 **BOX 4.5**

o0080 **(1)** Ungulates interact strongly with episodic disturbances. Many are attracted to severely burned areas because of increased forage palatability and availability, where in turn they influence vegetation regrowth.

o0085 **(2)** Elk, bighorn sheep, and mule deer generally increase after intense fire in shrublands and forests.

o0090 **(3)** The larger the area of high-severity fire, the lower the adverse impact on regrowth of aspen forests from ungulate herbivory.

o0095 **(4)** Caribou may be adversely affected when intense fire reduces lichen used for winter forage.

s0065 4.6 MANAGEMENT AND CONSERVATION RELEVANCE

p0345 The abundance of certain mammal species after fire has direct benefits to land managers in the form of irreplaceable ecosystem and economic services. Bats are voracious predators of insects—many of them consume crop and forest pests—and as such are important regulators of insect populations, including disease-carrying mosquitoes (Reiskind and Wund, 2009). Bats are also critical pollinators of many plants (Molina-Freaner and Eguiarte, 2003). The loss of bats in North America could cost the economy $3.7 billion per year in agricultural losses alone (Boyles et al., 2011). Small mammals aerate the soil and, along with many carnivores, are important dispersers of seeds and fungi (Maser et al., 1978; Rost et al., 2012). Large carnivores are top-down regulators of smaller carnivores and ungulates and are vital to the health and function of natural ecosystems. Ungulates help to cycle nitrogen and provide big-game hunting opportunities and food for humans. Indeed, in 2001 alone, hunting of ungulates and large carnivores in the United States contributed to approximately $25 billion in retail sales and $17 billion in salaries and wages and employed ~~of~~ 575,000 people (IAFWA, 2002). These animals include mule deer, bighorn sheep, moose, elk, and bear, all of which use or thrive within heavily burned habitats.

p0350 As described here, a great many mammals benefit from mixed- and high-severity fire and play essential roles in postfire ecosystem dynamics. Land managers rarely weigh these benefits when evaluating the impacts of large fires of mixed- and high-severity, however, thus undervaluing their ecological and economic importance. The vital ecosystem services of mammals in postfire areas should be quantified and carefully considered when planning potentially harmful management activities such as postfire logging and common management activities following postfire logging, such as the application of herbicides and rodenticides.

s0070 4.7 CONCLUSIONS

p0355 The extraordinary abundance and diversity of mammals using (e.g., American marten, Pacific fisher, grizzly bear) and even thriving (e.g., deer mice, kangaroo rats, bats, mule deer, elk, bighorn sheep) in severely burned grassland, shrubland, and forested habitats is an important indicator of the high habitat suitability of these areas. Prescribed burning does not provide the expected gains in biological diversity for a range of mammal, reptile, bird, and plant taxa (Pastro et al., 2014). Only large, severe wildfires create significant ecological changes associated with increases in fire-loving species, and, as demonstrated herein, only larger fires can "swamp" the effects of ungulate herbivory on postfire vegetation. ~~Mixed-severity and severe~~ fires globally have unique ecological value that must be weighed against the dominant paradigm that such natural disturbance events are "catastrophic" (Zwolak and Foresman, 2008; also see Chapters 1, 2, and 13). Mammals and other wildlife using intensely burned forests provide myriad ecological services that benefit people and ecosystems alike.

s0075 **APPENDIX 4.1 THE NUMBER OF STUDIES BY TAXA SHOWING DIRECTIONAL RESPONSE (NEGATIVE, NEUTRAL, OR POSITIVE) TO SEVERE WILDFIRE OVER THREE TIME PERIODS FOLLOWING FIRE. STUDIES CITED INCLUDE UNBURNED AREAS COMPARED ~~WITH~~ SEVERELY BURNED WITH NO POST-FIRE LOGGING, AND EXCLUDED PRESCRIBED BURNS. FOR SMALL MAMMALS, ONLY SPECIES WITH ENOUGH DETECTIONS TO DETERMINE DIRECTIONAL RESPONSE WERE REPORTED**

t9010

	1-5 yr post-fire			6-10 yr post-fire			>10 yr post-fire		
	Negative	*Neutral*	*Positive*	*Negative*	*Neutral*	*Positive*	*Negative*	*Neutral*	*Positive*
Bats[1]		1	3						
Small Mammals[2]									
Masked shrew									
White-toothed shrew			1						1
Tamias spp.		4							
Pacific kangaroo rat		1	2			1			
Dulzura kangaroo rat			2						
Merriam's kangaroo rat			1			1			1
California pocket mouse		1	1						
San Diego pocket mouse	1	2							
Bush rat	1								
Long-haired rat	1								

Continued

	1-5 yr post-fire			6-10 yr post-fire			>10 yr post-fire		
	Negative	*Neutral*	*Positive*	*Negative*	*Neutral*	*Positive*	*Negative*	*Neutral*	*Positive*
Red-backed vole	2								
California vole		1	2						
Canyon mouse	1			1			1		
Brush mouse		1	1						
Deer mouse		2	5		1				
California mouse	3		1			1			
Cactus mouse	1	1	1						
Pinyon mouse	1		1	1					
Harvest mouse	2	1	1						
Desert woodrat	2								
Big-eared woodrat	1	1							
Snowshoe hare			1						
Antechinus	1								
Potoroo		1				1			1
Bandicoot			1			1	1		
Wombat		1			1		1		
Macrocarps (3 spp)		1		1			1		

Carnivores[3]									
American marten									
Stone marten						1			
Pacific fisher						1			
Gray fox			2						
Red fox						1			
Black bear	1								1
Grizzly bear									1
Ungulates[4]									
Caribou				2			2		
Moose			1						
Bighorn sheep			3			2			2
Elk			1						
Mule deer			4			1			1

np9010 [1]*Bat citations: Schwab 2006, Malison and Baxter 2010, Buchalski et al. 2013. Bats are categorized by phonic groups: of 6 phonic groups in 3 studies, 5 phonic groups showed positive response and 1 showed neutral response.*

np9015 [2]*Small mammal citations: Keith and Surrendi 1971, Cook 1959, Wirtz 1977, West 1982, Price and Waser 1984, Torre and Díaz 2004, Converse et al. 2006, Zwolak and Forsman 2007, Zwolak and Forsman 2007, Vamstad and Rotenberry 2010, Brehme et al. 2011, Banks et al. 2011, Arthur et al. 2012, Borchert and Borchert 2013, Letnic et al. 2013, Borchert et al. 2014*

np9020 [3]*Carnivore citations: Paragi et al. 1996, McLellan and Hovey 2001, Cunningham and Ballard 2004, Cunningham et al. 2006, Soyumert et al. 2010, Borchert 2012, Hanson 2013*

np9025 [4]*Ungulate citations: Taber and Dasmann 1957, McCulloch 1969, Bendell 1974, Peek 1974, Davis 1977, Keay and Peek 1980, Smith 2000, Holl et al. 2004, Dalerum et al. 2007, Bleich et al. 2008, Biggs et al. 2010, Joly et al. 2010, Greene et al. 2012* p0365 Studies cited include unburned areas compared with severely burned areas with no postfire logging; they exclude prescribed burns. For small mammals, only species with enough detections to determine directional response are reported.

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Non-Print Items

Abstract

Effects of mixed and severe fire on mammals vary spatially and temporally, by habitat type, and by species. Tree voles, masked shrews and some mice decrease, at least temporarily, after severe forest fire, but most bats and ungulates and many small mammals—especially deer mice and kangaroo rats—are strongly attracted to severely burned habitats due to novel foraging opportunities. In heavily burned forests, more insect prey is available for bats, and seeds and sprouting plants feed small mammals. Vegetation re-growth after intense fire produces highly palatable browse for elk, mule deer, and bighorn sheep. Standing dead trees provide cover for deer in severely burned forests, whereas bighorn sheep can more easily perceive predators in heavily burned chaparral. Mesocarnivores, including foxes, martens, and fishers, often are detected in forests that burned intensely. Unburned refugia within larger severe burns, and the time-since-fire, are especially important factors for recolonization by small mammals.

Keywords: Severe fire; Mammal; Bat; Rodent; Lagomorph; Carnivore; Ungulate; Forage.

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Key Points:
- Ten years postfire, 85% of fire-killed necromass remain in the forest
- Ten years postfire, fire-killed trees emit 0.6 Mg C ha^{-1} yr^{-1}
- While decomposition from fire-killed trees last decades, their contribution to NEP is small

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Carbon emissions from decomposition of fire-killed trees following a large wildfire in Oregon, United States

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Abstract A key uncertainty concerning the effect of wildfire on carbon dynamics is the rate at which fire-killed biomass (e.g., dead trees) decays and emits carbon to the atmosphere. We used a ground-based approach to compute decomposition of forest biomass killed, but not combusted, in the Biscuit Fire of 2002, an exceptionally large wildfire that burned over 200,000 ha of mixed conifer forest in southwestern Oregon, USA. A combination of federal inventory data and supplementary ground measurements afforded the estimation of fire-caused mortality and subsequent 10 year decomposition for several functionally distinct carbon pools at 180 independent locations in the burn area. Decomposition was highest for fire-killed leaves and fine roots and lowest for large-diameter wood. Decomposition rates varied somewhat among tree species and were only 35% lower for trees still standing than for trees fallen at the time of the fire. We estimate a total of 4.7 Tg C was killed but not combusted in the Biscuit Fire, 85% of which remains 10 years after. Biogenic carbon emissions from fire-killed necromass were estimated to be 1.0, 0.6, and 0.4 Mg C ha^{-1} yr^{-1} at 1, 10, and 50 years after the fire, respectively; compared to the one-time pyrogenic emission of nearly 17 Mg C ha^{-1}.

1. Introduction

Forest fires have long been recognized as an important component of the global carbon cycle. Among natural processes, combustion ranks second after metabolic respiration in mineralizing terrestrial biomass to the atmosphere, fire mortality ranks second after litter production in transferring live aggrading biomass to decomposing necromass, and the pyrolysis of biomass by forest fires feeds a global pool of black carbon which is largely isolated from the biological cycle [*Singh et al.*, 2012]. The role of forest fire in the carbon cycle is especially important in today's changing climate, not only because of its direct contribution to greenhouse gas emissions but also because a warming climate is expected to increase frequency and intensity of wildfires [*Flannigan et al.,* 2000, 2009*; Moritz et al.,* 2012], pushing the terrestrial biosphere toward a new equilibrium wherein less carbon resides in forest biomass and more resides in the atmosphere. Furthermore, because forest fire behavior is viewed by many as manageable, its control is regularly included as part of comprehensive climate change mitigation strategies [*Campbell et al.,* 2012; *Bradstock et al.,* 2012].

Characterizing and quantifying the effects of fire on the flux of carbon from forests into the atmosphere requires an understanding of both pyrogenic emissions due to immediate combustion and the prolonged biogenic emissions due to the decomposition (heterotrophic mineralization of carbon) by fire-killed necromass. A recent wealth of empirical studies aimed at quantifying combustion across a range of forest fires has allowed us to both constrain estimates of pyrogenic emissions and predict how this flux may change under alternate fire regimes (see reviews by *Sommers et al.* [2014] and *Urbanski* [2014]). By comparison, less attention had been paid to the protracted loss of terrestrial carbon to the atmosphere through the decomposition of fire-killed trees and how this flux is expected vary in relation to fire behavior or change under alternate fire regimes [*Harmon et al.,* 2011a, 2011b; *Ghimire et al.,* 2012].

Carbon emissions via the decomposition of fire-killed trees differ from pyrogenic emissions in several important ways. First, we expect that pyrogenic emissions to be lower in magnitude and less tightly coupled to fire behavior than subsequent carbon emissions via decomposition of fire-killed trees. Since combustion of aboveground biomass in forest fires is typically confined to dead surface fuels and live foliage, pyrogenic carbon emissions in any given fire tend not to exceed 15% of a forest's live and dead biomass [*Campbell et al.*, 2007; *Urbanski*, 2014]. Moreover, since the majority of surface fuels are consumed in nearly all fire conditions, while standing biomass experiences little combustion even in a crown fire, it is difficult for a high-mortality fire to combust much more than twice the amount of carbon than does a low-mortality fire. By contrast, subsequent carbon emissions through decomposition of biomass killed in the fire but not consumed may range from none (e.g., low-severity fires when no trees are killed) to all of the prefire biomass (e.g., high-severity fires when all trees are killed). For this simple reason, cumulative carbon emissions through decomposition of fire-killed trees may exceed pyrogenic emissions and are more dependent on fire behavior than are pyrogenic emissions.

Emissions through decomposition of fire-killed biomass also differ from pyrogenic emissions in their influence on Net Ecosystem Production (NEP). While pyrogenic emissions necessarily contribute to net ecosystem carbon balance, the flux itself is concentrated in time. By contrast, the protracted decomposition of fire-killed trees can contribute to disequilibrium in stand-level NEP for decades [*Bond-Lamberty and Gower*, 2008; *Harmon et al.,* 2011a; *Ghimire et al.,* 2012]. Theoretically, fire-induced disequilibrium in NEP will balance out to zero over sufficiently long time frames or spatial extents (after all, no tree ever escapes death and mineralization, fire only aggregates this inevitable emission in time). However, like many natural disturbances, the majority of area subject to high-mortality forest fire is the result of relatively few, very large events [*Malamud et al.,* 1998; *Reed and McKelvey,* 2002]. As such, the extent required for spatial neutrality in NEP to emerge may easily exceed any meaningful geographic boundary, and the time frame required for neutrality in NEP to emerge may easily exceed the meaningful continuity of any fire regime. Consequently, assessing the effects fire on the carbon exchange between forests and the atmosphere demands not only a mechanistic understanding of combustion, mortality, and decomposition (which we largely have) but also the ability to quantify these processes with enough context specificity to accurately account for individual fire events.

In this study, we evaluate the current and future carbon emissions attributable to the decomposition of trees killed but not combusted in the 2002 Biscuit Fire. This exceptionally large wildfire burned over 200,000 ha of mixed-conifer forest in southwest Oregon. Due to its diversity of forest types, forest age-classes, and severity of fire effects, the Biscuit Fire has served as a valuable case study for evaluating the effects of wildfire on carbon dynamics, including the following: pyrogenic emissions [*Campbell et al.,* 2007], export of soil carbon through erosion [*Bormann et al.,* 2008], and charcoal formation [*Donato et al.,* 2009a; *Heckman et al.,* 2013]. In *Campbell et al.* [2007] we reported biomass combustion for 25 functionally distinct carbon pools. Then, using measures of prefire biomass and fire effects on 180 one hectare inventory plots, we estimated fire-wide pyrogenic emissions. In this current companion study, we report the 10 year decay status of various biomass pools killed, but not combusted, by the Biscuit Fire. Then, using measures of fire mortality on the same 180 inventory plots as before, we estimate current and future fire-wide emissions resulting from the decomposition of fire-killed trees. Our specific objectives are as follows:

1. Quantify mortality, dead tree fall rate, and decomposition rates specific to different species, parts (e.g., root, bole, and branch), physical setting, prefire stand history, and fire effects.
2. Using these stratified parameters, calculate the current cumulative flux of carbon from fire-killed trees into the atmosphere and model its attenuation into the future.
3. Evaluate the current and future carbon emissions from fire-killed trees in the context of commensurate forest regrowth and other regional carbon fluxes, including the pyrogenic emissions from the same fire.

2. Methods

2.1. Study Site

The Biscuit Fire burned at a mix of severities across 200,000 ha of forest in the Siskiyou Mountains of southwestern Oregon and northern California in the summer of 2002, making it the largest contiguous forest fire on record for Oregon (Figure 1). The Siskiyou Mountains are characterized by a wide variety of forest types, from Douglas fir/western hemlock/bigleaf maple communities on mesic sites, to Douglas fir/tanoak on drier sites, to Jeffrey pine on ultramafic substrates *Whittaker* [1960]. A general description of the Biscuit Fire and the forests it affected can be found in *Halofsky et al.* [2011].

2.2. Decomposition of Fire-Killed Trees

As illustrated in Figure 2, decomposition of fire-killed trees was computed as the collective mass loss to the atmosphere, over a specified period, from three primary pools representing different physical orientations:

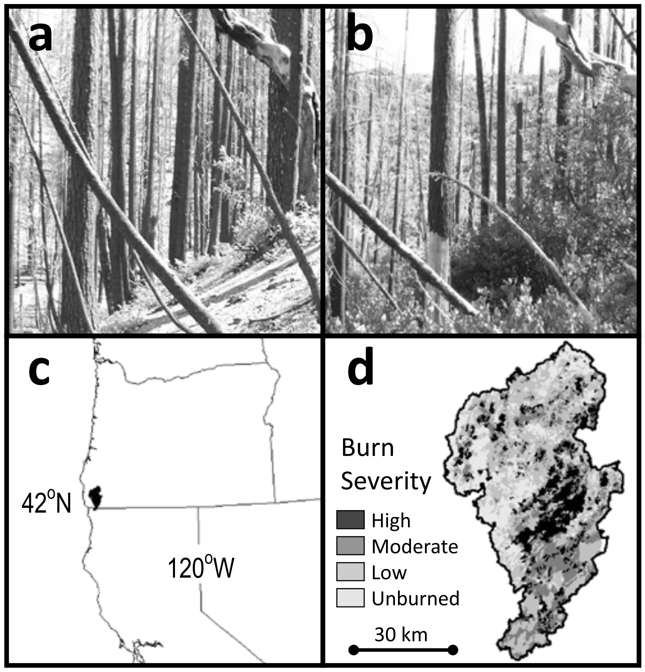

Figure 1. The 2002 Biscuit Fire showing (a) representative fire effects in 2004, (b) the same location in 2012, (c) location of the fire in the U.S. Pacific states of North America, and (d) remotely detected fire severity distribution. High = >90% overstory mortality, unburned = no overstory mortality but typically experiences surface fire.

standing necromass, fallen necromass, and buried necromass (i.e., dead root mass). Three separate rate constants defined mass loss to the atmosphere from standing, fallen, and buried necromass pools, respectively. Two additional rate constants defined transfer of mass from the standing to fallen pool by fragmentation and whole-tree fall, respectively. This three-pool, five-flux model was further stratified by tree part, namely, bole, branch, bark, and foliage (in the standing and fallen pools), and coarse root and fine root (in the buried pool). Boles were further stratified into three diameter classes, and all pools were stratified into three species groups (i.e., pines, non-pine conifers, and hardwoods) and three climatic zones (representing potentially different decomposition regimes) defined by aggregate plant association group and nominally corresponding to mesic, dry, and higher-elevation regions within the Biscuit Fire [*Donato et al.*, 2009b].

To estimate flux rates, we fit empirical observations of mass loss over time to a single-exponential model *Olsen* [1963] of the form:

$$M_t = M_0\left(e^{-kt}\right) \quad (1)$$

where M_t is the mass of a specified necromass pool at time t, M_0 is the mass of the same pool immediately following its death by fire and any assessed combustion, and t is the elapsed time since the fire (~10 years in this study). In this way, the rate constant k not only describes the cumulative mass loss at year t but can also be used to extrapolate mass loss into the future. The accuracy of such extrapolation does, however, depend on the assumption that loss rates remain constant over time, which may be violated if either the environment in which decay is occurring changes or if discriminating decay renders mixed substrates more recalcitrant over time. Extrapolation of our decay model does not account for climate-driven changes in the decay environment, but our model does account for important changes in decay that occur after wood

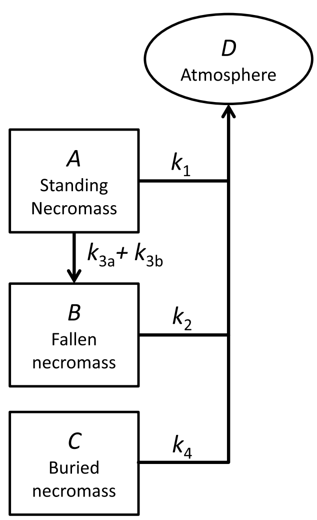

Figure 2. Approach to computing biogenic decomposition of fire-killed necromass. Decomposition was calculated separately for each plant tissue class according to five first-order exponential rate constants. The constant k_1 is the decomposition of necromass in its standing state; k_2 is the decomposition of necromass in its fallen state; k_{3a} and k_{3b} are the transfers between standing and fallen states, via whole-tree fall and fragmentation, respectively; and k_4 is the decomposition of buried roots.

Table 1. Methodology and Sampling Design for Determination of Rate Constants[a]

Aerial Decay: $k_1 = ln(D_0/D_t)/t$, where D_0 = live tree part density, D_t = density of standing fire-killed tree part circa 2012, and t = elapsed time since fire	
Bole	D_t measured for 198 trees, stratified by species group (Douglas fir, pine species, and pacific madrone), diameter class (range 7 to 146 cm DBH), and climatic zone (defined by aggregate plant association group, nominally corresponding to mesic, dry, and higher-elevation regions within the Biscuit fire). Tree-average density was calculated as the average density of three transverse samples (cookies) collected from the lower, middle, and upper third of each tree, weighted by a factor of 0.60, 0.36, and 0.04, respectively, to account for volume proportion by height (derived from the taper equations of *Arney* [2009]). D_0 assumed to be 0.39, 0.45, and 0.58 g cm^{-3} for sugar pine, Douglas fir, and pacific madrone, according to *Maeglin and Wahlgren* [1972], *USFS* [1965], and Wood Data Base, respectively.
Branch	D_t measured for 259 branches stratified by diameter (range 1 to 56 mm) collected from the 198 standing dead trees described above. D_0 measured for 55, similarly stratified live tree branches samples.
Bark	Bark density loss was not directly measured in this study. Based on *Allison and Murphy* [1963], we crudely assumed bark to decompose at one half the rate of bole wood of the same species. Anecdotally, bark from fire-killed trees in this study regularly showed evidence of charring and fragmentation but not any apparent density loss.
Foliage	Aerial decay rates of fire-killed foliage are computationally inconsequential, not only because fire mortality on the Biscuit most often entailed full foliage combustion [*Campbell et al.,* 2007] but also because fall rates of fire-killed foliage approach totality within the first year after mortality such that nearly all decay occurs on the ground. As such, foliage aerial decay rates were arbitrarily set to 0.5 year^{-1}.
Surface Decay: $k_2 = ln(D_0/D_t)/t$, where D_0 = live tree part density, D_t = density of fire-killed tree part having fallen to ground shortly after fire, and t = elapsed time since fire	
Bole	D_t measured on 60 fallen logs, deduced to have been killed in the Biscuit Fire (by presence of surface charring) and fell within the next year (saw cuts datable to known salvage operations); stratified by species group (see above), diameter class (range 7 to 146 cm DBH), and climatic zone (see above). Density was determined from a single transverse sample (cookie) taken from the center of each log. D_0 as described above for areal bole decay.
Branch	D_t measured for 86 branch samples, stratified by diameter (range 1 to 72 mm) collected from the 60 fallen logs described above. D_0 as described above for areal branch decay
Bark	Crudely assumed to be one half the rate of fallen bole wood (see above for aerial bark decay).
Foliage	Given the short residence times of leaf litter (relative to wood and bark), and the exceptionally small portion of fire-filled biomass represented by uncombusted foliage [*Campbell et al.,* 2007], we chose to avoid the hazard of false accuracy and simply assign foliage decomposition rates the arbitrarily rapid rate of 0.5 year^{-1}.
Whole-tree Fall Rate: $k_{3a} = ln(C_0/C_t)/t$, where C_0 = count of standing dead trees circa 2004, C_t = count of standing dead trees ca 2013, and t = elapsed time between samples	
Whole tree	Before-and-after stem surveys conducted at 44 independent and dispersed study plots, including a total sample size of >3000 fire-killed trees ranging in size from 2 to 198 cm DBH.
Fragmented Fall Rate: $k_{3b} = ln(M_0/M_t)/t$, where M_0 = mass of standing tree parts circa 2004, M_t = mass of standing tree parts circa 2012, and t = elapsed time between samples	
Bole	M_0 allometrically modeled from DBH with the assumption that each tree was live and entire. M_t is the same value, corrected to account height loss due to observed breakage. Assessed for each of the 3000 fire-killed trees described above.
Branch	M_0 allometrically modeled from DBH with the assumption that each tree was live and entire. Each fire-killed tree surveyed in 2014 was binned into one of four fragmentation classes through ocular assessment, corresponding to an M_t of 0.05 M_0, 0.15 M_0, 0.60 M_0, and 1.0 M_0, respectively.
Bark	M_0 allometrically modeled from DBH with the assumption that each tree was entire. Each fire-killed tree surveyed in 2014 was binned into one of four fragmentation classes through ocular assessment, corresponding to an M_t of 0.0, 0.25 M_0, 0.75 M_0, and 1.0 M_0, respectively.
Foliage	Practically all uncombusted foliage retained on fire-killed trees fell to the ground within the first year after the fire. To account for this in our decomposition model (constructed only of first-order exponential rate constants) we set the rate constant describing dead foliage fall to 5.0 year^{-1}.
Buried Decay: k_4 = first-order exponential decay constants according to named authors	
Coarse root	k = 0.02 year^{-1} according to *Janisch et al.* [2005] assessment of Douglas fir roots > 1.0 cm diameter.
fine root	k = 0.20 year^{-1} according to *Chen et al.* [2002] and *Fogel and Hunt* [1979] for various tree roots < 1.0 cm diameter.

[a]Dead wood density was determined after oven drying at 95°C to constant mass; an 8% downward correction was then applied to account for oven shrinkage and afford direct comparison with published green tree densities [*Glass and Zelinka*, 2010].

transitions from the aerial to surface environment. Furthermore, by disaggregating our necromass pools (i.e., into bole, branch, bark, foliage, root, species group, and size class) our model minimizes the changes in recalcitrance that any one pool may experience over time [*Freschet*, 2012]. The specific sampling methods used to determine M_t and M_0 for each necromass category are detailed in Table 1. Note that while the form of equation (1) was used in computing all flux rates, at times, density, volume, or count was operationally substituted for mass.

2.3. Initial Fire-Killed Biomass

Within the perimeter of the Biscuit Fire there are 180 regularly spaced permanent federal inventory plots, all of which received postfire measurements in 2003 or 2004 [*Azuma et al.,* 2004]. It is well established that injury caused by fire can sometimes contribute to tree death several years after being burned [*Filip et al.,* 2007].

Our assessment operationally defines fire mortality as trees which died within 1–2 years after the fire. Any subsequent mortality and ensuing decomposition, though perhaps related to fire, was not in this study directly attributed to the Biscuit Fire.

For each tree identified in the inventory plots as having been killed in the Biscuit Fire, we estimated the mass of its fine roots, coarse roots, bole, branch, bark, and foliage as if it were alive and whole. From each of these parts, we then subtracted the proportion estimated to have been combusted in the fire according to *Campbell et al.* [2007] to yield a tree-specific M_0 for each of its component parts. Bole mass was estimated using species- and site-specific allometric equations relating stem diameter to volume and species-specific wood density values [*van Tuyl et al.,* 2005]; foliage and bark mass were estimated directly from species- and site-specific allometric equations [*Means et al.,* 1994]; coarse root mass was assumed to be 0.31 times the bole mass (an average of regionally representative, plot-level ratios, allometrically estimated by *Campbell et al.* [2004a]); and fine root mass was assumed to be 0.16 times the bole mass (an average of regionally representative, plot-level ratios directly sampled by *Campbell et al.* [2004b]). Total biomass was converted to carbon mass assuming a carbon concentration of 0.5 for all woody parts and 0.45 for foliage. These tree-level values for M_0 were then summed across each inventory plot as to be expressed in carbon mass per unit ground area.

2.4. Fire Severity and Scaling Across the Fire

For evaluating the direct effects of fire severity on subsequent carbon emissions, fire severity was calculated, for each of the 180 inventory plots, as the fraction of initial live basal area (including all woody stems $>=$ 2.5 cm diameter breast high (DBH)) killed in the Biscuit Fire. For the purpose of scaling plot-level measurements to the entire Biscuit Fire it was necessary to use a mapped assessment of fire severity. Specifically, plot-level estimates of decomposition were scaled-up to the entire Biscuit Fire according to mapped fire severity classification and whether or not a site had burned in the Silver Fire (a major fire which burned 13 years prior to the Biscuit Fire). Such strata accounted only for variation in M_0 (tree mass killed in the Biscuit Fire), as the rate constants k were assumed to be the same among plots. We employed the same BAER (Burned Area Emergency Response) severity classification map used earlier by *Campbell et al.* [2007]. Since this time, improved maps of Biscuit Fire severity have been built [*Thompson and Spies,* 2009], but we felt it was more important to maintain consistency between our pyrogenic and biogenic accounting. Moreover, since the 180 inventory plots are distributed widely in space and randomly with respect to actual fire effects, misclassification by BAER, or any other severity map, does not bias fire-wide estimates of carbon flux.

2.5. Uncertainty Propagation

For this study, we assumed the inventory-based estimates of fire-killed necromass to be largely accurate and limited our uncertainty analysis to that associated with decomposition rates. To account for this uncertainty, we computed alternate estimates of total carbon emissions using an upper and lower values for the rate constants defining mass loss to the atmosphere. Uncertainty in mass loss from standing and fallen necromass pools (k_1 and k_2 in Figure 2) were based on the upper and lower 95% confidence intervals in dead wood density (among samples collected in 10 years after death). Since we relied on crude literature values for root decay, uncertainty in mass loss from buried necromass pools (k_4 in Figure 2) was generously set to plus and minus 20% density loss at 10 years after death.

3. Results

3.1. Fire Mortality

Prefire live aboveground and belowground biomass among the 180 inventory plots ranged from 1 to 502 (median = 161) Mg C ha^{-1} depending somewhat on site quality but largely disturbance history (i.e., whether sites had experienced late twentieth century fire). Fractional tree mortality, which was largely independent of prefire biomass, ranged from zero to totality. As a result the necromass killed but not combusted among the 180 inventory plots ranged from 0 to 352 (median = 24) Mg C ha^{-1} . Despite smaller trees being more abundant, more often killed, and only somewhat more combusted than larger trees, fire mortality in the form of large-diameter (>30 cm DBH) boles and their associate coarse roots made up greater than 40% of all other fire-killed biomass combined. The remaining uncombusted fire mortality is composed of smaller diameter wood, bark, fine roots, and foliage in that order (Table 2). Overall the Biscuit Fire killed and left uncombusted a total of 10.4 Tg C (an average of 51 Mg C ha^{-1}).

Table 2. Biomass Killed But Not Combusted in Biscuit Fire (kg C ha^{-1})

	Biscuit Fire Severity[a]							
	Not Burned 15 years Earlier in Silver Fire				Also Burned 15 years Earlier in Silver Fire			
Necromass Pool	High	Moderate	Low	Unburned Very Low	High	Moderate	Low	Unburned Very Low
				Foliage				
Small conifers	19	62	58	26	0	0	4	2
Small hardwoods	31	23	57	99	2	42	29	67
Medium conifers	135	367	232	131	0	1	32	29
Medium hardwoods	292	77	354	606	3	151	289	763
Large conifers	180	384	409	162	0	242	67	190
Large hardwoods	52	7	67	162	0	37	98	46
				Branch				
Small conifers	130	115	88	34	0	13	6	3
Small hardwoods	144	37	106	159	146	142	50	120
Medium conifers	1207	981	501	247	0	83	78	60
Medium hardwoods	1778	183	1026	1407	53	1130	806	2202
Large conifers	3279	1837	1438	523	0	2598	350	683
Large hardwoods	835	23	280	610	0	2540	387	211
				Bark				
Small conifers	111	109	86	34	0	12	6	3
Small hardwoods	76	22	65	100	75	83	31	76
Medium conifers	1284	1184	607	314	0	95	95	78
Medium hardwoods	1314	135	861	1207	44	955	701	1917
Large conifers	5019	3097	2641	962	0	4760	639	1281
Large hardwoods	877	23	318	748	0	2944	446	237
				Bole				
Small conifers	537	409	328	146	0	69	28	13
Small hardwoods	1220	250	876	1348	1333	1272	416	974
Medium conifers	7058	5254	2734	1425	0	555	462	401
Medium hardwoods	16733	1559	9027	12772	557	10730	7152	15988
Large conifers	31100	16967	13599	4950	0	28981	3632	6380
Large hardwoods	6885	186	2206	4947	0	21250	3576	1461
				Roots				
Small conifers	193	147	118	52	0	25	10	5
Small hardwoods	439	90	315	485	479	457	150	350
Medium conifers	2538	1890	983	512	0	199	166	144
Medium hardwoods	6017	561	3246	4593	200	3858	2572	5749
Large conifers	11184	6101	4890	1780	0	10422	1306	2294
Large hardwoods	2476	67	793	1779	0	7642	1286	525

[a]As determined by remotely sensed BAER severity classification. Values are the average of 24, 36, 42, and 34 inventory plots for high, moderate, low, and unburned very low severity plots not burned prior in the Silver fire, respectively; and the average of 1, 2, 14, and 5 inventory plots for high, moderate, low, and unburned very low severity plots burned prior in the Silver fire, respectively. Small trees are <10 cm DBH, medium trees are 10–20 cm DBH, and large trees are >20 cm DBH. For our decomposition calculations, conifers were further partitioned into pine and nonpine species (data not shown here), and roots were partitioned into coarse roots and fine roots, consistently computed as 0.66 and 0.34 total root mass, respectively.

3.2. Decomposition Rates

The measured densities of standing and fallen fire-killed wood, from which decomposition rates were calculated, are shown in Figure 3. An analysis of variance performed on the decomposition rates calculated for over 198 sampled tree boles revealed significant effects of species (with Douglas fir decomposing only slightly faster than pine species and pacific madrone) and condition (fallen logs decomposing only slightly faster than standing snags), but nonsignificant effects of geographic zone (mesic, dry, or high elevation) or size (diameter class). The single-exponent decomposition constants (fit to a single 10 year data point and used to subsequently model carbon emissions) are shown in Table 3.

3.3. Tree Fall Rates

As shown in Table 4, a greater fraction of fire-killed biomass fell from the canopy to the ground in 10 years through whole-tree fall than through fragmentation. The proportion of whole trees having fallen after

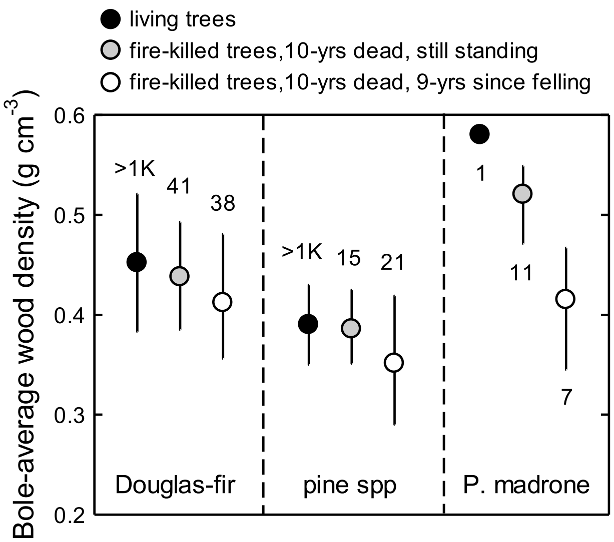

Figure 3. Wood density of green trees (live), fire-killed trees still standing 10 years after death (snags), and fire-killed trees 10 years after death and near immediate falling (logs). Sample size (shown near each symbol) is the number of independent trees sampled, with the density of each being determined as the taper-weighted average density of three cross-sectional subsamples taken along the length of each tree. Variability in wood density among trees is shown as the standard deviation (upper and lower error bars are the average positive and negative residuals of the mean, respectively; except for green trees where only a single symmetrical standard deviation was available from source literature, and green madrone where no variance was reported). Live wood densities are from *Maeglin and Wahlgren* [1972], *US Forest Service* [1965], and *Wood Data Base*, for pine species, Douglas fir, and Pacific madrone, respectively. Dead wood densities are those measured in the present study.

10 years was 20 times greater for smaller diameter trees (<20 cm DBH) than for larger diameter trees. Neither whole-tree fall rate nor fragmentation rate varied according to community type (used here as a proxy for decomposition regime). Across species, size class, and location, 57% of the trees killed in the Biscuit Fire are still standing 10 years after their death and have on average lost only 26% of their postcombustion necromass via fragmentation.

3.4. Biogenic Emissions

The amount of carbon released through the decomposition of fire-killed trees in the first 10 years following the Biscuit Fire is estimated to be 1.3 to 1.6 Tg C (or 6.5 to 7.8 Mg C ha^{-1}). As shown in Table 5, the largest contributing pools were those with the largest initial mass (i.e., bole wood and coarse root), not those with the highest decomposition rates (foliage and fine roots). Extrapolating our 10 year estimates of fall rates and decomposition rates back to the first year following fire and forward to 100 years after fire reveals several emergent patterns. Partitioning emission rates among necromass pools (Figure 4a) illustrates not only differential decay rates (responsible for the inflection point in collective emissions) but also an important 10 year lag in peak emissions from bole, branch, and bark, which results from a particular combination of aerial decay rates, fall rates, and surface decay rates. Total emissions from fire-killed necromass over time exhibit a distinct inflection point approximately five years following the fire (Figure 4b). Such inflection points are indicative of mixed substrate decay and in this case occur when the more labile foliage and fine root pools have become largely exhausted leaving the more recalcitrant wood and coarse roots. Overall, half of the Biscuit-killed necromass will still remain 50 years after the fire, at which time emissions from this single mortality cohort will be approximately 25 Mg C ha^{-1} yr^{-1} (Figure 4c).

The total amount of fire-killed necromass explained 99% of the variation in post fire decomposition among the 180 study plots (Figure 5a), indicating that variation in prefire species composition and tree size class was of

Table 3. Decomposition Constants for Fire-Killed Necromass[a]

	Decomposition Constant *k* (year^{-1})	
Necromass Pool	Aerial Decay(Standing Snags)	Surface Decay(Fallen Logs and Debris)
	Bole	
Nonpine conifers	0.010 (0.008–0.012)	0.016 (0.013–0.019)
Pines	0.001 (0.001–0.004)	0.010 (0.005–0.014)
Hardwoods	0.010 (0.008–0.012)	0.016 (0.014–0.018)
	Branch	
All species	0.014 (0.013–0.015)	0.010 (0.008–0.012)

[a]Decomposition constant $k = \ln(\text{Density}_{\text{live}}/\text{Density}_{11\ \text{years dead}})/11$ years. Upper and lower estimates shown in parentheses were computed using standard error of the mean $\text{Density}_{11\ \text{years dead}}$. See Table 1 for assumptions regarding decomposition of other fire-killed necromass pools such as foliage, bark, and roots.

Table 4. Fall Rate of Fire-Killed Necromass[a]

Necromass Pool	Number of Trees Sampled	Fraction Fallen After 10 years		Fall Rate k (year^{-1})	
		Via Whole-Tree Fall	Via Fragmented Fall	Via Whole-Tree Fall	Via Fragmented Fall
Bole					
Conifers (small)	156	0.86		0.177	
Conifers (medium)	407	0.36		0.041	
Conifers (large)	805	0.03		0.003	
Hardwoods (all sizes)	229	0.03	0.35	0.003	0.043
Nonpine conifers (all sizes)	1075		0.16		0.017
Pines (all sizes)	137		0.14		0.016
Branch					
Conifers (small)	156	0.86		0.177	
Conifers (medium)	407	0.36		0.041	
Conifers (large)	805	0.03		0.003	
Hardwoods (all sizes)	229	0.03	0.41	0.003	0.053
Nonpine conifers (all sizes)	1075		0.42		0.054
Pines (all sizes)	137		0.50		0.070
Bark					
Conifers (small)	156	0.86		0.177	
Conifers (medium)	407	0.36		0.041	
Conifers (large)	805	0.03		0.003	
Hardwoods (all sizes)	229	0.03	0.51	0.003	0.070
Nonpine conifers (all sizes)	1075		0.48		0.065
Pines (all sizes)	137		0.57		0.085

[a]Fall rate $k = \ln(\text{standing necromass}_{2004}/\text{standing necromass}_{2014})/10$ years. Small trees are <10 cm DBH, medium trees are 10-20 cm DBH, and large trees are >20 cm DBH.

little importance in dictating postfire decomposition. Moreover, since low-biomass stands often experienced high-fractional mortality and high biomass often experienced low-fractional mortality, fire severity (as assessed by fractional basal area mortality) was, by itself, an imprecise predictor postfire carbon emissions (Figure 5b).

4. Discussion

4.1. Fire Mortality

The necromass generated in high-severity portions of the Biscuit Fire (about 103 Mg C ha^{-1}) corresponds well to the 130–200 Mg C ha^{-1} biomass held in mature and old-growth forests of the Klamath ecoregion according to the regional assessment of *Hudiburg et al.* [2009]. Between the ages of 50 and 100, these particular forests are estimated to experience tree mortality rates of just over one-half percent annually [*Hudiburg et al.*, 2009]. As such, when mature forests burned at high severity in the Biscuit, somewhere between 100 and 200 years of future mortality was compressed into a single event. When individual fires of this size and severity occur in high biomass forests, like those of western Oregon, the generation of decomposing necromass is

Table 5. Biogenic Emissions From Fire-Killed Necromass by Carbon Pool and Burn Severity Class

Necromass Pool	Carbon Released (kg C ha^{-1} After 10 years)				Fire-Wide Emissions[b] (Tg C Across 202,642 ha, After 10 years)
	High Severity[a]	Moderate Severity[a]	Low Severity[a]	Unburned Very Low Severity[a]	
Foliage	676	887	949	1163	0.19
Branch	850	365	307	324	0.08
Bark	405	219	173	168	0.04
Bole	5450	2125	2095	2362	0.54
Roots	6040	2385	2164	2277	0.58
Total	13421	5982	5683	6294	1.44 (1.31–1.59)[c]

[a]As determined by remotely sensed BAER severity classification.
[b]Fire-wide emissions calculated by weighting the emissions from each burn class by the area of that burn class over the fire perimeter.
[c]Upper and lower estimates based on propagated uncertainty in woody decomposition rate constants.

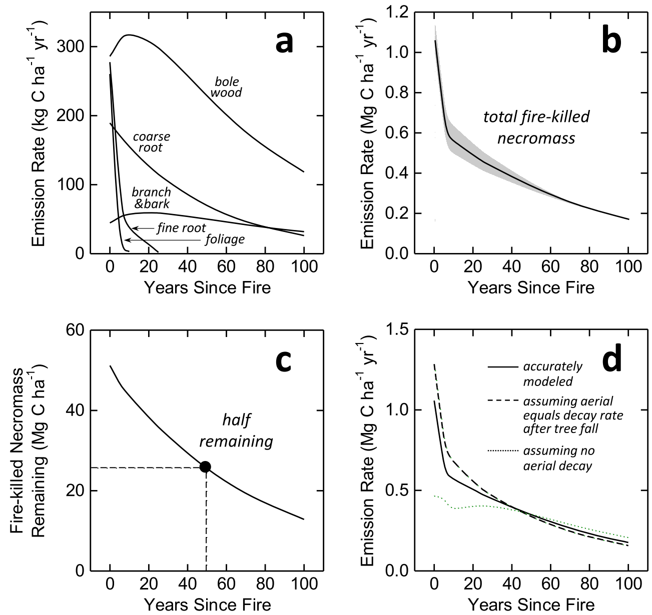

Figure 4. Temporal patterns of carbon emissions from fire-killed necromass, (a) partitioned by pool; (b) total, illustrating inflection point around year 5 and propagated uncertainty in decomposition rates (shaded band = 95% confidence interval); (c) approximate 50 year half-life; and (d) consequences of recognizing differential aerial and surface (fallen) decay rates.

notable at regional and even continental scales. The total amount of carbon transferred by the Biscuit Fire from aggrading living pools into decomposing dead pools was approximately three-quarters the average amount killed annually by wildfire throughout the entire western US (6 Tg C yr^{-1}) [*Hicke et al.,* 2013]. The distribution of fire mortality among different pools (Table 2) is a simple reflection of within-tree allometric proportions sans foliage which is commonly combusted in fire-killed trees. Understandably then, large-diameter wood made up the largest fire-generated necromass pool, more so in forests not recently burned where an even greater proportion of biomass was in the form of bole wood.

Due largely to the wide range of prefire biomass, fractional fire mortality (whether inferred through remote imagery, or direct ground measurement) was a poor predictor of absolute mortality and subsequent carbon emissions. While both intuitive and expected, this observation reminds us of the importance of accurately assessing preburn biomass in mapping and modeling fire effects on carbon dynamics.

4.2. Decay Rates

The wood density decomposition rates reported here fall comfortably within the range reported by other studies in the Pacific Northwest [*Sollins,* 1982; *Harmon et al.,* 1986; *Janisch et al.,* 2005; *Harmon et al.,* 2011b; *Dunn and Bailey,* 2012], which both validates our assessment and brings into question the need for additional field studies, at least those using single-exponent decay models fit to mass loss over a single time interval. In reality, necromass decay over time is expected to exhibit some initial lag (as substrates await decomposer colonization or fragmentation) and a decreasing proportional loss over time (as mixed substrates are reduced to their more recalcitrant fractions). By measuring mass loss across a chronosequence of dead wood, *Harmon et al.* [2000] demonstrated that dead wood decay can, in fact, exhibit such lags and tails in mass loss over time. Still, provided necromass pools are appropriately disaggregated (i.e., relatively recalcitrant and labile substrates assigned their own loss rate constants), single-exponent models like those used in this study fit empirical data just as well as multiparameter models [*Freschet,* 2012].

Given the recognized effects of moisture and temperature on decomposition, our inability to detect site effects on decomposition rate was likely a combination of measurement error (driven largely by our use of a single-species-specific green tree wood density in assessing mass loss for all wood fragments) and a wide variation in realized decay environments within the crude climate zones we recognized (Table 1). Given our samples were so widely distributed across our study area, our mean decomposition rates remain good estimates for our particular study. However, caution should be taken in applying these or any other landscape-average decomposition rates to any particular site, as decay rates of common substrates may vary across forest microenvironments by as much as 10 times, more so even than across large-scale climate gradients [*Vanderhoof*, 2013; *Bradford et al.,* 2014].

4.3. Fall Rates

The fall rates of standing necromass by fragmentation and whole-tree fall pertain to carbon emissions only to the degree that decomposition rates are different between the aerial and surface environments. It is commonly

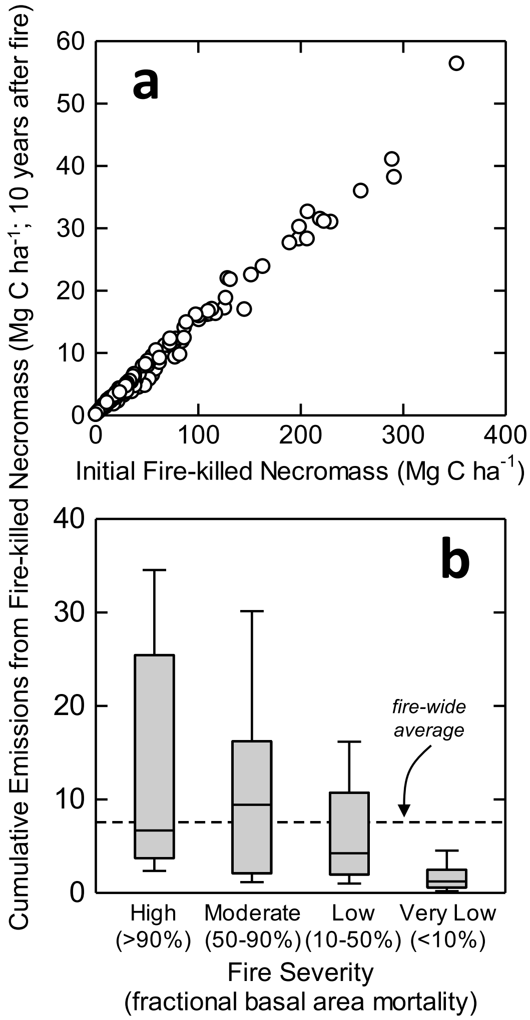

Figure 5. Carbon emissions from fire-killed necromass as a function of (a) absolute mortality and (b) fire severity among 180 inventory plots regularly stratified across the Biscuit Fire. Centerline, box, and whiskers, represent median, 25th percentiles, and range up to three-halves end quartiles (i.e., range excluding outliers), respectively. Fire severity (fractional tree basal area mortality) was directly determined for each plot (not remotely sensed).

assumed and consistently observed that decay rates of wood are slower in the drier aerial environment than in the moister surface environment [*Harmon et al.,* 2011b; *Yatskov et al.,* 2003; *Dunn and Bailey,* 2012]. Ecosystem models which apply the more commonly available surface decay rates to all fire mortality, without considering the decades many dead trees may spend in a standing condition, will inevitable overestimate initial emission rates and underestimate their duration. Similarly, models which assume negligible wood decay until a dead tree falls are prone to an inverse bias. The significance of tree fall rates in the timing of postfire carbon emissions is apparent in Figure 4a where peak emissions from branch, bark, and bole wood occur not immediately following the fire (when pool sizes are necessarily largest), but rather 10–20 years following the fire (after a requisite portion of the pool has fallen to the ground where it decays quicker). To further evaluate the relevance of tree fall on carbon emissions following the Biscuit Fire, we compared our fully parametrized model to others with alternate assumptions regarding fall rate and differential decay. As illustrated in Figure 4d, the largest bias occurred in the model which assumed wood remained undecayed until it fell to the ground. Applying a single surface decay rate to all wood did overestimate the near-term emission rates, but not as much as purported for other disturbed forests where both fall rates and the disparity between aerial and surface decay were determined to be higher than we observed in the Biscuit Fire [*Harmon et al.,* 2011b]. Moreover, once combined with the consistently attenuating emission from fire-killed roots and foliage, the fall-mediated lag in emissions from bole, branch, and bark did not produce a bimodal or "double-humped" emission pattern as it might have [*Harmon et al.,* 2011a].

Some authors have reported a brief (2 to 3 year) delay between tree mortality and the onset of measurable fall (see review by *Cluck and Smith* [2005]), suggesting that fall rates sometimes accelerate after passing some threshold in declining stability (e.g., root or basal decay). Since snag fall in this study is evaluated using stem attrition measured only at one-time point (10 years after death), we cannot resolve any early changes in fall rate. However, as a general rule, snag attrition measured over decades in prior studies conforms well to a first-order decay function as we have done here [*Everett,* 1999; *Cluck and Smith,* 2005]. Necromass decay over time is expected to exhibit some initial lag (as substrates await decomposer colonization or fragmentation) and a decreasing proportional loss over time (as mixed substrates are reduced to their more recalcitrant fractions).

4.4. Emission Rates

It is expected that dead wood dynamics operate over longer time scales in the Pacific Northwest than they do in other forests where environmental conditions or disturbance frequency prevent individual trees from growing as large. The analysis by *Spies and Franklin* [1988] suggests it would take >1000 years for woody

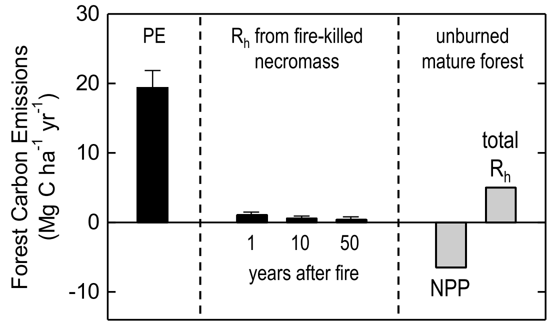

Figure 6. Forest carbon emissions from heterotrophic respiration (R_h) of necromass killed in the Biscuit fire, compared to the one-time pyrogenic emissions (PE) incurred during the fire and the biological fluxes typical of unburned mature forests of the Klamath region. Error bars on R_h are propagated 95% confidence intervals in decomposition rates. Pyrogenic emissions and uncertainty estimated by *Campbell et al.* [2007]. Net Primary Production (NPP) modeled by *Turner et al.* [2007] and consistent with empirical observations of *Hudiburg et al.* [2009]. Total R_h, which includes both the heterotrophic fraction of soil surface efflux and dead wood decay, modeled by *Turner et al.* [2007] and consistent with empirical observations of *Campbell et al.* [2004a, 2004b].

debris to reach a site-level steady state in Western Oregon, and as such, most forests in the region exist in a state of dead wood disequilibrium defined by site-specific disturbance history. In this study, the measured magnitude and modeled duration of carbon released to the atmosphere through the decomposition of fire-killed trees speaks to how disturbance-generated mortality shapes not only the amount woody debris present at any given time, but in the exchange of carbon occurring between forest and atmosphere at any given time.

As shown in Figure 6, 10 years after the Biscuit Fire the annual flux of carbon from fire-killed trees into the atmosphere is estimated to be 0.6 Mg ha^{-1} yr^{-1}, which is only 10% the total heterotrophic respiration rates to which these forests hypothetically equilibrate once mature [*Turner et al.,* 2007; *Campbell et al.,* 2004a] and only 3% the one-time pyrogenic emissions released during the fire [*Campbell et al.,* 2007]. Clearly, the capacity of this relatively modest carbon flux to shape carbon exchange between forest and atmosphere has not to do with its magnitude, but rather its duration and the fact that other ecosystem carbon fluxes such as net primary production, and potentially soil surface efflux, are greatly reduced in the initial period following wildfire.

Several studies suggest that high-severity wildfire, despite generating substantial additions to the dead wood pool, actually reduces total heterotrophic respiration by about one half [*Meigs et al.,* 2009; *Dore et al.,* 2012]. This is because wildfire typically consumes the forest floor (the substrate from which up to 30% of total heterotrophic respiration arises; *Campbell et al.* [2004b]) and temporally cuts off the supply of fine root turnover (a sizable contribution to belowground heterotrophic respiration). It was not the purpose of the paper to compute postfire NEP which would depend largely on uncertain patterns of forest regrowth and mineralization of soil carbon; however, NPP of regenerating and surviving vegetation need only reach 0.57 Mg ha^{-1} yr^{-1} by the 10 year following fire in order to compensate for the respiration from the remaining fire-killed necromass. Preliminary measurements (unpublished data) suggest that shrub production alone 10 years after the Biscuit Fire has already far exceeded this rate, consistent with other studies showing NPP over 1.5 Mg C ha^{-1} by 2 years postfire in dry forests [*Irvine et al.,* 2007].

4.5. Regional Carbon Disequilibrium

Single, large disturbances like the Biscuit Fire make for valuable examples because they provide a broad range of conditions over which to stratify measurements. The specificity with which we evaluated mortality, fall, and decay within the Biscuit Fire was limited only by resources, not by opportunity. But quantifying the impacts of single events such as the Biscuit Fire also sheds light on the unique importance of rare events in shaping regional carbon exchange and the need to accurately account for them when either upscaling terrestrial measurements or downscaling atmospheric measurements.

It is reasonable to postulate, as *Odum* [1969], that over a sufficiently large landscape, disturbance-induced disequilibrium in any one location will be balanced in other locations experiencing similar disturbances at different times, and as long as the region-wide frequency of such disturbances remains constant, this shifting mosaic will operate with mass neutrality (e.g., NEP). However, within many ecoregions forest fires may not occur at fine-enough grain and high-enough frequencies for such equilibriums to arise. In fact, the self-organizing behavior of fire across landscapes dictates that most of the area burned in any given fire regime is the result of relatively few, very large events [*Malamud et al.,* 1998; *Reed and McKelvey,* 2002]. This disproportional impact of large infrequent disturbances thwarts landscape equilibriums in two dimensions. First, it can extend the area required to balance disturbance effects at any given time beyond meaningful ecological boundaries. Second, it can extend the time horizon required for any bounded area to achieve equilibrium beyond the period we expect disturbance regimes to be reasonable stable. This second constraint on landscape equilibrium is especially relevant considering climate change may now be altering probabilistic fire regimes faster than the return interval of the most important events [*Zinck et al.,* 2011], rendering the realized impacts of fire on processes such as carbon emission wildly stochastic in space and time.

As illustrated in Figure 6, the carbon emissions attributed to the decomposition of trees killed in the Biscuit Fire documented in this study, as well as the pyrogenic emissions released by the Biscuit Fire documented in *Campbell et al.* [2007], attest to the importance single-disturbance events can have in regional carbon dynamics, especially in large biomass systems confined to relatively small ecological boundaries. Predicting the frequency of these rare events will be increasingly difficult in a changing environment, but our ability to accurately assess their impacts on regional carbon flux is slowly approaching sufficiency.

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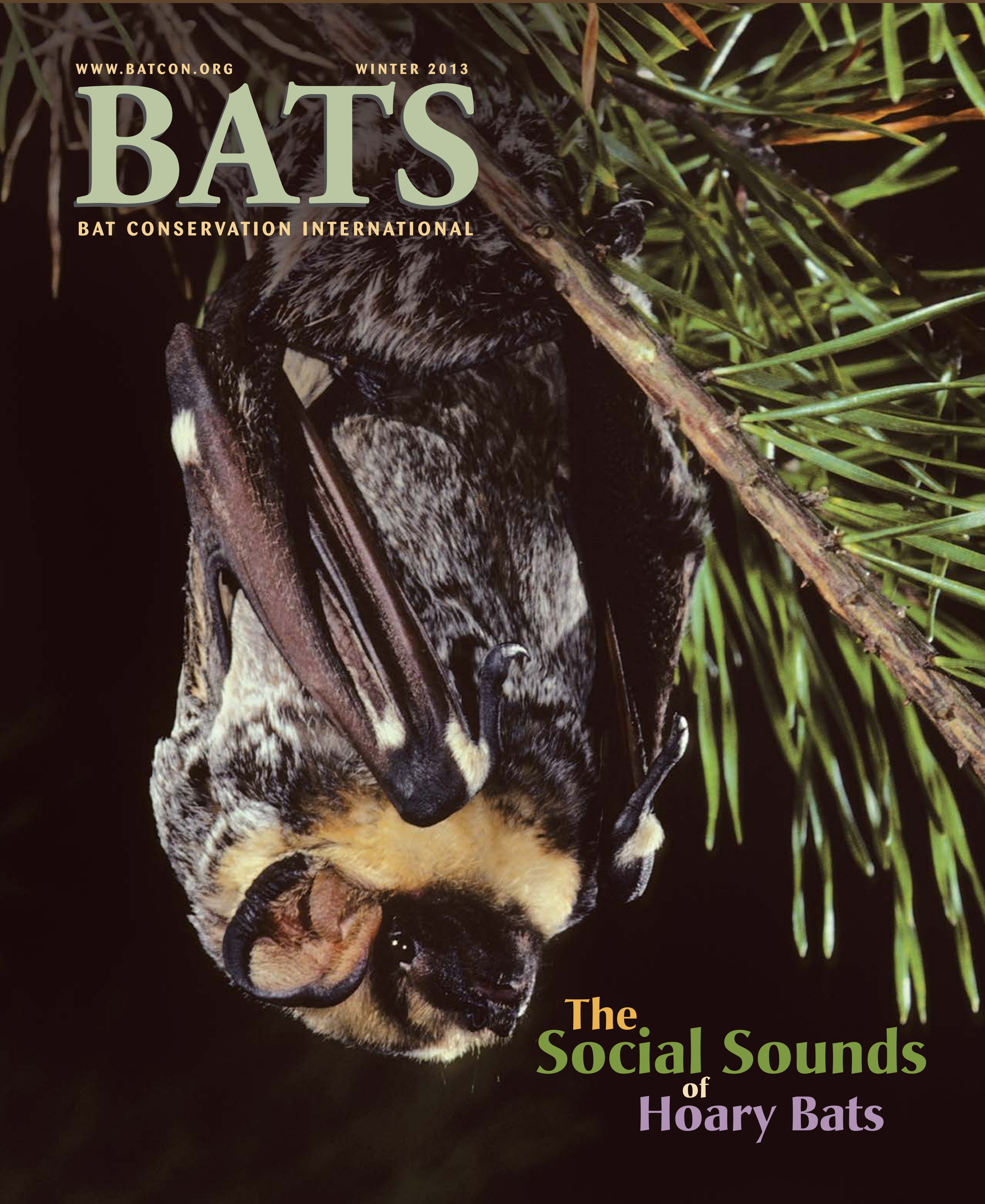
SURPRISING DIVERSITY OF BATS IN SENEGAL

BCI'S FIVE-YEAR PLAN FOR GLOBAL BAT CONSERVATION

HOW BATS ADAPT TO WILDFIRES & CLIMATE CHANGE



WWW.BATCON.ORG WINTER 2013

BATS

BAT CONSERVATION INTERNATIONAL

The Social Sounds of Hoary Bats

COURTESY OF ERIN SAUNDERS

Researchers captured bats in mist nets over this pond in a severely burned area of the Apache-Sitgreaves National Forests in Arizona during their study of how bats respond to forest fires.

Bats in the Burns

Studying the impact of wildfires and climate change

by Carol Chambers and Erin Saunders

The Wallow Fire began with an abandoned campfire on the Apache-Sitgreaves National Forests in Arizona's White Mountains on May 29, 2011. By the time it was controlled 40 days later, it had become the largest wildfire in the state's history. Flames blazed across 538,000 acres that range from high-country grasslands to the giant pine forests favored by bats. And bats, like most other wildlife, will likely face more and more charred habitat in the years to come. Thanks to decades of fire suppression and livestock grazing, plus the stirrings of climate change, wildfires are becoming bigger and more frequent throughout the American West.

Our field crew, a half-dozen biologists – plus 50 volunteers from Virginia to California who stepped up to help for a week – spent an intense and arduous summer within the boundaries of that immense fire last summer as part of a study into how bats adapt to a burned-over landscape. We captured bats in mist nets over ponds, attached tiny radio transmitters to reproductive females and tracked them back to often-surprising maternity roosts. We call our research project, a collaboration of Northern Arizona University and the National Forests, "Bats in the Burns," and we hope to expand into other wildfire-burned forests in the Southwest.

Our preliminary evidence suggests that, not surprisingly, bats prefer unburned areas for travel, foraging and drinking. Roost selection was a different story: bats of some species chose roosts in completely charred tree trunks, including some surrounded by burned-over forests.

The forests of the White Mountains range from short-statured piñon pine and juniper woodlands around 5,000 feet (1,500 meters) elevation to subalpine meadows above 9,000 feet (2,750 meters). In between are forests of tall ponderosa pine, quaking aspen, and Douglas-fir trees. During summers, the White Mountains are green, cool and lush with scattered ponds, lakes and streams. At least 10 bat species spend their summers here, roosting in live trees and the dead trees known as snags. Many of them gather by species into maternity colonies to give birth and raise pups.

Previous research has found that bats typically use snags of more than two feet (60 centimeters) in diameter. They roost in vertical cracks in the snags, but will also wedge themselves under patches of loose bark that can house anywhere from one bat to hundreds, depending on the species of bat and the size of the sheltering bark. More than 900 Arizona myotis (*Myotis occultus*) were once counted as they emerged from a single snag.

Wildfires, meanwhile, have been part of forest ecosystems of the southwestern United States for centuries. Until the mid-1800s, lightning-caused fires burned through the ponderosa pine forests every 2 to 20 years. The low flames of those fires burned grasses and shrubs, but moved too fast to kill large pine trees with their thick, fire-resistant bark. That changed when Euro-Americans arrived. Livestock grazing eliminated much of the understory vegetation that had maintained low-intensity fires in the past. Plus, these new settlers considered such fires destructive and eventually began to extinguish them quickly.

Then, in the early twentieth century following a bumper seed crop and a wet year, millions of pine seedlings germinated and, without low-intensity fires to kill many of the tiny seedlings, tree densities increased from tens to thousands per acre. And these now-dense forests are facing yet another stressor in the form of changing climate. The unusually dry summers and winters that the Southwest is now experiencing have changed the way fires burn in forests. Tall flames now reach forest canopies and incinerate whole trees and snags. The decades of accumulated needles and forest litter smolder on the ground, killing old pine trees that would usually survive the fast-moving, pre-settlement fires. Today's forest fires can be so hot they create their own weather and wind patterns: a virtual firestorm. In addition, humans are now one of the leading causes of fires.

The Wallow Fire scorched or incinerated many existing bat-friendly snags. Although new snags were created from trees killed by fire, many were smaller than the size preferred by bats. So the question becomes: would bats accept or reject these blackened snags?

To find out, we captured bats at 20 livestock ponds. Not all the area burned, so we split our efforts among ponds in areas of high severity (at least 75 percent of surrounding landscape burned) or low (25 percent or less). Despite some rainy nights, between mid-June and the end of July, we captured more than 650 bats of 13 species, including the uncommon Allen's big-eared bat (*Idionycteris phyllotis*). The long-legged myotis (*Myotis volans*) was the most common capture, accounting for 25 percent of the total. Arizona myotis, long-eared myotis (*M. evotis*), silver-haired bats (*Lasionycteris noctivagans*) and big brown bats (*Eptesicus fuscus*) rounded out the top five, which represented 83 percent of our captures for the summer.

With long days of driving over rough, rocky and muddy roads, plus rugged hikes into forested ravines, we tracked our radiotagged females back to their roosts. We also occasionally resorted to telemetry flights to locate roosts from the air. In all, we found 19 roosts, including one snag that was shared by an Arizona myotis and a long-legged myotis colony, each of which used a different part of the snag.

More than half the roosts (58 percent) were large ponderosa pine snags, while 21 percent were Douglas-fir, 16 percent quaking aspen and 5 percent white fir. The pine snags averaged 24 inches (62 centimeters) in diameter and the Douglas-fir snags were 17 inches (43 centimeters). The average height of the roost snags was 80 feet (24 meters).

Most of the bats roosted in unburned snags, and bats were mostly captured while foraging and drinking at ponds in habitat relatively untouched by fire. The Arizona myotis and long-legged myotis roosted in unburned snags surrounded by unburned forest. However, four individuals of three species (long-eared myotis, fringed myotis [*M. thysanodes*] and Allen's big-eared bat) used snags that were completely charred – picture a huge, black toothpick. And big brown bats, long-eared myotis, fringed myotis and the single Allen's big-eared bat roosted in the midst of burned-out forest. What causes these species to choose burned or unburned areas for roosting? Perhaps thermal properties of roosts at these high elevations are important. We hope to find out more next summer, when we will be back in the White Mountains to hunt down still more roosts.

This project has been full of surprises, not the least of which is that so many people are willing to volunteer to work at night in remote and challenging terrain. And we were amazed at how bats choose and use roosts in this wildfire-burned area. We were astonished when 70 bats emerged from a completely charred pine snag. We found species segregating the use of snags based on the severity of fire damage in the surrounding landscape. That bats can bear and raise pups at elevations above 8,000 feet (2,400 meters) in such cold temperatures shows how unique and tough these little animals can be.

We will continue our investigation next summer to expand our initial results into how bats are using the Wallow Fire zone. And we hope in the future to explore the remnants of large fires in Arizona and New Mexico. Given the certainty of climate change, it is imperative that we learn how this complex assemblage of bats in the Southwest responds to this transformed habitat.

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COURTESY OF ERIN SAUNDERS

Field Assistant Steven Granroth removes a bat from a mist net in a burned forest in Arizona.

Preventing DISASTER

Home Ignitability in the Wildland–Urban Interface

Wildland-urban interface (W-UI) fires are a significant concern for federal, state, and local land management and fire agencies. Research using modeling, experiments, and W-UI case studies indicates that home ignitability during wildland fires depends on the characteristics of the home and its immediate surroundings. These findings have implications for hazard assessment and risk mapping, effective mitigations, and identification of appropriate responsibility for reducing the potential far home lass caused by W-UI fires,

By Jack D. Cohen

Once largely considered a California problem, residential fire losses associated with wildland fires gained national attention in 1985 when 1,400 homes were destroyed nationwide (Laughlin and Page 1987). The wildland fire threat to homes is increasing and is commonly referred to as the wildland—urban interface (W-UI) fire problem. Since 1990, W-UI fires have threatened and destroyed homes in Alaska, Arizona, California, Colorado, Florida, Michigan, New Mexico, New York, and Washington. Extensive or severe fires in Yellowstone in 1988, Oakland in 1991, and Florida in 1998 attracted much media coverage and focused national attention on wildland fire threats to people and property

Federal, state, and local land management and fire agencies must directly and indirectly protect homes from wildfire within and adjacent to wildlands. Davis (1990) indicated that since the mid-1940s, a major population increase has occurred in or adjacent to forests and woodland areas. Increasing residential presence near fire-prone wildlands has prompted agencies to take actions to reduce W-UI fire losses.

When an apparently all-encompassing, seemingly unstoppable W-UI fire occurs, the rapid involvement of many homes over a wide area produces a surreal impression; some homes survive amid the complete destruction of surrounding residences. After the 1993 Laguna Hills fire, some termed this seemingly inexplicable juxtaposition a “miracle.’ Miracles aside, the characteristics of the surviving home and its immediate surroundings greatly influenced its survival.

Wildland fire and home ignition research indicates that a home’s exterior and site characteristics significantly influence its ignitability and thus its chances for survival. Considering home and site characteristics when designing, building, siting, and maintaining a home can reduce W-UI fire losses.

W-UI Fire Loss Characteristics

W-UI residential fire losses differ from typical residential fire losses. Whereas residential fires usually involve one structure with a partial loss, W-UI fires can result in hundreds of totally destroyed homes. Particularly during severe W-UI fires, numerous

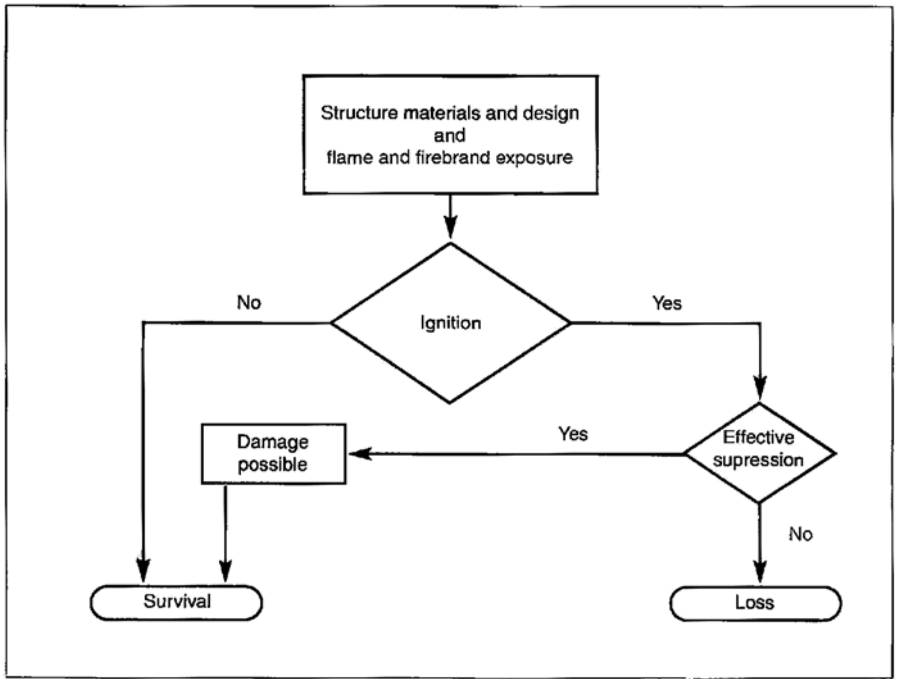

Figure 1. **The structure survival process**

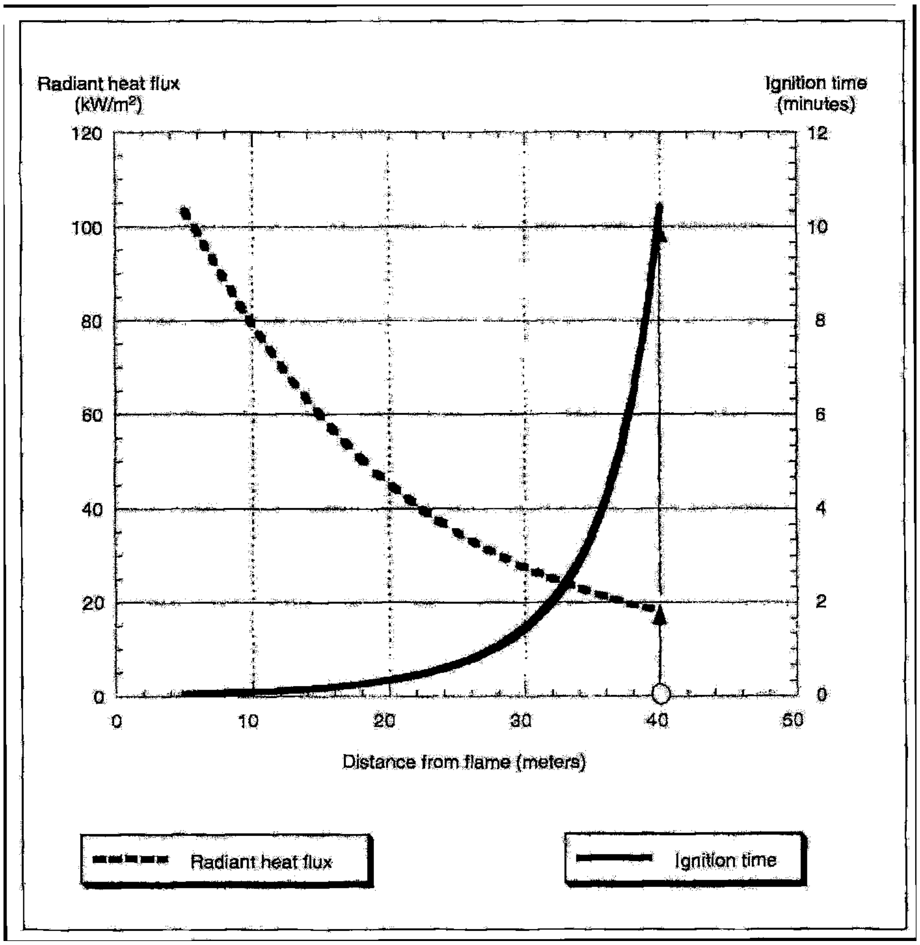

***Figure* 2.The incident radiant heat flux is shown as a function of a wall's distance from a flame 20 meters high by 50 meters wide, uniform, constant, 1,200 K, black-body. The minimum time required for a piloted wood ignition is shown given the corresponding heat flux at that distance.**

homes can ignite in a very short time. The usual result is that a home either survives or is totally destroyed; only a few structures incur partial damage (Foote 1994).

The W-UI Fire commonly originates in wildland fuels. During dry, windy conditions in areas with continuous fine fuels, a wildland fire can spread rapidly, outpacing the initial attack of firefighters. If residences arc nearby, a wildland fire can expose numerous homes to flames and lofted burning embers, or firebrands.

A rapidly spreading wildland fire coupled with highly ignitable homes can cause many homes to burn simultaneously. This multistructure involvement can overwhelm fire protection Capabilities and, in effect, result in unprotected residences. Severe W-UI fires can destroy whole neighborhoods in a few hours—much faster than the response time and suppression capabilities of even the best—equipped and staffed firefighting agencies. For example, 479 homes were destroyed during the 1990 Painted Cave fire in Santa Barbara, most of them within two hours of the initial fire report. The 1993 Laguna Hills fire in southern California ignited and burned nearly all of the 366 homes destroyed in less than five hours.

Whether a home survives depends initially on whether it ignites; if ignitions with continued burning occur, survival then depends on effective fire suppression. *Figure 1* shows that home survival begins with attention to the factors that influence ignition. These factors determine home ignitability and include the structure's exterior materials and design combined with its exposure to flames and firebrands. The lower the home ignitability the lower the chance of incurring an effective ignition.

Ignition: A local Process

Ignition and spread of fire, whether on structures or in wildland vegetation, is a combustion process. Fire spreads as a continuing ignition process whether from the propagation of flames or from the spot ignitions of firebrands. Unlike a flash flood or an avalanche, in which a mass engulfs objects in its path, fire spreads because the requirements for combustion are satisfied at locations along the path. The basic requirements for combustion—the fire triangle—are fuel, heat, and oxygen. An insufficiency of any one of the three components, which can occur over a relatively short distance, will prevent a specific location from burning. “Green islands” that remain after the passage of a severe, stand-replacement fire demonstrate this phenomenon. Commonly one can find a green, living tree canopy very close to a completely consumed canopy.

The requirements for combustion equally apply to the W-UI fire situation. In the wildland fire context, fire managers commonly refer to vegetation as fuel. However, for the specific context of W-UI residential fire losses, a house becomes the fuel. Heat is supplied by the flames of adjacent burning materials that could include firewood piles, dead and live vegetation, and neighboring structures. Firebrands from upwind fires also supply heat when they collect on a house and adjacent flammable materials. The atmosphere amply supplies the third necessary component, oxygen.

A wildland fire cannot spread to homes unless the homes and their adjacent surroundings meet those combustion requirements. The home ignitability determines whether these requirements are met, regardless of how intensely or fast—spreading distant fires are burning. To use an extreme example, a concrete bunker would not ignite during any wildland fire situation. At the other extreme, some highly ignitable homes have ignited without flames having spread to them. These homes directly ignited from firebrands.

Firebrands are a significant ignition source during W-UI fires, particularly when flammable roofs are involved. Foote (1994) found a significant difference in home survival solely based on roof f1ammability. Homes with nonflammable roofs had a 70 percent survival rate compared with 19 percent for homes with flammable roofs. Davis (1990) reported similar results related to roof flammability.

Reducing W-UI fire losses in the context of home ignitability involves mitigating the fuel and heat components sufficiently to prevent ignitions. However, the question of sufficiency (or efficiency) remains: How much, or perhaps more appropriately, how little fuel and heat reduction must be done to effectively reduce home ignitions? To answer this question, we must first quantify the heat source in terms of the fuel’s ignition requirements; specifically, how close can flames be to a home’s wood exterior before an ignition occurs?

Research Insights

Diverse research approaches are providing clues for assessing the fuel and heat requirements for residential ignitions. Structure ignition modeling, fire experiments, and W-UI fire case studies indicate that the fuel and heat required for home ignitions only involve the structure and its immediate surroundings—the home ignitability context.

Modeling. The Structure Ignition Assessment Model (SIAM) (Cohen 1995) is currently being developed to asses the potential for structure ignitions from flame exposure and firebrands during W-UI fires. One function of SIAM is to calculate the total heat transferred, both radiation and convection, to a structure for varying flame sizes and from varying distances. From the calculated heat transfer, SIAM calculates the amount of heat over time that common exterior wood products can sustain before the occurrence of a piloted ignition (Tran et al. 1992).

Piloted ignition When wood is sufficiently heated, it decomposes to release combustible volatiles. At a sufficient volatile—air mixture, a small flame or hot spark can ignite it to produce flaming; thus, a piloted ignition.

Based on severe-case assumptions of flame radiation and exposure time, SIAM calculations indicate that wildland flame fronts comparable to crowning and torching trees (flames 20 meters high and 50 meters wide) will not ignite wood surfaces at distances greater than 40 meters (Cohen and Butler, in press). *Figure 2* shows the radiant heat a wall would receive from flames depending on its distance from the fire. The incident radiant heat flux, defined as the rate of radiant energy per unit area received at an exposed surface, decreases as the distance increases.

Figure 2 also shows that the time required for ignition depends on the distance to a flame of a given size. At 40 meters the radiant heat transfer is less than 20 kilowatts per square meter *(kW/m^2)*, which translates to a minimum piloted ignition time of more than 10 minutes.

1 kW/m^2	Maximum for indefinite skin exposure.
10.4 kW/m^2	Pain after three seconds of skin exposure.
16.0 kW/m^2	Second-degree burn after five seconds of skin exposure.

Source: Drysdale 1985.

Ten minutes, however, is significantly longer than the burning time of wildland flame fronts at a location. Large flames of wildland fires typically depend on fine dead and live vegetation, which limits the intense burning duration at a specific location to less than a few minutes. Recent crown fire experiments have demonstrated a location-specific burning duration of 50 to 70 seconds.

Experiments. Field studies conducted during the International Crown Fire Modelling Experiment (Alexander et al. 1998) provide data for comparisons with SIAM model estimates. Total heat transfer (radiation and convection) and ignition data were obtained from heat flux sensors placed in wooden wall sections.

The instrumented walls were located on flat, cleared terrain at 10, 20, and 30 meters downwind from the edge of the forested plots. The wall section at 10 meters was 2.44 meters wide and 2.44 meters high with a 1.22-meter eave and roof section (*fig. 3a*). Exterior plywood (T-1-11) covered the wall with oriented-strand board covering the roof section and the eave soffit. Trim boards were solid wood with wood fiber composition board on the cave fascia. None of the materials were treated with fire retardant.

(a) 10-meter wood wall section before the crown fire.

(b) Experimental crown fire.

Figure 3. International Crown Fire Modelling Experiment.

The forest was variably composed of an overstory of jack pine *(Pinus banksiana)* about 14 meters high with an understory of black spruce *(Picea mariana).* The spreading crown fire produced flames approximately 20 meters high. *Figures 3b and 3c* show examples of the experimental crown fire.

Five burns were conducted where wall sections were exposed to a spreading crown fire. As the crown fires reached the downwind edge of the plot, turbulent flames extended into the clearing beyond the forest edge. In two of the five burns, flames extended beyond 10 meters to make contact with the 10-meter wall section. When flame contact occurred, the 10-meter walls ignited; however, without flame contact, only scorch occurred, as shown in *figure 3d.* The wooden panels at 20 meters experienced light scorch when flames extended beyond 10 meters from the experimental plot, and no scorch from the other burns. The 30-meter wall section had no scorch from any of the crown fires.

Figure 4 displays the average total incident heat flux (radiation and convection combined) corresponding to the wall at 10 meters *(fig. 3d)* and the crown fire shown in figures *3b and 3c.* The average total incident heat flux is calculated from two sensors placed 1 meter apart in the wall. The amount of heat received by the wall increased as the flame front approached and decreased as the fine vegetation was consumed. The initial heat flux "spike" was caused by a nonuniform crowning flame front.

The flux-time integral shown in *figure 4* indicates whether sufficient heating has occurred to pilot-ignite wood (Tran et al. 992). SIAM uses the flux-time integral for calculating ignition potential, a correlation of the incident heat flux and the time required for piloted wood ignition.

The flux-time correlation identifies two principal ignition criteria: (1) A minimum heat flux of 13 kW/m^2 must occur before a piloted ignition can occur for any exposure time, and (2) piloted ignition depends on attaining a critical heating dosage level (heat transfer and its duration). These criteria are graphed in *figure 4.* The flux-time integral only increases for incident heat fluxes greater than the minimum of 13 kW/m^2, and the flux-time integral threshold value of 11,500 is shown as the ignition threshold. As seen in the figure, the flux-time integral does not reach the ignition threshold, indicating an exposure insufficient for ignition and corresponding to no actual occurrence of a wall ignition. Therefore, a home at some distance from a large flame front, such as a crown fire, may not receive sufficient energy to meet the minimum for ignition over any time period. In addition, a home closer to a large flame front can receive a high heat flux (for example, 46 kW/m^2 as shown in *figure 4*), but without the necessary duration to meet the threshold for ignition.

The flux-time integral plot indicates the duration of the heat transfer relevant to ignition. The heat transfer duration relevant to ignition combines the heat transfer from the approaching crown fire plus the burning time of the fire after it has reached the end of the plot. The observed time required for the flux-time integral to increase from zero to its maximum value corresponds to the heat transfer duration significant for ignition. *Figure 4* indicates a duration of 65 seconds (flux-time plot from 75 seconds to 140 seconds).

Case studies. Case studies of actual W-UI fires provide an independent comparison with SIAM and the crown

(c) Experimental crown fire.

(d) After crown fire exposure the wall scorched but did not ignite. Note the lack of wall scorch under the eave because of the radiation "shading" from the eave.

fire experiments. The actual fires incorporate a wide range of fire exposures. The case studies chosen examine significant factors related to home survival for two fires that destroyed hundreds of structures. The Bel Air fire resulted in 484 homes destroyed (Howard et al. 1973) and the Painted Cave fire destroyed 479 homes (Foote 1994).

Analyses of both fires indicate that home ignitions depend on the characteristics of a structure and its immediate surroundings. Howard et al. (1973) observed 86 percent survival for homes with nonflammable roofs and a clearance of 10 meters or more.

Dicussion

A comparison of the SIAM model calculations in *figure 2* with the observed heat flux from the experimental crown fire in *figure 4* indicated that the model overestimates the heat flux. The model calculation at 10 meters reveals a radiant heat flux of 70 kW/m^2, which exceeds the highest total heat flux of 46 kW/m^2 observed

At the 10-meter wall section in *figure 4*. SIAM calculations

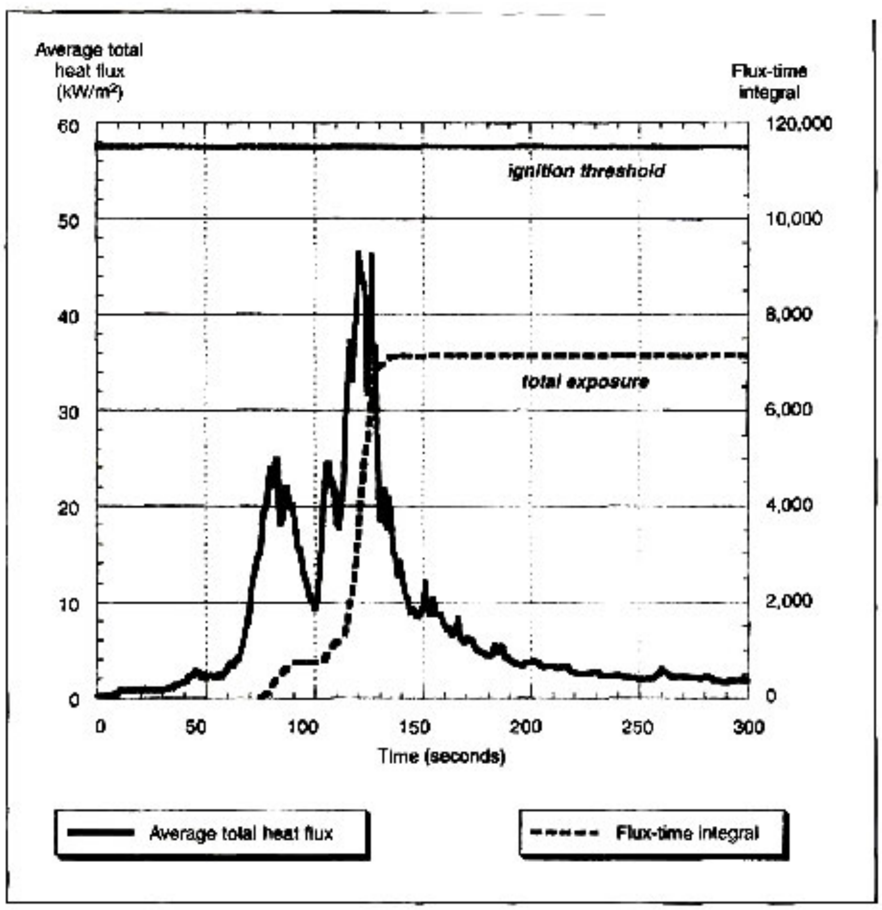

Figure 4. Actual average total incident heat flux and flux-time integral for the crown fire and 10-meter wall section shown in figure 3.

overestimate the heat transfer because the severe-case assumptions designate a homogeneous, black-body radiating flame front. Real flame fronts do not meet these assumptions and produce a significantly smaller radiant heat flux by comparison. For a given flame front, the SIAM calculations represent an extreme-case estimate of radiant heat transfer, and thus an extreme-case estimate of ignition potential.

Given the duration of the experimental heat flux (65 seconds), we can calculate the heat flux and corresponding distance required for ignition. At 65 seconds, the ignition time graph (*fig. 2)* indicates ignition at a flame distance of less than 30 meters. If the heat flux duration is extended by a factor of five to 325 seconds, the flame distance for ignition is less than 40 meters. By comparison, the 10-meter wall sections in the crown fire experiment did not ignite without flame contact and all burns produced little or no scorch to wall sections at 20 and 30 meters. The W-UI fire case studies indicated approximately 90 percent survival with a vegetation clearance on the order of 10 to 20 meters for homes with nonflammable roofs. Thus, the case studies support the general flame-to-structure distance range of 10 to 40 meters as found through modeling and experiments.

However, firebrands can also cause homes to ignite during wildland fires. Although firebrands capable of ignition can originate from a fire several kilometers away, homes can only be threatened if the firebrands ignite the home directly or ignite adjacent flammable materials that then ignite the home.

Analyses of potential home ignitions using modeling, experiments, and case studies did not explicitly address firebrand ignitions. However, firebrand ignitions were implicitly considered because of the firebrand exposures that occurred during the crown fire experiments and the case studies. The experimental crown fires provided a firebrand exposure that resulted in spot ignitions in the dead wood and duff around the wall sections hut not directly on the walls. In the case studies, firebrand ignitions occurred throughout the areas affected by the Bel Air and Painted Cave fires. The high survival rate for homes with nonflammable roofs and 10- to 20-meter vegetation clearances included fire-brands as an ignition factor, thus indicating that firebrand ignitions also depend on the ignition characteristics of the home and the adjacent flammable materials.

Conclusions

The key to reducing W-UI home fire losses is to reduce home ignitability. SIAM modeling, crown fire experiments, and case studies indicate that a home's structural characteristics and its immediate surroundings determine a home's ignition potential in a W-UI fire. Using the model results as guidance with the concurrence of experiments and case studies, we can conclude that home ignitions are nor likely unless flames and firebrand ignitions occur within 40 meters of the structure. This finding indicates that the spatial scale determining home ignitions corresponds more to specific home and community sites than to the landscape scales of wildland fire management. Thus, the W-UI fire loss problem primarily depends on the home and its immediate site.

Consequently if the community or borne site is not considered in reducing W-UI fire losses, extensive wildland fuel reduction will be required. For highly ignitable homes, effective wildland fire actions must riot only prevent fires from burning to home sites, but also eliminate firebrands that would ignite the home and adjacent flammable materials. To eliminate firebrands, wildland fuel reductions would have to prevent firebrand production from wildland fires for a distance of several kilometers away from homes.

Management Implications

Because home ignitability is limited to a home and its immediate surroundings, fire managers can separate the W-UI structure fire loss problem from other landscape-scale fire management issues. The home and its surrounding 40 meters determine home ignitability, home ignitions depend on home ignitability, and fire losses depend on home ignitions. Thus, the W-UI fire loss problem can be defined as a home ignitability issue largely independent of wildland fuel management issues. This conclusion has significant implications for the actions and responsibilities of homeowners and fire agencies, such as defining and locating potential W-UI fire problems (for example, hazard assessment and mapping), identifying appropriate mitigating actions, and determining who must take responsibility for home ignitability

W-UI fire loss potential. Because home ignitions depend on home ignitability, the behavior of wildland fires beyond the home or community site does not necessarily correspond to W-UI home fire loss potential. Homes with low ignitability can survive high-intensity wildland fires, whereas highly ignitable homes can be destroyed during lower-intensity fires.

This conclusion has implications for identifying and mapping W-UI fire problem areas. Applying the term wildland-urban interface to fire losses might suggest that residential fire threat occurs according to a geographic location. In fact, the wildland fire threat to homes is not a function of *where* it happens related to wildlands, but rather to *how* it happens in terms of home ignitability. Therefore, to reliably map the potential for home losses during wildland fires, home ignitability must be the principal mapping characteristic. The home threat information must correspond to the home ignitability spatial scale, that is, those characteristics of a home and its adjacent she within 40 meters.

Home fire loss mitigation. W-UI home losses can be reduced by focusing efforts on homes and their immediate surroundings. At higher densities where neighboring homes may occupy the immediate surroundings, loss reductions may necessarily involve a community. If homes have a sufficiently low home ignitability, a community exposed to a severe wildfire can survive without major fire destruction. Thus, there is a need to examine the reduction of wildland fuel hazard for the specific objective of home protection. There are various land management reasons for conducting wildland vegetation management. However, when considering the use of wildland fuel hazard reduction specifically for protecting homes, an analysis specific to home ignitability should determine the treatment effectiveness.

Responsibility for home ignitability. If no wildfires or prescribed fires occurred, the wildland fire threat to residential development would not exist. However, our understanding of the fire ecology for most of North America indicates that fire exclusion is neither possible nor desirable. Therefore, homeowners who live in and adjacent to the wildland fire environment most take primary responsibility for ensuring that their homes have sufficiently low home ignitability. Homes should not be considered simply as potential victims of wildland fire, but also as potential participants in the continuation of the fire at their location.

A change needs to take place in the relationship between homeowners and the fire services. Instead of home-related presuppression and fire protection responsibilities residing solely with fire agencies, homeowners must take the principal responsibility for ensuring adequately low home ignitability.

The fire services should become a community partner providing homeowners with technical assistance as well as fire response in a strategy of assisted and managed community self-sufficiency (Cohen and Saveland 1997). For this approach to succeed, it must be shared and implemented equally by homeowners and the fire services.

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Assessing the value of roadless areas in a conservation reserve strategy: biodiversity and landscape connectivity in the northern Rockies

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Summary

1. Roadless areas on United States Department of Agriculture (USDA) Forest Service lands hold significant potential for the conservation of native biodiversity and ecosystem processes, primarily because of their size and location. We examined the potential increase in land-cover types, elevation representation and landscape connectivity that inventoried roadless areas would provide in a northern Rockies (USA) conservation reserve strategy, if these roadless areas received full protection.
2. For the northern Rocky Mountain states of Montana, Wyoming and Idaho, USA, we obtained GIS data on land-cover types and a digital elevation model. We calculated the percentage of land-cover types and elevation ranges of current protected areas (wilderness, national parks and national wildlife refuges) and compared these with the percentages calculated for roadless and protected areas combined. Using five landscape metrics and corresponding statistics, we quantified how roadless areas, when assessed with current protected areas, affect three elements of landscape connectivity: area, isolation and aggregation.
3. Roadless areas, when added to existing federal-protected areas in the northern Rockies, increase the representation of virtually all land-cover types, some by more than 100%, and increase the protection of relatively undisturbed lower elevation lands, which are exceedingly rare in the northern Rockies. In fact, roadless areas protect more rare and declining land-cover types, such as aspen, whitebark pine, sagebrush and grassland communities, than existing protected areas.
4. *Synthesis and applications*. Landscape metric results for the three elements of landscape connectivity (area, isolation and aggregation) demonstrate how roadless areas adjacent to protected areas increase connectivity by creating larger and more cohesive protected area 'patches.' Roadless areas enhance overall landscape connectivity by reducing isolation among protected areas and creating a more dispersed conservation reserve network, important for maintaining wide-ranging species movements. We advocate that the USDA Forest Service should retain the Roadless Area Conservation Rule and manage roadless areas as an integral part of the conservation reserve network for the northern Rockies.

Key-words: conservation, elevation zones, land-cover types, landscape metrics, national forests, reserve design

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Introduction

A growing body of scientific evidence indicates that the current USA system of federal protected areas (designated wilderness areas, national parks and national wildlife refuges) may be too small and disconnected to protect against the decline and loss of native species diversity or to accommodate large natural ecosystem processes (Wright, Dixon & Thompson 1933; MacArthur & Wilson 1967; White 1987; Wilcove 1989; Baker 1992; Turner *et al*. 1993; Noss & Cooperrider 1994; Reice 1994; Newmark 1995; Sinclair *et al*. 1995; Soule & Terborgh 1999). Expanding road networks, human settlements, resource extraction and other encroachments on the landscape have increased the fragmentation and loss of natural areas. Such disturbances have isolated many protected areas, causing them to function as terrestrial 'islands' surrounded by a matrix of lower quality altered lands (Harris 1984; Pickett & White 1985; Wiens, Crawford & Gosz 1985; Turner 1989; Saunders, Hobbs & Margules 1991). The long-term persistence of many species within protected areas is dependent on the degree of human activities and land-use practices on lands adjacent to and near protected areas. There is a need to identify relatively undisturbed lands located outside protected areas that may increase the potential of protected areas in maintaining native biodiversity and certain ecological processes, and to include these lands within the conservation reserve system before they are lost or altered.

Inventoried roadless areas, large tracts of relatively undisturbed land on USA Forest Service lands, are often left out of landscape assessments for identifying functional conservation reserves. Only two studies (DeVelice & Martin 2001; Strittholt & DellaSala 2001) have analysed the contribution that roadless areas make to the current protected areas reserve network. However, more than one-third of inventoried roadless areas on national forests are adjacent to protected areas (DeVelice & Martin 2001). They hold the potential to increase the size and connectivity of designated wilderness areas, national parks and national wildlife refuges, thus increasing the ability of protected areas to maintain natural landscape dynamics and native species population viability over the long term. Smaller, isolated roadless areas are also important because they may contain rare species, capture more habitat variation, including underrepresented habitat types, and may function as 'stepping stones' that connect current protected areas across a landscape (Shafer 1995; Strittholt & DellaSala 2001).

There is a precedent for the protection of national forest roadless areas. The USA Congress has designated as wilderness more than half, 6 million ha, of roadless areas that the Forest Service inventoried in national forests in the 1970s. In 1998, the Forest Service began to devise regulations aimed at protection of roadless area characteristics in national forests. In May 2000, the agency released its proposed rule, familiarly known as the Roadless Rule, and draft environmental impact statement. Eight months later, the Forest Service adopted the rule. In July 2004, the Forest Service proposed to repeal the Roadless Rule and replace it with a state petition and rule-making process, which would offer less protection by presumably opening national roadless areas to all forest service activities and requiring state governors to 'opt in' Roadless Rule protections affirmatively for any roadless area.

Included in the Roadless Rule environmental impact statement was an evaluation of the potential contribution that protection of roadless areas could make to the conservation of biodiversity at a national scale (USDA Forest Service 2000b). In that evaluation, DeVelice & Martin (2001) found that the inclusion of roadless areas in the network of federal protected areas would expand representation of ecoregions in protected areas, increase the acreage of reserved areas at lower elevations, and increase the number of areas large enough to provide refuge for wide-ranging species.

Strittholt & DellaSala (2001) focused on similar questions at a regional scale for the Klamath-Sikiyou area in southern Oregon and northern California, USA. They found that roadless areas protect a wide range of ecological attributes, especially at mid- to lower elevations, important in this region. They also concluded that roadless areas increase the connectivity among ecoregions.

The northern Rocky Mountain states of Montana, Wyoming and Idaho comprise a region particularly rich in roadless areas, roughly 2·6 million ha, providing a unique opportunity to create a relatively intact reserve design that captures important elements of conservation for the northern Rockies. Using two key concepts in conservation biology, biodiversity representation and landscape connectivity, we investigated the potential contributions of national forest roadless areas to the protected areas reserve network across the northern Rocky Mountain region.

DIVERSITY REPRESENTATION

An important goal in the design and establishment of conservation reserves is to represent a full range of native biodiversity (Shelford 1926; Margules, Nicholls & Pressey 1988; Church, Stoms & Davis 1996; Possingham, Ball & Andelman 2000). Even though this goal has been articulated for some time, most protected areas are demarcated around areas with high scenic and recreational attributes (Davis *et al*. 1996). As a result, existing protected areas in the northern Rockies are, for the most part, concentrated at higher elevations, where other important elements of biodiversity are most likely to be poorly represented (Scott *et al*. 2001).

Representation of a full range of biodiversity in reserves requires an understanding of all species and ecosystem processes operating within a given landscape. However, many researchers have used ecological communities and elevation ranges as coarse-scale surrogates for native biodiversity in the design of conservation reserves (Scott *et al*. 1993; Host *et al*. 1996). This concept is based on the idea that if a full range of ecological communities and elevation ranges is protected, it is more likely that many ecological communities, wide-ranging species and ecosystem processes will be maintained in the reserves. In the northern Rockies, ecological communities are often associated with elevation gradients (Hansen & Rotella 1999). Hence, roadless areas situated at middle and lower elevations may make valuable contributions in protecting many elements of biodiversity that are currently not well represented in protected areas (DeVelice & Martin 2001).

LANDSCAPE CONNECTIVITY

Connectivity refers to the degree to which the structure of a landscape helps or hinders the movement of wildlife species or natural processes such as fire (Wiens, Crawford & Gosz 1985; Turner *et al*. 1993; Noss & Cooperrider 1994; Bascompte & Solé 1996; With 1999). A 'well-connected' area can sustain important elements of ecosystem integrity, namely the ability of species to move and natural processes to function, and is more likely to maintain its overall integrity compared with a highly fragmented area.

Roads are highlighted in the scientific literature as major causes of landscape fragmentation, and function as barriers to organism movements, resulting in a reduction of overall landscape connectivity for many native species. The effects of roads are broad and include mortality from collisions, modification of animal behaviour, disruption of the physical environment, alteration of chemical environments, spread of exotic and invasive species, habitat loss, increase in edge effects, interference with wildlife life-history functions and degradation of aquatic habitats through alteration of stream banks and increased sediment loads (Franklin & Forman 1987; Andrews 1990; Noss & Cooperrider 1994; Reice 1994; Reed, Johnson-Barnard & Baker 1996; Trombulak & Frissell 2000; McGarigal *et al*. 2001). Thus, the addition of roadless areas to existing protected areas reserve is likely to maintain or increase landscape connectivity, as well as increase the integrity of protected areas.

With the advent of landscape metrics, it is now possible to quantify connectivity for landscapes, land-cover types, species' habitats, species' movements and ecosystem processes across a given region (O'Neill *et al*. 1988; McGarigal & Marks 1995; Gustafson 1998; With 1999). Many different metrics that quantify spatial characteristics of patches or entire landscape mosaics have been described (Turner & Gardner 1991; McGarigal & Marks 1995; Ritters *et al*. 1995; Hargis, Bisonette & David 1998; Dale 2000; Jaeger 2000; McGarigal & Holmes 2002). We chose metrics that measure three elements of landscape connectivity: area, isolation and aggregation.

Area

It is known that larger areas (patches) generally contain more species, more individuals, more species with large home ranges and/or sensitive to human activity, and more intact ecosystem processes than smaller areas (Robbins, Dawson & Dowell 1989; Turner *et al*. 1993; Newmark 1995; Shafer 1995). Higher numbers of patches will usually contribute to greater resilience of populations and may also increase the utility of patches that act as 'stepping stones' or connectors across a landscape (Buechner 1989; Lamberson *et al*. 1992).

Isolation

The distance between patches plays an important role in many ecological processes. Studies have shown that patch isolation is the reason that fragmented habitats often contain fewer bird and mammal species than contiguous habitats (Murphy & Noon 1992; Reed, Johnson-Barnard & Baker 1996; Beauvais 2000; Hansen & Rotella 2000). As habitat is lost or fragmented, residual habitat patches become smaller and more isolated from each other, species movement is disrupted, and individual species and local populations become isolated (Shinneman & Baker 2000).

Aggregation

The spatial arrangement of patches may help to explain how certain species are found in patches located close together and are not found in patches that are more isolated, or vice versa (Ritters *et al*. 1995; He, DeZonia, & Mladenoff 2000). This concept generally follows the ideas developed in island biogeography theory (MacArthur & Wilson 1967) and metapopulation theory (Levins 1969, 1970).

For some species or natural processes, the isolation or aggregation of patches across the landscape may be more important, for others, area may be the key element. Together, these three elements offer a comprehensive assessment of the importance of roadless areas to the maintenance of overall landscape connectivity and ecosystem integrity of current protected areas in the northern Rockies.

In this study, we aimed to assess the extent to which roadless areas increase biodiversity representation and landscape connectivity when they are included in the protected areas reserve network for the northern Rockies.

Methods

STUDY AREA

Of the 84 million ha of land that stretch across Montana, Wyoming and Idaho in the USA, roadless areas cover 2·6 million ha and existing federal protected areas (wilderness areas, national parks, special management areas and national wildlife refuges) protect almost 8·7 million ha. Within this region, three large, relatively

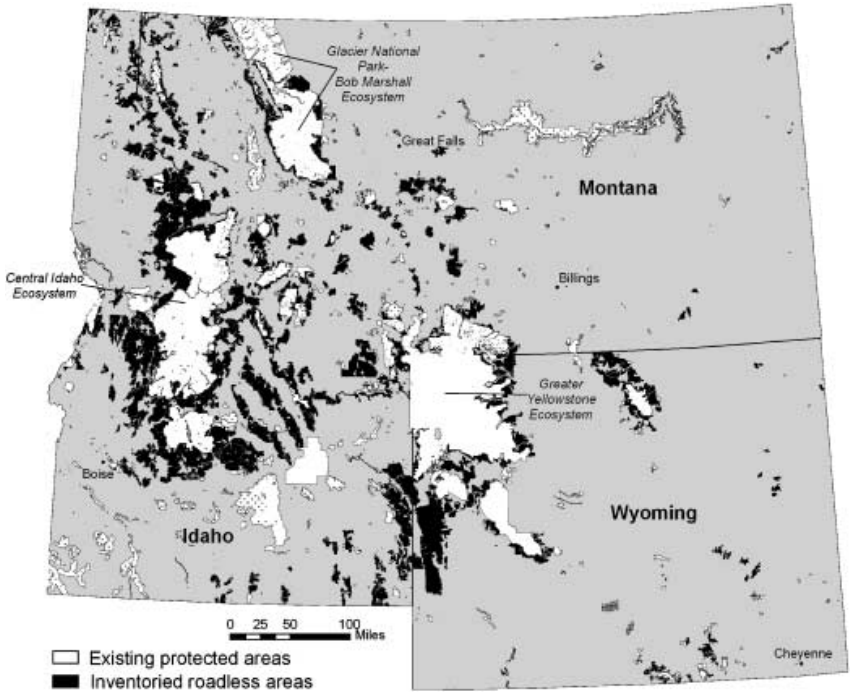

Fig. 1. Roadless areas and protected areas across the states of Idaho, Montana and Wyoming, USA.

undisturbed, mountain ecosystems are delineated around national parks and/or wilderness complexes. These are the Greater Yellowstone Ecosystem, Glacier National Park–Bob Marshall Ecosystem, and the Central Idaho Ecosystem (Fig. 1).

The topography of the northern Rocky Mountain states spans steep physical gradients in elevation, slope, aspect, temperature and precipitation that give rise to diverse vegetation types. Elevations range from 150 m to 4200 m. Average precipitation ranges from 28 cm to 51 cm (Franklin 1983). The northern Rockies comprise a variety of non-forested and coniferous forest types. Low-lying valleys are characterized by grasslands, sagebrush (*Artemisia* spp.) and desert shrublands, interspersed with juniper (*Juniperus* spp.) and riparian woodlands. Ponderosa pine *Pinus ponderosa* dominates lower elevation montane forests, while xeric coniferous forests of mainly Douglas fir *Psuedotsuga mensiezia*, ponderosa pine, grand fir *Abies grandis*, lodgepole pine *Pinus contorta* and aspen *Populus tremuloides* occur at mid-elevations. Mesic forests in the north and west largely contain western larch *Larix occidentalis*, grand fir, western red cedar *Thuja plicata* and mountain hemlock *Tsuga mertensiana*. Higher elevations are composed of Engelmann spruce *Picea engelmannii*, subalpine fir *Abies lasiocarpa*, alpine larch *Larix lyalli* and white-bark pine *Pinus albicaulis* intermixed with subalpine meadows. Herb lands, rock, alder *Alnus sinuata* shrub-fields and snowfields/ice occur at the highest elevations.

DATA COLLECTION

We used a land management status GIS coverage and classification system developed by the USA Geological Survey's Biological Resources Division in its nation-wide GAP Analysis Programme (Scott, Tear & Davis 1996) to delineate 'protected areas'. This programme devised a ranking scheme to represent various levels of protection, ranging from the least protected lands (category 4, e.g. private lands) to those with the highest level of protection (category 1, e.g. wilderness areas) for all public lands in the GIS spatial database. For this study, we assumed that categories 1 and 2 represent adequate protection as their primary management objective is conservation (Scott, Tear & Davis 1996), and selected these categories as our protected areas on all forest service lands located in the three states.

We used the federal inventoried roadless areas GIS database (USDA Forest Service 2000a). This includes areas that are greater than 2000 ha in size, where road building is prohibited under current National Forest Plan decisions and where road building is presently allowed. We recognize that our decision leaves out smaller roadless areas that were not considered during the inventory of federal roadless areas and that these areas serve important conservation goals (Strittholt & DellaSala 2001). For this study, the term 'roadless areas' refers to inventoried roadless areas.

We used three independently derived land cover maps for Montana, Wyoming and Idaho from the GAP Analysis Programme (Scott, Tear & Davis 1996). The Montana and Idaho GAP products were produced based on classification techniques by Redmond *et al.* (1998) for raw Landsat Thematic Mapper (TM) satellite imagery. Spatial resolution of the grid was 90 m for Montana and 30 m for Idaho. The Wyoming GAP Analysis Programme digitized land cover data in a vector format from Landsat TM satellite imagery at a scale of 1 : 100 000 (Gap Analysis Wyoming 1996). We converted Wyoming's vector map into a grid format and resampled the three data sets to 90-m resolution. Then we merged the three land cover maps into a single image and a common land cover classification scheme (Appendix 1).

Similar to most GIS databases, errors are associated with the land management status, inventoried roadless areas and land-cover grids. These grids represent a composite of data from many sources and include variations in mapping procedures and possible misclassifications that could potentially cause inconsistencies that are difficult to detect. However, we believe, based on professional judgement, that the error rate is not large enough to affect conclusions drawn from this large regional-scale analysis.

To investigate the representation of roadless areas at various elevation classes, we downloaded a digital elevation model from the 30-m National Elevation Dataset produced by the USA Geological Survey's EROS Data Center (Sioux Falls, SD). We reclassified the elevation range into 21 equal-interval classes ranging in 200-m increments from approximately 150 m to 4200 m.

DATA ANALYSIS

All data analyses were conducted in ARC/INFO and ArcView GIS software from Environmental Systems Research Institute (Redlands, CA).

Land cover representation

Using ARC/INFO, we combined the protected areas database with the land cover map. To calculate the percentage representation of each land-cover type, we divided the protected portion of each land-cover type by the total area of each land-cover type across the study area. Next, we appended the national forest inventoried roadless areas to the existing protected areas and repeated the same calculation described above to measure the additional representation of each land-cover type because of the inclusion of roadless areas. In addition, we calculated the percentage increase between each land cover percentage representation for protected areas alone and protected areas and roadless areas combined. This measure quantified the 'relative' ecological contribution from roadless areas for each land-cover type. We then ranked these land-cover types according to the level of representation within the existing protected areas.

Elevation representation

Using ARC/INFO, we combined the protected areas database with the 30-m digital elevation model. Similar to the procedure for land-cover types described above, we added the roadless areas to the existing protected areas, intersected this image with the elevation data, and calculated the change in representation for each elevation class provided by protection of roadless areas.

To examine the potential increase of landscape connectivity caused by roadless areas, we used ARC/INFO and FRAGSTATS (McGarigal & Marks 1995; McGarigal & Holmes 2002), a computer program developed to quantify heterogeneity of the landscape. We identified five landscape metrics available in FRAGSTATS to assess our three elements of landscape connectivity (McGarigal & Holmes 2002). To assess area, we used the metrics percentage land (PLAND), number of patches (NP) and patch size (AREA). We included the metrics NP and AREA to help explain the context of an increase in PLAND. For example, an increase in PLAND and AREA and a decrease in NP would indicate that the added roadless patches were located next to existing conservation patches, resulting in an increase in the size of patches and a decrease in the number of patches across the landscape. Conversely, a decrease in AREA and an increase in NP would indicate that the added patches were generally smaller and did not combine with existing patches.

To assess isolation we used nearest neighbour distance (ENN). A decrease or increase in ENN would indicate that patches are either located closer together or farther apart, respectively, across the landscape.

To assess aggregation, we used contagion (CONTAG). An increase in CONTAG would indicate that patches are, to a certain extent, aggregated together across the landscape.

Using FRAGSTATS, we selected and ran our five landscape metrics on the two grids described above (current protected areas only, and roadless areas and current protected areas combined). Each grid was a binary map where all grid cells that comprised the 'protected' and 'roadless' patches were classified as 1 and all other 'non-protected' grid cells were masked out as background (–99). For each landscape metric, we computed the mean, area-weighted mean and coefficient of variation where applicable. We then compared the differences in metrics between the two grids. In addition, differences in the mean, area-weighted mean and coefficient of variation helped to explain how the range of values for each metric were distributed when existing protected areas were compared with the conservation system including roadless areas.

Results

LAND COVER REPRESENTATION

In existing protected areas, burned forest and snowfields/ice had the highest land cover representation, 88% and 86%, respectively. Representation of other land-cover types, such as alpine meadows, whitebark pine, exposed rock/soil, subalpine meadows, wetlands, mixed subalpine forest and lodgepole pine, ranged from 31% to 71%.

The inclusion of roadless areas increased the representation of all land-cover types except for one, sand dunes (Table 1). Relative percentage increases ranged

Table 1. Additional representation and percentage increase in representation of each land-cover type across the northern Rockies when national forest roadless areas are added to existing protected areas

Land-cover type	Existing level of representation (%)	Potential level of representation including roadless areas (%)	Percentage increase including roadless areas
Burned forest	88·12	93·09	5·65
Snowfields/ice	86·12	97·48	13·19
Alpine meadow	71·51	94·18	31·70
Mixed whitebark pine	59·62	84·94	42·46
Exposed rock/soil	44·67	59·92	34·12
Subalpine meadow	40·49	68·85	70·05
Wetlands	37·34	38·68	3·61
Mixed subalpine forest	32·20	68·63	113·11
Lodgepole pine	31·35	59·42	89·54
Mixed barren lands	21·66	22·61	4·37
Sand dunes	18·44	18·44	0·00
Mixed conifer	16·97	37·24	119·44
Mesic upland shrub	10·74	26·14	143·44
Shrub-dominated riparian	7·98	12·77	59·91
Forest-dominated riparian	7·18	12·14	69·11
Sagebrush	6·33	9·91	56·55
Juniper	5·87	6·80	15·95
Xeric upland shrub	5·85	7·97	36·33
Vegetated sand dunes	5·69	6·03	5·89
Western red cedar	5·57	22·00	295·08
Mud flats	5·33	7·39	38·79
Ponderosa pine	4·94	9·88	99·97
Aspen	4·48	25·99	479·80
Shrub–grassland associations	4·25	5·89	38·46
Western hemlock	3·36	23·62	602·54
Grasslands	2·49	3·64	46·31
Grass-dominated riparian	2·15	3·07	43·01
Salt-desert shrub flats	1·58	1·71	8·63
Bur oak woodland	0·00	2·40	NA

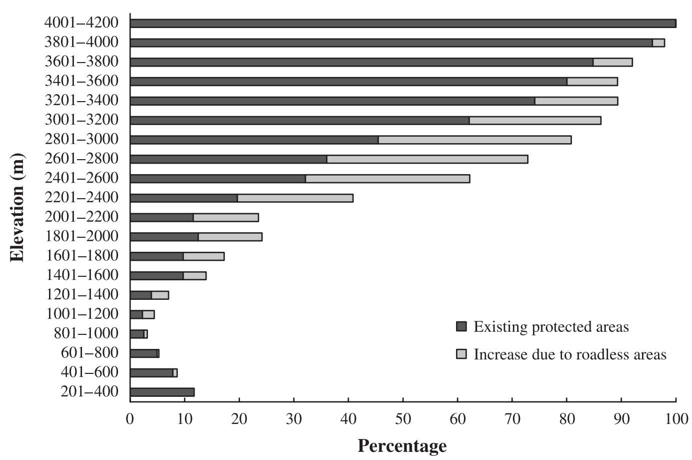

Fig. 2. Additional representation of elevation ranges resulting from the inclusion of roadless areas with protected areas for the northern Rockies. The *x*-axis represents elevation in 200-m increments and the *y*-axis shows absolute increase in percentage representation when roadless areas are added to protected areas. Black bars represent protected areas and grey bars represent roadless areas.

from 5% to 600%. Fifteen land-cover types increased by more than 40%, among them important ecological communities, western hemlock, aspen, ponderosa pine, western red cedar and sagebrush, each of which has less than 10% representation in current protected areas. Moreover, the addition of roadless areas represented one land-cover type, bur oak *Quercus macrocarpa* woodland, not present in protected areas.

ELEVATION REPRESENTATION

Our elevation analyses showed that elevations in the range of 2200–4200 m were well represented in protected areas (Fig. 2). The addition of roadless areas resulted in a large increase in representation of lands at elevations ranging from 1000 m to approximately 3400 m. For elevation ranges below 1000 m and above 3400 m, the

Table 2. Landscape metrics comparing the spatial pattern of protected areas alone with a scenario that includes protected areas and national forest roadless areas combined for the northern Rockies. + and – indicate an increase or decrease, respectively, in the metric value caused by the addition of roadless areas

Landscape Metrics	Protected areas	Protected and roadless areas	+/ –
Area			
Class area (ha)	8 814 900	15 673 600	+
Percentage land	9	16	+
Number of patches	770	722	–
Patch size (mean, ha)	11 447·92	21 708·59	+
Patch size (area-weighted mean)	1 105 055·78	2 505 909·11	+
Patch size (coefficient of variation)	977·39	1 069·74	+
Isolation			
Nearest neighbour (m)	7 013·72	5 353·11	–
Nearest neighbour (area-weighted mean)	3 153·73	2 518·75	–
Nearest neighbour (coefficient of variation)	122·47	134·16	+
Aggregation			
Contagion index	72·56	58·64	–

contribution of roadless areas was small. However, the proportion of area represented at lower elevations increased when we included roadless areas with protected areas.

CONNECTIVITY

Results from the landscape metrics showed that the addition of roadless areas increased regional connectivity for all three connectivity elements (Table 2). Area metrics demonstrated that the addition of roadless areas almost doubled the amount of area protected, rising from 9% to 16%, and the mean patch size in protected areas changed from 11448 ha to 21709 ha. The number of patches decreased from 770 to 722. Area-weighted mean patch size increases and the patch size coefficient of variation increased from 977 to 1070. Isolation metrics showed a decrease in the mean and area-weighted mean nearest-neighbour metrics when roadless areas were added. The mean distance between nearest protected patches decreased from 7014 m to 5353 m. The decrease in the area-weighted mean was less than the overall mean when patches of all sizes were considered. The coefficient of variation also increased for this metric. The aggregation metric (contagion) decreased from 72·56 to 58·64 when roadless areas were included, signifying more dispersion of patches across the landscape.

Discussion

BIODIVERSITY REPRESENTATION

A review of the literature suggests that a given vegetation community is adequately represented when 12–25% of it is included in a conservation area (World Commission on Environment & Development 1987; Noss & Cooperrider 1994), although it is not certain that these thresholds are truly adequate to protect vegetation communities. Based on this range, we define land-cover types above 25% as adequately protected, land-cover types within the range of 12–25% as minimally protected, and those below 12% as underrepresented, similar to DeVelice & Martin (2001).

Our results show that roadless areas make a substantial contribution in maintaining regional biodiversity. One of our most important findings is that roadless areas would protect a wider range of land-cover types and elevation ranges than protected areas alone, especially those characteristic of mid- to low elevations that are underrepresented in protected areas. These lands are among the last remnants of biologically productive lands that have not been significantly altered through human settlements, resource extraction and road construction (Scott *et al*. 2001; Strittholt & DellaSala 2001). We also found that protected areas adequately represent land-cover types that are characteristic of higher elevations. This finding supports the generally accepted notion that wilderness areas and national parks mainly protect higher elevation ecological communities (Davis *et al*. 1996; Possingham, Ball & Andelman 2000). Contrary to DeVelice & Martin (2001), whose study found that roadless areas mainly occurred at mid- to lower elevations, but similar to Strittholt & DellaSala (2001), we found that roadless areas considerably increase the protection of higher elevations and corresponding cover types as well. The different results are probably because of the scale at which the studies were implemented. DeVelice & Martin's (2001) study included all roadless areas across the nation, incorporating a wide range of elevations from sea level to the highest peaks. Our study, and that of Strittholt & DellaSala (2001), focused on smaller regions at higher elevations.

Across the northern Rockies region (Montana, Wyoming and Idaho), protected areas adequately represent nine land-cover types, whereas five biologically important land-cover types, western hemlock, aspen, ponderosa pine, western red cedar and mesic upland shrub, are underrepresented in protected areas. However, the addition of roadless areas increases representation of two cover types (western hemlock and western red cedar) to the minimally protected threshold and two cover types (aspen and mesic upland shrub) to the adequately represented threshold (greater than 25%). Ponderosa pine, even though it increases by nearly 100%, remains underrepresented. Overall, the magnitude of the increased representation, from 100% to 600%, indicates that roadless areas can make substantial contributions to the protection of land-cover types that are not well represented in protected areas.

Increased representation of certain rare ecological communities is particularly important in a northern Rockies conservation strategy. Aspen, for example, is thought to be declining in the northern Rockies (Gallent *et al*. 1998). When roadless areas are added to protected areas, aspen moves up two full categories: from underrepresented to adequately represented, a 480% increase in representation for this forest type, on which many avian species depend upon (Hansen & Rotella 2000). Representation of whitebark pine changes from 60% to 85% when roadless areas are added. Whitebark pine is declining throughout North America due to blister rust *Cronartium ribicola*, an introduced disease, and is a 'keystone species' important for many higher elevation species (Keane, Morgan & Menakis 1994).

Elevation representation results demonstrate that protected areas are mainly located at higher elevations. We also found that roadless areas are generally concentrated at mid- to high elevations and represent a wider range of elevations, especially low- to mid elevations, than protected areas. However, our results show that protected areas encompass more lower elevation lands than roadless areas. This situation is somewhat deceiving. Representation of lower elevations in protected areas is largely a result of two well-placed low-elevation conservation areas: Hell's Canyon National Recreation Area and Missouri Breaks National Monument. In fact, low-elevation lands below 1000 m are not well represented in either protected areas or roadless areas. As a majority of lower elevation lands in the northern Rockies have been converted to other uses, it is of utmost importance to increase representation of lower elevation sites in protected areas (Strittholt & DellaSala 2001). Protection of these lower elevation roadless areas would contribute greatly to the conservation of lower elevation species and ecological communities that are poorly represented in protected areas.

LANDSCAPE CONNECTIVITY

Our analyses of three elements of connectivity show that roadless areas increase connectivity across the northern Rockies, and increase both the area and size of protected area patches. In addition, the number of protected area patches decreases with the addition of roadless areas because they combine with protected areas to form one larger patch. Larger patches will protect more species and more individuals, species with large home ranges, species sensitive to human activity, and more intact ecosystem processes than smaller areas (Askins, Philbrick & Sugeno 1987; Robbins, Dawson & Dowell 1989; Turner *et al*. 1993; Newmark 1995; Shafer 1995). Roadless areas also reduce the distance between protected areas and create a more evenly dispersed reserve system, critical for maintaining many species' movements and a large distribution of local populations (MacArthur & Wilson 1967; Murphy & Noon 1992; Reed, Johnson-Barnard & Baker 1996; Ritters *et al*. 1996; Beauvais 2000; Hansen & Rotella 2000; He, DeZonia, & Mladenoff 2000; Shinneman & Baker 2000).

Our results show an increase in the coefficient of variation for patch size and isolation metrics, which may be an important consideration in delineating conservation reserve systems capable of maintaining movements of various species and ecological processes (Wiens & Milne 1989; Wilcove & Murphy 1991; Noss 1992; Noss *et al*. 1996; O'Neill *et al*. 1996). Smaller patches may supplement larger reserves by protecting rare species that occur only in certain areas (Franklin & Forman 1987; Hansen *et al*. 1991; Shafer 1995). The dispersion of roadless areas may also contribute to greater resilience or survival of island populations by allowing a greater chance for species exchange, essentially maintaining a metapopulation or source–sink population structure (Wiens, Crawford & Gosz 1985; Pullium 1988; Gilpin & Hanski 1991; Murphy & Noon 1992). Many studies are investigating how species move through landscapes and their use of stepping-stone habitats, especially in fragmented landscapes (Freemark *et al*. 1993; With 1999; Beauvais 2000; Hansen & Rotella 2000; Holloway, Griffiths & Richardson 2003; Johnson, Seip & Boyce 2004). Being relatively undisturbed and well-distributed among protected areas, roadless areas are top candidates for the delineation of high-quality 'habitat connections' across the northern Rockies, particularly those that target rare or declining species. The loss or alteration of roadless areas may further reduce the movement of species among interdependent island populations located in protected areas and roadless areas, resulting in greater isolation.

Moreover, the addition of roadless areas increases the effective size of the three largest wilderness and national park complexes in the northern Rockies: the Greater Yellowstone Ecosystem, the Glacier National Park–Bob Marshall Ecosystem and the Central Idaho Ecosystem, where management challenges include maintaining large-scale ecological processes such as species' movements and natural fire across jurisdictional boundaries (Pickett & White 1985; Turner *et al*. 1993). Roadless areas not immediately adjacent to these complexes are dispersed in the surrounding landscape, which helps to decrease the degree of isolation between the complexes and possibly allows for species movement among these ecosystems.

MANAGEMENT IMPLICATIONS

Using research to guide reserve design and develop land protection policies is the strongest approach in conservation. The importance of intact, functioning natural ecosystems to the maintenance of native biodiversity and ecological processes is unquestioned (Wright, Dixon & Thompson 1933; MacArthur & Wilson 1967; Usher 1987; White 1987; Shafer 1995; Noss, O'Connell & Murphy 1997). The negative impacts of roads in natural areas are well known (Andrews 1990; Foreman & Wolke 1992; Reed, Johnson-Barnard & Baker 1996; Spellerberg 1998; Trombulak & Frissell 2000; McGarigal *et al*. 2001). Our landscape assessment demonstrates how roadless areas, the remaining relatively undisturbed forested lands in the northern Rockies, are essential for maintaining biodiversity and landscape connectivity in a conservation reserve strategy for this area. This has direct bearing on management decisions regarding the protection of roadless areas in this region. Our results, along with the findings of DeVelice & Martin (2001) and Strittholt & DellaSala (2001), highlight the important role of roadless areas in USA conservation efforts and contribute to the larger policy dialogue surrounding roadless areas.

The methods used in this study can help land managers determine appropriate guidelines to identify and assess roadless areas that are critical in maintaining regional biodiversity, ecosystem processes, landscape connectivity and overall intact ecosystem integrity. Land managers should avoid activities such as road building, logging, spread of exotic species, off-road vehicle use and exurban development in roadless areas that would result in their degradation or loss. If roadless areas are not protected from these activities as a matter of priority, it is possible that their potential contribution to conservation effort in the future will be diminished and existing protected areas surrounded by or in close proximity to roadless areas will be negatively affected as well. We recommend that roadless areas receive full protection and are managed responsibly, so that they can function as an important part of the current conservation reserve system in the USA.

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Supplementary material

The following material is available from http://www.blackwellpublishing.com/products/journals/suppmat/JPE/JPE996/JPE996sm.htm.

Appendix 1. Land-cover types across the northern Rocky Mountain region reclassified from USA Geological Survey's Biological GAP Analysis Programme (Scott, Tear & Davis 1996).

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ERRATA

The preceding paper unfortunately was published with several minor errors that are corrected below:

Page 181: Gregory H. Aplet's address is The Wilderness Society, 1660 Wynkoop Street, Ste. 850, Denver, CO 80202, USA.

"Forest Service" should be capitalized throughout.

Page 183, col. 2: The first sentence of the Study Area section should read: Of the 84 million ha of land that stretch across Montana, Wyoming and Idaho in the USA, roadless areas cover 6.8 million ha, and existing federal protected areas (wilderness areas, national parks, special management areas and national wildlife refuges) protect 8.8 million ha.

Page 184, par. 1, line 13: The correct spelling is: Douglas-fir *Pseudotsuga menziesii*.

Page 185, col. 2: The top of the column should begin with the heading *Landscape connectivity*.

Page 187, col. 2, par. 1.: The last sentence should read: Our study, and that of Strittholt and DellaSala (2001), focused on smaller regions, where national forests are concentrated at higher elevations.

Page 188, col. 1, par. 1: The third sentence should read: When roadless areas are added to protected areas, aspen moves up two full categories: from underrepresented to adequately represented, a 480% increase in representation for the forest type, upon which many avian species depend (Hansen and Rotella 2000).

Page 188, col. 1, par. 2: The first two sentences should read: Elevation representation results demonstrate that higher elevations are well represented in existing protected areas. We also found that roadless areas would add substantially to protected areas at mid- to high elevations.

Page 189, col. 2: The reference to Beauvais (2000) should refer to F.W. Smith.

Page 191, col. 2, last line: The manuscript was received 30 December 2003.

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Assessing Crown Fire Potential in Coniferous Forests of Western North America: A Critique of Current Approaches and Recent Simulation Studies

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Assessing crown fire potential in coniferous forests of western North America: a critique of current approaches and recent simulation studies

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Abstract. To control and use wildland fires safely and effectively depends on creditable assessments of fire potential, including the propensity for crowning in conifer forests. Simulation studies that use certain fire modelling systems (i.e. NEXUS, FlamMap, FARSITE, FFE-FVS (Fire and Fuels Extension to the Forest Vegetation Simulator), Fuel Management Analyst (FMAPlus®), BehavePlus) based on separate implementations or direct integration of Rothermel's surface and crown rate of fire spread models with Van Wagner's crown fire transition and propagation models are shown to have a significant underprediction bias when used in assessing potential crown fire behaviour in conifer forests of western North America. The principal sources of this underprediction bias are shown to include: (i) incompatible model linkages; (ii) use of surface and crown fire rate of spread models that have an inherent underprediction bias; and (iii) reduction in crown fire rate of spread based on the use of unsubstantiated crown fraction burned functions. The use of uncalibrated custom fuel models to represent surface fuelbeds is a fourth potential source of bias. These sources are described and documented in detail based on comparisons with experimental fire and wildfire observations and on separate analyses of model components. The manner in which the two primary canopy fuel inputs influencing crown fire initiation (i.e. foliar moisture content and canopy base height) is handled in these simulation studies and the meaning of Scott and Reinhardt's two crown fire hazard indices are also critically examined.

Additional keywords: canopy base height, canopy bulk density, crown fire behaviour, crown fraction burned, crowning, Crowning Index, dead fuel moisture content, fire behaviour, fire behaviour modelling, fireline intensity, foliar moisture content, forest structure, rate of fire spread, Torching Index, wind speed.

Introduction

Crowning forest fires are exceedingly exciting to observe but like most natural phenomena, are dangerous as well. The safe and effective management of fire in most coniferous forest ecosystems is thus dependent to a very large extent on the ability to reliably assess or forecast crown fire potential based on predictive aids produced by research coupled with the skill and knowledge of the user.

Many advances have been made in crown fire behaviour research in recent years, including more intensively monitored experimental crown fires (Stocks *et al.* 2004) and physical-based modelling (Butler *et al.* 2004; Cruz *et al.* 2006*a*, 2006*b*). Nevertheless, crown fire behaviour is sometimes portrayed as a complex phenomenon for which we possess very limited knowledge and understanding of the exact physical processes involved (Cohen *et al.* 2006). Although this may very well be true, a substantial number of observations garnered from conducting outdoor experimental fires (Alexander and Quintilio 1990) and monitoring wildfires coupled with case study documentation (Cruz and Plucinski 2007) over the years have provided a solid foundation on several aspects of crown fire phenomenology as well as benchmark data on expected fire characteristics under certain environmental conditions, at least on an empirical basis.

Understanding the environmental conditions required for the onset or initiation and sustained propagation of crown fires is necessary to implement fuel management programs aimed at mitigating the likelihood of large, high-intensity crowning wildfires in the conifer-dominated forests found in western North America. Keyes and Varner (2006) have recently outlined just how complicated the processes involved are in using silvicultural methods to treat forest fuels in order to modify potential crown fire

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behaviour. The need for research into the effectiveness of fuel treatments in reducing crown fire potential has received considerable attention in recent years (Graham *et al.* 2004; Agee and Skinner 2005; Peterson *et al.* 2005). Roccaforte *et al.* (2008) classified research of this type into three categories: experimental, observational and simulation modelling.

Martinson and Omi (2008) have recently reported that more than half of the published studies aimed at quantifying fuel treatment effectiveness rely solely on modelling simulations. Commonly, these simulation studies characterise the fuel structure of distinct forest stands and through the use of fire modelling systems, coupled with specified fire weather, fuel moisture and slope conditions, attempt to integrate this information into a few fire behaviour descriptors in order to assess the relative 'flammability' of the fuel complex (McHugh 2006), and in turn, are able to gauge the effectiveness of fuel management strategies to mitigate the possibility of crown fires occurring (Graham *et al.* 1999; Keyes and O'Hara 2002).

Various fire modelling systems, such as NEXUS (Scott and Reinhardt 2001), Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS) (Reinhardt and Crookston 2003), FARSITE (Finney 2004), Fuel Management Analyst (FMAPlus®) (Carlton 2005), FlamMap (Finney 2006) and BehavePlus (Andrews *et al.* 2008), are extensively used in these simulation studies to assess potential crown fire behaviour in the western US (Keyes and Varner 2006; McHugh 2006; Varner and Keyes 2009) and to a lesser extent to date in western Canada (e.g. Bessie and Johnson 1995; Feller and Pollock 2006). The technical basis and intended uses of these modelling systems are contrasted elsewhere (McHugh 2006; Andrews 2007; Peterson *et al.* 2007).

All of the fire modelling systems referred to previously implement, link or integrate (or both) Rothermel's (1972, 1991) models for predicting surface and crown fire rates of spread with Van Wagner's (1977, 1993) crown fire transition and propagation models in various ways, and provide an output of several fire behaviour characteristics (e.g. rate of fire spread, fireline intensity, type of fire, crown fraction burned). Some of the systems also output two crown fire hazard indices – the Torching index (TI) and the Crowning Index (CI) as per Scott and Reinhardt (2001). The TI and CI represent the threshold wind speeds required for the onset of crowning and active crown fire propagation in coniferous forests respectively. Each TI and CI value is tied to a unique set of surface fuelbed characteristics (expressed in terms of a stylised or custom fuel model), dead and live moisture contents of surface fuels, crown fuel properties (canopy base height and bulk density, foliar moisture content), and slope steepness. This approach of using fire modelling systems to assess potential crown fire behaviour has gained widespread popularity within the US wildland fire research community, as evident by the number of published simulation studies over the past 10 years or so (e.g. Scott 1998*a*; Stephens 1998; Raymond and Peterson 2005; Harrington *et al.* 2006; Graetz *et al.* 2007; Mason *et al.* 2007; Battaglia *et al.* 2008). Scott and Reinhardt's (2001) two crown fire hazard indices are now being recommended for use in Canada (Gray and Blackwell 2008).

Our cursory critique of these simulation studies has revealed that many of them have produced unrealistic outcomes in terms of crowning potential, as evident by the resulting TI and CI values, given the specified environmental conditions and fuel characteristics. Quite often, critically dry fuel moisture levels are specified along with very low canopy base heights and relatively high canopy bulk densities and yet the simulations suggest that exceedingly strong winds are commonly required to initiate crowning and for fully developed or active crown fires to occur.

We have subsequently discovered that the fire modelling systems used in assessing crown fire potential in these simulation studies have an inherent underprediction bias associated with them as a result of the underlying models or the manner in which they have been implemented (Cruz *et al.* 2003*a*). The primary purpose of the present paper is to accordingly document the unrealistic nature of the outputs from these simulation studies and the level of underprediction bias involved in the models or modelling systems (or both), and then to explain the reasons for such results. Finally, comments are made on the manner in which two of the canopy fuel characteristics (i.e. foliar moisture content and canopy base height) involved in these simulation modelling studies are handled as well the interpretation of the two crown fire hazard indices.

Wind speeds quoted in this article are in terms of the international 10-m open standard (Lawson and Armitage 2008) unless otherwise stated. For the convenience of the reader, a summary list of the variables, including their symbols and units, referred to in the equations and text is given at the end of this article.

Evidence for underprediction of crowning potential in relation to environmental conditions

The notion of an underprediction trend associated with the modelling systems used in various simulation studies has also been hinted at by others. Hall and Burke (2006) found in applying the NEXUS modelling system to prefire fuel complex data collected in the area burned by the 2002 Hayman Fire in north-central Colorado (Graham 2003) that the system failed to simulate the crowning activity actually observed under the weather and fuel moisture conditions that prevailed. Similarly, Agee and Lolley (2006) noted that the low torching potential found in their simulations was 'contradictory to local and regional experience on recent wildfires'. Fulé *et al.* (2001*a*) also recognised that simulation outputs from the NEXUS modelling system appeared contradictory to actual wildfire experience, noting that 'simulated fires using our fuel and weather conditions proved nearly impossible to crown using realistic data, even though real fires had crowned under similar or even less severe conditions'. Here, we specifically discuss and provide evidence for the underprediction bias in terms of wind speed and dead fuel moisture content.

Wind speed and dead fuel moisture combinations

The simulations produced in several studies examining fuel treatment effectiveness reveal a rather low potential for crown fire behaviour relative to the specified environmental conditions (e.g. Scott 1998*a*; Graves and Neuenschwander 2001; Fulé *et al.* 2002; Perry *et al.* 2004; Raymond and Peterson 2005; Agee and Lolley 2006; Hall and Burke 2006; Harrington *et al.* 2006; Page and Jenkins 2007; Roccaforte *et al.* 2008). This is reflected in the threshold wind speeds required for the onset of crowning as represented by the TI and for active crown fire spread as represented by the CI. Both values are generally quite high considering that the simulations are generally based on extremely dry fuel moisture conditions. In many cases, these simulation studies have reported TI and CI values associated with gale-force winds (i.e. sustained winds greater than $\sim$100 km h^{-1}). Such winds seldom occur inland, but when they do, they generally result in trees and whole forest stands being blown down over large areas (List 1951). Scott (2006) has indicated that these very high wind velocities simply indicate 'a very low potential for initiating a crown fire' and that wind speeds at or in excess of 100 km h^{-1} 'occur so rarely that crown fire can be considered nearly impossible to initiate'. Stephens *et al.* (2009) suggest that such levels of wind strength should be 'interpreted as a characteristic of a forest structure that is extremely resistant to passive crown fire'. Although these are possible explanations, they aren't the only ones.

It can be argued that the outcomes of these simulation studies are realistic in that they simply reflect the fact that both strong winds and dry fuels are required to achieve any sort of torching or crowning activity. Although this may be intuitively true for areas that have undergone some form of fuel treatment, for control or untreated areas, the simulation results do not appear realistic based on general observation and experience (Fig. 1 and Table 1), thereby suggesting that the authors of these simulation studies have failed to compare their simulation outputs with empirical observation in order to gauge that their results are realistic (Alexander 2006). Empirical evidence from outdoor experimental crown fires (Stocks *et al.* 2004; Cruz *et al.* 2005) and from wildfire case study documentation (Alexander and Cruz 2006) provides a ready test of this assertion. Fig. 1*a* is a plot of the range in the fine dead fuel moisture (FDFM, %) as per Rothermel (1983) and 10-m open wind speed (U_{10}, km h^{-1}) associated with a dataset of 54 documented crowning wildfires from across North America as taken from a summary given in Alexander and Cruz (2006). FDFM is referred to as the 'estimated fine fuel moisture' in Cruz *et al.* (2004, 2005), Alexander and Cruz (2006), and Alexander *et al.* (2006).

Also plotted in Fig. 1*a* is the 1-h time-lag fuel moisture content (Fosberg and Deeming 1971; Deeming *et al.* 1977) – in lieu of the FDFM – and U_{10} pairs used in the control or no-treatment fuel complexes for a selected set of fuel treatment effectiveness simulation studies. It is apparent from Fig. 1*a* that the conditions used in these simulation studies are extremely severe and not representative of the conditions commonly encountered in large, high-intensity wildfire incidents that involve extensive crowning activity.

Fig. 1*b* illustrates the level of underprediction bias associated with crown fire rate of spread for nine simulation studies by comparing the resultant outputs with observed wildfire rates of spread in relation to U_{10}; some additional observations are given in Table 1. As a general trend, the simulation studies, even though they are relying on extremely dry fuel moisture conditions, require almost a doubling in the U_{10} to attain the level of fire spread rates contained within the wildfire dataset. It is evident from the plots of the TI and CI values (Fig. 1*c*) – the outputs sought by these studies in order to quantify stand or landscape 'flammability' – that the simulation results constitute a distinctly different population from the dataset compiled by Alexander and Cruz (2006) that is based largely, but not exclusively, on wildfires in the western and northern North American coniferous forests. The TI and CI values presented in Fig. 1*c* are applicable to stands with mostly low (i.e. $<$3 m) to moderately high (i.e. 3–8 m) canopy base heights. The various simulation studies generally indicate that exceptionally dry fuel conditions and very strong winds are required for passive and active crowning activity compared with the conditions associated with the documented wildfires.

Wind speed limits

Also noteworthy in Fig. 1*c* is the magnitude of simulated wind speeds, especially in respect to the TI, in several cases in excess of 100 km h^{-1}, given in some of these and other studies (e.g. Scott 1998*a*; Fiedler and Keegan 2003; Monleon *et al.* 2004; Perry *et al.* 2004; Fried *et al.* 2005; Ager *et al.* 2007; Moghaddas and Stephens 2007; Stephens *et al.* 2009). This is consentaneous with other studies aimed at quantifying the potential crown fire behaviour associated with specific fuel complex structures that have reported winds close to or in excess of 1000 km h^{-1} (e.g. Raymond and Peterson 2005; Hall and Burke 2006; Johnson 2008; Stephens *et al.* 2009; Vaillant *et al.* 2009*a*). Some authors have chosen to simply express their TI and CI (6.1-m open wind speeds) values as $\geq$40.2 km h^{-1} or the CI separately as $\geq$64.4 km h^{-1} (e.g. Skog *et al.* 2006; Huggett *et al.* 2008), thereby masking the possibility of very high speeds presumably required for crowning; $\geq$85 km h^{-1} has also recently appeared (Battaglia *et al.* 2008) and $>$145 km h^{-1} (Fiedler *et al.* 2010) have also recently appeared. More recently, some authors have elected to cite only the CI values (e.g. Ager *et al.* 2007; Brown *et al.* 2008; Finkral and Evans 2008).

In contrast to the winds reported in Fig. 1*c*, the 10-m open winds associated with the eight crown fire rate of spread observations used in the formulation of the Rothermel (1991) crown fire rate of spread model averaged 38 km h^{-1} and ranged from 20 to 83 km h^{-1}. The highest wind speed (i.e. 83 km h^{-1}) was associated with the later stages of the major run of the 1967 Sundance Fire in complex mountainous terrain in northern Idaho (Anderson 1968). If this one observation was removed, the winds would have averaged 32 km h^{-1}. Thus, based on all of the available evidence (i.e. Rothermel 1991; Alexander and Cruz 2006; Table 1), one can say with some degree of confidence that there has been no documented active crown fire of any size associated with sustained winds greater than $\sim$80 km h^{-1} reported to date.

Dead fuel moisture levels

In the development of his crown fire rate of spread model, Rothermel (1991) equated the FDFM of Rothermel (1983) to the 1-h time-lag fuel moisture content; this lack of distinction has undoubtedly led to some of the confusion now seen in several simulation studies. He then estimated the 10- and 100-h time-lag values by adding 1.0 and 2.0% to the FDFM value respectively. Some simulation studies (e.g. Cram *et al.* 2006), including many of those identified in Fig. 1*a* and 1*b*, have chosen to use the dead fuel moisture time-lags generated by the US National Fire Danger Rating System (NFDRS) (Deeming *et al.* 1977) rather than estimating the 1-h time-lag fuel moisture content from the FDFM or using the seasonal moisture condition scenarios (or both) presented in Rothermel (1991).

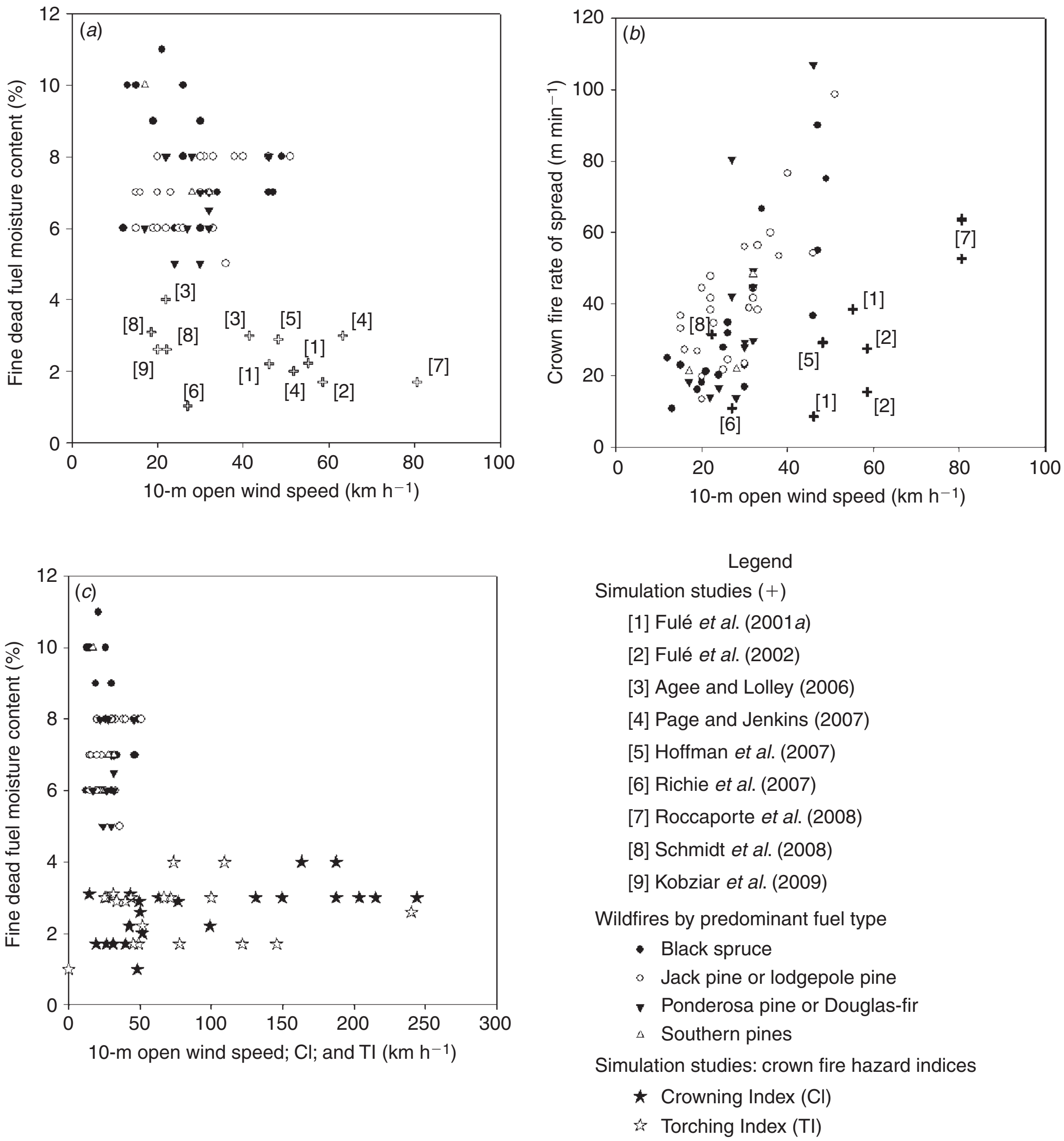

Fig. 1. Environmental conditions and associated crown fire rates of spread and indices of crown fire hazard for a dataset of actively crowning wildfires assembled by Alexander and Cruz (2006) and for a sample of selected simulation studies that have appeared in the scientific peer-reviewed literature: (*a*) fine dead fuel moisture *v.* 10-m open wind speed; (*b*) crown fire rate of spread *v.* 10-m open wind speed; and (*c*) fine dead fuel moisture *v.* 10-m open wind and Scott and Reinhardt's (2001) two crown fire hazard indices. Level terrain is assumed in all cases.

For the purpose of their simulations, Roccaforte *et al.* (2008) assumed 1-, 10- and 100-h time-lag fuel moisture contents of 1.7, 3.0 and 4.5% respectively, representing the 97th percentile level of fire weather severity based on 34 years of archived NFDRS calculations. DeRose and Long (2009) similarly applied values of 1.9, 2.1 and 3.2% respectively in their simulations. In calculating TI and CI values at the time that the 2002 Cone Fire in north-eastern California burned into their experimental fuel treatment plots, Ritchie *et al.* (2007) applied the NFDRS 1-h time-lag fuel moisture content of 1.0% as computed at a nearby fire weather station. The 10- and 100-h values both registered 2.0%. These three situations represent extremely low fuel moisture conditions for coniferous forests in all three categories.

Rothermel (1991) reported value ranges of 3–8, 4–9 and 5–9% respectively for the 1-h (i.e. FDFM was regarded as a surrogate), 10-h and 100-h time-lag fuel moisture contents associated with the wildfires used in the development of his crown fire rate of spread model. Even for his worst case 'late summer, severe drought' scenario, Rothermel (1991) only used 1-h (i.e. FDFM), 10-h and 100-h time-lag fuel moisture contents of 3.0, 4.0 and 6.0% respectively.

Table 1. Characteristics of some of the best-known or well documented (or both) crowning wildfires in conifer forests of North America and Australasia associated with exceptionally strong winds on level to gently undulating terrain not included in Fig. 1

Predominant fuel types are sand pine (*Pinus clausa*) and radiata pine (*P. radiata*). FDFM, fine dead fuel moisture; ROS, rate of spread; U_{10}, 10-m open wind speed. For U_{10}, the World Meteorological Organization standard to express wind speed at a height of 10 m in the open was followed here (Lawson and Armitage 2008). Winds measured at a height of 6.1 m in the open, as per the standard for fire danger rating and fire behaviour prediction used in the United States (Deeming *et al.* 1977; Rothermel 1983), were increased by 15% to approximate the U_{10} standard (Lawson and Armitage 2008)

Reference	Name of fire	Geographical location	Date (dd/mm/yy)	Predominant fuel type(s)	FDFM (%)	ROS (m min^{-1})	U_{10} (km h^{-1})	Type of crown fire
Stocks and Walker (1973)	Garden Lake	Ontario, CAN	02/06/30	Black spruce–jack pine–balsam fir	11	–[A]	48	Active
Folweiler (1937)	Big Henry	Florida, USA	12/03/35	Sand pine	–[B]	135–150	68	Active
Prior (1958)	Balmoral	New Zealand	26/11/55	Radiata pine	8	28	60	Passive and active
Schaefer (1957)	Dudley Lake	Arizona, USA	14/06/56	Ponderosa pine	6	–[C]	64	Active
Dieterich (1979)	Burnt	Arizona, USA	02/11/73	Ponderosa pine	9	30	74	Passive[D]
Geddes and Pfeiffer (1981)	Caroline	South Australia	02/02/79	Radiata pine	5	67	45	Active
Keeves and Douglas (1983)	Mount Muirhead	South Australia	16/02/83	Radiata pine	4	207	80	Active
NFPA (1992)	Spokane area	Washington, USA	16/10/91	Ponderosa pine	10[E]	30	66[E]	Passive and active[F]

[A]According to Stocks and Walker (1973), the extremely strong winds caused 'crowning and contributed greatly to the very fast spread of the fire' that saw two sustained runs of 24 and 64 km take place over a 26-h period on 1–2 June.

[B]According to Folweiler (1937), no measurements of relative humidity were available but it was 'probably low'.

[C]According to Schaefer (1957), the fire made a sustained run of 16 km. Dieterich (1976) estimated Byram's (1959) fireline intensity to have been 52 925 kW m^{-1}.

[D]According to Dieterich (1979), 'Damage from this fast-spreading fire was extremely variable ranging from complete destruction of crown material in patches of saplings and pole timber and an occasional mature tree, to large areas where the only evidence of fire was a blackened litter layer and slight scorch on the lowest portions of the crowns', and that much of the ponderosa pine was 'open grown, and tree crowns extended to within 4–5 feet [1.2–1.5 m] of the ground'. Alexander (1998) computed the fireline intensity at the head of the fire to be 5251 kW m^{-1} using Eqn 4 and the critical surface fire intensity for initial crown combustion to be just 343 kW m^{-1} using Eqn 1.

[E]Based on Alexander and Pearce (1992).

[F]According to the NFPA (1992) case study report, it was observed that: typically stands of ponderosa pine contain dead branches extending to the ground. In some cases, these 'ladder fuels' enabled the fire to reach the crowns of the 30- to 100-foot pine trees and would result in the fire spreading at extremely high rates. Unlike other severe wildland fires, however, this 'crowning' was fairly limited.

As illustrated in Fig. 1*a*, Alexander and Cruz (2006) found for a large database composed mainly of western and northern North American wildfires that the FDFM commonly varied between 6 and 10%. The moisture content of shaded needle litter in conifer forest stands very seldom is less than 2.5–3.0% (Countryman 1977; Harrington 1982; Rothermel *et al.* 1986; Hartford and Rothermel 1991; Wotton and Beverly 2007). The 1-h time-lag NFDRS fuel moisture content can easily be ~2.0% less than the shaded condition represented by the FDFM owing to the effects of solar radiation on fully exposed fuels. This is the reason for the very low fuel moisture conditions commonly associated with the simulation studies on fuel treatment effectiveness (Fig. 1*a*). Considering that the fine, dead fuels represented by the 1-h time-lag fuels are the principal carrier for surface fire spread, the use of the NFDRS computation in lieu of the FDFM represents a significant departure in the application of Rothermel's (1991) crown fire rate of spread model.

Reasons for underprediction of potential crown fire behaviour

The comparison of simulation results with actual observed data presented in Fig. 1 suggests there is a problem in the fundamental underlying models or the manner (or both) in which the models were implemented in the modelling systems. An in-depth analysis of the modelling system framework as dictated by the linkages between the Rothermel (1972, 1991) and Van Wagner (1977, 1993) models reveals that the underprediction bias in the assessment of potential crown fire behaviour arises from three principal sources: (1) incompatible model linkages; (2) use of surface and crown fire rate of spread models that have an inherent underprediction bias; and (3) the reduction in crown fire rate of spread based on the use of crown fraction burned functions. A further potential source of bias is the use of uncalibrated custom fuel models. All but one of these bias sources (i.e. the second one) arise from what we believe is unsubstantiated use of the cited models.

Rothermel (1972) surface fire–Van Wagner (1977) crown fire initiation model linkages

The implemented linkage between the outputs of the Rothermel (1972) surface fire model (i.e. rate of spread and intensity) and the Van Wagner (1977) crown fire initiation model overlooks an important assumption of the latter model. Through a combination of physical reasoning and empirical observation, Van Wagner (1977) defined quantitative criteria to predict the onset of crowning. He defined the critical surface fire intensity for initial crown combustion (I_o, kW m^{-1}) as a function of the canopy base height (CBH, m), and heat of ignition (h, kJ kg^{-1}):

$$I_o = (C \cdot \mathrm{CBH} \cdot h)^{1.5} \tag{1}$$

where h is in turn determined by the foliar moisture content (FMC, %) (Van Wagner 1989, 1993):

$$h = 460 + 25.9 \cdot \mathrm{FMC} \tag{2}$$

Van Wagner (1977) considered the quantity C in Eqn 1, the criterion for initial crown combustion, 'is best regarded as an empirical constant of complex dimensions whose value is to be found from field observations'. Van Wagner (1977) derived a value for the proportionality constant C using the following transformation of Eqn 1 on the basis of a blend of three experimental crown fires carried out in a red pine (*Pinus resinosa*) plantation:

$$C = \frac{I_o^{0.667}}{(\mathrm{CBH} \cdot h)} \tag{3}$$

The surface fire intensity at the onset of crowning was estimated to be ~2500 kW m^{-1} (Van Wagner 1968). Thus, for a CBH of 6.0 m and FMC of 100%, $C = 0.010$ (kW$^{2/3}$ kJ^{-1} kg m$^{-5/3}$).

Van Wagner (1977) equated I_o to Byram's (1959) fireline intensity (I_B, kW m^{-1}), which he calculated from measurements of fire spread rate and fuel consumption:

$$I_B = H \cdot w_a \cdot r \tag{4}$$

where H is the low heat of combustion (kJ kg^{-1}), w_a is the fuel consumed in the active flaming front (kg m^{-2}), and r is the rate of fire spread (m s^{-1}) (Alexander 1982). It is possible to express the requirements for the onset of crowning in terms of the surface fire spread rate by replacing I_o for I_B in Eqn 4 and working backwards (Van Wagner 1989, 1993; Forestry Canada Fire Danger Group 1992), giving the following result:

$$R_i = \frac{60 \cdot I_o}{H \cdot w_a} \tag{5}$$

where R_i is the critical surface fire rate of spread for crown fire initiation (m min^{-1}).

Modelling systems such as NEXUS, FlamMap, BehavePlus, FARSITE, FFE-FVS, and FMAPlus calculate fireline intensity from Rothermel's (1972) reaction intensity (I_R, kW m^{-2}) (Albini 1976):

$$I_B = I_R \cdot t_r \cdot r \tag{6}$$

where t_r is the flame-front residence time (s). Fireline intensities calculated in this manner are consistently lower than per the original Byram (1959) formulation (Cruz *et al.* 2003*a*, 2004). The extent of the differences is a function of the fuelbed characteristics. For the original 13 standard US fire behaviour fuel models as described by Anderson (1982), Byram's (1959) fireline intensity (Eqn 4) is larger than the Rothermel (1972) I_R-derived fireline intensity by a factor of 2 to 3 (Cruz *et al.* 2004).

The implication of these differences within a modelling system such as NEXUS is that higher simulated surface fire rates of spread, and consequently stronger wind speeds and hence larger TI values, are necessary to induce crowning than if the model linkages were to follow the original model assumptions. The end result is increasingly large TI values. Fig. 2 presents a graphical representation of the magnitude of this error for the Anderson (1982) Fuel Model 2 – Timber (grass and understorey) and Fuel Model 10 – Timber (litter and understorey) considering an I_o of 2935 kW m^{-1} per Eqns 1 and 2 based on a CBH of 5.0 m and an FMC of 140%; the output of Fuel Model 9 – Hardwood litter would be very similar to that of Fuel Model 10. The increase in mid-flame wind speed required for the onset of crowning is 72% (i.e. from 6.5 to 10.9 km h^{-1}) for Fuel Model 2 and 48% (i.e. from 8.2 to 12.1 km h^{-1}) for Fuel Model 10. The differences observed in this modelling exercise are considered as conservative in nature. The calculations of Byram's (1959) fireline intensity undertaken here assume that the fuels consumed in the flame front and thus contributing to the upward heat fluxes are the fine, dead and live fuels plus the 10-h time-lag fuels, whereas Van Wagner (1977) in his original formulation did not specifically differentiate between the fuels consumed during flaming as opposed to flaming and smouldering or glowing combustion. In other words, he assumed w_a was equivalent to the difference he obtained from pre- and post-burn fuel sampling – i.e. the fuel consumed in the active flaming front and by glowing or smouldering combustion following passage of the front (w, kg m^{-2}).

Conceptually, the two methods of computing Byram's (1959) fireline intensity should, in theory, yield nearly identical results. The main differences between these two arise from the use of the I_R and t_r models in the Rothermel (1972) model to calculate Byram's (1959) fireline intensity. I_R is estimated from an empirical model developed for homogeneous fuelbeds under no-wind/no-slope conditions in a laboratory setting. How well these assumptions hold for natural surface fuelbeds, with heterogeneous fuel particle and moisture content distributions is unknown, as the model has never been evaluated against field data to our knowledge other than the attempt by Brown (1972) involving simulated slash fuelbeds.

The use of Anderson's (1969) model to estimate t_r in Eqn 6 is the most likely source for the differences between the two methods of determining Byram's (1959) fireline intensity. Research on t_r in natural fuelbeds has identified fuel load, compactness, particle size and moisture as well as wind speed as the most influential variables (Cheney 1981; Nelson 2003). Anderson's (1969) model predicts t_r solely from the characteristic or average weighted size of individual fuel particles.

Nelson (2003) developed and evaluated a semi-physically based model to predict t_r that takes into account fuelbed structure and combustion zone properties. A comparison between the Anderson (1969) and Nelson (2003) t_r models reveals that the former model consistently yields lower t_r values when w_a exceeds ~0.5 kg m^{-2} (Fig. 3). Evaluation data for simulated fuelbeds of slash pine (*Pinus elliottii*) needle litter (Nelson and Adkins 1988) and ponderosa pine (*P. ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) slash (Brown 1972) reveal a marked underprediction of t_r by Anderson's (1969) model and general agreement with Nelson's (2003) model.

If Nelson's (2003) model is considered to provide an acceptable prediction of t_r, as supported by Fig. 3 and his own evaluation against an array of artificial fuelbeds, the Anderson (1969) model is underpredicting t_r in fuel beds with medium to high available fuel loads. This error is propagated within the modelling system and leads to low fireline intensities, and in turn, a low potential for crown fire initiation as illustrated in Fig. 2.

Underprediction bias in the Rothermel (1972) surface fire rate of spread model

In addition to the incompatibility between the various US fire modelling systems and Van Wagner's (1977) criteria for crown

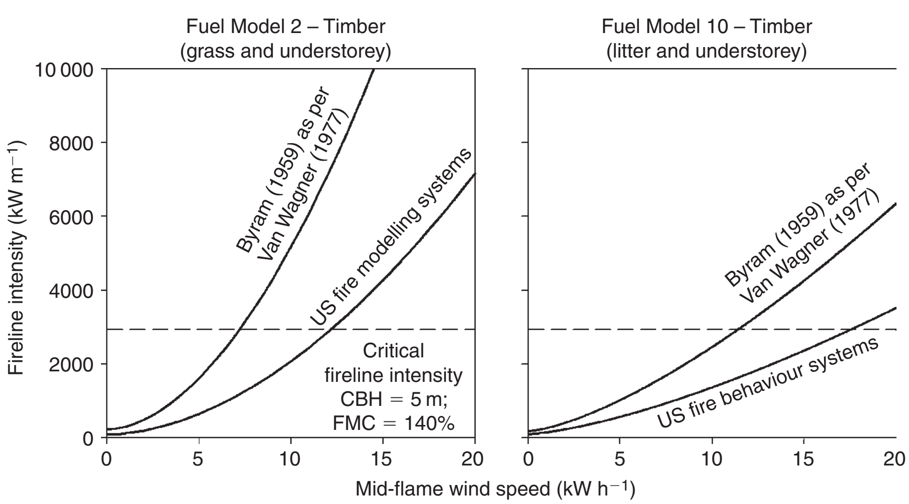

Fig. 2. Critical mid-flame wind speeds required for crown fire initiation as per Van Wagner's (1977) critical surface fire intensity criteria for two US fire behaviour fuel models (Anderson 1982) based on different methods of calculating fireline intensity (i.e. Byram (1959) as per Van Wagner (1977) *v.* US fire behaviour modelling systems) for a particular canopy base height (CBH) and foliar moisture content (FMC) equating to a critical surface fire intensity for initial crown combustion (I_o) of 2920 kW m^{-1}. The following environmental conditions were held constant: slope steepness, 0%; fine dead fuel moisture, 4%; 10- and 100-h time-lag dead fuel moisture contents, 5 and 6% respectively; live woody fuel moisture content, 75%; and live herbaceous fuel moisture content, 75%. The associated 10-m open winds would be a function of forest structure and can be approximated by multiplying the mid-flame wind speed by a factor varying between 2.5 (open stand) and 6.0 (dense stand with high crown ratio) (Albini and Baughman 1979).

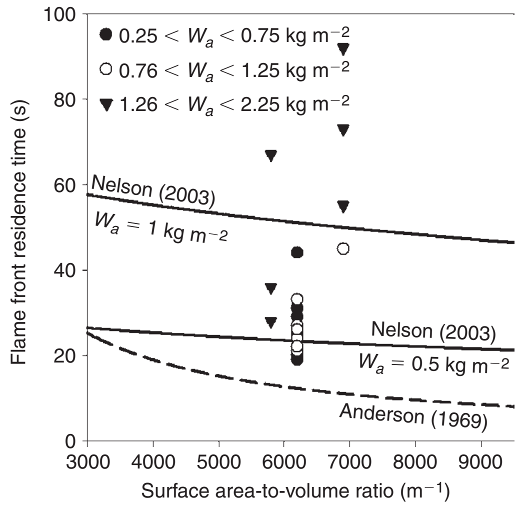

Fig. 3. Sensitivity of Anderson (1969) and Nelson (2003) flame front residence time models to surface area-to-volume ratio and fuel consumed in the active flaming front (w_a). For Nelson's (2003) model, the following environmental conditions were held constant: fuelbed depth, 0.1 m; fuel moisture content, 5%; and mid-flame wind speed, 5 km h^{-1}. Data points represent computed flame front residence times from experimental fires conducted in simulated fuelbeds of slash pine needle litter (Nelson and Adkins 1988) and ponderosa pine and Douglas-fir slash (Brown 1972), where it was implicitly assumed that w was $\sim w_a$.

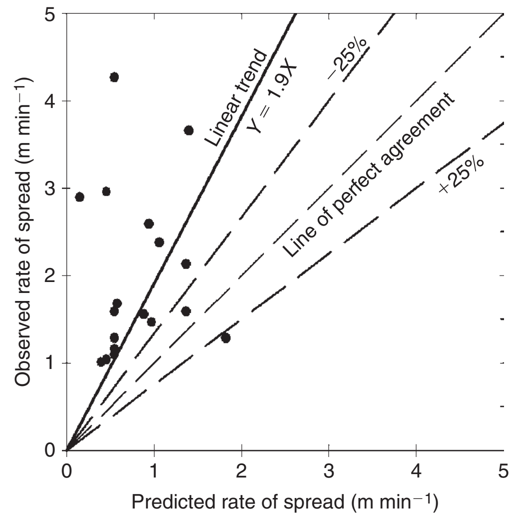

Fig. 4. Observed head fire rates of spread >1 m min^{-1} associated with prescribed burning experiments in ponderosa pine forests of Yosemite National Park, CA, *v.* predictions based on the Rothermel (1972) surface fire rate of spread model for Anderson (1982) Fuel Model 9 – Hardwood litter (adapted from van Wagtendonk and Botti 1984). The dashed lines around the line of perfect agreement indicate the ±25% error interval.

fire initiation with respect to determining w_a, a certain amount of uncertainty exists as to whether the Rothermel (1972) surface fire model can in fact reliably predict, in certain conifer forest stand types, the spread rate of moderate- and high-intensity surface fires that would lead to crowning. Studies that have evaluated Rothermel's (1972) fire spread model for any of the Anderson (1982) stylised 'timber' fuel models (numbers 2, 8, 9 and 10) have identified underprediction trends (Norum 1982; van Wagtendonk and Botti 1984; Grabner *et al.* 1997, 2001). This underprediction trend or bias arises from the sensitivity of the Rothermel (1972) fire spread model to the compactness of the horizontally oriented surface fuelbeds associated with these fuel models (Catchpole *et al.* 1993) and has been discussed in detail by Cruz and Fernandes (2008). Most investigators commonly develop an adjustment factor for rate of spread predictions on the basis of their performance testing (Rothermel and Reinhart 1983). Stephens (1998) for example used the adjustment factors derived by van Wagtendonk and Botti (1984) in his simulation study.

Modelling systems like NEXUS are widely applied to western US ponderosa pine forests (e.g. Johnson *et al.* 2007) and yet performance testing of Rothermel's (1972) model in such fuel complexes is limited to a single outdoor field study by van Wagtendonk and Botti (1984). The same underprediction bias seen in other studies is also evident in their study (Fig. 4 and Table 2). Considering that surface rate of fire spread is a factor in determining the onset of crowning in coniferous forests, the use of unadjusted predictions from stylised fuel models constitutes yet another source of underprediction bias in assessing crown fire potential.

Underprediction bias in the Rothermel (1991) crown fire rate of spread model

Until recently, the only comparison of observed crown fire spread *v.* predictions from Rothermel's (1991) model was that undertaken by Goens and Andrews (1998) on the 1990 Dude Fire that occurred in central Arizona. They found good agreement between predicted and observed spread distances. However, the Dude Fire was considered by Rothermel (1991) as a plume-dominated crown fire as opposed to a wind-driven crown fire, for which he considered his predictive methods were not applicable.

Several studies (Cruz *et al.* 2003*a*, 2005; Stocks *et al.* 2004; Alexander and Cruz 2006) have separately evaluated the Rothermel (1991) crown fire rate of spread model against outdoor experimental crown fire and wildfire datasets (Table 3). A composite summary of those evaluations is presented in Fig. 5. Rothermel's (1991) model underpredicted all 34 experimental observations, with a mean absolute error of 71% (Table 2).

A distinct underprediction bias was also evident in the wildfire observations (Fig. 5*b*). All 54 observations were underpredicted with a mean absolute error of 61%; 63 and 58% for the US and Canadian wildfires respectively (Table 2). The Rothermel (1991) model consistently underpredicted the four observed spread rates in ponderosa pine forests extracted from the 2002 Hayman Fire in north-central Colorado (Finney *et al.* 2003; Graham 2003) by a factor of 2.8 (Alexander and Cruz 2006).

Scott (2006) has acknowledged the underprediction trends evident in Fig. 5 and suggested the use of a correction or adjustment factor (1.7) to obtain what Rothermel (1991) defined as the near-maximum crown fire rate of spread derived on the basis of five 'chance' observations of temporary escalations in

Table 2. Model performance statistics for the Rothermel (1972), Rothermel (1991) and Schaaf *et al.* (2007) rate of fire spread models evaluated against different types of data sources

Statistic	Rothermel (1972)	Rothermel (1991)		Schaaf *et al.* (2007)
	Prescribed fires	Experimental fires	Wildfires	Wildfires
Number of observations	18	34	54	15
Root mean square error	1.54	27	30.7	22.2
Mean absolute error	1.23	22.2	26.0	15.2
Mean absolute percentage error	57	70.8	60.7	41.6
Mean bias error	−1.16	−22.2	−25.9	−15.7
Percentage within ±25% error	6	3	4	20
Over and under predictions	1, 17	0, 34	0, 54	1, 14

Table 3. Basic descriptive statistics associated with the experimental fire and wildfire datasets used in the evaluation of the Rothermel (1991) crown fire rate of spread model as shown in Fig. 5

For Experimental fires, refer to Table 1 in Cruz *et al.* (2005) and to Stocks *et al.* (2004) for the specific details on data sources. For Wildfires, refer to Alexander and Cruz (2006) for the specific details on data sources

Variable	Experimental fires ($n = 34$)				Wildfires ($n = 54$)			
	Mean	s.d.	Min.	Max.	Mean	s.d.	Min.	Max.
10-m open wind speed (km h^{-1})	15.6	5.9	5	35	28.2	9.92	12	51
Air temperature (°C)	25.7	3.9	18.5	31.4	26.6	4.2	20	36
Relative humidity (%)	36.1	7.5	23	52	28	10.6	5	56
Fine dead fuel moisture (%)	7.8	1.9	4	12	7.2	1.37	5	11
Rate of fire spread (m min^{-1})	29.2	16.9	10.7	69.8	39.8	22.1	10.7	107

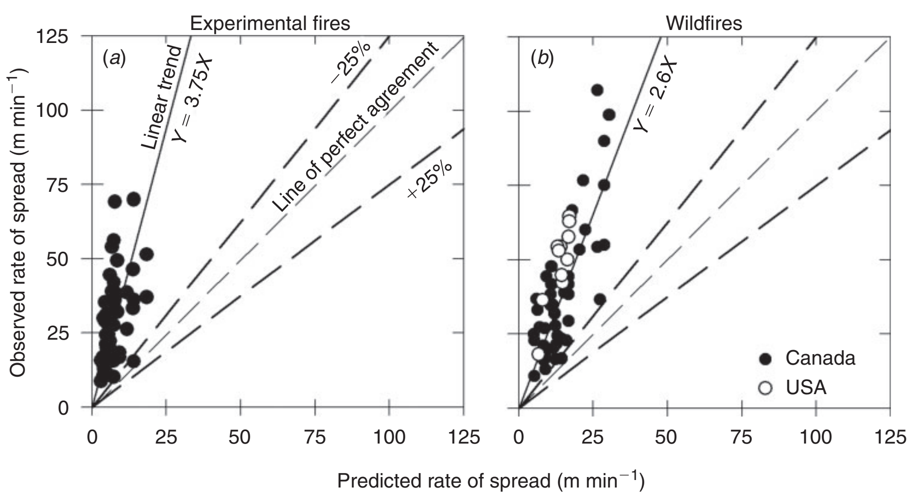

Fig. 5. Observed rates of spread of (*a*) experimental active crown fires; and (*b*) wildfires that exhibited extensive active crowning *v.* predictions based on the Rothermel (1991) crown fire rate of spread model. The dashed lines around the line of perfect agreement indicate the ±25% error interval.

crown fire spread but without any corresponding wind speed measurements. However, according to Rothermel (1991, p. 25), the near-maximum crown fire rate of spread adjustment was intended solely for predicting short bursts in crown fire spread that could be expected to occur during upslope runs and not as a general adjustment factor.

Why is the Rothermel (1991) model consistently under-predicting by a factor of ∼2.5–3.0 and why does it also appear to be relatively insensitive to burning conditions? It is likely due to a multitude of interacting factors (Alexander 2006).

The Rothermel (1991) model is a simple relationship consisting of a correlation derived between the observed average crown fire rate of spread based on eight observations involving seven western US wildfires and the output of the Rothermel (1972) surface fire spread model using Fuel Model 10 and a wind-reduction factor of 0.4 (R_{10}, m min^{-1}) in order to adjust the 6.1-m open wind speed to a mid-flame height value (Albini and Baughman 1979). The Rothermel (1991) model for predicting active crown fire rate of spread (R_a, m min^{-1}) is as follows:

$$R_a = 3.34 \cdot R_{10} \quad (7)$$

Only four of the eight observations used in the model development involved level terrain, so the difficulty of obtaining representative winds in complex terrain relative to observed spread rate can be called into question. Furthermore, the overall average observed rate of spread for five of the eight observations used in the model development was 43 m min^{-1}, which seems reasonable for active or fully developed crown fires in light of the wildfire database compiled by Alexander and Cruz (2006). However, three of eight observations had spread rates of only 14 m min^{-1}. Without knowing what the associated canopy bulk density (CBD) values were for these three observations, such spread rates are low for active crown fires (Cruz *et al.* 2005; Alexander and Cruz 2006). This raises the issue as to the stage of development or degree of crown fire activity (i.e. passive crowning *v.* active crowning) associated with these three crown fire observations and their relative magnitude in the derivation of the Rothermel (1991) model.

From a conceptual perspective, it can be argued that the underlying relationships in the Rothermel (1972) model (i.e. developed from shallow surface fuelbeds in a laboratory setting) do not apply to crown fire phenomena, where the dimension of the fuelbed sustaining fire propagation and the heat flux generated are orders of magnitude higher. Rothermel (1972) readily acknowledged this point and clearly stated in the preface of his publication that the nature and mechanisms of heat transfer in a crown fire are considerably different than those for a surface fire and therefore stated that 'the model developed in this paper is not applicable to crown fires'. Thus, using R_{10} as a correlative or independent variable in what amounts to a statistical model is questionable. The underprediction tendency associated with Rothermel's (1991) model shown in Fig. 5 has also been found to occur with the crown fire rate of spread model developed recently by Schaaf *et al.* (2007) as part of the Fuel Characteristic Classification System (Ottmar *et al.* 2007). The Schaaf *et al.* (2007) model, based on a reformulation of the Rothermel (1972) model by Sandberg *et al.* (2007), is specifically designed to predict the rate of spread of crown fires in coniferous forests. Schaaf *et al.* (2007) undertook to test model performance on the basis of data extracted from Alexander and Cruz (2006) for 15 actively crowning wildfires in black spruce (*Picea mariana*) forests of Canada (Fig. 6 and Table 2). Cronan and Jandt (2008) observed the same underprediction bias evident in Fig. 6 with the experimental fires they conducted in Alaskan black spruce forests.

Another possible reason for the underprediction trend in the Rothermel (1991) model is its low sensitivity to changes in wind speed. As noted, the Rothermel (1991) crown fire spread model is a direct function of Fuel Model 10. Considering that heat

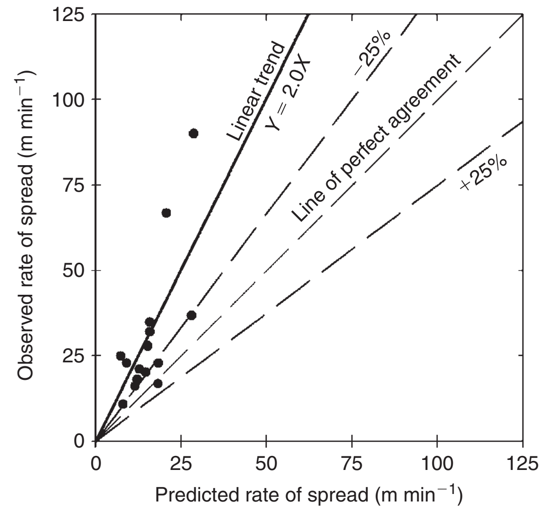

Fig. 6. Observed rates of spread of actively crowning wildfires in black spruce forests *v.* predictions based on the Schaaf *et al.* (2007) crown fire rate of spread model. The dashed lines around the line of perfect agreement indicate the ±25% error interval.

transfer is optimised for vertically oriented, high-porosity fuelbeds (Rothermel 1972), the wind speed–rate of spread relationship of a litter and understorey fuelbed may not be representative of phenomena occurring in deep, low-packing-ratio fuel layers such as canopy fuels in a conifer forest stand. Cohen *et al.* (2006) have described in some detail the inadequacies of the Rothermel (1972) model framework to represent the processes determining crown fire propagation in conifer forests.

The seven wildfires used in the development of the Rothermel (1991) crown fire rate of spread model encompass a wide range in fuel complex structure and composition, although it is difficult to critically assess this factor because formal case study documentation is only available for two of the seven wildfires (Anderson 1968; NFPA 1990) that Rothermel (1991) used in his model development. The Rothermel (1991) crown fire rate of spread model does not explicitly take into account any stand or canopy fuel structure variables as inputs (e.g. CBH, CBD). Hence, crown fire behaviour in the Rothermel (1991) model is independent of the physical fuel characteristics associated with conifer forest stands (Finney 2004).

Rothermel (1991) indicated that the correlation he obtained between the observed crown fire rate of spread and the prediction of surface fire rate of spread from Fuel Model 10 did 'give reasonable results'. However, he was also quick to point out that 'It is readily apparent that more research is needed to strengthen this analysis', and emphasised that his guide represented 'first-order approximations of crown fire behavior' designed to aid operational decision-making.

All 34 experimental fires and 39 of the 54 wildfire observations presented in Fig. 5 involve boreal or boreal-like forest fuel complexes. Thus, it could be argued that the fires selected for evaluation are not 'applicable to the Northern Rocky Mountains or mountainous areas with similar fuels and climate' as per one of Rothermel's (1991) assumptions. Strictly speaking, this is a valid comment.

However, the Rothermel (1991) model has been directly and also indirectly applied through the application of fire modelling systems like NEXUS, FlamMap, FARSITE, FFE-FVS, FMAPlus and BehavePlus, to other distinctly different forest stand types and in other regions of the western US, including for example, the Sierra Nevada (Stephens and Moghaddas 2005*a*, 2005*b*; Dicus *et al.* 2009), north-central (Kobziar *et al.* 2009) and north-eastern (Ritchie *et al.* 2007) regions of California as well as the whole state (Vaillant *et al.* 2009*a*, 2009*b*), south-central (Hummel and Agee 2003), north-eastern (Graves and Neuenschwander 2001) and western Washington (Agee and Lolley 2006), north-eastern (Williamson 1999; Ager *et al.* 2007), central (Fitzgerald *et al.* 2005) and western Oregon (Raymond and Peterson 2005), south-western Utah (Stratton 2004), central Arizona (Goens and Andrews 1998), northern Arizona (Fulé *et al.* 2001*a*, 2001*b*, 2002, 2004), south-central New Mexico (Mason *et al.* 2007), northern Arizona–north-central New Mexico (Clifford *et al.* 2008), and even the north-eastern US (Duveneck and Patterson 2007). In defence of the datasets incorporated in Fig. 5, the fuel characteristics associated with montane and subalpine forests in the Northern Rocky Mountains – namely, ponderosa pine, lodgepole pine (*Pinus contorta*), Englemann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) are not that dissimilar structurally from forests composed of pure and mixed stands of red pine, jack pine (*Pinus banksiana*), black spruce, white spruce (*Picea glauca*) and balsam fir (*Abies balsamea*).

Reduction of crown fire rate of spread due to use of crown fraction burned functions

All of the fire modelling systems mentioned here (i.e. NEXUS, FlamMap FARSITE, FFE-FVS and FMAPlus), with the exception of BehavePlus, that integrate or link the Rothermel (1972, 1991) and Van Wagner (1977, 1993) models to predict the full range of fire behaviour apply a reduction factor to the predicted crown fire rate of spread based on a crown fraction burned (CFB) function (Table 4) as used for example in the Canadian Forest Fire Behaviour Prediction (FBP) System (Van Wagner 1989; Forestry Canada Fire Danger Group 1992).

The CFB, which indicates the proportion of tree crowns involved in the spread of the fire, varies from 0.0 (surface fire with no crown fuel involvement) to 1.0 (fully developed crown fire). In the FBP System, passive crown fire spread or intermittent crowning and continuous crowning or active crown fire spread is judged to occur at CFB values ranging from 0.1 to 0.89 and $\geq$0.9 respectively (Forestry Canada Fire Danger Group 1992).

The final rate of fire spread (R, m min^{-1}), whether surface or crown, is computed as follows:

$$R = R_s + \mathrm{CFB} \cdot (R_a - R_s) \qquad (8)$$

where R_s is the predicted surface fire rate of spread (m min^{-1}) per Rothermel's (1972) model and R_a by Rothermel (1991) per Eqn 7.

The CFB adjustment scheme devised by Van Wagner (1993) provides for a gradual transition in a fire's spread rate from the initial onset of crowning (i.e. passive crown fire spread), as defined by Eqn 5, to the point of active crown fire development

Table 4. Description of computation procedures involved in predicting passive and active crown fire rate of spread in terms of crown fraction burned (CFB) within the various US fire modelling systems

FMAPlus performs the same crown fire computations as FARSITE and NEXUS

Spread regime	BehavePlus	NEXUS and FFE-FVS	FARSITE and FlamMap
Passive crown fire	Does not calculate a CFB for use in computations. Does not provide a spread rate output specifically associated with passive crown fires but does identify passive crown fires as a distinct type of fire.	Calculates CFB between 0.0 and 1.0 using a simple linear transition function between surface and active crown fire rates of spread (Scott and Reinhardt 2001). CFB values are higher than those produced by the CFB function used in FARSITE or FlamMap. Spread rate for passive crown fires is determined to be intermediate between the active crown fire and surface fire rates of spread based on the calculated CFB value.	Calculates CFB between 0.0 and 1.0 based on an exponential transition function between surface and active crown fire rates of spread developed by Van Wagner (1993). CFB values are lower than those produced by the CFB function used in NEXUS or FFE-FVS. Spread rate for passive crown fires assumed to be the same as the surface fire rate of spread.
Active crown fire	Uses Rothermel's (1991) model to predict the average active crown fire rate of spread.	Uses Rothermel's (1991) model to predict the average active crown fire rate of spread. There is also the option to apply the near-maximum crown fire rate of spread multiplier (i.e. 1.7).	Uses Rothermel's (1991) model to predict a reference active crown fire rate of spread that is then adjusted on the basis of the CFB. Because calculated CFB values are lower than those produced by the CFB function used in NEXUS and FFE-FVS, active crown fire spread rates remain less than that predicted by Rothermel's (1991) model even after an active crown fire is judged to have occurred.

based on Van Wagner's (1977) concept of a critical minimum spread rate for active crowning (R_o, m min^{-1}):

$$R_o = \frac{S_o}{\text{CBD}} \quad (9)$$

where S_o is the critical mass flow rate for solid crown flame (kg m^{-2} min^{-1}) and CBD is the canopy bulk density (kg m^{-3}). Van Wagner (1977) provided one estimate of S_o, namely 3.0 kg m^{-2} min^{-1} (Alexander 1988), based largely on a single experimental crown fire in a red pine plantation plot exhibiting a CBD of 0.23 kg m^{-3} (Van Wagner 1964). Cruz *et al.* (2005) have since confirmed the robustness of this estimate based on an examination of a relative large ($n = 37$) dataset of experimental crown fires carried out in several different conifer forest fuel complexes (Fig. 7*a*).

Dickinson *et al.* (2009) claim to have recalibrated Van Wagner's (1977) model represented by Eqn 9 on the basis of the foliar biomass per unit area or available canopy fuel load (CFL, kg m^{-2}) rather than the CBD:

$$R_o = \frac{23.4}{\text{CFL}} \quad (10)$$

This formulation implies that the propagation of active crown fire is not dependent in any way on the stand structure (i.e. height or crown depth) or, in other words, the vertical distribution of the available canopy fuel. It appears from the available experimental evidence that the Dickinson *et al.* (2009) modification of Van Wagner's (1977) R_o model is not as reliable at distinguishing active crown fires from passive crown fires as originally envisioned (Fig. 7*b*).

In deriving his estimate of S_o, Van Wagner (1977) computed the CBD as the available canopy fuel load divided by the canopy depth (Cruz *et al.* 2003*c*) and assumed that all the fuel was uniformly distributed. Admittedly, this is not always the case, for example, in multistoried stands (Reinhardt *et al.* 2006*b*) and even to a certain extent in red pine plantations (Sando and Wick 1972, pp. 6–7) such as Van Wagner (1964, 1968, 1977) worked in. Nevertheless, Alexander *et al.* (1991*b*) found that Van Wagner's (1977) simple model represented by Eqn 9 worked well at distinguishing between surface and crown fires in a black spruce–lichen woodland fuel complex that exhibited large gaps between clumps of trees and crowns that extended down to the ground surface. In their implementation of Eqn 9 in NEXUS, Scott and Reinhardt (2001) initially defined CBD as the maximum 4.5-m vertical running mean bulk density; this was later changed to a 3.0-m interval, although no reason was given (Peterson *et al.* 2005; Scott and Reinhardt 2005, 2007; Scott 2006). This represents a distinct departure from the manner in which Van Wagner (1977) calculated CBD and undoubtedly leads to higher CBD values and hence lower R_o values required for active crowning to occur. As such, it constitutes a violation of one of the fundamental assumptions of Van Wagner's (1977) active crown fire propagation model represented by Eqn 9.

The form of the CFB function varies among the fire modelling systems. FARSITE uses the original exponential form presented by Van Wagner (1993). NEXUS, however, assumes a linear adjustment when the rate of fire spread is between R_i and R_o (Scott and Reinhardt 2001). This gives distinctly different results even if the core models are the same (Fig. 8). Scott and Reinhardt (2001) explored the impact of Van Wagner's (1993) CFB function in FARSITE and found that even under extreme burning conditions, the crown fire rate of spread predicted by the Rothermel (1991) model was reduced by approximately one-third. Regardless of which CFB function is used, the result is a further increase in the underprediction bias (Stocks *et al.* 2004).

The BehavePlus modelling system (Andrews *et al.* 2008) has separately implemented the Rothermel (1972, 1991) surface and crown fire rate of spread and Van Wagner (1977) crown fire initiation and propagation models rather than attempt to directly link them using a CFB function. Thus, BehavePlus doesn't

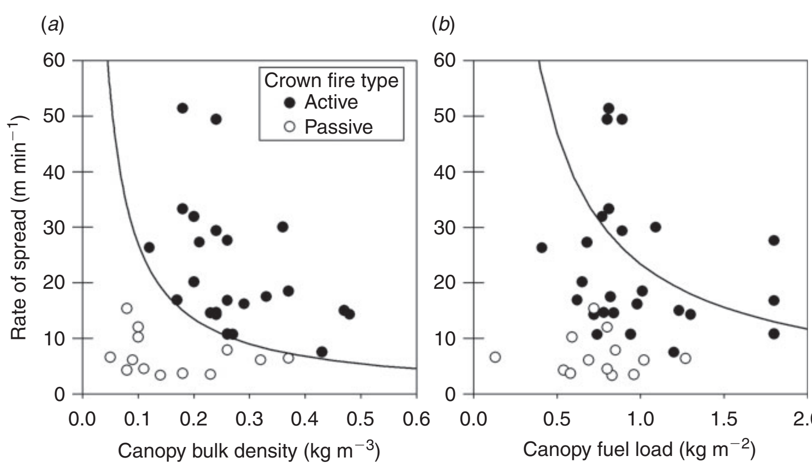

Fig. 7. Scatterplots of experimental crown fire rates of spread by type of spread regime *v.* two canopy fuel properties (adapted from Cruz *et al.* 2005). The curve in (*a*) represents Van Wagner's (1977) criterion for active crowning represented by Eqn 9, assuming an S_o value of 3.0 kg m^{-2} min^{-1}. The curve in (*b*) represents the Dickinson *et al.* (2009) recalibration of the Van Wagner (1977) criterion using canopy fuel load rather than canopy bulk density as represented by Eqn 10.

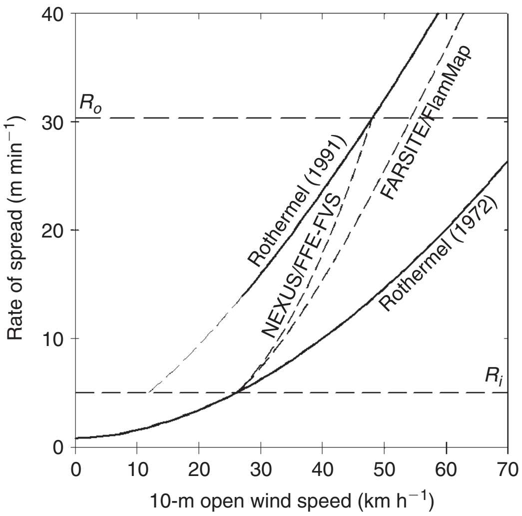

Fig. 8. Comparison of the effect of crown fraction burned functions on rate of fire spread used in the NEXUS and FFE-FVS (Scott and Reinhardt 2001; Reinhardt and Crookston 2003) *v.* FARSITE and FlamMap (Finney 2004, 2006) modelling systems in relation to the Rothermel (1972, 1991) surface and crown fire rate of spread models and Van Wagner's (1977) criteria for the critical minimum spread rates for crown fire initiation (R_i) and active crowning (R_o) for the Anderson (1982) Fuel Model 2 – Timber 2 (grass and understorey) with a canopy bulk density of 0.1 kg m^{-3}, canopy base height of 1.5 m, and a wind reduction factor of 0.2 (Albini and Baughman 1979). The following environmental conditions were held constant: slope steepness, 0%; fine dead fuel moisture, 6%; 10- and 100-h time-lag dead fuel moisture contents, 7 and 8% respectively; live woody fuel moisture content, 75%; live herbaceous fuel moisture content, 75%; and foliar moisture content, 140%. The dashed portion of the Rothermel (1991) curve represents output below the original dataset bounds for rate of spread.

provide a spread rate for passive or intermittent crowning but rather provides a transition to crowning ratio and an active crown fire spread ratio based on the values generated by Eqns 4 *v.* 1 and Eqns 7 *v.* 9 respectively in a manner analogous to Anderson's (1974) index of crowning potential as dictated by the ratio of predicted flame height *v.* an observed or measured CBH.

There is no experimental or sound theoretical evidence for a CFB effect on crown fire rate of spread. Furthermore, general observations of wildfires (e.g. Alexander *et al.* 1991*a*; Cohen *et al.* 2006) and documentation of experimental crown fires (e.g. Van Wagner 1964; Bruner and Klebenow 1979; Burrows *et al.* 1988; Fernandes *et al.* 2004; Stocks *et al.* 2004) indicate that a rather abrupt transition between surface and crown fire regimes is far more commonplace than a gradual transition as implied by a CFB function (Alexander 1998) and as illustrated in Fig. 8.

Use of uncalibrated custom fuel models

Understandably, the use of standard, stylised fuel models (Anderson 1982) in simulation studies examining fuel treatment effectiveness on potential crown fire behaviour limits the extent to which one can gauge the influence of surface fuelbed characteristics on the start and spread of crown fires. Furthermore, there is no empirical proof produced to date to substantiate that by simply increasing the number of fuel models (Scott and Burgan 2005) or reformulating Rothermel's (1972) surface fire rate of spread model (Sandberg *et al.* 2007) would greatly improve matters.

The use of calibrated custom fuel models to represent surface fuelbeds is thus seen by some as a more realistic alternative. However, the use of uncalibrated custom models (e.g. Bessie and Johnson 1995; Battaglia *et al.* 2008; Cheyette *et al.* 2008) can constitute another potential source of underprediction bias. Custom fuel models (Burgan and Rothermel 1984; Burgan 1987) are likely to be unsuccessful when developed without calibrating the predictions or tuning the parameters against field observations of fire behaviour (e.g. Hough and Albini 1978; Cruz and Fernandes 2008).

Studies that have evaluated custom fuel models in horizontally oriented fuels, such as found in conifer litter surface fuelbeds, have identified strong underprediction trends (e.g. Lawson 1972; McAlpine and Xanthopoulos 1989; Hély *et al.* 2001) and in other forest fuel complexes as well (e.g. Burrows 1994; Grabner *et al.* 1997). The effect of this underprediction trend or bias is noticeable in the studies of potential crown fire behaviour that rely on uncalibrated custom fuel models based on field sampling using methods such as those of Brown *et al.* (1982).

Agee and Lolley (2006), for example, predicted a flame height of 1.4 m for their control or untreated ponderosa pine–Douglas-fir fuel complex for simulations based on a 1-h time-lag fuel moisture content of 3% and 6.1-m open wind speeds of 36 km h^{-1}. Comparatively, the Hayman Fire in north-central Colorado (Finney *et al.* 2003; Graham 2003) went from ~5000 to 25 000 ha over a period of 12 h on 9 June 2002 under more moist fuel conditions (FDFM 6–7%) than that of the Agee and Lolley (2006) simulated situation and with a maximum U_{10} of 30–40 km h^{-1} at its peak (Alexander and Cruz 2006).

Similar unrealistic predictions of potential fire behaviour have been reported by others, for example by Page and Jenkins (2007) for lodgepole pine stands infested with mountain pine beetle (*Dendroctonus ponderosae*) in northern and north-eastern Utah and central Idaho (e.g. rates of spread of ~2.0 m min^{-1} for FDFM of 6% and 6.1-m open winds of 50 km h^{-1}) and by Stephens and Moghaddas (2005*a*, 2005*b*) for California mixed-conifer forests (rate of spread of 1.9 m min^{-1} for a 1-h time-lag fuel moisture content of 3.9% and 6.1-m open winds of 22 km h^{-1}). The low spread potential of these custom fuel model predictions explains the need for very dry fuels and high wind speeds in order to induce crown fire activity, as illustrated in Fig. 1*c*.

Other simulation modelling and interpretation issues

Selection of foliar moisture content levels

Van Wagner's (1977) crown fire initiation model is sensitive to FMC (Fuglem and Murphy 1980; Alexander 1988). Changing the FMC from 80 to 140% will almost double the surface fire intensity required for the onset of crowning (Alexander 1988). Within the simulation framework of the fire behaviour modelling systems like NEXUS, this will lead to a large increase in the critical surface fire rate of spread required for crown fire initiation and hence wind speed or fuel dryness (or both) necessary to initiate crown fire activity. Varner and Keyes (2009) recently pointed out that some modellers have assigned FMC 'values without justification or use values that lie on the extremes of published data'.

Scott and Reinhardt (2001) suggested using a constant or default FMC value of 100% as 'a reasonable approach' until better data exist. They also suggested that future research should be directed at compiling existing FMC data and then conducting field research to fill in data gaps. Keyes (2006) concluded on the basis of a review of FMC studies that a single FMC default value 'ignores established differences amongst tree species'. However, he also stated that 'For species lacking published FMC data, a low default value of 90 or 100% remains a prudently conservative assignment'. As a general rule of thumb, an FMC of 90% seems unduly low based on existing information. Chandler *et al.* (1983) regarded crown fire potential as 'high' when the FMC fell below 100%. Some authors have used an FMC of 100% in their simulation studies (e.g. Brown *et al.* 2008; Vaillant *et al.* 2009*b*), whereas others have elected to use much lower values.

Roccaforte *et al.* (2008) used an FMC of 80% in their simulations for ponderosa pine fuel complexes in north-western Arizona without any justification. Although this value might be appropriate for ponderosa pine forests in the south-western US, which typically experience their fire season much earlier in the year, it would be unduly low for other areas in the western US given the seasonal dynamics in FMC found to date in ponderosa pine. Several studies conducted in the western US indicate that the FMC typically ranges from 100 to 120% for 1-year-old ponderosa pine needles between July and September (Philpot and Mutch 1971; Agee *et al.* 2002; Finney *et al.* 2003; Faiella and Bailey 2007), the traditional peak burning period in the western US. Agee *et al.* (2002) and Faiella and Bailey (2007) in turn report FMC in the range of 250–335% and 180–220% respectively for new needle growth. Simulations should consider an aggregate or composite FMC taking into account the differences in moisture contents between new and old needles and the relative proportions of each as well as seasonal changes (cf. Van Wagner 1974). The proportion of new and 1-year and older needle growth is dependent on species, canopy position and site characteristics (Reich *et al.* 1995). Needle longevity for ponderosa pine has been reported to vary between 2 and 4 years in low to moderate elevation sites, but reaching 6 to 9 years in high-stress environments such as arid and alpine habitats (Ewers and Schmid 1981; Richardson and Rundel 1998). Assuming that new needle foliage makes up approximately one-third of the foliage biomass (Van Wagner 1967, 1974) and taking into account the midpoint of Faiella and Bailey's (2007) foliar moisture content ranges for 1-year and older needle foliage (i.e. 110%) and for new growth (i.e. 200%), a nominal FMC value for summertime conditions in ponderosa pine would be ~140%.

It appears the use of low FMC values is becoming commonplace in simulation studies examining potential crown fire behaviour. Stephens and Moghaddas (2005*a*, 2005*b*) used 75% for mixed conifers and Page and Jenkins (2007) used 70% for lodgepole pine. Neither study sampled FMC directly, referenced any previous studies of FMC or otherwise rationalised their FMC selection. Similarly, Stephens *et al.* (2009) used an FMC of 75% without any justification. In their study in ponderosa pine, Ritchie *et al.* (2007) indicated the FMC 'was estimated to be 75% since the Cone Fire burned under dry, north wind conditions following the long, dry summer'. Certainly FMC values this low have occasionally been observed (Keyes 2006). Van Wagner (1993) in fact computed FMC values that average 67% based on a weighting of the moisture contents of old needle foliage and fine, dead woody crown material relative to their separate fuel loadings (Van Wagner 1977). However, such low FMC levels have typically been reported in boreal coniferous tree species just before needle flushing in the spring (Van Wagner 1967, 1974; Fuglem and Murphy 1980).

The National Wildfire Coordinating Group (2008) recently recommended that in the absence of specific information on FMC, one should assume that the FMC is equal to the live woody fuel moisture content input given in BehavePlus, which presently allows for the FMC to vary from 30 to 300%. The moisture content of understorey shrub vegetation can reach 30% (Rothermel 1983) or less and thereby be treated as dead fuel. Existing information on the moisture contents of conifer trees and shrubs sampled at the same time and at the same location does not support this recommendation (e.g. Philpot 1963; Agee *et al.* 2002).

Some authors have selected FMC values below 30% in their application of fire behaviour modelling systems like NEXUS to insect-killed conifer forest stands (e.g. Cheyette *et al.* 2008). Given the empirical nature of Van Wagner's (1977) crown fire initiation model with respect to FMC, applying FMC values any lower than ~70% is not recommended, even if the computer software associated with modelling systems such as NEXUS or BehavePlus allow for it. What is needed is the derivation of a C value for use in Eqn 1 based on a carefully documented outdoor experimental fire(s) carried out at very low FMC levels in order to determine crown fire potential in canopy fuel layers comprised largely of fine, dead fuels (e.g. Kuljian and Varner 2010).

Canopy base height criteria

Another input in Van Wagner's (1977) crown fire initiation model, and one that readily favours the occurrence of crowning activity is the CBH. In fact, the natural variation in CBH would allow for a much greater effect on crowning potential than would the observed variation in FMC (Fuglem and Murphy 1980; Alexander 1988).

Van Wagner's (1977) crown fire initiation model has an empirical basis and was parameterised using the mean crown base height of the trees within a red pine plantation experimental plot (Van Wagner 1968). In their simulation studies, Ritchie *et al.* (2007) and Roccaforte *et al.* (2008) used the lowest quartile CBH value. We do not dispute the fact that the lowest quartile could possibly be a better descriptor of a fuel complex's vertical continuity than the average value when applying a physical-based model. Nonetheless, the use of the lowest quartile in the context of Van Wagner's (1977) crown fire initiation model, as represented by Eqn 1, violates one of the fundamental assumptions of this semi-empirical-based model.

Defining what constitutes an effective CBH can admittedly be difficult at times (Williamson 1999; Scott and Reinhardt 2001; Cruz *et al.* 2004; Menning and Stephens 2007; Mitsopoulos and Dimitrakopoulos 2007), especially in forest stands with highly complex vertical fuel distributions. Muraro (1971) was the first to suggest a threshold CBD value (i.e. 0.320 kg m^{-3}) as a means of quantitatively defining the CBH. Sando and Wick (1972) indicated that 'little is known about the amount of fuel required to support combustion vertically'; they ended up selecting an arbitrary threshold value as well (i.e. 0.037 kg m^{-3}), which

Williams (1977) simply doubled for his application (i.e. $0.074\,kg\,m^{-3}$). Roussopoulos (1978) arbitrarily defined CBH as the height separating the lower 5.0% of the total needle foliage load from the upper 95%.

In determining CBH, the majority of simulation studies examining potential crown fire behaviour have followed Scott and Reinhardt's (2001) definition – i.e. 'the lowest height above ground at which there is a sufficient amount of canopy fuel to propagate fire vertically into the canopy'. Scott and Reinhardt (2001) also selected an arbitrary CBD value ($0.011\,kg\,m^{-3}$) as the basis for determining CBH. In the intervening years, this approach has come to be an accepted standard with little or no questioning of its origin. Reinhardt *et al.* (2006*a*) readily admit that this threshold value is 'not based on any kind of combustion physics, but it seems to perform well', although they offer no details regarding their performance testing. Thus, the lack of an objectively defined threshold CBD value for determining CBH remains a continuing research need (Alexander 2006).

Meaning of the two crown fire hazard indices

TI and CI values are outputs of NEXUS, FFE-FVS and FMA-Plus but not of the BehavePlus, FARSITE or FlamMap modelling systems. The TI and CI concept were initially introduced by Scott (1998*b*) and later elaborated on by Scott and Reinhardt (2001) for the purpose of assessing crown fire hazard in coniferous forests. Scott (2008) has also extended the methodology to shrubland and open forest woodland fuel complexes. The TI might have been more appropriately termed the 'passive or intermittent crowning index' as torching is more commonly associated with calm to light winds (e.g. Lawson 1972; Dyrness and Norum 1983) and a single tree torching does not make for even a passive crown fire (Forestry Canada Fire Danger Group 1992). Similarly, the CI could have been labelled the 'active or continuous crowning index'.

Although the TI and CI are to be regarded as relative numerical values (Fulé *et al.* 2004; Roccaforte *et al.* 2008; Stephens *et al.* 2009), Scott and Reinhardt (2001) chose to express both indices in terms of the wind speed (in either $km\,h^{-1}$ or miles h^{-1}) as taken at a height of 6.1 m (20 feet) above open ground per the standard for fire danger rating and fire behaviour prediction used in the US (Deeming *et al.* 1977; Rothermel 1983). Later on, Scott (2006) expressed TI and CI in terms of the 10-m open wind standard used for fire danger rating and fire behaviour prediction in Canada (Lawson and Armitage 2008) and elsewhere (e.g. Australia and New Zealand).

The present practice of calculating TI and CI values by various authors does not readily allow for direct comparison between different studies or assessments. For example, the fuel moisture contents selected are based on one of the various scenarios presented by Rothermel (1991) or on percentile values derived from a fire weather database, each of which has value. Added to this is the fact that both the FDFM (Rothermel 1983) and the NFDRS 1-h time-lag fuel moisture content (Fosberg and Deeming 1971; Deeming *et al.* 1977) are used in computing the two crown-fire hazard indices and they do not result in the same numerical value for a given set of weather conditions. Some authors have failed to specify the associated environmental conditions (e.g. Graves and Neuenschwander 2001; Fiedler *et al.* 2004; Monleon *et al.* 2004; Mason *et al.* 2007) or the description remains vague (e.g. Moghaddas and Craggs 2007). Furthermore, some authors have failed to explicitly specify the FMC applied in their simulations (e.g. Stephens 1998; Monleon *et al.* 2004; Johnson *et al.* 2007; DeRose and Long 2009). The situation is further complicated by the lack of standardisation of the index scale as dictated by the use of two different units of measure (i.e. $km\,h^{-1}$ and miles h^{-1}) and to a much lesser extent, two different open wind-speed exposure heights (i.e. 6.1 and 10 m). To make matters worse, some authors have now chosen to express TI and CI outputs in $m\,s^{-1}$ (e.g. Ritchie *et al.* 2007; Finkral and Evans 2008). The basic premise of any index is that it has a consistent scale.

Summary and concluding remarks

The ready availability of a multitude of fire modelling systems in the US in recent years has led to their widespread use in numerous simulation studies aimed at assessing various fire behaviour characteristics associated with specific fuel complex structures, including the propensity for crown fire initiation and spread (McHugh 2006). The results of these simulations, often aimed at evaluating fuel treatment effectiveness, are in turn utilised in a whole host of applications (e.g. Scott 2003; Fiedler *et al.* 2004; Skog *et al.* 2006; Johnson *et al.* 2007; Finkral and Evans 2008; Huggett *et al.* 2008; Johnson 2008; Reinhardt *et al.* 2010) and thus have significant implications for public and wildland firefighter safety, community fire protection, fire management policy-making, and forest management practices. As Cheney (1981) has noted, 'The reality of fire behaviour predictions is that overestimates can be easily readjusted without serious consequences; underestimates of behaviour can be disastrous both to the operations of the fire controller and the credibility of the person making the predictions'.

A critical review of several of these simulation studies, as documented here, has found that the results are often unrealistic for a variety of reasons. It's recognised that the authors of these studies commonly point out the limitations of the models and modelling systems being used through a customary disclaimer concerning the unknowns regarding crown fire behaviour (e.g. Stephens *et al.* 2009). Nevertheless, the fact that the fuel treatment evaluation studies referenced here are based on modelling systems that utilised model linkages for gauging potential crown fire behaviour that have not previously undergone any form of performance evaluation against independent datasets or any empirical observations should be of concern. There appears, however, to be an aversion within an element of the fire research community to do so (e.g. Scott and Reinhardt 2001; Scott 2006; Stephens *et al.* 2009). Nevertheless, such testing is now generally regarded as a basic tenet of modern-day model development and evaluation (Jakeman *et al.* 2006).

Fire modelling systems like NEXUS (Scott and Reinhardt 2001), FFE-FVS (Reinhardt and Crookston 2003), FARSITE (Finney 2004), FMAPlus (Carlton 2005), FlamMap (Finney 2006), and BehavePlus (Andrews *et al.* 2008) that are based on separate implementations or linkages between Rothermel's (1972, 1991) rate of fire spread models and Van Wagner's (1977, 1993) crown fire transition and propagation models have been shown to have a marked underprediction bias when used to assess potential crown fire behaviour. What has been allowed to evolve is a family of modelling systems composed of independently developed, linked models that were never intended to work together, are sometimes based on very limited data, and may propagate errors beyond acceptable limits.

We have documented here the sources of the bias based on empirical evidence in the form of published experimental fire and wildfire datasets. By analysing model linkages and components, we have described the primary sources of such bias, namely: (1) incompatible model linkages; (2) use of surface and crown fire rate of spread models that have an inherent underprediction bias; and (3) reduction in crown fire rate of spread based on use of unsubstantiated CFB functions. The use of uncalibrated, custom fuel models to represent surface fuelbeds is considered another potential source of bias.

Our analysis has also shown that the crown fire initiation underprediction bias inherent in all of these fire modelling systems could possibly be rectified by modifying the method used to calculate the surface fireline intensity for the purposes of assessing crown fire initiation potential, namely using Nelson's (2003) model to estimate t_r in place of Anderson's model (1969). Other modelling systems exist for predicting the likelihood of crown fire initiation and other aspects of crown fire behaviour (Alexander *et al.* 2006; Cruz *et al.* 2006*b*, 2008). Mitsopoulos and Dimitrakopoulos (2007) have, for example, made extensive use of this suite of models in their assessment of crown fire potential in Aleppo pine (*Pinus halepensis*) forests in Greece. These systems are based on models that have undergone performance evaluations against independent datasets and been shown to be reasonably reliable (Cruz *et al.* 2003*b*, 2004, 2006*b*; Cronan and Jandt 2008). Resolving the underprediction bias associated with predicting active crown fire rate of spread inherent in the Rothermel (1991) model would require substantial changes, including a reassessment of the use of a CFB function, if not complete replacement with a more robust empirically developed model (Cruz *et al.* 2005) that has been extensively tested (Alexander and Cruz 2006) or a physically based one that has undergone limited testing (Butler *et al.* 2004).

Alexander (2007) has emphasised that assessments of wildland fire potential involving simulation modelling must be complemented with fire behaviour case study knowledge and by experienced judgment. This review has revealed an overwhelming need for the research users of fire modelling systems to be grounded in the theory and proper application of such tools, including a solid understanding of the assumptions, limitations and accuracy of the underlying models as well as practical knowledge of the subject phenomena (Brown and Davis 1973; Albini 1976; Alexander 2009*a*, 2009*b*).

List of symbols, quantities and units used in equations and text

C, criterion for initial crown combustion (kW$^{2/3}$ kJ^{-1} kg m$^{-5/3}$)
CBD, canopy bulk density (kg m^{-3})
CBH, canopy base height (m)
CFL, canopy fuel load (kg m^{-2})
CFB, crown fraction burned
CI, crowning index (km h^{-1})
FDFM, fine dead fuel moisture (%)
FMC, foliar moisture content (%)
h, heat of ignition (kJ kg^{-1})
H, low heat of combustion (kJ kg^{-1})
I_B, fireline intensity (kW m^{-1})
I_o, critical surface fire intensity for initial crown combustion (kW m^{-1})
I_R, reaction intensity (kW m^{-2})
r, rate of fire spread (m s^{-1})
R, final rate of fire spread, surface or crown (m min^{-1})
R_a, active crown fire rate of spread (m min^{-1})
R_i, critical surface fire rate of spread for crown fire initiation (m min^{-1})
R_s, surface fire rate of spread (m min^{-1})
R_o, critical minimum spread rate for active crowning (m min^{-1})
R_{10}, predicted surface fire rate of spread for Fuel Model 10 using a 0.4 wind reduction factor (m min^{-1})
S_o, critical mass flow rate for solid crown flame (kg m^{-2} min^{-1})
t_r, flame front residence time (s)
TI, torching index (km h^{-1})
U_{10}, 10-m open wind speed (km h^{-1})
w, fuel consumed in the active flaming front and by glowing or smouldering combustion following passage of the front (kg m^{-2})
w_a, fuel consumed in the active flaming front (kg m^{-2})

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esa ECOSPHERE

Does increased forest protection correspond to higher fire severity in frequent-fire forests of the western United States?

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Abstract. There is a widespread view among land managers and others that the protected status of many forestlands in the western United States corresponds with higher fire severity levels due to historical restrictions on logging that contribute to greater amounts of biomass and fuel loading in less intensively managed areas, particularly after decades of fire suppression. This view has led to recent proposals—both administrative and legislative—to reduce or eliminate forest protections and increase some forms of logging based on the belief that restrictions on active management have increased fire severity. We investigated the relationship between protected status and fire severity using the Random Forests algorithm applied to 1500 fires affecting 9.5 million hectares between 1984 and 2014 in pine (*Pinus ponderosa*, *Pinus jeffreyi*) and mixed-conifer forests of western United States, accounting for key topographic and climate variables. We found forests with higher levels of protection had lower severity values even though they are generally identified as having the highest overall levels of biomass and fuel loading. Our results suggest a need to reconsider current overly simplistic assumptions about the relationship between forest protection and fire severity in fire management and policy.

Key words: biodiversity; climate; fire frequency; fire severity; fire suppression; Gap Analysis Program levels; logging; protected areas.

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Introduction

It is a widely held assumption among federal land management agencies and others that a lack of active forest management of some federal forestlands—especially within relatively frequent-fire forest types such as ponderosa pine (*Pinus ponderosa*) and mixed conifers—is associated with higher levels of fire severity when wildland fires occur (USDA Forest Service 2004, 2014, 2015, 2016). This prevailing forest/fire management hypothesis assumes that forests with higher levels of protection, and therefore less logging, will burn more intensely due to higher fuel loads and forest density. Recommendations have been made to increase logging as fuel reduction and decrease forest protections before wildland fire can be more extensively reintroduced on the landscape after decades of fire suppression (USDA Forest Service 2004, 2014, 2015, 2016). The concern follows that, in the absence of such a shift in forest management, fires are burning too severely and may adversely affect forest resilience (North et al. 2009, 2015, Stephens et al. 2013, 2015, Hessburg 2016). Nearly every fire season, the United States Congress introduces forest management legislation based on this view and aimed at increasing mechanical fuel treatments via intensive logging and weakened forest protections.

However, the fundamental premise for this fire management strategy has not been rigorously tested across broad regions. We broadly assessed the influence of forest protection levels on fire severity in pine and mixed-conifer forests of the western United States with relatively frequent-fire regimes to test this assumption. We used vegetation burn severity data from all fires >405 ha over a three-decade period, 1984–2014, in forests with varying levels of protection.

Study area

Pine and mixed-conifer forests at low/mid-elevations, where historical fires were relatively frequent, are broadly distributed across several ecoregions in the western United States (Fig. 1; Appendix S1: Table S1). Although ponderosa pine often dominates these forests, they can also include Jeffrey pine (*Pinus jeffreyi*), which in places intermix with, and are similar to, ponderosa pine forests, and Madrean pine–oak (*Quercus* spp.) forests with a diversity of pines. Mixed-conifer forests at low/mid-elevations are also broadly distributed across multiple ecoregions (Fig. 1). They can include additional pines (e.g., lodgepole pine, *Pinus contorta*; sugar pine, *Pinus lambertiana*), true firs (*Abies* spp.), Douglas-fir (*Pseudotsuga menzeisii*), and incense-cedar (*Calocedrus decurrens*).

Methods

We used Gap Analysis Program (GAP) protection classes (USGS 2012), as described below, to determine whether areas with the most protection (i.e., GAP1 and GAP2) had a tendency to burn more severely than areas where intensive management is allowed (i.e., GAP3 and GAP4). We compared satellite-derived burn severity data for 1500 fires affecting 9.5 million hectares from years for which there were available data (1984–2014) among four different forest protection levels (Fig. 1), accounting for variation in topography and climate. We analyzed fires within relatively frequent-fire forest types comprised of pine and mixed-conifer forests mainly because these are the predominant forest types at low to mid-elevations in the western United States, there is a large data set on fire occurrence, and they have been a major concern of land managers for some time due to decades of fire suppression. We defined geographic extent of forest types from the Biophysical Settings data set (BpS) (Rollins 2009; *public communication*, http://www.landfire.gov) that derived forest maps from satellite imagery and represents plant communities based on NatureServe's Ecological Systems classification. Baker (2015) noted that some previous work found ~65% classification accuracy of this system with regard to specific forest types and, accordingly, he analyzed groups of related forest types in order to improve accuracy. We followed his approach (see Appendix S1: Table S1). The categories selected from the Biophysical Settings map were ponderosa/Jeffrey pine and mixed-conifer forest types with relatively frequent-fire regimes (e.g., Swetnam and Baisan 1996, Taylor and Skinner 1998, Schoennagel et al. 2004, Stephens and Collins 2004, Sherriff et al. 2014), compared to other forest types with different fire regimes such as high-elevation forests and many coastal forests not studied herein. Forest types in our study totaled 29.2 million hectares (Fig. 1; Appendix S1: Table S1). We used the BpS data to capture areas that were classified as forests before fire, because postfire vegetation maps can potentially show these same areas as temporarily changed to other vegetation types. We sampled our response and predictor variables on an evenly spaced 90 × 90 m grid within these forest types using ArcMap 10.3 (ESRI 2014). This created a data set of 5,580,435 independent observations from which we drew our random samples to create our models. The 90-m spacing was chosen because it was the smallest spacing of points that was computationally practical with which to operate.

Fires

The Monitoring Trends in Burn Severity project (MTBS, *public communication*, http://www.mtbs.gov) is a U.S. Department of Interior and Department of Agriculture-sponsored program that has compiled burn severity data from satellite imagery, which became available in 1984, for fires >405 ha, and was current up to 2014 (Eidenshink et al. 2007). The MTBS Web site allows bulk download of spatial products that include two closely related indices of burn severity: differenced normalized burn ratio (dNBR) (Key and Benson 2006) and relative differenced normalized burn ratio (RdNBR) (Miller and Thode 2007). Both indices are calculated from Landsat TM and ETM satellite imagery of reflected light from the earth's surface at infrared wavelengths from before and after fire to

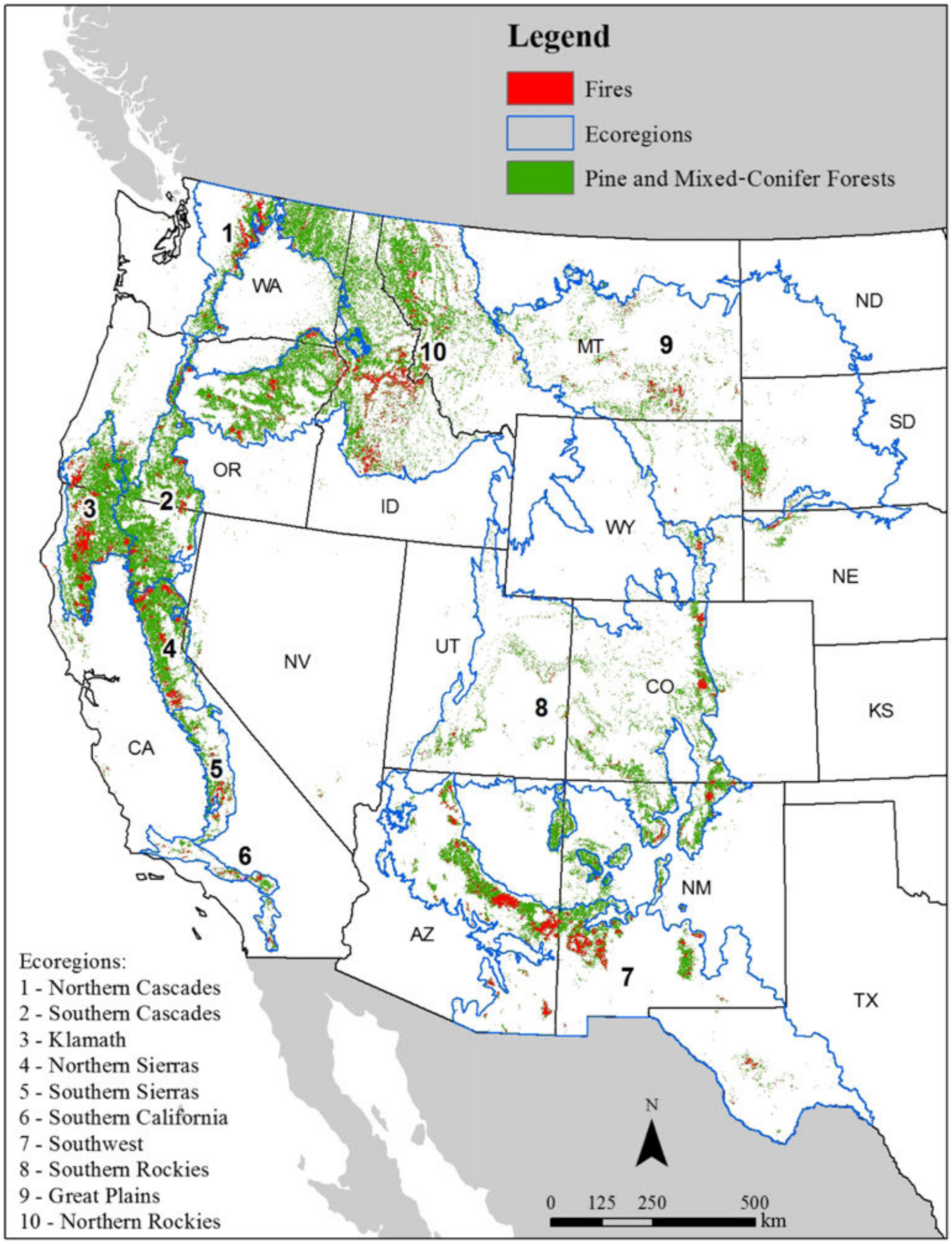

Fig. 1. Pine and mixed-conifer forests, fires, and ecoregions analyzed in this study.

measure associated changes in vegetation cover and soil characteristics. We defined burn severity with the RdNBR index because it adjusts for prefire conditions at each pixel and provides a more consistent measure of burn severity than dNBR when studying broad geographic regions with many different vegetation types (Miller et al. 2009*a*, Norton et al. 2009). RdNBR values typically range from negative 500 to 1500 with values further away from zero representing greater change from prefire conditions. Negative values represent vegetation growth and positive values increasing levels of overstory vegetation mortality. The RdNBR values could be used to classify fires into discrete burn severity classes of low, medium, and high but this was not performed in our study, as we desired to have a continuous response variable in our models.

We intersected forest sampling points with fire perimeters downloaded from MTBS to determine fires that occurred in our analysis area, and censored fires with <100 sampling points (81 ha). The remaining points represented sampling locations from 2069 fires (Fig. 1). We extracted RdNBR values at each sampling point as our response variable as well as predictor variables that included topography, geography, climate, and GAP status. These sampling points were used to investigate the relationship between forest protection levels and burn severity (Appendix S1: Tables S2 and S3). We chose topographic and climatic variables based on previous studies that quantified the relationship between burn severity, topography, and climate (Dillon et al. 2011, Kane et al. 2015).

Topographic and climatic data

To account for the effects of topographic and climatic variability, we derived several topographic indices (Appendix S1: Table S2) from seamless elevation data (*public communication*, http://www.landfire.gov/topographic.php) downscaled to 90-m^2 spatial resolution due to computational limits when intersecting sampling points. These indices capture categories of topography, including percentage slope, surface complexity, slope position, and several temperature and moisture metrics derived from aspect and slope position. We used the Geomorphometry and Gradient Metrics Toolbox version 2.0 (*public communication*, http://evansmurphy.wix.com/evansspatial) to compute these metrics. We also computed several temperature and precipitation variables (Appendix S1: Table S3) by downloading climatic conditions for each month from 1984 to 2014 from the PRISM climate group (*public communication*, http://prism.oregonstate.edu). Climate grids record precipitation and minimum, mean, and maximum temperature at a 4-km grid scale created by interpolating data from over 10,000 weather stations. To determine the departure from average conditions, we subtracted each climate grid by its 30-yr mean monthly value. These "30-yr Normals" data sets were also downloaded from the PRISM Web site and reflected the mean values from the most recent full decades (1981–2010). We determined mean seasonal values with summer defined as the mean of July, August, and September of the year before a given fire; fall being the mean of October, November, and December of the previous year; winter the mean of January, February, and March of the current year of a given fire; and spring the mean of April, May, and June of the current year.

Protected area status and ecoregion classification

We used the Protected Areas Database of the United States (PAD-US; USGS 2012) to determine forest protection status, which is the U.S. official inventory of protected open space. The PAD-US includes all federal and most State conservation lands and classifies these areas with a GAP ranking code (see map at: http://gis1.usgs.gov/csas/gap/viewer/padus/Map.aspx). The GAP status code (herein referred to interchangeably as GAP class or protection status) is a metric of management to conserve biodiversity with four relative categories. GAP1 is protected lands managed for biodiversity where disturbance events (e.g., fires) are generally allowed to proceed naturally. These lands include national parks, wilderness areas, and national wildlife refuges. GAP2 is protected lands managed for biodiversity where disturbance events are often suppressed. They include state parks and national monuments, as well as a small number of wilderness areas and national parks with different management from GAP1. GAP3 is lands managed for multiple uses and are subjected to logging. Most of these areas consist of non-wilderness USDA Forest Service and U.S. Department of Interior Bureau of Land Management lands as well as state trust lands. GAP4 is lands with no mandate for protection such as tribal, military, and private lands. GAP status is relevant to the intensity of both current and past managements.

We made one modification to GAP levels by converting Inventoried Roadless Areas (IRAs) from the 2001 Roadless Area Conservation Rule (S_USA.RoadlessArea_2001, *public communication*, http://data.fs.usda.gov/geodata/edw/datasets.php) to GAP2 unless these areas already were defined as GAP1. We considered most IRAs as GAP2 given they are prone to policy changes and because they allow for certain limited types of logging (e.g., removal of predominately small trees for fuel reduction in some circumstances).

However, we note that very little logging has occurred within IRAs since the Roadless Rule, although there occasionally have been proposals to log portions of some IRAs pre- and postfire, and fire suppression often occurs.

We modified level III ecoregions (U.S. Environmental Protection Agency (EPA) 2013) to create areas of similar climate and geography (Fig. 1). We did this by extracting ecoregions and combining adjacent provinces in our study region.

Random Forests analysis

We investigated the relationship between protection status and burn severity using the data-mining algorithm Random Forests (RF) (Breiman 2001) with the "randomForestSRC" add-in package (Ishwaran and Kogalur 2016) in R (R Core Team 2013). This algorithm is an extension of classification and regression trees (CART) (Breiman et al. 1984) that recursively partitions observations into groups based on binary rule splits of the predictor variables. The main advantage of using RF in our study is that it can work with spatially autocorrelated data (Cutler et al. 2007). It can also model complex, nonlinear relationships among variables, makes no assumption of variable distributions (Kane et al. 2015), and produces accurate predictions without overfitting the available data (Breiman 2001).

Our independent observations were a random subset of our 5.5 million points, from which we drew three random samples of 25,000 points each. Each sample consisted of 500 fires randomly selected without replacement from the pool of 2069 fires. Fifty points were then randomly selected within each of the 500 fires. Our dependent variables were all continuous (Appendix S1: Tables S2 and S3) except for the main variable of interest, protected area status, which included the four GAP levels. The three observation samples were used to create three RF model runs, each consisting of 1000 regression trees. We conducted three RF model runs to assess whether our random samples of 25,000 points produced fairly consistent results.

The RF algorithm samples approximately 66% of the data to build the regression trees, and the remaining data are used for validation and to assess variable importance. We used this validation sample to determine the amount of variance explained and variable importance. The algorithm also produces individual variable importance measures by calculating differences in prediction mean-square-error before and after randomly permuting each dependent variable's values. Variable importance is a measure of how much each variable contributes to the model's overall predicative accuracy.

Unlike linear models, RF does not produce regression coefficients to examine how a change in a predictor variable affects the response variable. The analogy to this in RF is the partial dependence plot which is a graphical depiction of how the response will change with a single predictor while averaging out the effects of the other predictors, such as the climatic and topographic variables (Cutler et al. 2007). We used this approach, in addition to using RF to determine overall variable importance as described above, in order to determine the effect of GAP status, in particular, on fire severity, while averaging out effects of climate and topography.

Mixed-effects analysis

We performed a linear mixed-effects analysis using the "nlme" add-on package in R (Pinheiro et al. 2015). We used a random intercept model and identified year of fire (n = 31) and ecoregion (n = 10) as random effects. Similar to our RF models, our independent observations were a random subset of our 5.5 million points but for these models we drew three random samples of 50,000 points each. Each sample consisted of 500 fires randomly selected without replacement, and within each of those fires, 100 points were randomly selected. Our dependent variables were the same used in our RF models, and we log-transformed the non-normal variables of slope, surface roughness, and topographic radiation aspect index. We removed dependent variables that were correlated with each other (Pearson's r > 0.5), retaining 21 of 45 candidate dependent variables, and centered these on their means. Model reduction was performed in a stepwise process using bidirectional elimination with Bayesian information criterion selection criterion.

Spatial autocorrelation analysis

Spatial autocorrelation (SA) is the measure of similarity between pairs of observations in relationship to the distance between them. Ecological variables are inherently autocorrelated because

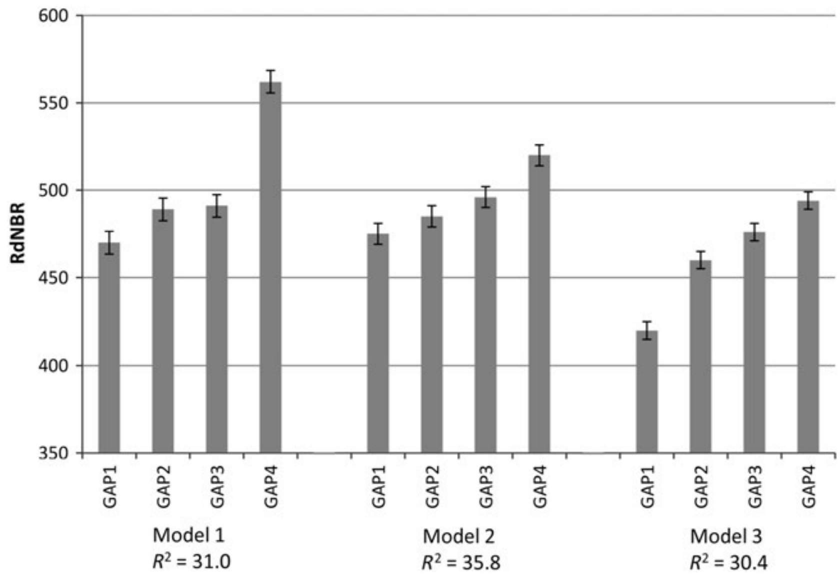

Fig. 2. Random Forests partial dependence of protection status vs. RdNBR burn severity for each model (n = 25,000). The variance explained is shown as pseudo R^2.

landscape attributes that are closer together are often more similar than those that are far apart.

We assessed the SA in the Pearson residuals with inspection of Moran's I autocorrelation index using the "APE" package add-in in R (Paradis et al. 2004) after removing points that shared the same x and y coordinates. Moran's I is an index that ranges from −1 to 1 with the sign of the values indicating strength and direction of SA. Values close to zero are considered to have a random spatial pattern. Our mixed-effects models all had a Moran's I values statistically different from 0 at the 95% confidence level ($P < 0.001$) so we included a spatial correlation structure in our model using the "nlme" package in R. Of Gaussian, exponential, linear, and spherical spatial correlation structures, we determined that the exponential structure produced the lowest Akaike's information criterion (AIC). Despite these additions, our second measurements still found relatively small, but significant, autocorrelation (Moran's I for model runs 1, 2, 3 = 0.10, 0.08, 0.10, all $P < 0.001$).

Results

With regard to ranking of variables in the model runs, variable importance plots from the three RF model runs show that protection status was consistently ranked as one of the 10 most important of the 45 variables in explaining burn severity (Appendix S1: Table S4). The most important variable explaining burn severity was ecoregion for models 1 and 2 and maximum temperature from the previous fall for model 3.

With regard to the GAP status variable in particular, after averaging out the effects of climatic and topographic variables, the RF partial dependence plots show an increasing trend of fire severity with decreasing protection status (Fig. 2). Fires in GAP4 had mean RdNBR values greater than two standard errors higher than all other GAP levels. Fires in GAP3 had mean RdNBR values two standard errors higher than GAP1 in all model runs. GAP3 differences with GAP2 were less pronounced with only one model showing differences greater than two standard errors. Fires in GAP1 were consistently the least severe, being two standard errors less than GAP3 in all model runs and two standard errors less than GAP2 in two of three model runs.

Our mixed-effects models validated these findings with similar results (Fig. 3, Appendix S1: Table S5). Like our RF models, our linear mixed-effects models showed GAP4 fires to have significantly higher RdNBR values and GAP1 fires to have significantly lower RdNBR values when compared to all other GAP classes. Fires in GAP

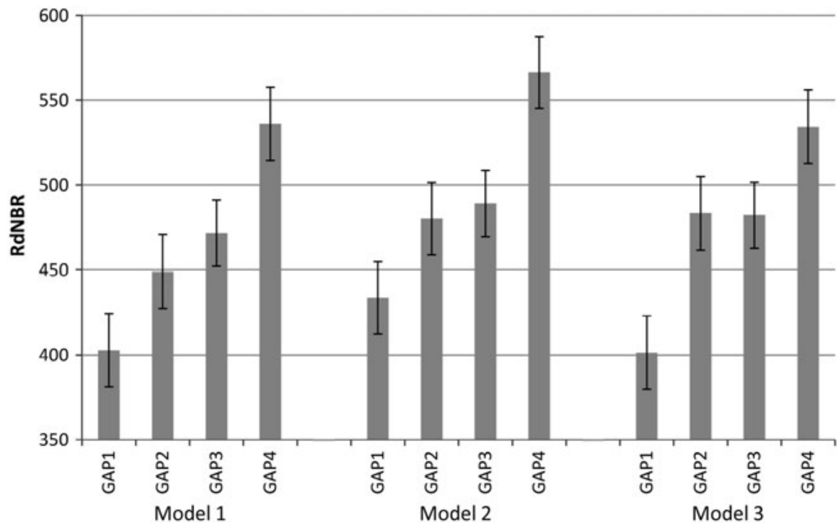

Fig. 3. Linear mixed effects models of protection status vs. RdNBR burn severity (n = 50,000).

status levels 2 and 3 were not significantly different in the mixed-effects models. Although the level of autocorrelation was significant, it was small in our model (Moran's I ~0.1) and not enough to account for such a substantial difference in burn severity among protection classes.

Discussion

Protected forests burn at lower severities

We found no evidence to support the prevailing forest/fire management hypothesis that higher levels of forest protections are associated with more severe fires based on the RF and linear mixed-effects modeling approaches. On the contrary, using over three decades of fire severity data from relatively frequent-fire pine and mixed-conifer forests throughout the western United States, we found support for the opposite conclusion—burn severity tended to be higher in areas with lower levels of protection status (more intense management), after accounting for topographic and climatic conditions in all three model runs. Thus, we rejected the prevailing forest management view that areas with higher protection levels burn most severely during wildfires.

Protection classes are relevant not only to recent or current forest management practices but also to past management. Millions of hectares of land have been protected from logging since the 1964 Wilderness Act and the 2001 Roadless Rule, but these areas are typically categorized as such due to a lack of historical road building and associated logging across patches >2000 ha, while GAP3 lands, for instance, such as National Forests lands under "multiple use management," have generally experienced some form of logging activity over the last 80 yr.

We expect that the effects of historic logging from nearly a century ago to gradually lessen over time, as succession and natural disturbance processes reestablish structural and compositional complexity, but it was beyond the scope of this study to attempt to assess the relative role of recent vs. historical logging. Similarly, industrial fire suppression programs that intensified in the 1940s influenced fire extent across forest protection classes. While more recent let-burn policies have been applied in GAP1 and GAP2 forests in some circumstances, evidence indicates that protected forests nevertheless remain in a substantial fire deficit, relative to the prefire suppression era (Odion et al. 2014, 2016, Parks et al. 2015). Thus, we believe it is unlikely that recent decisions to allow some backcountry fires to burn, largely unimpeded, account for much of the differences in fire severity among protection classes that we found, simply because such let-burn policies have not been extensive enough to remedy the ongoing fire deficit.

While forests in different protection classes can vary in elevation, with protected forests often occupying higher elevations, our results indicate that protection class itself produced notable differences in fire severity after averaging out the effects of elevation and climate (see Fig. 2 and *Results* above). In our study, GAP1 forests were 284 m on average higher in elevation than GAP4 forests, while GAP1 forests experienced lower fire severity. This is the opposite of expectations if elevation was a key influence because higher elevation forests are associated with higher fire severity (see, e.g., Schoennagel et al. 2004, Sherriff et al. 2014). We note that we are not the first to determine that increased fire severity often occurs in forests with an active logging history (Countryman 1956, Odion et al. 2004).

Prevailing forest–fire management perspectives vs. alternative views

An extension of the prevailing forest/fire management hypothesis is that biomass and fuels increase with increasing time after fire (due to suppression), leading to such intense fires that the most long-unburned forests will experience predominantly severe fire behavior (e.g., see USDA Forest Service 2004, Agee and Skinner 2005, Spies et al. 2006, Miller et al. 2009*b*, Miller and Safford 2012, Stephens et al. 2013, Lydersen et al. 2014, Dennison et al. 2014, Hessburg 2016). However, this was not the case for the most long-unburned forests in two ecoregions in which this question has been previously investigated—the Sierra Nevada of California and the Klamath-Siskiyou of northern California and southwest Oregon. In these ecoregions, the most long-unburned forests experienced mostly low/moderate-severity fire (Odion et al. 2004, Odion and Hanson 2006, Miller et al. 2012, van Wagtendonk et al. 2012). Some of these researchers have hypothesized that as forests mature, the overstory canopy results in cooling shade that allows surface fuels to stay moister longer into fire season (Odion and Hanson 2006, 2008). This effect may also lead to a reduction in pyrogenic native shrubs and other understory vegetation that can carry fire, due to insufficient sunlight reaching the understory (Odion et al. 2004, 2010).

Another fundamental assumption is that current fires are becoming too large and severe compared to recent historical time lines (Agee and Skinner 2005, Spies et al. 2006, Miller et al. 2009*b*, Miller and Safford 2012, Stephens et al. 2013, Lydersen et al. 2014, Dennison et al. 2014, Hessburg 2016). However, others have shown that this is not the case for most western forest types. For instance, using the MTBS (www.mtbs.gov) data set, Picotte et al. (2016) found that most vegetation groups in the conterminous United States exhibited no detectable change in area burned or fire severity from 1984 to 2010. Similarly, Hanson et al. (2009) found no increase in rates of high-severity fire from 1984 to 2005 in dry forests within the range of the northern spotted owl (*Strix occidentalis caurina*) based on the MTBS data set. Using reference data and records of high-severity fire, Baker (2015) found no significant upward trends in fire severity from 1984 to 2012 across all dry western forest regions (25.5 million ha), nearly all of which instead were too low or were within the range of historical rates. Parks et al. (2015) modeled area burned as a function of climatic variables in western forests and non-forest types, documenting most forested areas had experienced a fire deficit (observed vs. expected) during 1984 to 2012 that was likely due to fire suppression.

Whether fires are increasing or not depends to a large extent on the baseline chosen for comparisons (i.e., shifting baseline perspective, Whitlock et al. 2015). For instance, using time lines predating the fire suppression era, researchers have documented no significant increases in high-severity fire for dry forests across the West (Williams and Baker 2012*a*, Odion et al. 2014) or for specific regions (Williams and Baker 2012*b*, Sherriff et al. 2014, Tepley and Veblen 2015). Future trends, with climate change and increasing temperatures, may be less simple than previously believed, due to shifts in pyrogenic understory vegetation (Parks et al. 2016).

This is more than just a matter of academic debate, as most forest management policies assume that fire, particularly high-severity fire, is increasing, is in excess of recent historical baselines, and needs to be reduced in size, intensity, and occurrence over large landscapes to prevent widespread ecosystem damages (policy examples include USDA Forest Service 2002, Healthy Forests Restoration Act 2003, USDA Forest Service 2009, HR 167: Wildfire Disaster Funding Act 2015). However, large fires (landscape scale or the so-called megafires) produce myriad ecosystem benefits underappreciated by most land managers and decision-makers (DellaSala and Hanson 2015*a*, DellaSala et al. 2015). High-severity fire patches, in particular, provide a pulse of "biological legacies" (e.g., snags, down logs, and native shrub patches) essential for complex early seral associates (e.g., many bird species) that link seral stages from new forest to old growth (Swanson et al. 2011, Donato et al. 2012, DellaSala et al. 2014, Hanson 2014, 2015, DellaSala and Hanson 2015*a*). Complex early seral forests are most often logged after fire, which, along with aggressive fire suppression, exacerbates their rarity and heightens their conservation importance (Swanson et al. 2011, DellaSala et al. 2014, 2015, Hanson 2014).

Limitations

One limitation of our study is that, due to the coarseness of the management intensity variables that we used (i.e., GAP status), we cannot rule out whether low intensities of management decreased the occurrence of high-severity fire in some circumstances. However, the relationship between forest density/fuel, mechanical fuel treatment, and fire severity is complex. For instance, thinning without subsequent prescribed fire has little effect on fire severity (see Kalies and Yocum Kent 2016) and, in some cases, can increase fire severity (Raymond and Peterson 2005, Ager et al. 2007, Wimberly et al. 2009) and tree mortality (see, e.g., Stephens and Moghaddas 2005, Stephens 2009: Figure 6)—the effects depend on the improbable co-occurrence of reduced fuels (generally a short time line, within a decade or so) and wildfire activity (Rhodes and Baker 2008) and can be over-ridden by extreme fire weather (Bessie and Johnson 1995, Hély et al. 2001, Schoennagel et al. 2004, Lydersen et al. 2014). Empirical data from actual fires also indicate that postfire logging can increase fire severity in reburns (Thompson et al. 2007), despite removal of woody biomass (tree trunks) described by land managers as forest fuels (Peterson et al. 2015). While our study did not specifically test for these effects, such active forest management practices are common on GAP3 and GAP4 lands. Recognizing these limitations, researchers have stressed the need for managers to strive for coexistence with fire by prioritizing fuel reduction nearest homes and allowing more fires to occur unimpeded in the backcountry (Moritz 2014, DellaSala et al. 2015, Dunn and Bailey 2016, Moritz and Knowles 2016).

Follow-up research at finer scales is needed to determine management emphasis and history in relation to fire severity. However, we believe our findings are robust at the subcontinental and ecoregional scales.

Conclusions

In general, our findings—that forests with the highest levels of protection from logging tend to burn least severely—suggest a need for managers and policymakers to rethink current forest and fire management direction, particularly proposals that seek to weaken forest protections or suspend environmental laws ostensibly to facilitate a more extensive and industrial forest–fire management regime. Such approaches would likely achieve the opposite of their intended consequences and would degrade complex early seral forests (DellaSala et al. 2015). We suggest that the results of our study counsel in favor of increased protection for federal forestlands without the concern that this may lead to more severe fires.

Allowing wildfires to burn under safe conditions is an effective restoration tool for achieving landscape heterogeneity and biodiversity conservation objectives in regions where high levels of biodiversity are associated with mixed-intensity fires (i.e., "pyrodiversity begets biodiversity," see DellaSala and Hanson 2015*b*). Managers concerned about fires can close and decommission roads that contribute to human-caused fire ignitions and treat fire-prone tree plantations where fires have been shown to burn uncharacteristically severe (Odion et al. 2004). Prioritizing fuel treatments to flammable vegetation adjacent to homes along with specific measures that reduce fire risks to home structures are precautionary steps for allowing more fires to proceed safely in the backcountry (Moritz 2014, DellaSala et al. 2015, Moritz and Knowles 2016).

Managing for wildfire benefits as we suggest is also consistent with recent national forest policies such as 2012 National Forest Management Act planning rule that emphasizes maintaining and restoring ecological integrity across the national forest system and because complex early forests can only be produced by natural disturbance events not mimicked by mechanical fuel reduction or clear-cut logging (Swanson et al. 2011, DellaSala et al. 2014). Thus, managers wishing to maintain biodiversity in fire-adapted forests should appropriately weigh the benefits of wildfires against the ecological costs of mechanical fuel reduction and fire suppression (Ingalsbee and Raja 2015) and should consider expansion of protected forest areas as a means of maintaining natural ecosystem processes like wildland fire.

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Are Wildland Fires Increasing Large Patches of Complex Early Seral Forest Habitat?

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Abstract: High-severity fire creates patches of complex early seral forest (CESF) in mixed-severity fire complexes of the western USA. Some managers and researchers have expressed concerns that large high-severity patches are increasing and could adversely impact old forest extent or lead to type conversions. We used GIS databases for vegetation and fire severity to investigate trends in large (>400 ha) CESF patches in frequent-fire forests of the western USA, analyzing four equal time periods from 1984 to 2015. We detected a significant increase in the total area of large patches relative to the first time period only (1984–1991), but no significant upward trend since the early 1990s. There was no significant trend in the size of large CESF patches between 1984 and 2015. Fire rotation intervals for large CESF patches ranged from ~12 centuries to over 4000 years, depending on the region. Large CESF patches were highly heterogeneous, internally creating ample opportunities for fire-mediated biodiversity. Interior patch areas far removed from the nearest low/moderate-severity edges comprised a minor portion of high-severity patches but may be ecologically important in creating pockets of open forest. There was ample historical evidence of large CESF patches but no evidence of increases that might indicate a current risk of ecosystem-type shifts.

Keywords: complex early seral forest; conifers; biodiversity; high-severity fire; western USA

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1. Introduction

High-severity fire patches represent the component in fires that kill all or nearly all of the overstory trees within mixed-severity fire areas in conifer forests of the western USA [1,2], creating a unique forest habitat type known as the complex early seral forest (CESF) [3]. CESFs are distributed as small (<1 ha) to large patches (>400 ha) in mixed-severity burns in the lower/middle-montane conifer forests of the Sierra Nevada [2] and within other frequent-fire forest types of the western USA [4–6]. Unlike early seral produced by a clear-cut or otherwise intensively logged area, a CESF is more complex in its structure, and is characterized by a heterogeneous mix of abundant standing dead trees (snags) and downed logs, naturally regenerating conifers, other trees, shrub patches, and abundant wildflowers [3].

Whether high-severity fire is increasing and the ultimate causes of presumed increases (e.g., climate change, increase in tree densities) is the subject of much recent debate. For instance, the areal extent and proportion of high-severity fire within large fire complexes have not changed markedly in recent decades in most forested regions of the West [4,7–11], but results are equivocal in the Rocky Mountains and Southwestern US, e.g., see [9,11,12]. In the Sierra Nevada, some studies have reported increasing trends for high-severity fire, e.g., [13,14], whereas subsequent research [15,16] indicated no increases. Moreover, the size of CESF patches within large fire complexes has been used as a key metric to hypothesize whether fire regimes are operating within historical bounds [6,17–21]. Some have expressed concerns that large high-severity patches are increasing as a component of a recent increase in so-called megafires and that this may signal ecosystem-type shifts and the loss of old-growth forests [6,18,20,22], while others have predicted potential overall decreases in the future occurrence of high-severity fire in general [23]. Concern over high-severity fires and the resulting large patches of CESF has been a catalyst for fundamental changes to federal forest management policies (e.g., Healthy Forest Restoration Act of 2003, 2012 National Forest Management Act Planning Rule) and has been recently used to promote proposed congressional legislation that would substantially curtail environmental protections and dramatically increase logging in federal forests (e.g., The Resilient Federal Forests Act of 2019). Concerns over high-severity fires overall are missing a biodiversity perspective that is necessary to fully evaluate fire management proposals in the context of ecosystem benefits from such fires and not just their potential impacts on people [24,25].

Notably, patches of CESF support unique fire-adapted communities, including many plants [26], avifauna [27,28], mammals [29], bats [30], terrestrial [31] and aquatic invertebrates [32]. The Black-backed Woodpecker (*Picoides arcticus*) is associated with large CESF patches (typically ~100–800 ha for a single pair, depending on habitat quality) for nesting and foraging [33–36]. The California Spotted Owl (*Strix occidentalis occidentalis*), which is being petitioned for federal listing under the Endangered Species Act, actively forages in CESF patches [37,38]. Thus, policies aimed at suppressing large fires that otherwise would maintain and replenish CESF patches may have unintended consequences for fire-mediated biodiversity [24,25].

Our objectives were to determine whether there has been a recent trend (increase or decrease) in large CESF patches in fire areas within frequent-fire conifer forests of the western USA [4,39], to evaluate the spatiotemporal extent of such patches in these forests, assess their internal heterogeneity, and investigate historical evidence for the occurrence of such patches. Our study is the first to analyze the occurrence of large high-severity fire patches by distinct time periods. Additionally, our findings may have relevance to policy makers and forest-fire managers seeking to integrate biodiversity benefits of large CESF patches with wildfire risk reduction to people and natural resource management [24,25].

2. Methods

We analyzed the same western USA frequent-fire forest types, and used the same vegetation databases as in our related study [40] (Figure 1). These areas are dominated by mixed-conifer forests, as well as ponderosa pine (*Pinus ponderosa*) and Jeffrey pine (*Pinus jeffreyi*) forests.

We downloaded burn severity maps derived from satellite imagery from the Monitoring Trends in Burn Severity project (MTBS; http://www.mtbs.gov). Within the conifer forests of our study area, we defined CESF patches as areas experiencing high-severity fire, using a threshold of Relative Delta Normalized Burn Ratio (RdNBR) values ≥641 [41]. The same or similar thresholds have been used to define high-severity fire in multiple forest regions of western USA [21,42–44] and thus, our findings are directly applicable with consistent use of MTBS across studies. Although there is no accepted or standard definition of large CESF patches, we chose to analyze patches >400 ha in order to address concerns expressed by researchers that CESF patches hundreds of hectares or larger may not have occurred historically [6,18,21], may create homogeneity and inhibit post-fire forest regeneration due to lack of seed sources [20,22] and/or may reduce forest resilience to climate change [45–47]. We used an inclusive approach such that any high-severity fire pixels of conifer forest (30 × 30-m each) with sides touching were considered to be part of the same patch.

We used a Mann–Kendall test to determine whether there is any trend in (a) the combined total annual area of CESF patches >400 ha, and (b) the size of individual CESF patches >400 ha, for the years 1984–2015 (the period for which consistently mapped MTBS datasets were available for the US), analyzing both the annual area of large CESF patches, and the size of individual large CESF patches, as continuous variables. Mann–Kendall is a non-parametric test for monotonic upward or downward trends over time and has been used in similar studies [9,15,48]. Compared to other tests, including parametric tests, the Mann–Kendall has been found to have an equal or greater statistical power to detect trends in environmental time series data when the data are non-parametric, such as wildland fire trend data [15].

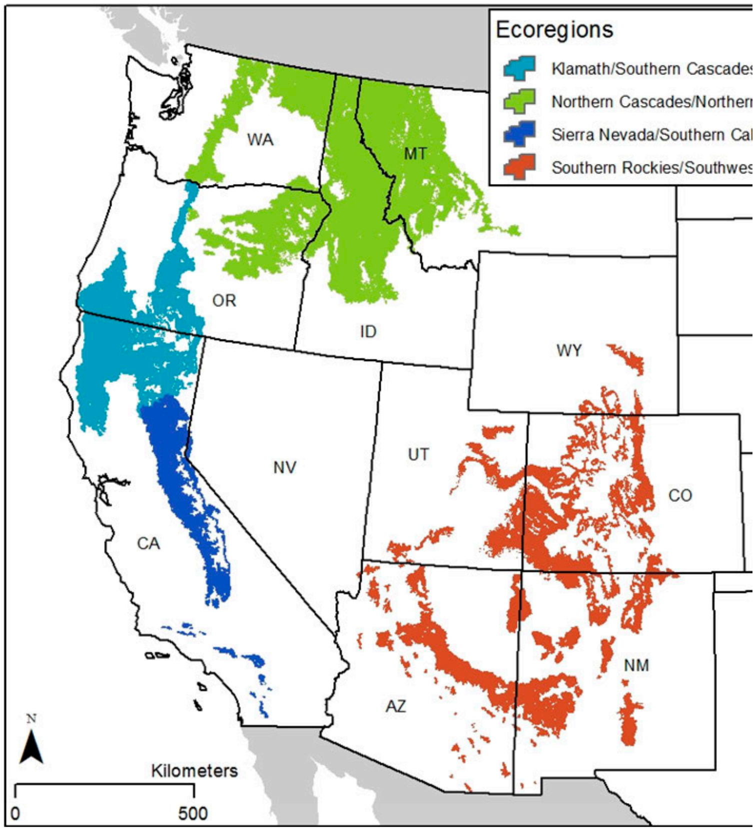

Figure 1. Ecoregions with pine and mixed conifer forests analyzed for large high-severity fire patches in our study modified from [40]. Two-letter acronyms shown on the map represent different U.S. states.

Since we were interested in determining the specific timing of any differences in occurrence in large CESF patches, we used a Nemenyi non-parametric test for multiple comparisons among groups with an equal sample size [49] to analyze whether there have been increases or decreases in large (>400 ha) patches of CESF, created by high-severity fire, for total annual area across four equal time periods (1984–1991, 1992–1999, 2000–2007, 2008–2015). To determine which specific time groups were significantly different with regard to individual patch sizes, we used a Dunn non-parametric test for multiple comparisons with unequal sample sizes [49]. In all analyses, significance was assessed at α = 0.05. We conducted this analysis because we wanted to determine whether any trend in the occurrence of large CESF patches is current and ongoing or happened at some point in the past, during the 1984–2015 time series, but may not be ongoing. This is not possible when large CESF patch occurrence is analyzed as continuous variables across the entire time series. For these two multiple comparison analyses, we chose to assess four groups of eight years each, rather than, for example, eight groups of four years each because the latter reduces sample size within each group to levels considered to be statistically inadvisable, and because using eight groups of four years increases the critical threshold to determine differences among groups, thus making it more difficult to reveal such differences when they exist [49].

In order to understand the spatiotemporal extent and context of large CESF patches across the forested landscape, we calculated fire rotation intervals [9] for high-severity fire patches >400 ha in each of four regions in the western USA: Sierra-Nevada/Southern-California, Klamath/Southern-Cascades, Northern-Cascades/Northern-Rockies, and Southern-Rockies/Southwest. The rotation interval for the occurrence of large CESF patches is equal to the average interval between occurrences of large patches across the study landscape [9].

We also analyzed the internal heterogeneity of CESF patches >400 ha in the four western USA regions by determining the percentage of the total area of such patches that was 1–100 m, 101–200 m, 201–300 m, and >300 m from the nearest unburned, low, or moderate-severity pixel (from either outside or inside the patch) within the frequent-fire conifer forest types analyzed in this study [40]. We included a specific analysis of internal heterogeneity of large high-severity patches because some authors have hypothesized that such patches would be internally homogeneous and have expressed concern about the potential for natural succession in this regard [6,20]. The distance intervals selected for this analysis were based on biologically meaningful relationships in levels of natural post-fire conifer regeneration at increasing distances from seed sources. We assumed lower levels of conifer recruitment at greater distances from live trees, consistent with natural succession to more open forest conditions [45,50–53].

Finally, although it was beyond the scope of this study to attempt to compare current versus historical rates of occurrence of large CESF patches, we included a table summarizing evidence for historical occurrence of patches >400 ha, focusing on low/middle-montane, frequent-fire forest types, given questions expressed about whether large CESF patches occurred historically in these forests [6,18,21].

3. Results

Over the entire time series, 1984–2015, there was a significant increasing trend in the combined total area of CESF patches >400 ha in each year (τ = 0.407, p = 0.001), but no trend in patch size (τ = 0.009, p = 0.802). However, when the data were analyzed by time periods, there was only one significant difference in the annual area of CESF habitat created by high-severity fire relative to the earliest time period (1984–1991), but no significant differences were detected among time periods since the early 1990s (Table 1, Figure 2). With regard to the size of individual large CESF patches, there were no significant differences detected among time periods (Table 2). Figure 3 shows the distribution of individual large CESF patches over the entire time series.

Table 1. Critical values ($q_{0.05,\,4}$), absolute difference between mean of ranks ($|R_A\text{-}R_B|$), standard errors (SE), and test statistics (q) to assess statistical significance, at α = 0.05 of any differences among the four time groups (1 = 1984–1991, 2 = 1992–1999, 3 = 2000–2007, and 4 = 2008–2015) for total annual area of CESF patches >400 ha using the Nemenyi non-parametric test for multiple comparisons between groups with an equal sample size (n = 8 years for each time group). The statistical significance of the levels of q are shown as "Y" (significant) or "N" (not significant).

Time Group Comparison	$q_{0.05,4}$	$\lvert R_A\text{-}R_B\rvert$	SE	q	Significant? (Is q > $q_{0.05,4}$?)
1–2	3.63	45.0	26.53	1.70	N
1–3	3.63	108.0	26.53	4.07	Y
1–4	3.63	107.0	26.53	4.03	Y
2–3	3.63	63.0	26.53	2.37	N
2–4	3.63	62.0	26.53	2.34	N
3–4	3.63	1.00	26.53	0.04	N

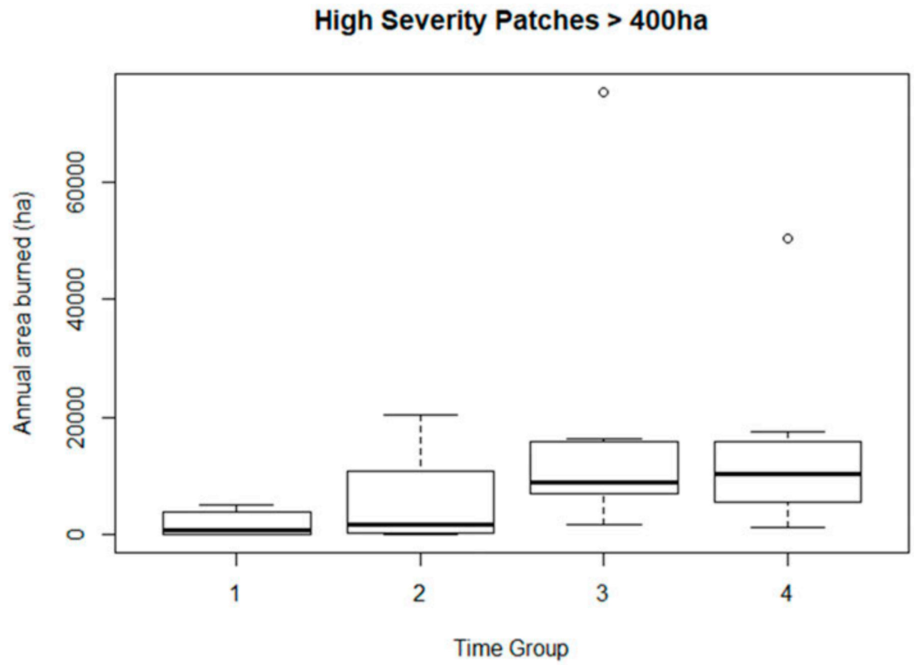

Figure 2. Annual area of large (>400 ha) CESF patches in the four time periods (see Tables 1 and 2 for time periods).

Table 2. Critical values ($q_{0.05,\,4}$), absolute difference between mean of ranks. ($|_{A-B}|$), standard errors (SE), and test statistics (Q) to assess statistical significance at α = 0.05 of any differences among the four-time groups (1 = 1984–1991, 2 = 1992–1999, 3 = 2000–2007, and 4 = 2008–2015) for the size of individual CESF patches >400 ha using the Dunn non-parametric test for multiple comparisons. The statistical significance of levels of Q are shown as "Y" (significant) or "N" (not significant). For time groups 1, 2, 3, and 4, n = 17, 46, 134, and 130 CESF patches >400 ha, respectively.

Time Group Comparison	$Q_{0.05,4}$	$\lvert_{A-B}\rvert$	SE	Q	Significant? (Is Q > $Q_{0.05,4}$?)
1–2	2.64	2.73	26.91	0.10	N
1–3	2.64	26.50	24.37	1.09	N
1–4	2.64	15.08	24.42	0.62	N
2–3	2.64	23.77	16.23	1.46	N
2–4	2.64	12.35	16.29	0.76	N
3–4	2.64	11.42	11.60	0.98	N

Over the 32-year study period, high-severity fire patches >400 ha occurred on ~0.7% to ~2.7% of the total area of frequent-fire conifer forest, depending on the region, such that the rotation intervals for occurrence of large (>400 ha) CESF patches, created by high-severity fire, ranged from 1181 years to 4354 years (Table 3).

Table 3. Total area and fire rotation interval for occurrence of CESF patches >400 ha in the four regions of the study area from 1984 to 2015.

Region	Area of Forest (ha)	Area (ha) of Patches >400 ha (% of Ecoregion)	Rotation Interval [1] (Years)
Sierra Nevada/Southern California	2,395,288	64,895 (2.709)	1181
Klamath/Southern Cascades	5,741,930	100,112 (1.744)	1835
Northern Cascades/Northern Rockies	10,057,451	73,936 (0.735)	4354
Southern Rockies/Southwest	6,956,201	72,851 (1.047)	3056

[1] Rotation intervals for high-severity patches were calculated by dividing the total area of the conifer forest by the average area of large high-severity patches per year.

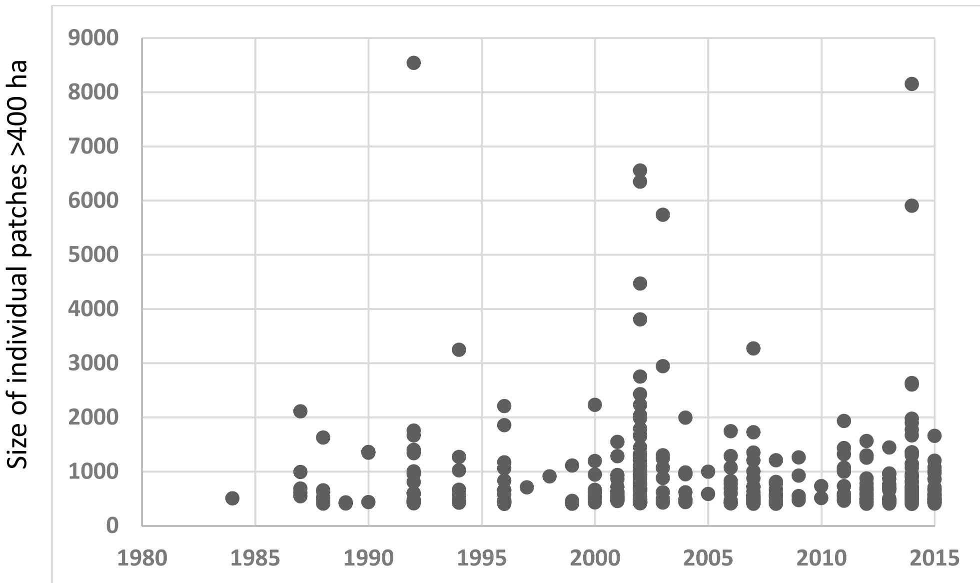

Figure 3. Scatter plot of size of individual large (>400 ha) CESF patches, 1984–2015.

Overall, 52% of the area within the boundaries of CESF patches >400 ha was within 100 m of unburned, low, or moderate-severity edges/inclusions, and 78% of the total area was within 200 m of such edges and inclusions. The results were similar in all four western USA regions (Table 4). Figure S1 is an example illustration of various distances from potential seed sources in very large (>1000 ha) high-severity patches in two areas: Rim fire 2013 (Stanislaus National Forest, Sierra Nevada, CA) and Hayman Fire 2002 (northwest Colorado Springs area).

Table 4. Percentages of the total area within the boundaries of CESF patches >400 ha, created by high-severity fire, that were at increasing distances from unburned or low/moderate-severity edges and inclusions.

Distance (m)	Sierra-Nevada/ Southern-California	Klamath/ Southern-Cascades	Northern-Cascades/ Northern-Rockies	Southern-Cascades/ Southwest
<100	49.3	55.6	46.8	54.7
101–200	27.6	25.5	25.2	26.0
201–300	13.5	11.2	12.8	10.6
>300	9.6	7.7	15.3	8.7

There is historical evidence of numerous large CESF patches created by high-severity fire prior to widespread fire suppression in every region of the western USA in low/middle-montane forests (Table 5). Historical patches >400 ha ranged from ~400 ha to >20,000 ha for our study area.

Table 5. Examples of historical occurrence of CESF patches >400 ha, created by high-severity fire, in low/middle-montane forests of the western USA [1].

Source	Region	Forest Type	Evidence Type	Patch Size/s (ha)	Time Period
[54,55]	Northern Sierra Nevada	Mixed-conifer and ponderosa pine	Historical USGS mapping, and current GIS analysis	400–~9000	19th century
[8]	Sierra Nevada	Mixed-conifer and ponderosa pine	Reconstruction, using 19th-century General Land Office data	Largest = 8050 (northern) and 9400 (southern)	19th century
[56]	Eastern Washington Cascades	Mixed-conifer	Reconstructions of past high-severity from historical aerial photos	400–10,500	19th century, and early 20th
[57]	Eastern Oregon Cascades	Mixed-conifer and ponderosa pine	Reconstruction from 19th-century General Land Office data	400–~5000	19th century
[58]	Oregon Klamath	Mostly ponderosa pine	Historical account, early 20th century U.S. Geological Survey report	~14,000	19th century
[59]	Colorado Front Range	Mostly ponderosa pine	Reconstruction from 19th-century General Land Office data	400–~22,000	19th century
[59]	Blue Mountains, Oregon	Ponderosa pine	Reconstruction from 19th-century General Land Office data	400–~12,000	19th century
[59]	Central/eastern Arizona	Ponderosa pine	Reconstruction from 19th-century General Land Office data	400–~40,000	19th century
[60]	Black Hills, South Dakota	Ponderosa pine, some lodgepole pine	Historical account	~19,000	mid-19th century
[61,62]	Northern Rockies	Ponderosa pine, some Douglas-fir	Reconstruction from historical aerial photos	~35,000	1910

[1] Some patches may have resulted from more than one fire. This represents all available data on historical occurrence of high-severity fire patches >400 ha known to currently exist within western US frequent-fire conifer forest types. For context, the largest individual high-severity fire patches in each of the four current time periods analyzed in this study are (in chronological order, by time period) 2109, 8539, 6554, and 8153 ha.

4. Discussion

Despite concerns about there being too many large CESF patches produced by big fires, we found that while an increase in the total area of such patches did occur initially in the time series, this happened over two decades ago and there has been no subsequent increase since the 1990s. We did not find an increase in the size of individual CESF patches >400 ha at any point during the time series (1984–2015)—i.e., patches >400 ha did not get significantly larger in more recent time periods. The rotation intervals for large patches ranged from about twelve centuries to over four millennia, depending on the region. A posteriori, we conducted the same analyses regarding whether there had been an increase in the area of large high-severity fire patches, but with a smaller patch size threshold (>100 ha), and we found the same result—i.e., significant differences between the first time period and the third and fourth time periods, but no other significant differences (Table S1, Figure S2). We did not conduct a posteriori analysis for patches >100 ha regarding the question of whether individual high-severity patches had been getting larger, since there were no significant or marginally significant differences with the >400 ha threshold.

Importantly, in large CESF patches, within-patch heterogeneity was high, with the great majority of patch area occurring within 200 m of the potential seed sources of unburned, low, or moderately burned conifer forest. In this regard, our findings are similar to those in the Northern US Rockies [63]. Depending on site factors, natural post-fire conifer regeneration generally occurs most quickly and abundantly within 100 m of low/moderate-severity and unburned recruitment areas, and secondarily at 100–200 m from unburned or low/moderate-severity areas [45,50–53,64]. It also occurs—typically more slowly and less densely—in the portion of large CESF patches that are >200 m from unburned or low/moderate-severity areas [51,53,64]. However, in these more distant areas, we can expect pockets of more open conifer forest or dense vegetation dominated primarily by oaks (*Quercus* spp.) and aspen (*Populus* spp.) and secondarily by conifers [51,64]. This internal patch heterogeneity indicates that large CESF patches play an important role in creating and maintaining pockets of open forest stands and increasing the heterogeneity (beta diversity) of forest structure across the landscape [64].

We also found considerable evidence of historical occurrence of large CESF patches in all regions, indicating that such patches are a component of natural fire regimes in low/middle-elevation, frequent-fire conifer forests of the western USA. More research is needed to compare current versus historical extents of such patches.

Modeling studies regarding wildland fire in western forests project overall increases [65], or more complex mixes of increases and decreases within and among regions, mediated by interactions between climate and vegetation shifts [65–67]. Thus, it will be important to continue to monitor high-severity fire occurrence and patch sizes periodically to understand any patterns that emerge in patch dynamics and conifer recruitment rates. Our findings also differ from some previous work regarding high-severity fire trends in western U.S. conifer forests. Some researchers [13,14], for instance, noted increasing trends in overall high-severity fire occurrence in mixed-conifer forests of the Sierra Nevada. Subsequent analyses [15,16] found that the use of a vegetation database by these researchers post-dated the time series being analyzed and led to an unintended omission of much of the high-severity fire in the earlier years of the time series, causing the appearance of an upward trend where no such trend existed. In other words, it was later found that the vegetation database used by these studies often did not reflect the vegetation that existed at the time of the fires analyzed, since much of the conifer forest that experienced high-severity fire in the earlier years of the time series was later reclassified as chaparral or other non-conifer vegetation—a phenomenon that occurred less for more recent fires in the time series.

Others [46,68,69] reported an increasing trend in the interior area of high-severity fire patches in the Sierra Nevada, but also used a vegetation database that post-dated the time series and omitted more of the high-severity fire in the earlier years of the time series [15]. They did not account for small low/moderate-severity inclusions within large high-severity fire patches, while inclusions of this size were common in our analyses.

Our results indicate that large CESF patches have high levels of heterogeneity (beta diversity), even within the most interior portions, which may facilitate heterogeneous natural forest regeneration in ecologically beneficial ways [25,53,55,70]. Some delayed tree mortality can, of course, occur in the years following a fire in low/moderate-severity inclusions, and this could potentially influence the internal patch complexity along with conifer seedling establishment. Yet, even in such cases, individual trees experiencing delayed mortality would provide seed source in the interim years, and research into delayed post-fire mortality indicates fairly modest levels of such occurrences in low/moderate-severity pixels [71].

Some researchers have expressed concern about type conversion to non-forest following fires, especially high-severity fires, e.g., [47,72]. Although a detailed discussion of this issue is beyond the scope of our study, we note that areas described as examples of possible post-fire type conversion nevertheless had substantial post-fire conifer regeneration, generally within the described natural range of variability for the specific forest type [72], and the areas with no regeneration occurred at the spatial scale of very small plots [47,72]. Thus, we suggest that there may be a scale-of-observation issue at work here, and much larger plots indicate more consistent post-fire regeneration [64]. Moreover, while recent research has suggested somewhat lower regeneration in more recent fires, time-since-fire was not accounted for, and far fewer years of post-fire succession had occurred at the time of field sampling in the more recent fires, which might account for the difference [47]. Nevertheless, some researchers have predicted that in a hotter and drier climate in certain areas, such as the Klamath region of northwestern California, recurrent high-severity fire could limit the recruitment of some conifer species in future decades [73]. Thus, more research is needed to address this question after taking spatial and temporal scale and time-since-fire into account.

5. Conclusions

Our findings have specific management and policy relevance. In particular, we counter claims made by some researchers, and often used by decision-makers, to justify large-scale forest "thinning" and post-fire logging projects—specifically, the assumption that such logging projects are needed to prevent type conversion in response to a perceived increase in CESF patch sizes and conifer regeneration failures in "megafires" (see [6,18,20,22]). Lack of a biodiversity perspective has created underlying tensions among researchers over the role of high-severity fires in maintaining CESF, and we hope that our findings will now inform this ongoing discussion. Additionally, contrary to assumptions made by land managers in the course of proposing extensive post-fire logging and creation of artificial tree plantations following large fires, we found ample evidence of patch heterogeneity–and presumably natural conifer establishment–in large severely burned patches, in addition to the occurrence of large high-severity patches in the historical record. This finding has key relevance to current forest management policy, since the assertion that current large CESF patches are unprecedented is not substantiated by our data but is being used to justify legislative and regulatory proposals to severely weaken environmental laws on U.S. federal lands.

Notably, numerous studies have found high levels of native plant and animal richness and abundance in large fires of mixed severity that produce CESF patches in severely burned areas, see [3,24–31,70,74,75]. Such fires facilitate high levels of beta diversity at landscape scales, providing a broad suite of habitat for both fire-seeking and fire-avoiding species [25], including many early seral birds that have been declining due to a lack of "diverse early seral habitat" [76]. Thus, far from being indicative of "catastrophic" (or "megafire") ecosystem shifts, large CESF patches have consistently been found to support a unique ecological community that is otherwise most often post-fire logged because of perceptions that this forest type has limited wildlife value, see [25,75]. Instead, we found that large CESF patches are extremely infrequent at landscape scales in ponderosa/Jeffrey-pine and mixed-conifer forests of the western U.S., and whether high-severity fire that produces this important seral stage is increasing in western USA forests remains debatable, e.g., [4,9–11,13–16,19,21,23].

Regarding the human implications of our findings, we recommend that land managers focus limited resources on community fire safety and defensible space of homes as a means of getting to coexistence with wildfire [77–79] and for managing wildfire under safe conditions for a myriad of ecosystem benefits.

Supplementary Materials: The following are available online at http://www.mdpi.com/1424-2818/11/9/157/s1: Figure S1: Example of CESF patches >1,000 ha, showing distances from areas of unburned, low, and moderate severity fire within the patch boundaries in the Rim (Stanislaus National Forest, CA) and Hayman fires (Colorado Front Range). Figure S2: Annual area of large patches (>100 ha) of CESF in the four time periods; Table S1: Critical values ($q_{0.05,\infty,4}$), absolute difference between mean of ranks ($|R_A\text{-}R_B|$), standard errors (SE), and test statistics (q) to assess statistical significance, at $\alpha = 0.05$ of any differences between the four time groups (1 = 1984–1991, 2 = 1992–1999, 3 = 2000–2007, and 4 = 2008–2015) for total annual area of CESF patches >100 ha using the Nemenyi non-parametric test for multiple comparisons between groups with an equal sample size (n = 8 years for each time group). The statistical significance of levels of q is shown as Y (significant) or N (not significant).

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An Ecologically Based Strategy for Fire and Fuels Management in National Forest Roadless Areas

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During the challenging 2000 fire season, the local and national headlines trumpeted daily news about the "worst fires in recent memory." The media showered us with the latest statistics on wildland fires in the West: "More than 6 million acres charred in 13 Western States…more than 25,000 firefighters deployed…over 80 blazes raging out of control…hundreds of homes consumed."

Amid the media frenzy, one Presidential candidate—George W. Bush—sought to improve his position in the public opinion polls by stating that greatly reduced logging levels on national forests during the previous decade had "made the forests more dangerous to fire." The implication was that the USDA Forest Service's proposed policy for protecting roadless areas was akin to putting a lit match into a tinderbox.

Others called for massive logging, roadbuilding, and a rash of prescribed fires as a quick fix for the previous 50-100 years of fire suppression. While conservationists advocated for roadless area protection on the grounds that roadless areas are the last remnants of formerly large and intact forests, critics asserted that fiery conflagrations would inevitably occur if the same forest remnants were not intensively managed. The rest of us pondered: Where is the science in all this? Is every acre doomed to “catastrophic” fire if not intensively managed? Is it appropriate to treat all forests the same, regardless of whether or not they contain existing road systems?

After all the hyperbole – a combination of media hype, electoral politics, and misinformation spread to promote special interests – it’s time to take a sober look at the questions raised by the 2000 fire season. Specifically, what evidence exists on the relationship between wildland fire and timber management in roaded vs. roadless areas? What effects might silvicultural treatments and prescribed fire have on ecosystems in roadless areas? Is there an ecologically based strategy for identifying, on a case-by-case basis, where active management might be appropriate for maintaining fire-dependent forest ecosystems?

Fire and Roadless Areas

Level of Fire Hazard. Scientists widely agree that protecting roadless areas on the national forests from roadbuilding, logging, and other forms of development will greatly enhance biodiversity and ecosystem conservation (Ercelawn 1999; Henjum and others 1994; Noss and Cooperider 1994; Strittholt and DellaSala [in press]). However, some critics of roadless area protection (Bernton 1999; Hansen 1999; Schlarbaum 1999) have repeatedly made two assertions:

- Road building prohibitions in roadless areas will restrict access and timber management, which in turn will increase the frequency of large, intense fires.
- Widespread silvicultural treatments (such as low thinning and crown thinning) in roadless areas will be necessary to reduce the fire hazard.

Does the relevant scientific literature support these claims?

Broad scientific assessments were completed in 1996 and 1997, respectively, for Federal lands in the Sierra Nevada in California and the Interior Columbia River Basin in portions of Idaho, Montana, Nevada, Oregon, Washington, and Wyoming. These studies provide the most comprehensive analysis to date for comparing fire, fuel, and vegetation conditions in intensively managed areas to conditions in roadless areas. Both assessments found the fire hazard to be significantly higher in intensively managed areas.

According to the Sierra Nevada assessment, "Timber harvest, through its effects on forest structure, local microclimate and fuel accumulation, has increased fire severity more than any other recent human activity" (SNEP 1996). The Interior Columbia Basin assessment similarly concluded that "fires in unroaded areas are not as severe as in roaded areas because of less surface fuel….Many of the fires in the unroaded areas produce a forest structure that is consistent with the fire regime, while the fires in the roaded areas commonly produce a forest structure that is not in sync with the fire regime. Fires in the roaded areas are more intense, due to drier conditions, wind zones on the foothill/valley interface, high surface-fuel loading, and dense stands" (Hann and others 1997).

Even within the forest types most altered as a result of fire suppression (such as dry forests with a regime of frequent low-intensity fires), intensively managed forests on federal lands in the Interior Columbia Basin are denser and carry higher fuel loads than do roadless areas. Accordingly, intensively managed lands were found to be at higher risk of tree mortality from fire, insects, disease, and other disturbance agents (Hann and others 1997).

Others have reported similar findings for portions of the interior West. In the Sierra Nevada, McKelvey and others (1996) and Weatherspoon (1996) identified timber harvest as the single most important factor responsible for an increase in potential fire severity. In the Klamath Mountains of northwestern California, Weatherspoon and Skinner (1995) found that partial-cut stands with fuels treatment (lop and scatter or broadcast burning) burned more intensely and suffered higher levels of tree mortality than adjacent areas left uncut and untreated. Fire and fuel models also suggest that mechanical treatments alone, including silvicultural thinning and biomass removal, are not likely to be effective at reducing fire severity in dense stands (van Wagtendonk 1996).

In eastern Oregon and Washington, Lehmkuhl and others (1995) and Huff and others (1995) reported a positive correlation between logging, on the one hand, and fuel loadings and predicted flame lengths, on the other hand. They attributed the increased fire hazard in intensively managed areas to leftover slash fuels from tree removal activities (including thinning) and to the creation of dense, early-successional stands through overstory removal. A postfire study of the effectiveness of fuels treatments (including thinning) on previously nonharvested lands on the Wenatchee National Forest in Washington found that harvest treatments likely exacerbated fire damage (USDA Forest Service 1995).

Overall, the scientific literature shows that forests in areas without roads are less altered from historical conditions and present a lower fire hazard than forests in intensively managed areas, for three reasons:

1. Timber management activities often increase fuel loads and reduce a forest's resilience to fire.
2. Areas without roads have been less influenced by fire suppression than intensively managed lands.
3. Widespread road access associated with intensively managed lands raises the risk of human-caused ignitions.

As summarized in a recent review of national forest management organized by the Ecological Society of America, "There is no evidence to suggest that natural forests or reserves are more vulnerable to disturbances such as wildfire than intensively managed forest stands. Indeed, there is considerable evidence to the contrary, evidence that natural forests are actually more resistant to many types of both small- and large-scale disturbances" (Aber and others 2000). Assertions about increased wildfire made by critics of roadless area protection are not based in fact, as the evidence is clear that the forests most in need of fuels treatment are not roadless areas but areas that have already been roaded and logged, "where significant investments have already been made" (USDA/USDI 1997).

Effectiveness of Fire Suppression. Some evidence exists that fire suppression activities have had a lower impact on roadless areas than on roaded portions of the national forests (Hann and others 1997; SNEP 1996). The lower impact may be attributable to limited access and steep terrain, which prevent the application of large, ground-based suppression strategies in roadless areas (Agee 1993; Fuller 1991; Pyne 1996; Schroeder and Buck 1970).

Fires in roadless areas tend to be more remote from human habitations than are fires on roaded lands. Accordingly, they are often the lowest priority for suppression during years when firefighting resources are in short supply. Although data are limited, findings from the Interior Columbia Basin assessment on this topic might apply to other regions as well. The assessment concluded that a "combination of past harvest practices and more effective fire suppression moved the roaded landscapes much further from their unaltered biophysical templates, as measured by dominant species, structures, and patterns, relative to unroaded areas….In general, all forests which show the most change from their historical condition are those that have been roaded and harvested" (Hann and others 1997). Furthermore, the forests that are most susceptible to moisture stress, insects, disease, and unnaturally intense fire tend to be at the lowest elevations, which typically border private, state, tribal, or other landownerships (Everett and others 1994).

Another reason why fire suppression has had less impact on forests in roadless areas is associated with differences in vegetation and fire regimes. Most roadless areas on the national forests, particularly in the interior West, are at mid- to high-elevations (Beschta and others 1995; Henjum and others 1994; Merrill and others 1995). The exceptions are in the Eastern United States, where elevational gradients are limited, and the Klamath–Siskiyou ecoregion in northwest California and southwest Oregon, where very steep slopes at lower elevations have limited road construction (Strittholt and DellaSala [in press]).

Higher elevations are cooler, receive more moisture, and have a shorter summer dry season than lower elevations. They are typically characterized by a regime of low frequency, high-intensity fires (Agee 1993; Baker 1989; van Wagner 1983). Roadless areas are therefore less likely to have current fire regimes that are significantly different from historical conditions (Agee 1997; Beschta and others 1995).

For fires in high-elevation forests, weather rather than fuels is often the primary variable determining fire severity and extent (Agee 1997; Bessie and Johnson 1995; Flannigan and Harrington

1988; Johnson and Wowchuck 1993; Turner and others 1994). Under severe fire weather, the efficacy of fire suppression decreases dramatically in forest types characterized by high-intensity fires (Agee 1998, SNEP 1996). Even substantial investments of financial and human firefighting resources often fail to control large fires; they are extinguished only when the weather changes (Romme and Despain 1989).

Risk of Human-Caused Ignitions. Roadless areas have a lower potential for high-intensity fires than roaded areas partly because they are less prone to human-caused ignitions (DellaSala and others 1995; USDA Forest Service 2000; Weatherspoon and Skinner 1996). Roads constructed for timber management and other activities provide unregulated motorized access to most national forestlands and are heavily used by the general public.

In the Western United States, many of the more than 378,000 miles of national forest roads traverse heavily managed forests with the greatest potential for high-severity fire. According to the Forest Service, more than 90 percent of wildland fires are the result of human activity, and ignitions are almost twice as likely to occur in roaded areas as they are in roadless areas (USDA Forest Service 1998, 2000). While it can be argued that roads provide improved access for fire suppression, this benefit is more than offset by much lower probabilities of fire starts in roadless areas.

The Case Against Mechanical Fuels Treatments in Roadless Areas

Some land managers and policy makers advocate the widespread use of silvicultural treatments (often mechanical thinning of merchantable trees) in western roadless areas to reduce fuel loads and tree stocking levels and thereby decrease the probability of large, intense fires. Although thinning has long been a part of intensive forest management, its efficacy as a tool for fire hazard reduction at the landscape scale is controversial, largely unsubstantiated, and fundamentally experimental in nature (DellaSala and others 1995; FEMAT 1993; Henjum and others 1994; SNEP 1996; USDA Forest Service 2000).

Few empirical studies have tested the relationship, even on a limited basis, between thinning or other fuels treatments and fire behavior. These studies, supported by anecdotal information and the analysis of recent fires, suggest that thinning treatments have highly variable results. In some instances, thinning intended to reduce the fire hazard appeared to have the opposite effect (Huff and others 1995; van Wagtendonk 1996; Weatherspoon 1996). Thinning might reduce fuel loads, but it also allows more solar radiation and wind to reach the forest floor. The net effect is usually reduced fuel moisture and increased flammability (Agee 1997; Countryman 1955).

Moreover, mechanical treatments fail to mimic the ecological effects of fire, such as soil heating, nutrient cycling, and altering forest community structure (Chang 1996; DellaSala and others 1995; Weatherspoon and Skinner 1999). In fact, according to the SNEP (1996), "although silvicultural treatments can mimic the effects of fire on structural patterns of woody vegetation, virtually no data exist on their ability to mimic the ecological functions of natural fire. Silvicultural treatments can create patterns of woody vegetation that appear similar to those that fire would create, but the consequences for nutrient cycling, hydrology, seed scarification, non-woody vegetation response, plant diversity, disease and insect infestation, and genetic diversity are almost unknown."

Although our current understanding of the ecological effects of thinning is incomplete, evidence indicates that mechanical treatments, even when carefully conducted, can have additional environmental impacts:

- Damage to soil integrity through increased erosion, compaction, and loss of litter layer (Harvey and others 1994; Meurisse and Geist 1994);
- Increased mortality of residual trees due to pathogens and mechanical damage to boles and roots (Filip 1994; Hagle and Schmitz 1993);
- Creation of sediment that might degrade streams (Beschta 1978; Grant and Wolff 1991);
- Increasing levels of fine fuels and near-term fire hazard (Fahnestock 1968; Huff and others 1995; Weatherspoon 1996; Wilson and Dell 1971);
- Disruption of mycorrhizal fungi – plant relationships that are important to ecosystem function – and shrubs and perennial native bunchgrasses involved in fungal linkages (Amaranthus and Perry 1994, Massicotte and others 1999, pers. comm. D. Southwort and L. Valentine, Southern Oregon University);

- Dependence on roads, which have numerous adverse effects of their own (Henjum and others 1994; Megahan and others 1994); and
- Reduced habitat quality for sensitive species associated with cool, moist microsites or closed-canopy forests (FEMAT 1993; Thomas and others 1993).

These adverse impacts of mechanical treatments should be of particular concern in managing roadless areas, where ecological values are especially high. Moreover, roadless areas are often in steep, unstable terrain that is highly sensitive to human disturbance (Henjum and others 1994; Wilderness Society 1993). According to the Forest Ecosystem Management Assessment Team, most existing roadless areas "are considered inoperable because timber harvest and road construction would result in irretrievable loss of soil productivity and other watershed values. These lands consist of erosion- and landslide-prone landforms such as inner gorges, unstable portions of slump earthflow deposits, deeply weathered and dissected weak rocks, and headwalls" (FEMAT 1993).

Similarly, the Interior Columbia Basin assessment found "a high risk to watershed capabilities from further road development in these [roadless] areas. In general, the effects of wildfires in these areas are much lower and do not result in the chronic sediment delivery hazards exhibited in areas that have been roaded. In contrast, the already roaded areas have high potential for restoration action" (USDA/USDI 1997). Given the potential for adverse impacts from silvicultural treatments in roadless areas, many scientists recommend limiting experimental treatments to previously managed lands already degraded by fire suppression and logging (Aber and others 2000, Beschta and others 1995; DellaSala and others 1995; Franklin and others 1997; Hann and others 1997; Henjum and others 1994; McKelvey and others 1996; Perry 1995).

In summary, scientific assessments of federal lands in several western regions generally conclude that previously roaded and logged areas should be the highest priority for fuels reduction and forest restoration treatments (FEMAT 1993; Hann and others 1997; SNEP 1996). Silviculture has a role to play in a scientifically based approach to fire and fuel management on federal lands, but current evidence indicates that widespread mechanical treatments in roadless areas would most likely increase rather than decrease ecosystem degradation. Therefore, experimentation with mechanical treatments for fire hazard reduction should proceed primarily in areas with road access and adjacent to private lands where the ecological risks are lower and the threat of fire to human lives and property is far greater.

Roadless areas should only be considered for mechanical treatment after all other, higher priority areas are addressed and only if it can be demonstrated that such treatments will not degrade ecological values. Any experimental treatments in roadless areas should occur in small roadless areas (less than 5,000 acres (2,000 ha)) that have relatively good access, are near the wildland-rural interface, and exhibit high fire hazard due to past suppression. Only small trees (generally less than 12" diameter) should be considered for removal and under no circumstances should new or temporary roads be built to conduct mechanical treatments.

The Case for Prescribed Fire in Roadless Areas

The Forest Service should treat roadless areas primarily by reintroducing fire, both natural and prescribed. Restoration of ecological processes is key to ecosystem integrity and biological diversity (Samson and Knopf 1993), particularly in unroaded areas. Use of prescribed fire has been successful in restoring wildland fire regimes to many fire-adapted ecosystems (Wright and Bailey 1982), and a widespread consensus exists that additional burning is necessary (Arno 1996; Mutch 1994, 1997; USDA/USDI 1995; Walstad and others 1990).

Prescribed fire has important advantages over mechanical treatments in areas where ecological integrity and biodiversity conservation are important management objectives (Hann and others 1997; SNEP 1996; Weatherspoon and others 1992). Prescribed fire also appears to be the most effective treatment for reducing fire severity and rate of spread (Stephens 1998; van Wagtendonk 1996). In addition to reducing fuel loading and continuity, prescribed fire may decrease pest outbreaks, provide germination sites for shade-intolerant species, release nutrients, and create wildlife habitat (Agee 1993; Biswell 1999; Chang 1996; Walstad and others 1990).

Positive outcomes associated with prescribed fire are, of course, contingent on detailed site-specific planning, adequate budgetary support, and careful execution by trained personnel. In roadless areas with forests characterized by low-intensity, high-frequency fire regimes, repeated prescribed burns within a relatively short timeframe might be required to sufficiently reduce fuels and ensure that fire intensities remain within an acceptable range (Biswell 1999). After initial treatment, the frequency of prescribed burns can be designed to reflect the inherent disturbance regime and range of variability associated with particular forests. Data from the Sierra Nevada suggest that prescribed burning is likely to be considerably cheaper for treating fuels than either mechanical treatments or fire suppression (Husari and McKelvey 1996; see Deeming (1990) for a summary of the literature on the cost-effectiveness of prescribed burning versus other fuel treatments).

In addition to prescribed fire, ecological benefits could flow from allowing some naturally ignited fires to burn in roadless areas under specific environmental conditions. Traditionally, the Forest Service has suppressed most wildland fires without adequately considering the potential resource benefits of a "confine-and-contain" strategy. However, Federal policies introduced in 1995 encourage careful management of naturally ignited wildland fires if they meet resource objectives and are consistent with historical fire regimes (USDA/USDI 1995). Less than full control strategies for fire suppression could be employed, provided the strategy chosen is projected to incur the least cost of suppression and the least loss of resource values (McKelvey and others 1996).

Carefully planned wildland fire use should be fully considered for roadless areas, based on fire regime, expected fire behavior, and other variables, as an alternative to costly firefighting in remote areas where there is little or no danger to lives and property. In 2000, the Forest Service spent more than $91 million fighting two large fires in Idaho, the Burgdorf Junction Fire and the Clear Creek Complex Fire. Together, the fires burned more than 280,000 acres, mostly in remote roadless and wilderness areas (Morrison and others 2000; NIFC 2000a). On such fires, wildland fire is likely to be the most sensible as well as ecologically appropriate strategy.

Roadless areas could instead benefit from proactive fuels management using fire. Fire management in roadless areas should be based on (1) a standard set of guidelines for identifying and prioritizing roadless areas based on their fire hazard and risk at the national or regional level (see sidebar); and (2) a subsequent step-down process for planning fire treatments at the local level, designed to allow fire to play a more important role while minimizing risks to ecological values.

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Integrated Management Strategies are Needed

Roadless areas do not exist in isolation from other land designations. It follows that an effective fire and fuel management strategy should be developed at the landscape scale. This means first identifying areas of highest priority for fire/fuels treatments and then planning treatments that are consistent with management standards to ensure protection of soil, water, wildlife and other ecological values. For roadless areas, high-priority treatment areas should first be identified at the national and regional scale. Then site-specific burn plans can be developed for individual roadless areas, or for complexes of areas, by integrating spatial information on fire hazard (fuel load, fuel continuity, and topography); fire risk (ignition history and weather); and ecosystem values (old-growth forests, wildlife habitat, and sensitive watersheds) (Agee 1995; Bunting 1996; Crutzen and Goldhammer 1993; Johnson and others 1997; Weatherspoon and Skinner 1996). By employing this kind of tiered prioritization, limited resources can be directed to areas that are most in need of fire and fuels reduction.

Over time, as fire is reintroduced into roadless areas – coupled with fire and other fuels treatments on adjacent, intensively managed lands – the occurrence of large, high-intensity wildland fires might become of less concern. In rare cases, limited low thinning (removal of small understory trees) may be appropriate in some roadless areas as a prerequisite for prescribed fire. However, more experimentation and research on the efficacy of mechanical treatments should first be conducted in intensively managed forests before broadly applying them to roadless areas. Such a cautious approach is warranted, given that a mere 4 percent of roadless lands present a high fire hazard; the vast majority of areas at risk of uncharacteristically intense fire are in the intensively managed, roaded landscape (USDA Forest Service 2000).

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Although much can be done to reduce fire hazards, there is no “magic bullet” to reverse many decades of fire suppression activities. Despite our best intentions, the fire situation may yet worsen as more homeowners build cabins deeper into fire-prone forests and climate change potentially produces hotter and drier conditions in some areas. Moreover, it is important to note that despite all the media hype, the 2000 fire season was relatively light by historical standards: In the 1930’s, more than 39 million acres (15.6 million ha) burned on average each year (NIFC 2000b).

The strategy outlined here is consistent with the Clinton Administration’s recent policy recommendations that emphasize treatment of the highest priority areas first in non-controversial areas – the wildland-rural interface and designated municipal watersheds (Council on Environmental Quality 2000). To ensure that current fire management policy avoids ecological risks associated with the logging of large trees and other ecosystem values, we recommend that thinning in priority areas target only the removal of small, non-commercial material that has most likely increased as a result of fire exclusion and is of greatest concern for hazardous fuel reduction. This is consistent with Chief Dombeck’s letter (5/23/00 file code 1500) to Senator Bingaman emphasizing that emergency appropriations be used to remove small trees <12 inch dbh (30 cm) from priority areas.

In contrast, timber industry representatives such as Butch Bernhardt of the Western Wood Products Association insist that “cutting some larger trees” is “the incentive” needed to “markedly improve forest health” by allowing “more sunlight and nutrients to reach the remaining growth” (Associated Press 2000). Commercial harvest is designed for profit, not to address ecological need; the timber industry’s claims to the contrary are inconsistent with the available science on fire and fuels management. Only through an integrated approach that emphasizes protection of roadless values and focuses treatment where it is most needed – in the roaded landscape – are we likely to make significant progress in restoring the resiliency of western forest ecosystems.

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[DESIGNER: Please set up the following as the first sidebar.]

Prioritizing Roadless Areas for Prescribed Fire

Land managers need a comprehensive set of criteria for prioritizing roadless areas for prescribed fire treatments. The following list provides a preliminary guidepost for determining high-priority areas for treatment. Prescribed fire should be considered for roadless areas where:

- Most of the area is covered by dry forest types that are characterized by low-intensity, high-frequency fire regimes;
- A long interval has passed since the last major fire (for example, more than three natural fire cycles have been missed);
- The topographic and elevational gradients are relatively gentle, permitting relatively low-risk prescribed fire treatments and raising the likelihood that past firefighting efforts have increased the fire hazard;
- Areas of high fire risk are nearby, such as the wildland–rural interface, major population centers, transportation routes, or residential developments and other infrastructure; and
- Ecological risk factors are absent or low, such as—
  - Populations of threatened and endangered species or rare communities that are known to be adversely affected by fire;
  - Vegetation changes that would predictably result from fire treatments; or
  - Fish refugia where burning could impair hydrological processes or degrade critical fish habitat through sedimentation.

[DESIGNER: Please set up the following as the second sidebar.]

Principles for Fire and Fuels Management

Land managers need a comprehensive, landscape-level strategy for fire/fuels management that takes into account the important values associated with roadless areas and directs treatments where they are needed the most. The strategy should be based on the following principles:

- Limit mechanical treatments to high-priority areas, primarily roaded areas of dense, dry forest within the wildland–rural interface.
- Define the wildland–rural interface by treating areas immediately adjacent to rural settlements as a first line of defense. Provide homeowners with assistance grants to reduce the fire hazard on private land by creating a defensible space around homes.
- Conduct watershed or landscape-scale assessments that identify restoration priorities before fire/fuel treatments are initiated.
- Eliminate commercial incentives for mechanical removal of merchantable trees by decoupling goods from services (that is, pay a fixed fee for tree removal services that is not tied to timber volume).
- Restrict thinning to small-diameter trees (e.g., less than 12 inches (30 cm) in diameter at breast height or less than the average stand diameter) where it can be demonstrated that current forest stand densities are outside the historical range of variability.
- Minimize impacts to soils, below-ground processes and related species, accumulation of surface fuels from thinning, and exposure to solar radiation and reduction of soil moisture retention.
- Conduct mechanical treatments in priority areas in compliance with all relevant environmental statutes (e.g., National Environmental Policy Act, National Forest Management Act, Endangered Species Act, etc).

FEATURE

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Roadless areas and clean water

Dominick A. DellaSala, James R. Karr, and David M. Olson

Clean water, like biodiversity, is most closely linked to undisturbed natural ecosystems. When undisturbed watersheds in roadless and protected areas (e.g., national parks, state parks, wilderness areas, national monuments) are fragmented by roads, logging, and intensive recreation development, both water quality and biodiversity decline as hydrological integrity is lost (USFS 1972, 1979, 2001; Alexander and Gorte 2008; Anderson 2008). In the United States, inventoried roadless areas (IRAs) are lands without roads exceeding 2,000 ha (5,000 ac) that have been inventoried by the USDA Forest Service. IRAs collectively amount to approximately one third of the 77 million ha (193 million ac) of the 155 national forests but are disproportionately concentrated in western states (figure 1) (Trout Unlimited 2004; Anderson 2008). The roaded, intensively managed landscapes of the other national forest lands have been closely correlated with heavily sediment-laden streams and dramatic changes in flow regimes (Espinosa et al. 1997; Trombulak and Frissell 2000; CBD et al. 2001; Coffin 2007; Frissell and Carnefix 2007). While the biodiversity benefits of IRAs are well documented (DeVelice and Martin 2001; Strittholt and DellaSala 2001; Loucks et al. 2003; Strittholt et al. 2004; Gelbardi and Harrison 2005), little has been made of the importance of IRA water for downstream users and wildlife.

In this paper, we assess the importance of IRAs from a water quality perspective, including the likely water quality effects of developing IRAs. We provide conservative estimates of the economic impact of intact unroaded watersheds on national forests for clean water and associated water resource benefits. In particular, rising demand and shrinking water supply associated with changing climate will likely make intact areas in drought-prone regions of the West even more valuable and crucial to protect. Thus, our findings are especially relevant to drought-prone states considering development of IRAs. The state of Colorado, for example, with approximately 1.7 million ha (4.2 million ac) of IRAs, has been seeking federal permission to develop its IRAs for logging, expanding ski areas, coal-bed mining, and producing oil and gas (figure 2) (Anderson 2008; Colorado Division of Wildlife 2010; Colorado, State of 2010; Straub 2010, USFS 2011). Although we focus on IRAs throughout the western United States, we also emphasize the importance of uninventoried roadless areas (unroaded) <2,000 ha (Henjum et al. 1994; Greenwald 1998; Beschta et al. 2004) that collectively cover an area roughly 1.5 times that of the total IRA network (USFS 2000; Strittholt et al. 2004). Those smaller unroaded areas also play a strategic role in maintaining reliable supplies of high-quality water and protecting aquatic ecosystems.

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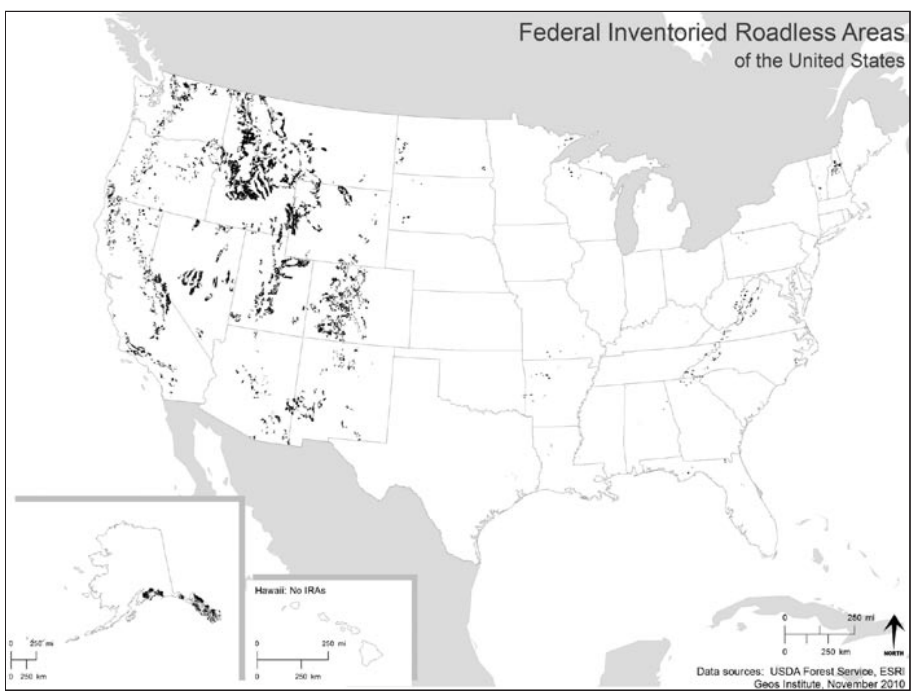
ROADLESS AREAS PROVIDE SUBSTANTIAL WATER RESOURCE BENEFITS

IRAs benefit society in many ways, including providing a valuable and increasingly rare natural supply of abundant, clean, and naturally reliable water (Sedell et al. 2000); affordable drinking water for municipal and rural communities; water for agricultural and industrial uses; flood control; in-stream aquatic recreation; aquifer recharge; flood protection; reliable water supply; diverse and productive fisheries; healthy aquatic ecosystems; resident and migratory waterfowl habitat; recovery of endangered species; and, increasingly, the vitality and sustainability of local economies (table 1). These benefits accrue nationally and at the local and regional levels.

National Benefits of Clean Roadless-Area Water. At least 124 million Americans directly benefit from water

Figure 1

Federal inventoried roadless areas (IRAs) of the United States (Source: USDA Forest Service).



originating from national forests (Sedell et al. 2000). In fact, national forests provide about 15% of the nation's runoff with an estimated net value of $3.7 (Sedell et al. 2000) to $27 billion (Krieger 2001). The water treatment value alone of National Forests ranges from $490 million (Loomis 2005) to $18 billion (Krieger 2001).

Because IRAs represent roughly a third of national forestland, by inference they contribute significantly to the overall runoff volume and value (Anderson 1997, 2008) estimated in billions of dollars annually (Loomis and Richardson 2001; Sechhi et al. 2005). For instance, using Forest Service data (USFS 2000), IRAs make up 661 of the 914 national forest watersheds, with 55% of the 914 watersheds acting as source areas for facilities that treat and distribute drinking water to the public. The cost-savings to water treatment plants and highway departments from avoiding sedimentation caused by logging in IRA watersheds is estimated at up to $18 billion annually (Loomis 1988). IRAs provide $490 million annually in waste treat-

Figure 2

Colorado's 2001 inventoried roadless areas (IRAs) are shown in light gray, the 2011 proposed Colorado roadless areas (CRAs) are shown in gray, and overlap between CRAs and IRAs is shown in black. Water quality will be most impacted by changes of allowable activities within existing IRAs relative to changes in designated areas (USFS 2011).

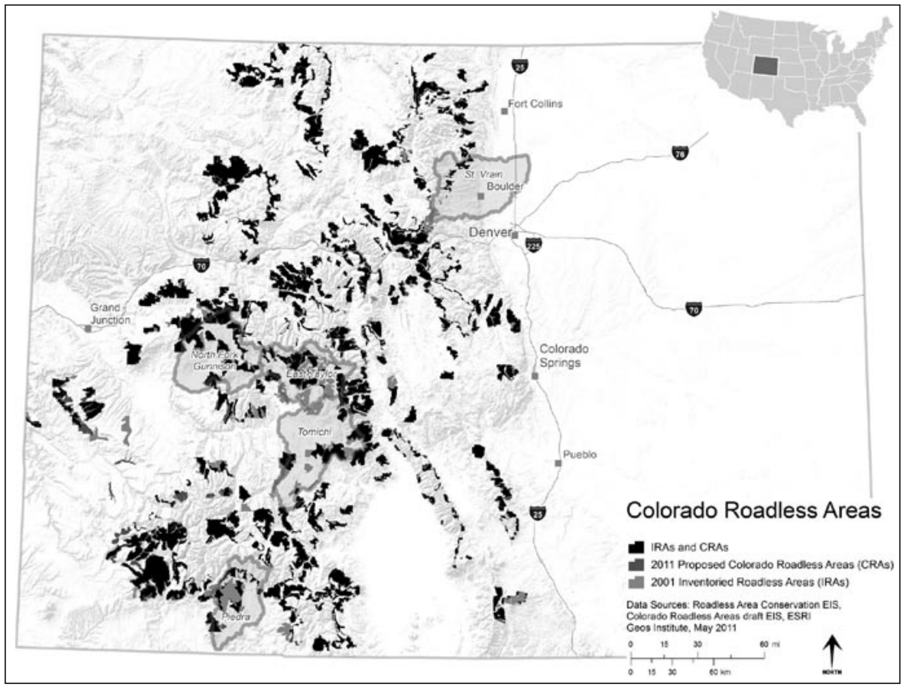


Table 1

General ecosystem services and benefits related to water that are provided by undisturbed IRAs and watersheds (derived from Greenway 1996; Costanza et al. 1997; Talberth and Moskowitz 1999; GAO 2000;Heal 2000, Loomis and Richardson 2001; Sedell et al. 2000; Krieger 2001; Dombeck 2003; Berrens et al. 2006).

	Benefits
Off-stream benefits	Low treatment costs for water for all beneficiaries
	Low price per unit volume costs for water for all beneficiaries
	High-quality and abundant drinking water for rural communities and municipal water supplies
	High-quality water for agricultural and industrial purposes
	High-quality water for downstream livestock production
	High-quality water for reduced health care and epidemic control
	Reduced costs of flood damage and flood control; enhanced local economies and property values
	Community benefits, including jobs, income, favorable trends for key economic indicators, and economic sustainability and stability
	Recharging of groundwater aquifers
	Healthy terrestrial and riparian ecosystems and their component species, sustained ecological and evolutionary processes, and resilient ecosystems
In-stream benefits	Healthy aquatic ecosystems
	Recovery of endangered species and protection of refugia
	Diverse and productive fisheries
	High-quality habitat for wildlife, including migratory waterfowl and game and nongame species
	Aquatic recreation such as swimming, rafting, and boating; enhancement of hiking and camping
	The inherent value of wild rivers and wilderness (including passive use benefits such as option, bequest, and existence values)
	Moderation of runoff and streamflows (e.g., lower peak flows, higher low flows, year-round water)
	Soil stabilization and erosion control
	Scientific value (intact watersheds are very rare today)
	Maintaining sediment production to streams at normal background rates
	Reducing potential for damage to downstream properties and water users during periods of high flow
	Breakdown and containment of waste and toxins (e.g., atmospheric, prior use)

ment services through recovering mobile nutrients and cleansing the environment, both processes that involve water flow through intact watersheds (Loomis and Richardson 2001).

Regional Benefits of Clean Roadless-Area Water. In the US Rocky Mountains, roughly one third of utilized streamflow is derived directly from IRAs (which cover a quarter of Colorado's headwaters), with cities like Denver receiving about 30% of their water supply from IRA watersheds. Annually, IRAs in Colorado are estimated to provide an equivalent of nearly 2.5 times Denver's annual water use (Doyle and Gardner 2010; Denver Water 2010). Similarly, IRAs in New Mexico provide an estimated water quality benefit up to \$42 million annually (Berrens et al. 2006).

Flood Control Protection and Inventoried Roadless Areas. The intact watersheds of IRAs are especially important for ameliorating the frequency and intensity of flooding, saving millions of dollars annually from averted floods and associated sedimentation, a service that will only increase in value as climate change drives more floods (Seeds 2010). Dredging reservoirs to increase capacity and channels to enable navigation costs cities, states, and ultimately taxpayers millions annually. Salem, Oregon, spent approximately \$100 million on new treatment facilities after logging in upper watersheds created conditions leading to mass sedimentation in its watershed following storms in 1996 (Schwickert and Mauldin 1997; Talberth and Moskowitz 1999). In addition, Seattle, Washington, deferred a \$150 million filtration plant expenditure through an intensive watershed rehabilitation program that will decommission 480 km (300 mi) of roads over a 10-year period, fix road erosion problems, and limit access and high-risk activities for fire and sedimentation within their watersheds (Seeds 2010).

Recreation Benefits and Strong Local Economies. IRA water benefits outdoor recreation and the people that either engage in or earn their living from outdoor recreation. The nation's IRAs generate \$600 million annually from recreation (Loomis and Richardson 2001). Passive-use values (i.e., the intrinsic value of wilderness, wildlands, and benefits for the future) are estimated at an additional \$280 million annually. At the regional scale, New Mexico IRA water provides an estimated \$27 million active outdoor recreation benefit and a \$14 million passive-use benefit annually (Berrens et al. 2006). For many visitors, much of the attraction to wildlands is associated with the presence of clean and abundant water—a dwindling resource as logging, grazing, and road-building continues across mountain landscapes and droughts from a changing climate intensify in much of the West (Saunders et al. 2008).

Freshwater Biodiversity and Healthy Fisheries. Clean water from IRAs also maintains healthy fisheries, such as salmon and trout fisheries, sustains viable aquatic ecosystems, and helps protect threatened species and ecosystems (Abell et al. 2000; Trout Unlimited 2004). Indeed, IRAs may act as important refugia for many salmon and trout populations, as well as for a diversity of endangered freshwater species (Henjum et al. 1994; Huntington 1998; NRC 1996; Trombulak and Frissell 2000; CBD et al. 2001; Strittholt and DellaSala 2001; Oechsli and Frissell 2002; Strittholt et al. 2004; Petersen 2005). Restoration of salmon and trout fisheries in places with high road densities will likely fail without the pivotal role provided by IRAs as fishery strongholds.

ROADLESS AREAS ARE IMPORTANT SOURCES FOR DRINKING WATER

The distribution of IRAs across prime hydrologic real estate—headwaters and upper watersheds—makes them particularly valuable for providing reliable supplies of clean water. In Colorado, IRAs occur in the headwaters of all major drainages, covering roughly a third of upper watersheds in the state. Indeed, most IRAs are located in mountainous terrain in western states, including Oregon, Idaho, New Mexico, Utah, Montana, California, and Washington. This extensive coverage of IRAs in headwaters, and because they are often the last minimally disturbed watersheds within larger landscapes of degraded lands, makes them hydrologic hotspots—areas with relatively small spatial extent that have a disproportionately important role in producing abundant and reliable clean water (Frissell and Carnefix 2007).

For many major drainages (entire watersheds of major rivers, such as the Columbia River Basin), IRAs and other wilderness areas represent the last few percentages (typically 1% to 5%) of the landscape with a minimally disturbed, or near natural, hydrology. As in many other ecological contexts, losing the last relatively natural systems typically results in major losses in water resource benefits, losses that can only be compensated by very expensive actions. The known relationship between watershed degradation and water quality decline deserves to be more rigorously incorporated as a central foundation for decisions on watershed management and protection.

Developing Roadless Areas Degrades Water Quality. In addition to their keystone location within watersheds, roadless areas typically encompass the most fragile of natural landscapes—montane forests and meadows. Road building and other intensive management in these otherwise intact areas damage their ability to provide clean water for downstream communities and biodiversity over both short and long terms (Beschta 1978; Forman and Alexander 1998; Lugo and Gucinski 2000; Trombulak and Frissell 2000; Gucinski et al. 2001; Coffin 2007). Logging, including post-disturbance, fire-risk reduction, forest health, and insect control; livestock grazing; mining; and road building are responsible for chronic and acute sedimentation of aquatic ecosystems, alter overland flow and stream structure, and change a range of physical and biological features by causing more frequent and intense floods, decreasing available water throughout the year, increasing stream and ambient temperatures, and elevating turbidity and nutrient levels (Beschta 1978; Fleischner 1994; Trombulak and Frissell 2000; DellaSala et al. 2006; Coffin 2007). Logging roads have been linked to great increases in erosion rates and sediment delivery to streams—up to 850% over rates in undisturbed habitat—with long-term and often catastrophic impacts on stream biota, aquatic ecosystems, and water quality (Fredricksen 1970; Megahan and Kidd 1972; Amaranthus et al. 1985; Bilby et al. 1989; King 1989, 1993; Haynes and Horne 1997; Jones et al. 2000; Wemple and Jones 2003).

Depending on severity and duration of impacts, disturbance can elevate average turbidity levels well above background levels (Seeds [2010] provides examples from Oregon), along with triggering more frequent and intense turbidity spikes that are a major source of excess costs to municipal water supply departments. Relative to roadless watersheds with intact natural vegetation, intensively managed watersheds also produce less available water (i.e., average monthly usable raw water) due to intensified high flows with very high turbidity and exacerbated low flow conditions (Seeds 2010). The monthly reliability of water is also diminished.

Even small disturbances in upper watersheds can result in significant, cumulative, and long-term impacts to downstream water and aquatic ecosystems (Platts and Nelson 1985; Boise National Forest 1993; McIntosh et al. 1994, 1995). In unstable terrain, for instance, small areas (e.g., less than 10% of a watershed's area) of low-intensity disturbance, including roads, may greatly increase the frequency and size of mass erosion events, with subsequent acute and chronic reduction in downstream water quality. Management activities that damage natural vegetation typically result in loads of suspended solids that exceed background levels and more frequent and intense spikes in suspended solids stemming from an increase in mass erosion events like landslides, debris flows, and bank failures. These impacts are strongly correlated with roads, as well as with logging and grazing (Amaranthus et al. 1985; Fleischner 1994; Trombulak and Frissell 2000; Coffin 2007).

Rising Demand and Climate Change Diminish Water Supply. Population in the West is projected to increase by 300% within just 30 years, with similar increases in demand for water (Sedell et al. 2000). Urban and exurban areas are growing exponentially, including communities adjacent to wilderness areas and IRAs (Theobald 2005). The demand for water in Colorado is expected to triple by 2050. Similarly, the number of people relying on national forest water has doubled in Oregon in the last 30 years, and 86% of the population of Washington rely on national forest water to some degree (Sedell et al. 2000).

The dramatic population growth in the West is concurrent with a warming and drying climate in many places. Temperatures are increasing, snow pack is declining and melting sooner, and drought and summer water deficits are more frequent and longer (Barnett et al. 2008; Mohammed and Tarboton 2008; Saunders et al. 2008; Miller et al. 2010). Streamflow reductions ranging from 10% to 35% are likely for the western states over the next half century as a consequence of climate change (Barnett and Pierce 2009). A 10% drop in streamflow is considered calamitous by municipal water districts. More frequent and intense flood events are also likely in places (Raff et al. 2009), despite drying conditions. Costs for flood control, repair and reconstruction, and insurance rates will also increase (GAO 2007). These events will worsen the severe and unprecedented droughts already afflicting much of the West (Drechsler et al. 2006; Saunders et al. 2008).

SOLUTION: A LIGHT HYDROLOGICAL FOOTPRINT IN ROADLESS AREAS

IRAs should be managed in the same way many municipalities manage their watersheds—sustaining a light ecological and hydrological footprint and hydrologic restoration through decommissioning or, even better, obliteration of roads (Barten et al. 1998; NRC 2000; Payne et al. 2004; Gallo et al. 2005; Postel and Thompson 2005; Seeds 2010). The most cost-effective and prudent approach to maintain water supplies and high-quality fresh water in the face of population growth and climate change is to manage upper watersheds in a roadless condition with undisturbed natural vegetation. The high, long-term economic cost of degrading clean water for millions of people, by itself, is argument strong enough to continue protection of the current roadless areas network either at national or state levels. Development of IRAs, as proposed in Colorado, would primarily provide opportunities for short-term gains, but the substantial and long-term impacts on water quality and availability will come at a time of increasing demand and shrinking supply. Managers should, therefore, treat IRAs as natural reservoirs of high quality water for downstream users before approving development projects. Cost-benefit analyses should include regionally and locally specific estimates of water quality to better inform project management decisions that may reduce the value of high-quality water in the short and long run.

CONCLUSIONS

Roadless areas and the relatively intact ecosystems they maintain provide many important biodiversity benefits, including acting as strongholds for threatened freshwater species. Beyond these important values, their role in producing clean and reliable water for people and economies is more likely to compel decision-makers to leave roadless areas undeveloped. We reviewed the importance of inventoried roadless areas on national forests in the United States to determine their importance in providing clean water for downstream users. We concluded that (1) many intact watersheds are in headwaters, (2) they supply downstream users with high-quality drinking water, and (3) developing these watersheds comes at significant costs associated with declining water quality and availability. Several case studies from the western United States, particularly Colorado, demonstrated the importance of assessing the diverse consequences of developing roadless areas. Managers should perform comprehensive cost-benefit analyses when weighing development options. A light-touch hydrological footprint is recommended to sustain the many values that derive from roadless areas, especially clean and abundant water.

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Complex Early Seral Forests of the Sierra Nevada: What are They and How Can They Be Managed for Ecological Integrity?

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RESEARCH NOTE

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Complex Early Seral Forests of the Sierra Nevada: What are They and How Can They Be Managed for Ecological Integrity?

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ABSTRACT: Complex early seral forests (CESFs) occupy potentially forested sites after a stand-replacement disturbance and before re-establishment of a closed-forest canopy. Such young forests contain numbers and kinds of biological legacies missing from those produced by commercial forestry operations. In the Sierra Nevada of California, CESFs are most often produced by mixed-severity fires, which include landscape patches burned at high severity. These forests support diverse plant and wildlife communities rarely found elsewhere in the Sierra Nevada. Severe fires are, therefore, essential to the region's ecological integrity. Ecologically detrimental management of CESFs, or unburned forests that may become CESF's following fire, is degrading the region's globally outstanding qualities. Unlike old-growth forests, CESFs have received little attention in conservation and reserve management. Thus, we describe important ecological attributes of CESFs and distinguish them from early seral conditions created by logging. We recommend eight best management practices in CESFs for achieving ecological integrity on federal lands in the mixed-conifer region of the Sierra Nevada.

Index terms: complex early seral forests, ecological integrity, mixed-severity fire, Sierra Nevada

INTRODUCTION

Early seral forests are ecosystems that occupy potentially forested sites after a stand-replacement disturbance and before re-establishment of a closed forest canopy (Swanson et al. 2011). Such forests are generated by disturbances that reset successional processes and follow a pathway that is influenced by biological legacies (e.g., large live and dead trees, downed logs, seed banks, resprout tissue, fungi, and other live and dead biomass) that were not removed during the initial disturbance (Franklin et al. 2000; Donato et al. 2012). Where these legacies are intact, complex early successional forests (CESFs) develop with rich biodiversity due to the function of the remaining biomass in providing resources to many life forms and because of habitat heterogeneity provided by mixed-severity fires that generated them (Odion and Sarr 2007; Swanson et al. 2011). In general, mixed-severity fires, which include patches of high-severity fire, create coarse-grained, high-contrast heterogeneity that results in CESFs, and, over time, a complex mosaic of seral stages at the landscape and local scales. Low to moderate fire severities create fine-grained, lower contrast heterogeneity that generate very little if any CESFs, although they create other conditions favorable to biodiversity. Many effects of fire cannot be mimicked by land-use disturbances (Odion and Sarr 2007). Suppression of fire and removal of biomass after a fire are thus causes of reduced biodiversity and ecological integrity.

While the unique "floral phoenix" that follows stand-replacing fire in many vegetation types such as the California chaparral has long inspired botanists in the United States (Brandegee 1891; Howell 1946) and elsewhere (Bond and van Wilgen 1996), similar attention has not been given to stand-replacing fire in Sierran forests. Instead, fire has been suppressed in these forests for many decades. Traditionally, stand-replacement processes have also been considered historically unimportant in these forests, simply because they occur less frequently than surface fires, which are largely non-lethal (Skinner and Chang 1996). Stand-replacing fire also has a negative connotation in resource management disciplines because of their narrow focus on impacts to timber values, and such fires frequently receive negative coverage from the mass media.

While much of the conservation attention in the Sierra Nevada has rightfully focused on iconic conifers like the giant sequoia (*Sequoiadendron giganteum*) and other old-growth forest types, even in the context of multiple-use management and conservation, there is still little appreciation for CESFs, which do not have the charismatic old-growth species and living structures (Swanson et al. 2011). Thus, for a variety of reasons, there is a paucity of literature on, or appreciation of, CESFs. Indeed, CESFs are not even recognized as a distinct habitat type in any current vegetation mapping used by the U.S. Forest Service in the Sierra Nevada (e.g., California Wildlife Habitat Relations). However, in terms of their contribution to biodiversity and vital life-history stages of many species, CESFs have disproportionately important ecological roles in the overall ecological integrity of forested landscapes. Thus, we call attention to this successional stage (Swanson et al. 2011) and the need for its inclusion in conservation strategies in the Sierra Nevada ecoregion.

It is timely to consider CESFs in Sierra conservation strategies because the Sequoia, Sierra, and Inyo National Forests (Figure 1) are undergoing forest plan revisions as part of the "early adopters" of the forest-planning rule (36 Code of Federal Regulations Part 219). The forest-planning rule directs the U.S. Forest Service to maintain or improve ecological integrity, defined as "the quality or condition of an ecosystem when its dominant ecological characteristics (for example, composition, structure, function, connectivity, and species composition and diversity) occur within the natural range of variation and can withstand and recover from most perturbations imposed by natural environmental dynamics or human influence" (Forest Planning Rule 36 CFR 219.19). Given the global importance of the Sierra Nevada ecoregion (Ricketts et al. 1999), many scientists and the public expect a high level of protection and stewardship in forest-planning decisions and they support managing for ecological integrity. But, as an often-overlooked seral stage, the role of CESFs in ecological integrity and conserving biodiversity has not been addressed.

We address three questions of management relevance to CESFs in the Sierra Nevada: (1) what are CESFs and why are they important to ecological integrity; (2) are there tradeoffs for managing species of conservation concern that occur at opposite ends of the successional continuum such as Black-backed Woodpeckers (*Picoides villosus*; avian taxonomy follows American Ornithologists' Union checklist of North and Middle American birds; http://checklist.aou.org/; active May 20, 2013) and California Spotted Owls (*Strix occidentalis occidentalis*); and (3) what are the principal threats to these forests? We also provide general recommendations for conserving, restoring, and researching the ecological integrity and biodiversity of Sierran CESFs.

STUDY AREA

The Sierra Nevada ecoregion spans some 63,111 km^2 along a north-south axis in California, and the USDA Forest Service manages the majority of montane forests in this region (Davis and Stoms 1996; Figure 1). The ecoregion is among the most diverse temperate conifer forests in the world and its conservation status is considered critically endangered due to extensive forest fragmentation and other land-use stressors (Ricketts et al. 1999). An extraordinary assortment of vegetation types and diverse forest successional stages occur across the region. For instance, based on potential vegetation mapping, 25 conifer, 23 hardwood forest/woodland types, 34 shrub and chaparral, and 5 herbaceous alliances are distributed across elevations, slopes, aspects, and soil types (USDA Forest Service 2008). Plant alliances mix together at zones of overlap resulting in high levels of beta diversity (change in numbers of species across environmental gradients). There are exceptional levels of endemic plants (e.g., approximately 405 vascular plants are endemic and 218 taxa are rare; Shevock 1996), especially in the southern Sierra, and some of the highest levels of mammal endemism in North America (Ricketts et al. 1999). Notably, areas with high concentrations of endemic species are a conservation priority because the restricted distribution of endemics predisposes them to extinction from habitat losses.

Mixed-conifer forests are the predominant forests in the Sierra that are typically found at middle elevations (760–1400 m) in the northern Sierra, higher elevations south (915–3050 m), and, to a lesser extent, on upper elevations (2130 m to 3040 m) along the east slopes (Chang 1996). They are replaced at higher elevations by pure red fir (*Abies magnifica,* Andr. Murray) and red and white fir (*A. concolor,* Gordon & Glend.) (Barbour et al. 2007). There are three forest types that comprise mixed-conifer forests in this region: (1) white fir/Jeffrey pine (*Pinus jeffreyi,* Grev. & Balf.) /lodgepole pine (*P. contorta,* Loudon); (2) Pacific Douglas-fir (*Pseudotsuga menziesii menziesii,* Franco), and ponderosa pine (*P. ponderosa*; at lower elevations); and (3) mid-elevation Douglas-fir (does not occur south of Yosemite National Park). These more typical conifers are associated with sugar pine (*P. lambertiana,* Douglas), incense cedar (*Calocedrus decurrens,* Torrey), black oak (*Quercus kelloggii,* Newb.), and patches of giant sequoia. Mixed-conifer forest types also support shrubs such as greenleaf manzanita (*Arctostaphylos patula,* E. Greene), huckleberry oak (*Q. vaccinifolia*, Kellogg), curleaf mountain mahogany (*Cercocarpus ledifolius,* Nutt.), snowbrush (*Ceanothus velutinus,* Dougl.), mountain alder (*Alnus incana* ssp. *tenuifolia,* Nutt.), mountain sagebrush (*Artemisia tridentate* ssp. *vaseyana,* Rydb.), and bitterbrush (*Purshia tridentata,* Pursh) (USDA Forest Service 2013a). Most of these forests consist of mid-sized trees that average 30–60 cm dbh and include areas with larger trees (>60 cm dbh; North 2013); nearly half of the mixed-conifer forest in the giant sequoia type is late seral (USDA Forest Service 2013a).

Very-long-interval, stand-replacement fire occurs in a patchwise fashion within low- and mixed-severity fires in moist mixed-conifer and white fir forests in this region, and variable (both short- and long-interval) stand-replacement fires occur in Douglas-fir and lodgepole pine (*Pinus contorta,* Loudon) forests (Leiberg 1902; Chang 1996). Prior to fire suppression, drier low-elevation forests burned relatively frequently and often at a low severity; but they also had significant mixed-severity effects, including occasional large high-severity fire patches (USDA Forest Service 1911).

What are Early Seral Forests and Why Are They Important?

In general, CESFs are rich in post-disturbance legacies (Photo Plates 1a, 1b, 1c) and post-fire vegetation (e.g., native fire-following shrubs/herbs, resprouting broad-leaved trees, and natural conifer regeneration) (Photo Plates 2a, 2b, 2c). We identify 12 ecological attributes that contribute to the prolific biological response common in CESFs and which are, therefore, key to the ecological integrity present in CESFs

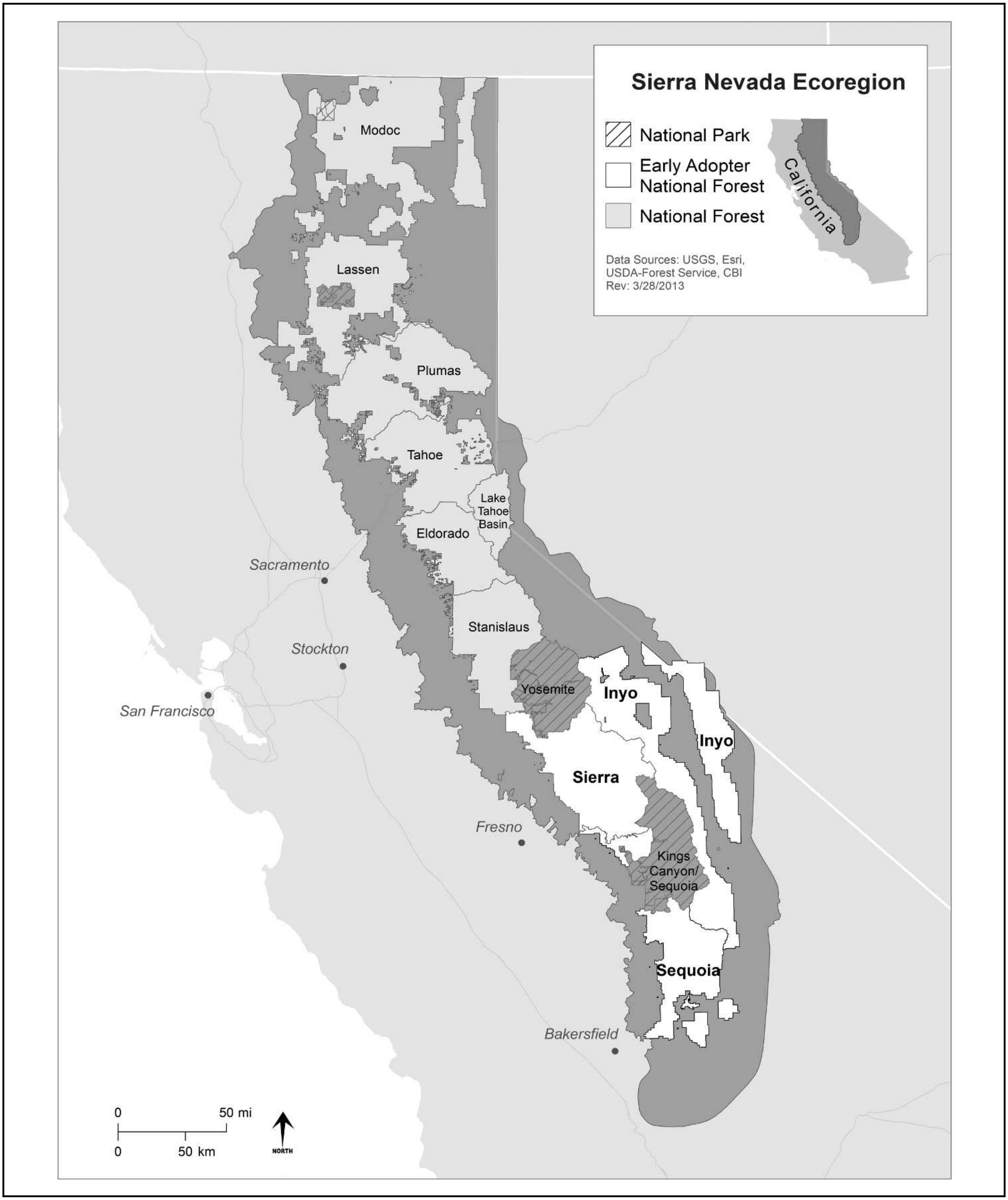

Figure 1. Location of Sierra Nevada ecoregion, northern California, and "early adopters" of the forest-planning rule involved in forest plan revisions.

Photo Plates. Extensive biological legacies, abundant forb cover, and abundant conifer regeneration present in complex early seral forests vs. early seral that has been post-fire logged. Post-fire logging in the Sierra Nevada and elsewhere sets back ecosystem processes creating a successional debt.

Plate 1a Plate 1b Plate 1c

Photo Plate 1. Star Fire of 2001, Northern Sierra, CA. (a) unmanaged with forbs (Doug Bevington, 2008); (b) natural conifer re-establishment (Chad Hanson, 2012); Storrie fire of 2000, Southern Cascades, CA. (c) unmanaged with snags and forbs (Chad Hanson, 2007).

Plate 2a Plate 2b Plate 2c

Photo Plate 2. Postfire logged portions of Fred's fire in the Eldorado National Forest, CA, showing lack of nitrogen-fixing shrubs (a) and presence of Klamath weed (*Hypericum perfoliatum*) and many readily ignitable, invasive grasses (b) (Dennis Odion, August 2011); (c) simplified system from Dinkey post-fire thin on west slopes of Southern Sierra (Chad Hanson, 2012).

(Table 1). When logging compounds the natural disturbance that created a CESF (Photo Plates 3a, 3b, 3c), each of these attributes is reduced or eliminated (Table 1). Such multiple disturbances often lead to alternative successional pathways, or loss of resilience (Paine et al. 1998; Odion and Sarr 2007), as has been documented in the Sierra Nevada following post-fire logging, which leads to dominance by the non-native ecosystem transformer, cheatgrass (*Bromus tectorum,* Linnaeus) (McGinnis et al. 2010).

Overall, compared to logged areas, CESFs are structurally more complex, contain more large trees and snags that originated from the pre-disturbed forest, have more diverse understories, functional ecosystem processes, and more diverse gene pools that, theoretically, should provide greater resilience in the face of climate change than that provided by the simplified early seral forests produced by logging. CESF attributes promote a high level of species richness, particularly bird communities that utilize these forests extensively (Hutto 1995; Kotliar et al. 2002; Fontaine et al. 2009; Appendix). The residual biomass of CESFs reduces disturbance stressors and provides for the rapid proliferation of new life (Odion and Sarr 2007). For example, seed banks and vegetation tissues give rise to dense, often rampant, forb cover, abundant grasses, and shrubs – especially nitrogen fixers (e.g., *Ceanothus* spp.) (Conard 1985; Busse et al. 1996; Busse 2001) and ectomycorrhizal associates (e.g., *Manzanita* spp.) that facilitate conifer growth (Zavitovsky and Newton 1968; Horton et al. 1999). Serotinous (closed cone) conifers like giant sequoia (Stephenson et al. 1991) also do well in these forests. Other plants that can abundantly colonize burns, such as conifers and fireweed (*Epilobium angustifolium,* Linnaeus), arrive by wind or animal dispersed seed. Thus, plant species richness of CESFs can be much higher than in unburned forests (Donato et al. 2009).

Other bird and small mammal communities that utilize CESFs forage extensively on the abundant insects and increased abundance of seeds from the post-fire flora (Lawrence 1966; Fontaine et al. 2009). These species, in turn, support an increase in raptors (Lawrence 1966). Bird species such as the Black-backed Woodpecker, Olive-sided Flycatcher (*Contopus cooperi*), Mountain Bluebird (*Sialia currucoides*), Chipping Sparrow (*Spizella passerina*), and Mountain Quail (*Oreortyx pictus*) (Appendix)

Table 1. Differences between early seral systems produced by natural disturbance processes vs. logging. For natural disturbances, assume that a disturbance originates from within a late-successional forest as legacies are maintained throughout succession. For logged sites, assume site preparation includes conifer plantings but no herbicides, which, if also applied, would magnify noted differences.

Attribute	Regeneration Harvest or Postfire Logged	Natural Disturbance
Large trees	rare	abundant and widely distributed
Large snags/downed logs	rare	abundant and widely distributed
Understory	dense conifer plantings followed by sparse vegetation as conifer crowns close (usually within 15-20 years depending on site productivity)	varied and rich flora
Species composition	few species mostly commercially stocked, deer initially abundant then excluded as conifer crowns close	varied and rich flora, rich invertebrates and birds, abundant deer
Structural complexity	simplified	highly complex; many biological legacies
Soils and below-ground processes	compacted and reduced mycorrhizae	complex and functional below ground mats
Genetic diversity	low due to emphasis on commercial species and nursery genomes	complex and varied
Ecosystem processes (predation, pollination)	moderate initially then sparse as conifer crowns close; limited food web dynamics	rich pollinators and complex food web dynamics
Susceptibility to invasives	moderate to high depending on site preparation, soil disturbances, livestock, road densities (see McGinnis et al. 2010)	low due to resistance by diverse and abundant native species and low soil disturbances
Disturbance frequency	commercial rotations (40-100 years or so)	varied and complex
Landscape heterogeneity	low	high; shifting mosaics and disturbance dynamics
Resilience/resistance to climate change	low due to nursery stock genomes but conifer plantings can be adjusted for locally anticipated climate envelopes	varied and complex genomes allow for resilience and resistance to climate change

achieve highest abundances in CESFs. In fact, in the Sierra Nevada, CESF habitat is comparable or higher in bird species richness and total bird abundance relative to unburned mature forest (Burnett et al. 2010). Bats (*Myotis, Idionycteris, Lasionycteris*, and *Eptesicus)*, which are an increasing conservation concern, are also favored by CESFs, likely because of greater insect prey as well as suitable roosts (Buchalski et al. 2013). Stand-replacing fires stimulate a flux of aquatic prey to terrestrial habitats, driving increases in riparian consumers (Malison and Baxter 2010). The trees killed by fire are highly beneficial to the ecological integrity of stream communities because they are a main source of large woody debris inputs (Minshall et al. 1997). There is also reproduction by some forest fungi species that are restricted to burns (e.g., morels, *Morchella* spp.) and the dead wood provides substrate for fungal growth that supports many arthropod species, including unique fire-following native beetles (Lindsey 1943; Bradley and Tueller 2001). Beetles, in general, colonize fire-killed trees in CESFs and their abundant larvae support species like Black-backed Woodpeckers (Hutto 2008).

Indicator Species for CESF Biodiversity (Figure 3)

Indicator species are valuable tools for conservation management because it is not practicable to monitor all biodiversity. When burned forests are logged after fire, one species that serves well as an ecological indicator for post-fire biodiversity, the Black-backed Woodpecker, declines substantially (Hutto 2008). Given that this woodpecker already is an indicator of the biodiversity supported by CESFs in the Sierra Nevada (USDA Forest Service 2013b), and is a fire specialist, we propose it as a Species of Conservation Concern. Designated Species of Conservation Concern are those whose population viability, or continued representation within a particular plan area, is of management concern. The forest-planning rule provides guidance to forest managers to use Species of Conservation Concern as a means for maintaining species diversity and wildlife population viability.

CESF habitat represented by Black-backed Woodpeckers is biologically unique (Hutto 1995; Bond et al. 2012). The Black-backed Woodpecker is an important primary excavator of nesting holes for many other cavity-nesting birds and mammals because it discards cavities after excavating them, and it uses a given cavity for one year (Tarbill 2010). Under a scenario with stand-replacing fire operating in a patchwise fashion in a landscape containing healthy populations of Black-backed Woodpeckers, the availability of nesting cavities across the landscape over time may be greatly enhanced compared to where fire is suppressed and/or fire-killed trees are removed. Black-backed Woodpeckers use CESFs for only several years (typically seven or eight) after fire and they depend upon the regular creation of CESFs to replenish their habitat (Hanson and North 2008; Tarbill 2010; Dudley et al. 2012; Siegel et al. 2013). When this does not occur, many other species that rely on nesting cavities are likely to be negatively affected. Thus, many species probably depend directly, or indirectly, on the continued occurrence of high-intensity natural disturbance across large landscapes to maintain their populations (Hanson and North 2008; Tarbill 2010; Dudley et al. 2012; Siegel et al. 2013).

Black-backed Woodpeckers have become increasingly rare because their optimal habitat has shrunk to a fraction of its historical extent (Figure 2 a – d); populations are estimated at <700 nesting pairs in burned forests (Bond et al. 2012). Importantly, the CESF habitat that the remaining pairs depend on has little or no protection on public lands managed by the U.S. Forest

Figure 3. Black-backed woodpecker – a fire dependent species in the Sierra (Photo – Monica Bond).

Service. Much of this CESF habitat is under mounting pressure from fire suppression and both pre- and post-fire logging (Hutto and Gallo 2006; Hanson and North 2008; Hutto 2008; Siegel et al. 2013), which prevent high-quality woodpecker habitat. That, in turn, may affect the biodiversity for which this woodpecker serves as an indicator.

Are There Management Tradeoffs for Species of Conservation Concern at Opposite Ends of the Successional Continuum?

Wildlife management often involves trade-offs when habitat for a particular species is emphasized. That is a problem with single-species management (managing for what one species needs), but is not a problem when managing for the maintenance of natural systems that a species may indicate. In the latter case, we would not enhance but would maintain natural levels of habitat for CESF indicators like the Black-backed Woodpecker, and for the biodiversity associated with its presence.

However, the California Spotted Owl is also a management indicator species but for late-seral forests in this region. Notably, all three subspecies (Mexican, California, Northern; Bond et al. 2002; Jenness et al. 2004; Roberts 2008; Bond et al. 2009; Clark et al. 2011; Roberts et al. 2011; Lee et al. 2012; Clark et al. 2013) appear to tolerate, or even benefit, from some degree of moderate- to high-severity fire within territories.

Managing CESFs for high levels of ecological integrity may provide important prey habitat (e.g., dusky-footed woodrat *Neotoma fuscipes*; Munton et al. 2002) for the spotted owl. In fact, the owl is known to reproduce in territories burned at all fire severities in this region, and preferentially selects high-severity fire areas for foraging (Bond et al. 2009). Owl reproduction has been found to be 60% higher in unmanaged mixed-severity fire areas than in unburned forests (Roberts 2008), and mixed-severity fire (with an average of 32% high severity) (Lee et al. 2012) does not reduce owl occupancy, though post-fire logging may precipitate territory abandonment (Clark et al. 2011, 2013; Lee et al. 2012). Moreover, because high-severity fire has been reduced by fire suppression, and current high-severity fire rotations are very long in the Sierra Nevada, if high-severity fire rates increased by even two- or three-fold, it would benefit CESF-associated species like the Black-backed Woodpecker, but would only reduce current old forest by a very small amount given old forest recruitment from ingrowth (Odion and Hanson 2013). Thus, protecting CESFs from post-fire logging and maintaining the spatial heterogeneity created by mixed-severity fires should provide habitat for all seral associates – there really are no management trade-offs when we manage for the maintenance of natural processes and systems.

What are Principal Threats to CESFS?

Management of CESFs has most often included post-fire (salvage) logging followed by tree planting, including burning of slash piles and associated soil disturbances, reseeding with grasses (often introducing invasive species inadvertently), use of straw-bales and other erosion prevention methods, herbicides to reduce shrub competition with conifers,

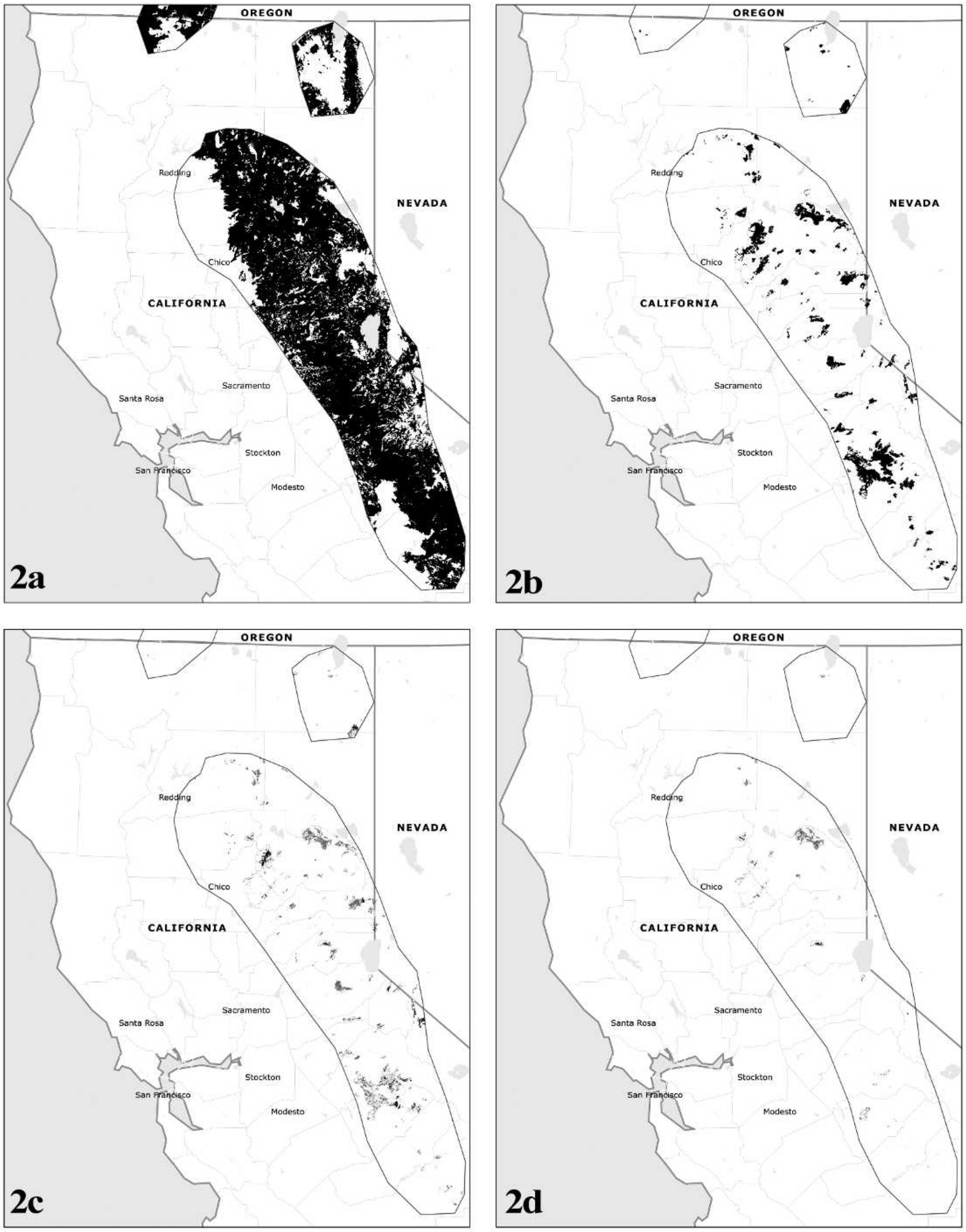

Figure 2. (a) Forest types used by Black-backed Woodpeckers in the Sierra Nevada management region; (b) fires since 1984 within the relevant forest types (private lands not included since they are rapidly logged); (c) moderate/high-severity fires resulting in >50% mortality (RdNBR >574 – see Hanson et al. 2010) of forests on public lands within the relevant forest types in the most recent decade for which there are fire severity data (2001–2010) (i.e., both high quality Black-backed Woodpecker habitat and moderate/low quality (older) habitat combined); and (d) moderate/high-severity fire on public lands within the relevant forest types in the most recent 5-year period for which fire severity data are available.

planting with conifer nursery stock, and livestock grazing (Swanson et al. 2011; Long et al. 2013; Table 1). These activities remove, or severely degrade, CESFs or, at a minimum, can narrow the window of duration for CESFs (Swanson et al. 2011), contributing to "landscape traps," whereby entire landscapes are shifted into, and then maintained in, a highly altered state as the result of cumulative impacts (Lindenmayer et al. 2011).

Climate change and forest fragmentation also have been identified as threats to biodiversity in the Sierra Nevada (USDA Forest Service 2013b). Since the 1980s, the region has experienced a decrease in annual number of days with below-freezing temperatures at higher elevations with more rain and less snowfall mainly in northern latitudes, more extreme heat days at lower elevations, earlier (5 to 10 days) snowmelt than decades ago, earlier (5–15 days) peak stream flows (Safford et al. 2012; Harpold et al. 2012), as well as an increase of approximately 1 °C since the early twentieth century, though some areas of the northern Sierra Nevada have seen a decrease in temperature (North 2012). Some regional climate models project further decreases in mountain snowpack, earlier snowmelt and peak stream flows, and greater drought severity (Overpeck et al. 2012). Such climatic changes are likely to affect the low-elevation ponderosa pine, which is projected to extend upward, while red fir and subalpine communities are projected to lose much of their climate envelope in the coming century (USDA Forest Service 2013b). It is unclear how such changes will affect CESFs. If fire increases in severity or frequency (Miller et al. 2009; Miller and Safford 2012), this could provide more opportunities for development of CESFs. This assumes there is not a concomitant increase in post-fire logging, and that fire suppression activities either cannot keep pace with climate-related fire events or prove ineffective due to the increasing influence of climate as a top-down driver of fire behavior. On the other hand, a number of climate models predict decreasing fire activity in these forests – even as temperatures rise – due to increasing precipitation, including summer precipitation and changes in vegetation (McKenzie et al. 2004; Krawchuk et al. 2009), and recent research using the largest fire severity data set to date has found no increase in fire severity in the Sierra Nevada since 1984 (Hanson and Odion, 2014; also see Odion et al. 2014 for related discussion).

Land-use stressors also magnify climate change effects on forest communities. For instance, Thorne et al. (2008) documented significant regional changes due to climate and land-use practices resulting in greater levels of disturbance compared to historical. Millar (1996) identified three paramount influences on Sierra Nevada ecosystems: (1) climate change and shifting hydrological patterns; (2) dense forests; and (3) rapidly expanding human populations. It is not known, however, whether these changes will act in concert to make CESFs more vulnerable to invading species, particularly those more suited to the changing climate and land-use disturbances.

Suggested Best Management Practices for CESF

For all the reasons outlined above, CESFs represent a neglected seral stage subject to multiple stressors that compromise ecological integrity. We, therefore, propose eight "best management practices" for stimulating conservation, restoration, and research interests in these unique forests. These principles can serve as appropriate guidelines where management goals include the maintenance of ecological integrity.

Conservation Focus

Principle 1 – "Rehabilitation" Is Not Needed After Fire Creates a Complex Early Seral Forest (Beschta et al. 2004; Swanson et al. 2011).

Fire acts as a natural restorative agent by resetting the successional clock and providing habitat for disturbance-dependent species. Although CESFs lack live trees initially and are populated by dead ones, this does not mean they require site rehabilitation or are "unhealthy" forests. In the context of ecological integrity, a functional forest system is one where the natural fire regime is of mixed-severity and has all stages of succession following stand-replacing fire. CESFs should be mapped and managed as a distinct forest habitat type.

Principle 2 – Protect Large, Old Forest Structures Across Seral Stages, and Retain Dense, Old Forests to Improve Ecological Integrity at Landscape Scales.

Large old-forest structures take decades to centuries to develop, and forest management has created a deficit through extraction. Dense, old forests provide high-quality habitat not only when they are green, but also when they experience mixed-severity fire (Hutto 2006, 2008), or snag pulses from beetles (Bond et al. 2012), as biological legacies remaining also serve to connect seral stages along the successional gradient.

Principle 3 – Mixed-severity Fire Should Be a Management Goal for Reserves.

Robust, reserve-based conservation strategies are needed to maintain the suite of seral stages and allow for climate-forced wildlife dispersals into suitable habitat. Thus, managers should allow fires to run their course in the backcountry and in reserves when not a risk to people or dwellings. This includes maintaining a landscape that includes diverse seral stages across environmental gradients (elevation, latitudinal).

Restoration and Management Focus

Principle 4 – Adopt Comprehensive Approaches to Restore Ecological Integrity in CESFs.

This starts with a restoration needs assessment (DellaSala et al. 2003) to evaluate and prioritize drivers of ecosystem degradation and best practices aimed at reducing specific stressors (see Principle 6). Most importantly, forests restored through fire usually do not need "restoration" otherwise.

Principle 5 – Limit Post-fire Management to Early Seral Forests Previously Degraded by Logging, Grazing, and Other Stressors.

Restoration approaches should identify comparable areas of high ecological integrity (e.g., unmanaged CESFs, DellaSala et al. 2003) to serve as a baseline or reference condition from which to restore degraded areas (e.g., burned plantations), and then surveillance, implementation, effectiveness, and ecological effects monitoring (Hutto and Belote 2013) should always be an integral part of the restoration activity.

Principle 6 – Reduce Land-use Stressors That Compromise the Ecological Integrity of CESFs.

Restorative measures can be active or passive depending on site-specific needs and should always be followed with well-funded monitoring (DellaSala et al. 2003). Examples include removal of livestock, invasive species abatement, road closures and obliteration, and reintroduction of fire.

Research Focus

Principle 7 – Determine Historical, Current, and Projected Future Distributions and Spatio-temporal Extent of CESFs as Well as Other Seral Stages Across the Planning Area.

This can be informed through "back-casting" approaches that reconstruct an historical baseline from combining age-structure reconstructions (e.g., from either FIA plot data or General Land Surveys from the 1800s; see techniques in Baker 2012; Williams and Baker 2012) with studies that link stand structure, disturbances and fire scar data (e.g., Sherriff and Veblen 2006), or other sources of information (e.g., USDA Forest Service 1911). Historical baselines can then be compared to current and future projected conditions under a changing climate in order to determine appropriate representation levels of CESFs and other seral stages in a planning area.

Principle 8 – Designate the Black-backed Woodpecker a "Species of Conservation Concern."

Continue, and expand upon, current monitoring efforts and, in partnership with the U.S. Fish and Wildlife Service and other experts, determine how best to meet population viability and habitat needs of this important CESF species. Although Black-backed Woodpecker populations decline as this seral stage advances (within seven years following fire), this species still functions as an indicator of early successional species because stable woodpecker populations would mean a steady supply of CESFs over time. Olive-sided Flycatcher, Chipping Sparrow, Mountain Bluebird, and other early seral species that have population peaks after declines in woodpeckers, may need to be monitored to ensure CESF conservation.

CONCLUSIONS

The forest-planning rule and its emphasis on ecological integrity, plant and animal community diversity, and Species of Conservation Concern provides the Forest Service with a unique opportunity to revise forest plans in the Sierra Nevada to meet the primary and cumulative threats that these forests now face – climate change and land-use stressors. Where the region's forests are to be managed for ecological integrity, managers will need to determine spatio-temporal occurrence of CESFs (historical and current) to allow for adequate representation of all seral stages across planning areas, particularly the rare ones that occupy opposite ends of the successional continuum (CESFs and late seral). This also means conducting field inventories in CESFs to better describe their unique attributes and ecological importance, treating CESFs as a distinctive wildlife habitat type in habitat classifications, and incorporating mixed- to high-severity fire into management goals at middle to upper elevations.

Clearly, climate change introduces uncertainties regarding how fire and other disturbance agents will operate on these forests in the future. Whether this will increase or further reduce CESFs remains to be seen. While managing for resilient ecosystems is a desired ecological objective of climate adaptation planning on the national forest system (36 Code of Federal Regulations § 219.5), it is important for managers to go beyond mechanical fuel reduction as a means for maintaining resilient ecosystem properties, and this includes acceptance of mixed- and high-severity fires as important ecosystem processes. However, resilient to natural disturbance does not necessarily mean resistant to disturbance. Sierran forests are disturbance dependent; they require severe fire for the production of CESFs.

The eight principles recommended for best management practices in CESFs in the Sierra Nevada would promote ecological resilience and allow the National Forests in this globally outstanding ecoregion to better adapt to climate change and increasing human development in the surroundings. We encourage conservationists and park managers to emphasize CESFs in reserve design and related conservation strategies as these forests are at least as important as their late-successional counterparts.

AUTHORS ENDNOTE

At the time of this publication, the Stanislaus National Forest was proposing extensive (~18,000 ha) post-fire logging of live (injured) and dead trees (including "roadside-hazard trees"), conifer re-planting, and shrub-eradication in the wake of the 2013 Rim Fire along the border of Yosemite National Park. The agency also proposes to plant conifers in high severity patches, thereby leap frogging important non-conifer dominant stages. Post-fire logging is incompatible with the needs of legions of species that depend on the presence of standing dead trees and montane chaparral.

Because of the significance of the Rim Fire as a pulse disturbance for generating CESFs, its proximity to an iconic national park, and the opportunity to educate the public about the importance of burned forest habitat, we believe the area warrants consideration for a national monument designation as did Mount St. Helens after the historic 1980 eruption. We urge managers and conservationists to give more attention to the ecological importance of CESFs in new protected areas proposals. This is especially important as we see the threat to these unique forests escalating due to increasing emphasis by federal agencies on extensive and intensive post-fire management projects.

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Chad Hanson, Ph.D., is the staff Ecologist and Director of the John Muir Project of Earth Island Institute. He has authored and co-authored papers published on topics as diverse as fire history, current fire patterns and trends, post-fire conifer response, and wildlife habitat selection in post-fire forests. Hanson focuses his research in the Sierra Nevada, with a particular emphasis on the Black-backed Woodpecker and the Pacific fisher.

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Appendix. Bird species present in complex early seral forests in the Sierra Nevada based on comparisons of burned and unburned plots (Raphael et al. 1987: east slopes of Sierra, University of California Sagehen Creek Field Station, pine-fir forests, ridgetop at 2100-m elevation, Burnett et al. 2012: Plumas National Forest, northeastern CA, mixed conifers, elevations 1094–2190 m: Storrie, Moonlight, and Cub mixed-severity fires). Only the Burnett et al. (2012) performed statistical analyses on bird abundances between burned and unburned plots. Taxonomy follows American Ornithologists' Union checklist of North and Middle American birds (http://checklist.aou.org/; active May 20, 2013).

Species	Present in both studies	Difference in abundance burned vs. unburned[2]
Mountain quail[3,4] *Oreortyx pictus*	+	+
American kestrel[3] *Accipiter cooperii*		
Mourning dove[3] *Zenaida macroura*		
Common nighthawk[3] *Chordeiles minor*		
Calliope hummingbird *Selasphorus calliope*	+	NS
Williamson's sapsucker *Sphyrapicus thyroideus*		
Red-breasted sapsucker *Sphyrapicus ruber*		
Hairy woodpecker *Picoides villosus*	+	+
White-headed woodpecker *Picoides albolarvatus*	+	+
Black-backed woodpecker *Picoides arcticus*	+	+
Northern flicker *Colaptes auratus*		
Olive-sided flycatcher[3] *Contopus cooperi*	+	+
Western wood-pewee *Contopus sordidulus*	+	+
Hammond's flycatcher *Empidonax hammondii*		
Dusky flycatcher *Empidonax oberholseri*	+	-
Cassin's vireo *Vireo cassinii*		-
Warbling vireo *Vireo gilvus*		NS
Stellar's jay *Cyanocitta stelleri*	+	NS
Mountain chickadee *Poecile gambeli*	+	-
Red-breasted nuthatch *Sitta canadensis*		-
White-breasted nuthatch *Sitta carolinensis*		

Cont'd.

Appendix. (Continued)

Species	Present in both studies	Difference in abundance burned vs. unburned[2]
Pygmy nuthatch[3] *Sitta pygmaea*		
Brown creeper *Certhia Americana*	+	+
Golden-crowned kinglet *Regulus satrapa*		-
Mountain bluebird[3] *Sialia currucoides*	+	+
Townsend's solitaire *Myadestes townsendii*		
Hermit thrush *Catharus guttatus*		-
American robin *Turdus migratorius*	+	NS
House wren[3] *Troglodytes aedon*	+	+
Nashville warbler *Oreothylypis ruficapilla*	+	-
MacGillivray's warbler *Geothlypis tolmiei*		NS
Yellow warbler[3] *Setophaga petechia*		NS
Yellow-rumped warbler *Setophaga coronata*	+	-
Hermit warbler *Setophaga occidentalis*		-
Green-tailed towhee[3] *Pipilo chlorurus*	+	+
Spotted towhee *Pipilo maculatus*		+
Chipping sparrow *Spizella passerina*	+	+
Brewer's sparrow[3] *Spizella breweri*		
Fox sparrow *Passerella iliaca*	+	+
Dark-eyed junco *Junco hyemalis*	+	+
Western tanager *Piranga ludoviciana*	+	-
Black-headed grosbeak *Pheucticus melanocephalus*		-
Lazuli bunting[3] *Passerina amoena*	+	+

Cont'd.

Appendix. (Continued)

Species	Present in both studies	Difference in abundance burned vs. unburned[2]
Brown-headed cowbird[3] *Molothrus ater*		
Cassin's finch *Carpodacus cassinii*	+	+
Red crossbill *Loxia curvirostra*		
Pine siskin *Carduelis pinus*	+	-
Evening grosbeak *Coccothraustes vespertinus*	+	-

[1] No fire severity estimate was given by Raphael et al. (1987) other than the Donner Ridge fire "consumed about 18,000 ha of pine-fir forests," a "few scattered mature pines and firs survived," and the fire area was "dominated by brush (primarily Ceanothus velutinus) and by regenerating conifers" (thus probably a high severity burn).

[2] (+) indicates species was significantly more abundant in burned, (-) means it was significantly less abundant, and NS means non-significant (P>0.05; Burnett et al. 2012).

AUTHOR QUERY FORM

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AU:1, page 380	should the brackets really be between "n" and "o"? Kindly advise.	☐
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Chapter 13

c0065 Flight of the Phoenix: Coexisting with Mixed-Severity Fires

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s0010 ECOLOGICAL PERSPECTIVES ON MIXED-SEVERITY FIRE

p0090 We have presented compelling evidence of fire's beneficial ecological role mainly in western North America but with relevant case studies in other regions. Even though most people recognize the importance of maintaining fire on the landscape, few realize the myriad ecosystem benefits associated with large fires of mixed severity. Habitat heterogeneity, which may be maximized by mixed-severity fire that includes large patches of high severity, and the successional mosaic such fire creates, is one of the most dependable predictors of species diversity (Odion and Sarr 2007, Sitters et al., 2014). This ecological tenet has yet to be fully realized in management circles. If such fires are operating within historical bounds, then ecosystems will remain resilient to them; indeed, deficits of these fires relative to the natural range of variability, in places such as montane forests of western North America, are degrading to fire-dependent biodiversity (Odion et al., 2014a; Sherriff et al., 2014). This is particularly the case when reductions in fire extent and/or severity occur in combination with forest management practices, such as postfire logging, that undermine development of complex early seral forests (Chapter 11).

p0095 Natural heterogeneity in vegetation types, stand structures, and successional age classes at all spatial scales and environmental settings is emerging as a strategy for enhancing forest ecosystem resilience to climate change, at least in North America (Moritz et al., 2014). This will help ensure that there will be enough habitat for species with varying postfire habitat requirements. The fire dynamic is changing in places, however, with climate change now poised in some systems to recalibrate fire behavior (Chapter 9). With the addition of ongoing pre- and postfire logging in forests and other development pressures, particularly in shrublands, this is having a combined negative impact on native biodiversity associated with both complex early seral and old-growth forest and chaparral ecosystems (e.g., Chapters 2–5).

s0015 Beneficial Fire Effects Often Take Time to Become Fully Realized

p0100 In general, for ecological acceptance of postfire landscapes to translate into improved management practices, as a prerequisite fire ecologists, land managers, and the general public all must recognize both pre- and postfire landscapes as irreplaceable habitat for fire-associated biodiversity. To a large extent, this depends on how one views the postfire landscape.

p0105 When considering the effects of fire, patience is clearly a virtue; postfire processes may take years, decades, or longer to unfold. However, land managers often rely on quick indices to assess fire effects, and this can have negative consequences. For instance, in the western United States, the US Forest Service's "burn area emergency response" (http://www.fs.fed.us/eng/rsac/baer/; accessed February 22, 2015) uses satellite images and other geospatial data in real time to classify soil "damages" immediately after fire. Similarly, the US Forest Services' Rapid Assessment of Vegetation Condition (RAVG) after Wildfire (http://www.fs.fed.us/postfirevegcondition/whatis.shtml; accessed February 22, 2015) provides estimates of "basal area losses" in forests 30-45 days following fires >400 ha. We saw in Chapter 11 that these types of rapid assessments can overestimate tree mortality given their immediate timeline compared with the delayed response of fire-affected trees. In forests, particularly pine and mixed conifer, this can lead to premature conclusions about fire "damages" and fire "catastrophes," as well as erroneous notions about high-severity fire patch size, along with a rush to "take action" at any cost and to advance "restoration" or "recovery" approaches that do far more harm than good (Box 13.1; see also DellaSala et al., 2014; Hanson, 2014).

p0110 Notably, differences in whether postfire vegetation is viewed as fuel or habitat (Haslem et al., 2011) most often are at the heart of heated conflicts between natural resource managers and conservationists. Witness these polar opposites: fire suppression (including both mechanical thinning and actions to halt active fires) versus [illegible]rn approaches for wildlife habitat (Chapter 12); postfire logging versus the pulse of biological legacies (Chapter 11); thinning versus habitat for closed-canopy species; and reseeding/replanting and shrub removal versus b0010 **BOX 13.1 Rapid Assessment of Vegetation Condition after Wildfire "Treatments" as Defined by the US Forest Service**

p0010 According to the US Forest Service RAVG assessments, the term *treatment* "describes any of a set of management activities that can assist the prompt recovery of forestlands. Management actions include any combination of live, dead, and dying wood removal, or disposal (with or without commercial value) by any feasible method, including but not limited to logging, piling, masticating, and burning, for site preparation. In addition, planting, seeding, and monitoring for natural regeneration without site preparation are appropriate management activities designed to foster the prompt recovery following wildfire. Treatments also include follow up activities to control vegetation that is believed to compete with desired trees during the early establishment period, usually 1 to 5 years after establishment, using any viable method that meets Land and Resource Management Plan direction."

the montane chaparral component of complex early seral forest. Where one stands on this debate can be a matter of principle and perspective, but can also stem from a lack of a comprehensive understanding of the effects of mixed-severity fire and successional processes after fire (see, e.g., Chapters 2–5). Further, while the public may consider fire to be a necessary change agent (see "Understanding the Public's Reaction to Fire," below), this seems to be tempered by whether fire is operating within "safe limits," constrained by prescribed (or "controlled") fire or reduced in intensity by tree thinning and shrub mastication. While prescribed fire is most appropriate for low-severity, high-frequency fire systems, it is not a replacement for the ecosystem benefits produced by large and higher-severity fire because prescribed fire does not mimic patch mosaics or pulses of biological activity that higher-severity fires provide (Moritz and Odion 2004, DellaSala et al., 2014). Thus, understanding one's perspective is a starting point for potentially settling differences and developing ways to coexist safely and beneficially with fire. Being willing to respond competently to the cognitive dissonance created when perspectives do not align with new scientific information is also vital to the development of successful and ecologically sound fire management strategies (e.g., Chapter 7).

s0020 13.2 UNDERSTANDING THE PUBLIC'S REACTION TO FIRE

p0115 If ecologists and conservationists want a new discourse on fire that improves ecological understanding and fire management practices, then informed and sustained communications with the public, land managers, the media, and decision makers are vital. A common understanding is needed to move the public and land management agencies from a view of fire as the harbinger of death (Kauffman, 2004) to fire as nature's phoenix. Here we provide some insights from a public poll on fire attitudes in the United States that reaffirms our personal experiences about the prevailing attitudes of the public and of land managers when it comes to fire.

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s0025 Attitudes Toward Fire

p0120 In 2008 The Wilderness Society and The Nature Conservancy got together to construct a 10-year fire communications framework that was informed by a large national sample of public attitudes ($n=2000$ respondents), focus groups in six regions of the United States where fire was a concern, and communications experts (Metz and Weigel, 2008). The task was to develop ecological messaging on fires that would "complement Smokey Bear's message" about being careful with fire.

p0125 Based on a summary of the survey findings, important messages on fire can be gleaned from survey data, some of which are remarkably aligned with fire ecology, whereas others are at odds with basic ecological principles. Most notably, the poll demonstrated the public's sophistication regarding the role of fire in ecosystems, but it was clearly tempered by safety concerns (Smokey Bear), notions regarding the importance of "controlled" burns, and a desire to let "some" fires burn in "natural areas." Education (higher levels) was associated with positive attitudes toward fires, and gender was a factor, with men being more risk tolerant and women more risk averse. Some of the poll's most relevant findings are displayed in Box 13.2. We

b0015 **BOX 13.2 Key Findings on Public Fire Attitudes from the Study by Metz and Weigel (2008)**

u0010 • Some fires can be beneficial, and a history of fire suppression has led to more large and destructive fires. (*Note that dramatic changes in fire behavior actually are associated with very few forest types in western North America* (Odion et al., 2014a)).

u0015 • Strong negative emotional reactions to fire persist based on safety issues (most view fire as "scary").

u0020 • Public understanding of fire's ecological role has increased over time.

u0025 • Public concerns about wildfire rank very low compared with other conservation issues.

u0030 • The most significant fire concerns pertain to effects on people and firefighters rather than ecosystem benefits.

u0035 • Allow fire teams to use "controlled burns" when and where doing so will safely reduce the amount of fuel for fires (*controlled burns are most relevant in low-severity rather than mixed-severity systems*).

u0040 • Cut and remove overgrown brush and trees in natural areas that act as fuel for fires (*this is largely true for low-severity systems, not higher-severity fires that are largely controlled by extreme weather*).

u0045 • Allow naturally started fires that do not threaten homes, people, or the health of natural areas to take their natural course, rather than putting them out.

u0050 • Shift some government funds from putting out practically all fires to proactively cutting and removing overgrown brush and trees and using controlled burns to reduce the amount of fuel for fires (*removing brush/trees and controlled burns are mostly ways to reduce fire severity in low-severity systems*).

also highlight in parentheses those beliefs that seem to be at odds with the ecological literature on mixed-severity fires.

p0130 Communication experts then advised the conservation groups that successful fire messaging should have the following five fundamental communication themes:

o0010 **1.** Protect people, property, and communities
o0015 **2.** Safeguard the health and regeneration of natural areas
o0020 **3.** Safely manage controlled burns to clear fuels (*this management is appropriate in low-severity systems only during the natural fire season*)
o0025 **4.** Save taxpayer money through controlled burns
o0030 **5.** Protect air and water by protecting the health of forests and natural areas and giving plants and wildlife the exposure to fire they need to survive

p0160 From focus groups and polling results, according to communication experts the following cogent messages are likely to reach the public:

u0075 • Safety is always the number one priority when it comes to fire. By putting out every single fire, however, we are actually creating more dangerous conditions (*in western North America, higher-severity fires are operating at an historical deficit*). Using controlled burns to thin out overgrowth and carefully managing natural fires help ensure the safety of neighborhoods in outlying areas.
u0080 • Forests and natural areas are important to our health; they act as natural filters to give us clean air and are the source of clean drinking water. We must ensure the health of forests and natural areas by allowing some fires to take their natural course.
u0085 • Taxpayer money is being wasted putting out fires that are far from people and their property. A far more cost-effective approach is to use controlled burns to prevent large, severe fires from spreading into areas where people live and to allow some fires to take their natural course (*and they are ecologically inappropriate when applied outside the natural fire season*).

p0180 For higher-severity fires, a good portion of this messaging may work to bridge the divide between science and public attitudes, whereas some of the recommendations of the communications experts in 2008 (refer to the italicized text in the parentheses above) do not incorporate the ecological importance of maintaining, and managing for, complex early seral forest created by mixed-severity fires. In particular, the poll's findings that fire safety matters most is still very much relevant; thus putting out fires that are dangerous to human communities is still of primary importance. From a safety standpoint, Smokey Bear's cautionary fire safety tale needs to be updated so that the focus of fire management is ~~clearly~~ on creating "defensible space" around homes, the home ignition zone (HIZ), and introducing land use zoning to allow fire to run its course unimpeded in natural areas under safe conditions (Making Homes Fire Safe, see below). And, while the poll found the public generally agreed that fire is necessary in natural areas, how far this tolerance would go in relation to large or higher-severity fires is unclear given that the poll's questions were ~~clearly~~ geared toward low-severity fires that can be either "controlled" or suppressed (through thinning or the use of fire retardants). Notably, in Chapter 12 we discussed how runaway expenditures in fire suppression have been ecologically damaging and fiscally irresponsible, and the public seems to agree with these fiscal concerns. In combination with economics, whether public attitudes will change, or are changing, regarding large or higher-severity fires is still unknown; this will require polling that is more specific to these kinds of fires along with enhanced public education (e.g., the videos referenced in the preface) regarding ways to coexist with large fires.

p0185 A core message—and one that will most certainly be difficult for much of the public to accept despite being fact based—is that large fires in any given location each year, at least in western North America, cannot be stopped no matter what we do. We at least need to be honest about that and clearly state the damages that can ensue from large-scale pre- and postfire management that attempts to control large, mainly climate-driven fires that are uncontrollable. We also need to clearly communicate to the public the current state of scientific knowledge regarding the ecological benefits and values of the habitats created by mixed-severity fire. This is especially so given the still all-too-common notions that such areas have been categorically damaged by fire, which in turn leads to misguided assumptions that ~~such areas~~ are in need of "restoration" or "recovery" management actions.

s0030 13.3 SAFE LIVING IN FIRESHEDS

p0190 Based on public attitudes toward fire there ~~clearly~~ are important challenges to coexistence with fire. These can be overcome, however, if we not only increase public education about current fire ecology but also act responsibly in reducing risks where they matter most. We note that by far the biggest challenge to coexistence with fire is the explosion of exurban sprawl in many rural communities triggered by those moving out of congested cities.

p0195 A case in point is Kalispell, Montana, the gateway to Glacier National Park. A November 17, 2014, article in *Greenwire*, the online source of information on the environment ("Where property rights are king, development continues despite growing wildfire threat"), reported that during the 1990s the county's population grew at twice the state's average as more and more people seeking a rural quality of life purchased 16-ha "ranchettes" scattered across Big Sky fire country. They were able to do so as a result of lax and often resisted land use zoning standards. Based on data provided by Headwaters Economics (2014), 11,000 houses in this Montana county lie within the wildland-urban interface (where towns, homes, and other built structures abut fire-prone wildlands)—more than any other county in Montana—and this number is growing at a phenomenal rate. As reported in the online article, public attitudes included the notion that fire will not directly affect them and strong views about private property rights (i.e., "don't tell me what to do on my land"). Some of the same people vocally oppose government actions in general then demand that public money be spent to remove "fuels" from wildlands. In essence, the lack of homeowner fire risk reductions and inappropriate fuel treatments is setting in motion the perfect storm of land use and fire conflicts.

p0200 To minimize these kinds of conflicts, landowners need to practice fire-safe (also known as "fire-wise" in the United States) planning to protect home structures. We suggest that landowners first declare a common "fireshed" boundary, as they do for watersheds. Firesheds are multidimensional spaces. They begin at the scale of a watershed and encompass the residential community with similar fire risks (Figure 13.1a). Within a fireshed, homeowners can take fire risk reduction measures together (preferably) or on their own (Figure 13.1b).

s0035 Making Homes Fire-Safe

p0205 Probably no research results are as relevant to fire safety science than those of Dr. Jack Cohen, whose seminal fire safe research recommendations are now standard risk reduction measures taken by many homeowners[1] and have caught on with risk-averse insurance companies[2]. The work of Syphard et al. (2012, 2014) on home loss in chaparral systems of southern California is strikingly similar.

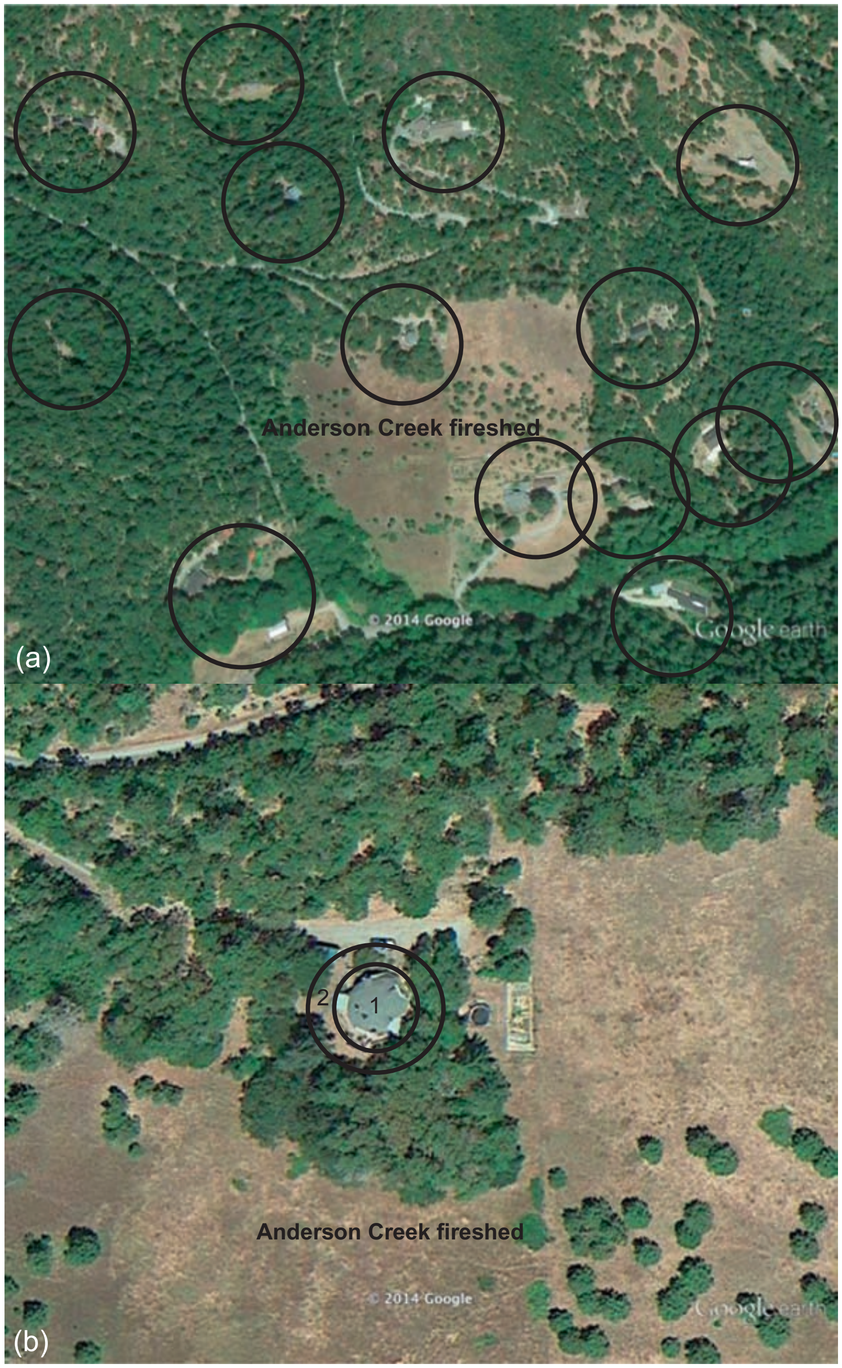
p0210 According to Dr. Cohen, fire planning within an HIZ begins with defensible space nearest the home. Notably, research on HIZ risks shows that homes whose owners reduced vegetation and flammables within 10-18 m of the structure and built with nonflammable roof materials had an 86% (Foote, 1996) to 95% (Howard et al., 1973) "survival" rate when fires swept through an area (cf. Syphard et al. (2014) for more recent and similar home structure protection distances). Combined with home fire simulations by the insurance industry (http://www.extension.org/pages/63495/vulnerabilities-of-buildings-to-wildfire-exposures#.VHUr00snRNs; accessed February 15, 2015), Box **13.3** provides measures that are most critical for living safely in firesheds.

p0215 An example from a town in Idaho during an intense 2007 fire is instructive regarding the importance of the HIZ and fireshed management. As the *Idaho Statesman* newspaper reported (Druzin and Barker, 2008):

dq0020 *We spend billions attacking almost every wildfire, but scientists say that's bad for the forest, can put firefighters in unnecessary danger and doesn't protect communities as well—or as cheaply—as we now know how to do. A wall of fire barreled through the forest with a jet-engine roar near Secesh Meadows last August, and local fire chief Chris Bent knew his work was about to be tested.*

np0010 1. http://www.firewise.org/wildfire-preparedness/firewise-toolkit.aspx?sso=0; accessed November 25, 2014.

np0015 2. http://www.extension.org/pages/63495/vulnerabilities-of-buildings-to-wildfire-exposures#.VHUr00snRNs; active November 26, 2014.



f0010 **FIGURE 13.1** (a) Google Earth image of the Anderson Creek watershed and community fireshed in Talent, Oregon, showing a housing development (circled; the center house is depicted in b). Most members of this community reduced lower-strata fuels via thinning small trees in the surroundings, although tree densities are beginning to fill in and require repeat treatments. (b) Two fire-safety zones where the landowner built with fire-resistant material in the inner most zone (home ignition zone 1) and cleared most vegetation within a 10 m radius around the structure (zone 2). Tree crowns are touching in zone 2; however, lower branches were pruned to 3 m, and there are few ladder fuels to carry fire from the ground into tree crowns. Downslope grasses may pose a fire hazard but may not crown out given the precautions taken in zones 1 and 2.

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b0020 **BOX 13.3 Prudent Fire Risk Reduction Measures for Homeowners**

u0055
- Build homes with noncombustible roof covering and siding; keep roof and gutters clear of leaves/needles; keep firewood away; keep vegetation adjacent to homes to a minimum; cut overhanging limbs of trees closest to the home; and install ember-resistant attic vents.

u0060
- Clearing vegetation within 5-20 m of a home is the most effective treatment: Carefully space plants, reduce wood plant cover to <40% around the structure, and use varieties that grow low and are free of resins, oils, and waxes that burn easily; mow the lawn regularly and prune trees up to 3 m from ground; space conifer crowns ~3 m apart and remove lower limbs; trim back trees overhanging the house; create a "fire-free" area within 1.5 m of the house using noncombustible landscaping; remove dead vegetation; use fire-resistant furniture; remove firewood and propane tanks; and water plants or use xeriscaping.

u0065
- Additional measures include low-growing, well-irrigated, and relatively noncombustible vegetation in low planting densities; a mix of deciduous and conifer trees; fuel breaks like driveways and gravel walkways and lawns.

u0070
- Treatments >30 m from the home structures offer no additional protection (Syphard et al., 2014).

Flames danced atop lodgepole pines, smoke darkened the sky, and residents of the tiny mountain hamlet north of McCall prepared for the worst. Just a month earlier, a forest fire had burned 254 homes near Lake Tahoe and the 2007 fire season appeared ready to claim its next community. But as the raging East Zone Complex fire reached the cluster of loosely-spaced homes, the flames dropped to the ground, crackling and smoldering. The fire crept right up to doorsteps. But without the intense flames that spurred the fire just moments before, no homes burned—a feat fire managers attributed largely to Bent's push to clear flammable brush from around houses in the community. "It just blew through the area," Bent said. "We were well prepared." The town's ability to withstand a frontal assault by a major wildfire demonstrates what fire behavior experts have been saying for more than a decade. Clearing brush and other flammables and requiring fireproof roofs will protect houses even in an intense wildfire—without risking firefighters' lives. More provocatively, the research suggests that fighting fires on public lands to protect homes is ineffective and, in the long run, counter-productive. It is also far more expensive.

p0225 Importantly, clearing vegetation nearest a home is not enough, as fire risk reduction also needs to include the home structure itself (Figure 13.2). This is often missed in discussions about homeowner fire safety, and it is a crucial step in responsible fire risk reduction, as we illustrate in the following examples.

p0230 In a recent research paper concerning why homes burn in wildfires, Syphard et al. (2014) concluded that geography is key: where the house is located and where houses are placed on the landscape. Syphard and her coauthors gathered data on 700,000 addresses in the Santa Monica Mountains and part of San Diego

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f0015 **FIGURE 13.2** Homes burn because they are flammable. Many homes with adequate defensible space still burn in wildland fires because embers land on flammable materials around the home or enter through openings such as attic vents. These two homes burned during the 2014 Poinsettia Fire in Carlsbad, California, despite fire-safe landscaping, a firewall, and thinned wildland vegetation. Focusing exclusively on wildland vegetation clearing ignores the main reasons homes burn: they are flammable. *(Photo credit: Richard W. Halsey.)*

County. They then mapped the structures that had burned in those areas from 2001 to 2010, a time of significant wildfire activity in the region. Buildings on steep slopes, in Santa Ana wind corridors, and in low-density developments intermingled with wildlands were the most likely to have burned. Nearby vegetation was not a major factor in home destruction.

p0235 Looking at vegetation growing within roughly 800 m of structures, Syphard et al. (2014) concluded that the exotic grasses that often sprout in areas cleared of native habitat like chaparral could be more of a fire hazard than shrubs. Interestingly, they found that homes that were surrounded mostly by grass actually ended up burning more than homes with higher fuel volumes such as shrubs.

p0240 Similarly, during the 2007 Witch Creek Fire (San Diego County, CA), houses in Rancho Bernardo started burning by ember contact when the fire front was nearly 6 km away. Two-thirds of the burning homes were set on fire by embers (Maranghides and Mell, 2009).

p0245 During the 2007 Grass Valley Fire near Lake Arrowhead in California's San Bernardino Mountains, approximately 199 homes were destroyed or damaged. This was despite the fact that the US Forest Service had thinned the surrounding forest. The main cause of the losses was that individual homeowners failed to understand that vegetation management is only one part of the fire risk reduction equation. Fire will exploit the weakest link—and it did so in Grass Valley. In the detailed report of the fire, Forest Service researchers (Rogers et al., 2008) concluded: "Post-fire visual examination indicated a lack of substantial fire effects on the vegetation and surface fuels between burned homes. Lack of surface fire evidence in surrounding vegetation provides strong evidence that house-to-house ignitions by airborne firebrands were responsible for many of the destroyed homes."

p0250 Investments in making homes and communities fire safe are clearly fiscally prudent and ~~represent~~ responsible homeownership that can save lives and homes by reducing risks to all, especially firefighters. Moreover, proper land use zoning that reduces housing densities in firesheds is key to the survival of home structures over the larger area (Syphard et al., 2014).

p0255 In sum, these recent studies show that overcoming misperceptions about homeowner losses is urgently needed because those misconceptions are a driving factor in many inappropriate fuel reduction projects in wild areas. We hypothesize that with stepped-up planning directed at proper homeowner safety (as demonstrated in the above studies), public attitudes about large and intense fires may begin to shift from fear-based primal responses to more of a neocortex-like awareness of fire as nature's phoenix. This could be tested using before-and-after polling about large, higher-severity fires with and without proper public safety measures in places.

s0040 13.4 TO THIN OR NOT TO THIN?

p0260 One of the most significant challenges involved in changing the way land managers think about fire in the forests is how the US Forest Service views forest fires. The agency is deeply invested in continuing the fire management trajectory of the past—a situation compounded by the budgetary issues associated with the agency's direction of much, and often most, of their tax-based support to selling timber from public lands, and the agency's retention of most of the revenue from such timber sales to fund staff salaries and operations. Though in recent years we have learned much about the ecological benefits of higher-severity fire and the risks to fire-dependent wildlife species from further suppressing these fires, which are deficient in most western US conifer forests (Chapters 1–5), the Forest Service continues to aggressively promote landscape-level mechanical thinning (North, 2012; Stine et al., 2014) and postfire logging (Collins and Roller, 2013) ostensibly to reduce fuels and prevent and mitigate future fire. These forest management policies are promoted based on the assumption that decades of fire suppression have created forests "overloaded with fuel, priming them for unusually severe and extensive wildfires" (Stine et al., 2014; see also North, 2012). The basic concept being articulated by the Forest Service is that, because of decades of fire suppression and "fuel accumulations," we cannot simply allow wildland fires to burn because long-unburned forests will "uncharacteristically" burn almost exclusively at higher severities (North, 2012; Stine et al., 2014). Under this premise, recommendations focus on how to manage forests through logging and fire suppression to further reduce and prevent the significant occurrence of mixed-severity fire (North et al., 2009; North, 2012; Stine et al., 2014). Yet these sources do not include a discussion of the current deficit of these fires in most forests of western North America (Odion et al. 2014a; see also Chapters 1, 2, and 9) or meaningful

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content on the ecological importance of mixed-severity fire for many rare and imperiled wildlife species (Chapters 2–5). Nor do they explore the validity of the basic premise that long-unburned forests will burn much more severely.

p0265 Studies that empirically investigated the "time-since-fire" issue in the Sierra region of northern California and the Klamath Mountains of Oregon and California tended to find that, contrary to popular assumptions, the most long-unburned forests experience mostly low- and moderate-severity fire and do not have significantly higher levels of higher-severity fire than more recently burned forests (Odion et al., 2004, 2010; Odion and Hanson, 2006, 2008; Miller et al., 2012; van Wagtendonk et al., 2012). One modeling study predicted a modest increase in fire severity with increasing time since fire, but the strength of inference was limited by a lack of data for all but long-unburned stands, especially in the largest forest types, such as mixed-conifer forest. Even the most long-unburned forests were predicted to have ~70-80% low/moderate-severity effects (Steel et al., 2015), well within the range of natural variability (see Chapter 1). In fact, long-unburned forests sometimes have the lowest levels of higher-severity fire; understory vegetation and the lower limbs of conifers self-thin as canopy cover increases and available sunlight in the understory decreases with increasing time since fire (Odion et al., 2010). Therefore the argument that we cannot allow more wildland fires to burn without suppression in natural areas is not valid for many dry montane forests in western North America (Odion et al., 2010).

s0045 Problems with Fuel Models and Fire Liabilities

p0270 Government programs that aim to make forests safe places for people to live are based on theory rather than actual evidence about historical forests. As discussed above, the common argument has been that fuels have unnaturally accumulated from fire exclusion and land uses, and if fuels are restored to low levels, fires will burn primarily at low intensity rather than as high-intensity crown fires (e.g., Agee and Skinner, 2005). Thus forests can be restored while also making them safe places to live—a win-win solution that is appealing to the public. Little evidence about actual historical fuel amounts in forests to support this argument was available, however; instead, evidence is mostly based on the idea that frequent fires would have kept fuels at low levels. When records from land surveys before fire exclusion were examined (Baker, 2012, 2014; Baker and Williams, 2015; Hanson and Odion, in press), understory fuels (shrubs, small trees) that would naturally have promoted intense fires were found to have been common and often abundant in many areas, and small trees were dominant, not rare. This direct evidence suggests that fuel treatments would typically have to artificially remove natural shrubs and small trees and adversely alter habitat for native species in a quest to make forests safer places for people to live.

p0275 Fuel reduction also has been overpromised to be effective, using questionable logic and unvalidated models. First, fire intensity in most forest types is

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much more strongly affected by wind than by fuel. High fire-line intensity, the primary fire characteristic that promotes crown fires, is the product of the energy released by burning fuel and the rate of spread of fire (Alexander, 1982). Energy release by fuel varies over perhaps a 10-fold range, however, whereas rate of spread can vary over more than a 100-fold range; thus a high rate of spread caused by strong winds can easily overcome the limited reductions in fuel that are feasible (Baker, 2009). This was confirmed by a recent analysis of the 2013 Rim Fire in California, which concludes: "Our results suggest that even in forests with a restored fire regime, wildfires can produce large-scale, high-severity fire effects under the type of weather conditions that often prevail when wildfire escapes initial suppression efforts. . . . During the period when the Rim fire had heightened plume activity. . . [n]o low severity was observed [in thinned areas], regardless of fuel load, forest type, or topographic position" (Lydersen et al., 2014, p. 333). Second, common fire models used to show that forests would be fire-safe after fuel reductions have an underprediction bias and are not validated. These flawed models include NEXUS, FlamMap, FARSITE, FFE-FVS, FMAPlus, and BehavePlus (Cruz and Alexander, 2010; Alexander and Cruz, 2013; Cruz et al., 2014). The underprediction bias means that these models often predict that fuel reductions would reduce or eliminate the potential for crown fires in forests, when in fact fuel reductions do not achieve this effect. Fixing these models would be difficult and has not yet occurred (Alexander and Cruz, 2013). Also, these models have not been sufficiently tested and validated using a suite of actual fires, in which case they would likely be shown to fail (Cruz and Alexander, 2010). Alternative validated models are available and could be further developed, but they are not being used (Cruz and Alexander, 2010). Further, studies of tree mortality in thinned areas following fire do not typically take into account the mortality caused by the logging itself before the fire, leading to further biased results. Au1

p0280 These concerns should raise red flags about the effectiveness of fuel treatments, as well as issues regarding liability and responsibility. Imagine if a company sold airplanes with identified flawed designs and without adequate test flights, which then crashed. There are thus sound scientific reasons to closely scrutinize government wildland fuel-reduction programs. Meanwhile, we need to be honest and warn the public that living within or adjacent to natural forests prone to burn is inherently hazardous. Only treating fuels in the immediate vicinity of the homes themselves can reduce risk to homes, not backcountry fuel reduction projects that divert scarce resources away from true home protection (Cohen, 2000; Gibbons et al., 2012; Calkin et al., 2013; Syphard et al., 2014).

p0285 Finally, another land management liability that is frequently overlooked when assessing fire-related economic losses is the role of silviculture. For instance, before the 2013 Rim Fire, a significant portion of the Stanislaus National Forest in central California's Sierra Nevada Mountains consisted of even-aged monoculture tree plantations (following past clearcuts) distributed across large landscapes (Figure 13.3). Land managers often claim that clearcutting over large landscapes

f0020 **FIGURE 13.3** Google image of the Stanislaus National Forest, central Sierra Nevada, taken on July 8, 2012, before the August 25, 2013, Rim Fire. The red boundary is where the Rim fire burned. Note numerous clearcuts within the burn area, where the fire later burned intensely. Figure provided by J. Keeley.

like this reduces fire spread, yet based on preliminary findings from the Rim Fire, clearcutting did nothing to stop the fire. In fact, the area with the most clearcutting had the largest contiguous area of high-severity fire of any portion of the Rim Fire (see Figure 13.3 and compare with Figure 11.11). In other areas with large portions of the landscape in tree plantations past clearcutting, fires have a tendency to burn uncharacteristically severely, presumably because of homogenized fuel loads (e.g., Odion et al., 2004). Despite these observations, in postfire assessments land managers rarely discuss this effect or the liabilities it creates for economic losses related to intense burns.

s0050 13.5 FIRE SAFETY AND ECOLOGICAL USE OF WILDLAND FIRE RECOMMENDATIONS

p0290 Based on the ecological importance of higher-severity forest fires (e.g., Reinhardt et al., 2008; DellaSala et al., 2014; Hanson, 2014; Moritz et al., 2014) and home safety concerns (e.g., Cohen, 2000; Headwaters Economics, 2014), there are ways for people to live safely in firesheds and still allow fire to perform its vital ecosystem service. Below we provide some summary recommendations that, if widely implemented, would allow fire to take its natural course (i.e., ecological use of wildland fire) while reducing risks to people.

s0055 Fire Safety Recommendations

u0090 • Prepare to live safely with fire so that it can perform its ecologically beneficial functions. (The bulk of fire risk reduction should occur immediately adjacent to homes.)

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- u0095 Develop negative financial consequences for landowners who increase fire risk within firesheds by not taking precautionary measures versus providing financial incentives for those who reduce risks (e.g., cost sharing for fire safety). As an example, mortgage and/or insurance rates could be increased for high risks from lack of fire safety and discounted for those who practice fire risk management principles. In this manner, planning for home fire safety would become as routine as taking out a mortgage to buy a home.
- u0100 Include HIZ and fire-safe principles in rural land use planning, including zoning restrictions that limit housing densities in firesheds deemed too risky for development.
- u0105 Require mandatory disclosure of fire risks to homebuyers.
- u0110 Have local and state governments contribute to firefighting costs to create a powerful incentive for improved land use planning, including zoning restrictions, which reduce fire suppression needs.
- u0115 Offer technology transfer to local governments and financial assistance to plan communities that are fire safe.
- u0120 Map high-risk areas where fire-safe standards are most prudent within a local county or other land use unit.
- u0125 Discourage rebuilding in the same high-risk place or require that building occurs with risk management conditions.
- u0130 Redirect funding away from backcountry fire suppression and fuel reduction programs and toward aiding willing homeowners in creating defensible space and reducing the ability of homes to ignite.
- u0135 Initiate strategies to reduce human-caused fire ignitions, especially along roadsides. Many wildland fires start along highways and streets.

s0060 Wildland Fire Recommendations

- u0140 Postfire "salvage" logging is especially damaging to complex early seral Au2 forests. If such forests were ecologically valuable or protected before fire, then they should also be recognized as uniquely valuable and protected after fire.
- u0145 Wildlands cannot be fireproofed by suppression (mechanical thinning or aerial retardants) or clearcutting; fuel treatments (thinning) are more likely to work in low-severity frequent fire systems and much less so in mixed- and higher-severity fire systems that tend to burn under extreme conditions, when suppression is least effective.
- u0150 Large fires, including high-severity patches, are the most efficient means of restoring fire-dependent ecosystems and natural heterogeneity where fire has been excluded for decades. When a fire burns under these conditions, fire-dependent communities are therefore restored. This should be encouraged, with public safety assured.
- u0155 The best way to buffer fire-dependent ecosystems from climate change is to increase ecological resilience, particularly in areas where a fire deficit

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exists, by allowing fires to burn naturally under safe conditions. This will require relatively large protected landscapes with proper land use zoning and logging restrictions.

u0160 • Implement strategies to reduce human-caused fires in ecosystems with excessive fire frequencies, such as the chaparral in southern California.

s0065 13.6 LESSONS FROM AROUND THE GLOBE

s0070 Africa

p0380 Of the five communication themes that arose from the polling in North America, the one most applicable to attitudes in sub-Saharan Africa is number 5, a broad statement to protect natural resources for the ecosystems services they provide (see Chapter 8). The public in South Africa, for example, assumes number 3, safety in controlled burns, because the public is already attuned to the widespread use of fire for habitat management, and when accessible, fuel wood is collected for heat and cooking. Of course, the South African public is not deluged by media reports of catastrophic losses caused by wildfire, so items 1, 2 and 4 are not part of a daily discourse in countries where wildfires in large forests are rare and most of the managed habitat is the much thinner type of woodland associated with savanna (see Chapter 8).

p0385 In terms of such issues as woodland thinning (directed silviculture or ad hoc management), in African savanna the public and policy makers are more concerned with maintaining herbivore populations as part of ecotourism and for the love of Africa's "big five" megafauna wildlife species. South Africa practices extensive silviculture, and it often is blended into wilderness areas (Tsitsikama National Forest lies adjacent to extensive tracts of forest plantation, where fire suppression is practiced because of economics of the wood industry). It seems the "fear" of fire so prevalent in North America is absent from rural areas of Africa for multiple reasons, but this results in a more sane approach to fire ecology. In Kruger Park managers learned over time that allowing wildfire is acceptable, and it is now a tool (although not frequent) integrated with controlled burns. They even seek to achieve as hot a fire as they can in certain habitat conditions to clear the invasive vegetation or just to suppress woody growth. The lesson learned in South Africa over 50 years of "experimenting," and from many decades of following the Serengeti system, is that monitoring is critical, and adapting to those results (adaptive management) is imperative.

s0075 Australia

p0390 In Australia prescribed burning is considered a staple part of the land management tool kit and is routinely applied with the aim of reducing the risk of large, unplanned wildfires to property and infrastructure (Clarke, 2008). In some cases fire is applied to the landscape in efforts to "restore" ecosystems or to

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create fine-scaled fire mosaics of mixed successional stages to encourage greater faunal and floral diversity (Bradstock et al., 2005). In response to the perceived need to apply fuel-reduction burns, the Victorian state government implemented a policy that mandated that 5% of the total land area under state jurisdiction be burned each year. This policy did not discriminate fire prescriptions between ecosystems and has been subject to widespread criticism from fire ecologists in Australia; it is currently under review (DELWP, 2015a). Although appropriate fire regimes have positive ecological outcomes in many systems, application of prescribed burning can lead to species declines and in some cases can cause irreversible changes in ecosystem state (Pardon et al., 2003, Pennman et al. 2011, Pastro et al., 2011).

p0395 Recent large wildfires in Australia have spurred new policies to address the growing public concern over the dangers presented by these fires (McLennan and Handmer, 2012; Whittaker et al., 2013). The royal commission that followed the 2009 "Black Saturday" fires suggested the implementation of new policies to encourage clearing around homes and to shift public perceptions toward recognition of bushfires as defensible events (i.e., homes can be effectively protected) that require early planning and avoidance actions (Teague et al., 2010). Residents in areas of high fire risk are now able to clear all vegetation within 50 m of their homes. These new measures, coupled with the 5% burn target, aim to reduce the potential of a repeat of the 2009 fires. This home protection approach is partially supported by science. Gibbons et al. (2012) highlighted that houses with vegetation cleared within 50 m were 70% more likely to survive a fire than those with no clearing. They revealed, however, that there was no effect of fuel reduction burning in nearby state forest or ecological reserves on house preservation following the 2009 fires in Victoria, Australia. Furthermore, in some of the most potentially pyrogenic systems, such as mountain ash forests, fuel reduction burns are rarely applied because moisture levels are normally high, and risk of fire spread is considered unacceptable when conditions are dry (DELWP, 2015b). A growing body of literature indicates that inappropriate fire regimes are contributing to species declines globally (Driscoll et al., 2010). In response to the increased fire risk caused by climate change, policy makers should seek to implement strategies with a proven ability to protect homes, while avoiding ineffective actions that detrimentally impact biodiversity.

s0080 Central Europe

p0400 In central Europe forest fires are relatively infrequent and mainly limited to regions with pine forest plantations growing on sands, gravel-sands and sandstone rocks. Any burned areas are mandatorily reclaimed within just 2 years of their formation; exceptions are possible in forests protected as national parks or nature reserves. The option to request avoidance of logging and replanting is used only rarely, however, and nearly all forests affected by fires are quickly logged and replanted.

p0405 Available evidence suggests that fire-induced bare soil patches, charred trunks, and dead wood resulting from the postfire dieback represent unique nesting resources for numerous species. The areas subject to mixed- and high-severity fires are associated with dynamic assemblages of plant and animal species, many of which are rare or even absent in the surrounding landscape. The burned forests serve as key habitats, particularly for aculeate Hymenoptera associated with cavities in dead wood (such as *Dipogon vechti*). Such cavities are considered limiting nesting resources, and their absence (and targeted removal of any newly emerging snags, which is mandatory by law) causes numerous specialized cavity adopters to be red-listed or extinct. Mounting evidence suggests that specific groups of organisms are strictly dependent on the occurrence of repeated fires. As long as sites of natural disturbances become extremely rare in the intensively cultivated landscape of central Europe, bare soil specialists and species that specialize in cavities of decaying wood will be completely absent where forests are subject to intense cultivation and rigorous dead wood removal. Dead wood thus should be considered an important habitat resource deserving conservation measures. Mosaic management of burned forest sites and retaining charred trunks are suggested as management measures supporting biodiversity at the sites of recent forest fires (Bogusch et al., 2015).

s0085 Canadian Boreal

p0410 There is emerging a new paradigm about the role of fire in the Canadian boreal forest. Historically, it was perceived as a simple system where “catastrophic” fire created landscapes of young, even-aged stands and where species diversity was poor. The reality is much more complex. There is an impressive range of fire cycle estimates—some as long as several centuries—suggesting that for at least part of the boreal forest region the abundance of old-growth forests in pre-industrial times was much greater than expected (see Chapter 8). Associated with these old-growth forests is high understory diversity in black spruce (*Picea mariana*) stands and a number of rare species of nonvascular plants associated with balsam fir (*Abies balsamea*) stands. Similar findings have been made in boreal forests of Europe and Asia.

p0415 At the other end of the disturbance spectrum, there is now compelling evidence showing the importance of early seral burned habitats for the pyro-community, led by saproxylic insects (dependent on dead or decaying wood) and followed by primary cavity nesting birds (see Chapter 8). The retention of a wide range of burn conditions enhances saproxylic insect diversity. A link between this saproxylic community and nutrient cycling has been found, indicating a connection between biodiversity and ecosystem function in Canadian boreal forests. Large fires produce significant pulses of dead wood, which drive biodiversity and ecosystem processes through natural succession over time. Fire skips, or remnants left after large burns, also are critically important for biodiversity, species persistence, and recolonization and ecosystem recovery.

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p0420 For a long time, forest management was driven with a strong focus on timber extraction and developed a jargon that infiltrated the dialect of forestry, with words like "decadent" for old-growth forests, "waste wood" for trees that had been killed by natural disturbances, and "salvage" as the practice used to recover that "wasted" timber. Today, management in the boreal forest is increasingly driven by themes like ecosystem-based management and sustainable development. The new era will require conservation of boreal forests at different ends of the disturbance spectrum from newly created, postfire habitat to multicentury, old-growth forests.

s0090 13.7 ADDRESSING UNCERTAINTIES

p0425 Even though most people recognize the importance of maintaining fire on the landscape, there remain important questions about what might be the optimal postfire conditions for the broad suite of species with varying fire tolerances. For instance, we do not know whether there is a certain amount of burned forest or spatial distribution of burned forest patches, patch sizes, and fire frequencies necessary to maintain species at polar ends of the successional gradient. However, we hypothesize that in large, intact forested landscapes where fire is allowed to burn and logging is restricted (e.g., wilderness areas, large national parks, and other protected ecosystems) there should be ample habitat for all seral species over the long term and the best opportunities for coexistence with fire as a process (see Chapters 3–5). By contrast, in highly degraded landscapes, particularly those close to towns and homes, an optimal condition of recently burned and long-unburned patches is more difficult to ascertain because it may involve tradeoffs for public safety reasons (DellaSala et al. 2004).

p0430 Currently, megafires in western North American forested landscapes burn in mixed-severity patterns and seem to provide the necessary patch mosaics for a broad array of species (Chapters 2–6). Fire-related change of late seral habitat to complex early seral forest (Swanson et al., 2011; DellaSala et al., 2014; Hanson, 2014) has not been a threat to species dependent on such mature forest habitat, particularly given that there is generally much less high-severity fire in mixed-conifer and pine forests of western North America than there was historically (Odion et al., 2014a). Rates of old forest recruitment, as a result of growth, also outpace rates of high-severity fire in old forest by several times (Hanson et al., 2009; Odion and Hanson, 2013; Odion et al., 2014b). The situation is less clear in portions of Australia, however, where fewer vertebrate species have thus far been found to be fire dependent (see Chapters 3 and 4) and there are more species associated with late seral conditions that are especially at risk (Kelly et al., 2015). By contrast, other Australian research found bird species richness to be highest where there is the most successional diversity from higher-severity fire (Sitters et al., 2014) (see Chapter 8). Human-caused fires in North American chaparral, the Great Basin, and many desert ecosystems, which mostly replace stands, have exceeded historical bounds, adversely affecting this diverse shrubland

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community (Chapter 7). Thus, whether or not fire mosaics are correlated with high levels of biodiversity (cf. Martin and Sapsis, 1991 versus Parr and Andersen, 2006; Taylor et al., 2012; Kelly et al., 2015) depends on differences in biogeography, fire histories, land use histories, and life history requirements (including fire tolerances and dependencies) of species over long time lines and large landscapes (e.g., Scott et al., 2014; see Chapters 3–5).

p0435 In addition, climate change introduces uncertainty in how forests will respond to changes in fire extent, longer fire seasons, higher severities in places, how soon the current fire deficit in places will remain that way before exceeding historical bounds, and whether existing deficits will be exacerbated in some forests with increasing precipitation driven by climate change (see Chapter 9). Nonetheless, at least for mixed-severity fire systems there is no magic thinning or suppression bullet to forestall climate-mediated fire changes. Changes in fire behavior are a consequence of human-caused climate change. It is best to treat the cause—climate change—rather than the symptom (fire behavior) if we are truly concerned about climate effects on ecosystems and people.

s0095 13.8 CLOSING REMARKS

p0440 When viewing the natural world, as a matter of perspective, we are reminded of discussions we have often had with foresters regarding how we each see the value of postfire landscapes. Clearly, we see the world differently depending on our professional judgment and value system.

p0445 A professional forester views the fruits of his or her labor, imagining what the future "production" forest will look like after decades of growing wood fiber, and then being frustrated by nature run amuck when the forest goes up in flames.

p0450 For the fire-trained ecologist, the initiating fire is but a glimpse into a vibrant community that begins with a pulse of biological activity and ensures successional events, just one of the many important links to follow in a long chain of ecosystem changes. Even the most charred forest is transformed by fire on one of nature's grandest stages. Among the first actors to arrive on the postfire stage are the biological legacies that provide the supporting foundation for other postfire actors to enter with the passage of time. If we imagine what the stage will look like years after a severe burn (often only 1 year), we see a floral phoenix arising from the ashes, we hear a cacophony of songbirds and drumming woodpeckers, and the rhythmic buzzing of bees and other insects as they go about their business of pollinating the next explosion of flowering plants. Up close and personal, we see tiny native beetle larvae tucked neatly into galleries beneath the outer charred tree bark, wood-boring scorpion wasps recoiling long abdomens after depositing eggs into open crevices in tree bark, centipedes and millipedes working charred humus, and ravenous insect-loving bats and fly-catching birds feasting on all the buzz.

p0455 The postfire landscape is indeed a transformative place if we humans are willing to have the patience to look beyond the brief snapshot in time right after the initiating event. Only then will the postfire esthetic become apparent. Our human world of instant gratification pales in comparison to nature's seemingly infinite horizon. Meticulous observations by trained ecologists too often are drowned out by the noise of a fast-paced society preoccupied with one-size-fits-all solutions, do something at any cost, myopic economic benefits, and a fear-based media blitz of fire catastrophe reporting. But if we wait for the ecosystem actors to emerge in synchronicity, the postfire habitat unveiled is remarkably resilient, brilliant like the mythical phoenix, and even musical if we know how to listen. We hope that we have sufficiently portrayed an ecological awareness for this postfire symphony in the chapters of this book.

p0460 In this closing chapter we also have discussed the importance of education and outreach for a communications framework and improved ecological understanding of fire that follows fundamental ecological and safety principles.

p0465 From a communications standpoint, fire operates very much like an apex predator, thinning out and culling its prey, sometimes in large numbers, sometimes not. Apex predators are indeed vital to fully functioning ecosystems, yet they are either loved or hated based on one's perspective, which simply boils down to either an appreciation for wild things or a fear of being attacked or of losing a commodity. People view fire in much the same way. Decades of public outreach and campaigns in many places (most notably Europe and North America) have shifted public opinion to be more accepting of predators, and even to relish them in national parks and other protected landscapes where predators roam free and tourists flock to witness nature primeval. Clearly, fires, like apex predators, cannot be restricted to inside national parks, as the parks are not big enough to sustain them.

p0470 There is a lesson to be learned regarding the message of fear in both instances: As with predators, the risks of losses to people and property can be successfully mitigated by taking precautionary measures (e.g., just don't feed the bears, and remember to make loud noises while hiking in grizzly bear country!). In the case of fire, public safety of those living in firesheds is based on prudent fire risk reduction that with stepped-up outreach one day may become common knowledge. With a shift in this direction, we envision a move toward fire tolerance, and eventually coexistence, so that fire, in all its severities and forms, can continue to shape ecosystems into the next millennium. This will take a concerted effort of sophisticated and sustained message framing, an infusion of funds for stepped-up education that at least rivals predator-friendly campaigns, a commitment from land management agencies and the media to become more ecologically literate (including replacing Smokey Bear with nature's phoenix), conservation groups to see the value in mixed-severity and not just low-severity fire, and politicians to see the big picture that the postfire landscape has irreplaceable ecological value and is not just a money tree to be ravaged for short-term profit. Then nature's phoenix will truly take flight, reborn out of the ashes of a postfire landscape mosaic that is alive and well!

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Non-Print Items

Abstract

Throughout this book we present a compelling case for the ecological importance of mixed-severity wildfires in forests (though some chaparral systems currently experience too much fire), including, in many cases, megafires from western North America. Stand-replacing fire disturbances are underappreciated natural events that have been shaping fire-dependent ecosystems for millennia, and their ecosystem benefits are being compromised by management actions that carry unintended consequences. Mimicking the spatial, temporal, and structural heterogeneity of these fire effects through management is not possible. Moreover, fire management actions such as forest thinning, mastication, and postfire logging are creating novel fire regimes at the expense of historical ones. Dramatic improvements in fire management and public perceptions of wildfire are needed to accommodate wildfires where they are beneficial. We provide several closing recommendations for addressing public safety concerns and ecological use of fire in natural areas.

Keywords: Fire-dependent forests; Fire safety; Forest thinning; Habitat conservation; Mixed-severity fires; Public attitudes.

Forum: Issues, Management, Policy, and Opinions

Accommodating Mixed-Severity Fire to Restore and Maintain Ecosystem Integrity with a Focus on the Sierra Nevada of California, USA

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ABSTRACT

Existing fire policy encourages the maintenance of ecosystem integrity in fire management, yet this is difficult to implement on lands managed for competing economic, human safety, and air quality concerns. We discuss a fire management approach in the mid-elevations of the Sierra Nevada, California, USA, that may exemplify similar challenges in other fire-adapted regions of the western USA. We also discuss how managing for pyro-

RESUMEN

La política de fuego actual fomenta la permanencia de la integridad del ecosistema en el manejo del fuego. Sin embargo esto es difícil de implementar en tierras manejadas con multiplicidad de objetivos (económicos, de seguridad humana, o relacionados con la calidad del aire). Nosotros debatimos un enfoque sobre el manejo del fuego en las elevaciones medias de la Sierra Nevada en California, EEUU, que podría extenderse a casos similares que ocurren en otras regiones adaptadas al fuego en el oeste de los EEUU. También discutimos como el manediversity through mixed-severity fires can promote ecosystem integrity in Sierran mixed conifer and ponderosa pine (*Pinus ponderosa* Laws) forests. To illustrate, we show how coarse-filter (landscape-level) and complementary fine-filter (species-level) approaches can enhance forest management and conservation biology objectives as related to wildfire management. At the coarse-filter level, pyrodiverse mixed-severity fires provide landscape heterogeneity. Species and ecosystem characteristics associated with pyrodiversity can be maintained or enhanced by accommodating moderately severe fires, which hasten restoration by recreating a complex vegetation mosaic otherwise at risk from suppression. At the fine-filter level, managers can select focal species and species of conservation concern based on the degree to which those species depend on fire and accommodate their specific conservation needs. The black-backed woodpecker (*Picoides arcticus* [Swainson, 1832]) is an ideal focal species for monitoring the ecological integrity of forests restored through mixed-severity fire, and the California spotted owl (*Strix occidentalis occidentalis* [Xantus de Vesey, 1860]) is a species of conservation concern that uses post-fire habitat mosaics and is particularly vulnerable to logging. We suggest a comprehensive approach that integrates wildland fire for ecosystem integrity and species viability with strategic deployment of fire suppression and ecologically based restoration of pyrodiverse landscapes. Our approach would accomplish fire management goals while simultaneously maintaining biodiversity.

jo para lograr la pirodiversidad a través de fuegos de severidad mixta podrían promover la integridad del ecosistema boscoso de coníferas mixtas de estas Sierras y de bosques de pino ponderosa (*Pinus ponderosa* Laws). Para ilustrarlo, mostramos cómo los enfoques a gran escala (a nivel de paisaje) y complementariamente a pequeña escala (a nivel de especie), pueden favorecer los objetivos del manejo forestal y de la conservación biológica en relación al manejo del fuego. A nivel de gran escala, la pirodiversidad de los fuegos de severidad mixta resultó en la heterogeneidad del paisaje. Las características de las especies y del ecosistema asociadas a la pirodiversidad pueden ser mantenidas o favorecidas cuando se admite la ocurrencia de algunos fuegos moderadamente severos, los cuales aceleran la restauración recreando un mosaico complejo de la vegetación, lo que no ocurriría en caso de ser suprimidos. A nivel de pequeña escala, los gestores pueden seleccionar especies focales y especies relacionadas con la conservación, basados en el grado sobre el cual esas especies dependen del fuego y se adaptan a sus necesidades de conservación específicas. El pájaro carpintero negro (*Picoides arcticus* [Swainson, 1832]) es una especie focal ideal para monitorear la integridad ecológica de los bosques restaurados a través de fuegos de severidad mixta, y la lechuza moteada de California (*Strix occidentalis occidentalis* [Xantus de Vesey, 1860]) es una especie de interés para la conservación que utiliza mosaicos de hábitat post fuego y es particularmente vulnerable al aprovechamiento forestal. Nosotros sugerimos un enfoque comprensivo que integre los fuegos naturales para la integridad del ecosistema y la viabilidad de las especies, con la implementación estratégica de la supresión del fuego y la restauración de paisajes pirodiversos basada en principios ecológicos. Nuestro enfoque podría cumplir con los objetivos de manejo del fuego, manteniendo simultáneamente la biodiversidad.

Keywords: coarse filter, ecosystem integrity, fine filter, focal species, mixed-severity fire, pyrodiversity, Sierra Nevada, species of conservation concern

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INTRODUCTION

Pyrodiversity, the mean spatial variability in wildfire effects, results in complex post-fire vegetation mosaics that are associated with high levels of biodiversity. Large fires that produce a variety of severities (i.e., mixed-severity fires) in ponderosa pine (*Pinus ponderosa* Laws) and mixed-conifer forests of the western USA are increasingly recognized for their importance in generating pyrodiverse landscapes (e.g., Perry *et al.* 2011, Williams and Baker 2012, Odion *et al.* 2014, Marcoux *et al.* 2015). Top-down processes such as extreme fire weather, regional climate (which influences fuel moisture and ignitions), and bottom-up processes such as topographic relief, vegetation, and disturbance history govern the distribution and size of fire patches in mixed-severity fires (Perry *et al.* 2011, Dunn and Bailey 2016). Regional drought, high winds and temperatures, and other factors (e.g., surface fuel loading, crown base height, and crown bulk density; Cruz and Alexander 2010) drive crown fire behavior in these systems, producing small and large patches of high tree mortality within a predominantly surface-fire matrix of mostly surviving trees. Mixed-severity fires therefore generate complex stand structures and landscape heterogeneity—characteristics not typically produced by low-severity fire (Table 1). Low-severity fire, while also important ecologically, is preferred by many managers due to lower risks to economic values. Here, we focus on mixed-severity fires because they have received less attention by managers, but they result in pyrodiverse landscapes (DellaSala and Hanson

Table 1. Pyrodiversity attributes produced by mixed-severity fires associated with high levels of biodiversity and ecosystem functions.

Mixed-severity fire attribute	Ecological importance
Landscape heterogeneity	Habitat for wide array of species—early to late seral associates Mixture of foraging and nesting habitat for spotted owls
Complex stand structures	Biological legacies: large snags, down wood, shrubs, flowering plants Habitat for black-backed woodpeckers
Food web dynamics	Complex trophic structure connected across seral stages with abundant food for certain taxa (e.g., beetle larvae for woodpeckers) Pulsed nutrient inputs (aquatic and terrestrial)
Ecosystem processes	Nutrient cycling and soil nutrient exchange, energy transfer from live to dead material, pollination, predator-prey (owls-mice)
Species composition	Rich and varied, compared to old growth

2015). We demonstrate how ecosystem integrity can be met by managing for pyrodiverse landscapes mediated by mixed-severity fires in the biodiverse region of the Sierra Nevada, California, USA.

Although it is the subject of ongoing research and debate (Odion *et al.* 2016), it has been suggested that mixed conifer, ponderosa pine, and Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.) forests in this region historically experienced a mix of fire severities, including areas of high overstory tree mortality (DellaSala *et al*. 2014, Stevens *et al.* 2016). There is considerable variability in reported proportions and sizes of high-severity fire patches, with the greatest differences found in relatively smaller study areas or studies in which shorter time periods were analyzed (Table 2). High-severity patches commonly ranged from 0.4 ha to >50 ha, but the historical frequency of patches >1000 ha is still debated (e.g., Baker 2014, Stevens *et al.* 2016). While uncertainty remains on some issues, there is general agreement that most forests of the Sierra Nevada currently have less high-severity fire, in terms of annual or decadal area burned, than they did historically, prior to fire suppression (Mallek *et al.* 2013, Odion *et al.* 2014, Baker 2015). Additionally, drier low-elevation pine forests burned most frequently at low to moderate severity (Stephens *et al.* 2015), but those fires also contained variably sized high-severity patches (Leiberg 1902, Baker 2014, Hanson and Odion 2016*a, b*). Douglas-fir (*Pseudotsuga menziesii* Mirbel) (Odion *et al.* 2014) and Sierra lodgepole pine (*Pinus contorta* var. *murrayana* Grev. & Balf.) forests experienced mixed-severity fires as well (Caprio 2008).

Tree mortality is also an important component of mixed-severity fire effects characterized mostly by low-mortality levels (0% to 20% tree basal area), highly variable moderate-mortality levels (20% to 70%), and high-mortality levels (>70% tree mortality) (Perry *et al.* 2011; Figure 1). Agee (2005) noted that mixed-severity fires are not merely an intermediate state between low and high severity but, rather, are a unique type of disturbance that warrants careful study by ecologists.

While there are winners and losers in the immediate aftermath of any disturbance event, the net effect of mixed-severity fire is that it

Table 2. Historical fire severity proportions and maximum high-severity fire patch sizes in mixed-conifer and ponderosa pine forests, Sierra Nevada management region.

Study	Study area size (ha)	Fire severity (%)			Time period (yr)	Maximum high-severity patch size (ha)
		Low	Moderate	High		
Beaty and Taylor (2001)[1]	1 587	1 to 60	14 to 47	6 to 86	43	no data
Bekker and Taylor (2001)	2 042	2 to 4	35 to 44	52 to 63	75	no data
Baker (2014)	330 000	13 to 26	42 to 48	31 to 39	110	9 400
Hanson and Odion (2016*a,b*)	65 296	no data	no data	22	60	697
Leiberg (1902)[2]	1 193 166	no data	no data	20	100	~16 000
Stephens *et al.* (2015)	11 500	no data	no data	1 to 6	~20 to 30	no data

[1] Fire severity percentages vary by slope position and aspect.
[2] Does not include high-severity fire patches <32.4 ha, so actual percent high-severity fire would be higher, if patches <32.4 ha had been mapped. Historical high-severity fire mapped polygons are from Leiberg (1902), and analysis of high-severity fire percent by forest type is from Hanson (2007), based on Leiberg (1902).

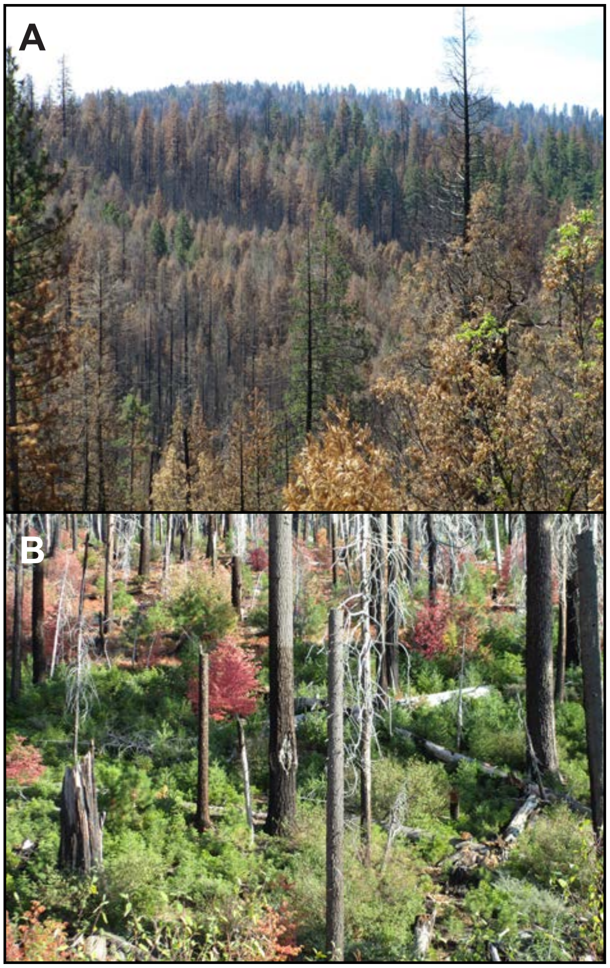

Figure 1. (A) Landscape view of mixed-severity fire effects in the Rim Fire 1 year post fire. The spatial pattern of fire severity patches and patch sizes results in a pyrodiverse landscape that provides habitat for wildlife across a post-fire vegetation gradient of low or unburned vegetation patches to severely burned vegetation patches. (B) Close-up of large patch of complex early seral forest created by high-severity fire in juxtaposition with abundant and varied "biological legacy" trees (complex structures, such as snags, logs, and shrubs that survive fire). Photos by C. Hanson.

provides a mosaic of habitat for a broad suite of species. For instance, songbirds have high levels of species richness and abundance in post-fire vegetation at mid elevations (Fontaine *et al.* 2009, Tingley *et al.* 2016). Black-backed woodpeckers (*Picoides arcticus* [Swainson, 1832]), mountain bluebirds (*Sialia currucoides* [Bechstein, 1798]), tree swallows (*Tachycineta bicolor* [Vieillot, 1808]), and numerous shrub-nesting birds preferentially use recently burned forests in the Sierra Nevada and other regions, presumably due to increased shrub cover and presence of snags (Fontaine *et al.* 2009, DellaSala *et al.* 2014, Hutto *et al.* 2015, Tingley *et al.* 2016). California spotted owls (*Strix occidentalis occidentalis* [Xantus de Vesey, 1860]) and olive-sided flycatchers (*Contopus cooperi* [Nuttall, 1831]) forage in severely burned patches where prey are abundant, and nest in unburned to moderately burned portions of the same fire mosaic (Bond *et al.* 2009, 2016; Hutto *et al.* 2015; Comfort *et al.* 2016). Bats make use of high snag densities (Buchalski *et al.* 2013) and fire-recruiting plants are associated with severely burned patches (Donato *et al.* 2009). Even mature-forest carnivores such as the Pacific fisher (*Pekania pennanti* [Erxleben, 1777]) actively forage in severely burned patches (Hanson 2015).

The high-severity patches within the mixed-severity mosaic provide a unique pulse of biological legacies—complex structures such as snags, downed logs, and native shrub patches from seed that survive fire and that are important in connecting seral stages through time (Franklin *et al.* 2000, Fontaine *et al.* 2009, Donato *et al.* 2012, DellaSala *et al.* 2014). The economic value of large dead and live trees within these patches means that commercial trees are most often targeted for harvest soon after fire. In addition, nursery-grown young trees are planted soon after fire and, to promote the crop of young trees, herbicides are often sprayed to kill competing vegetation (Lindenmayer *et al.* 2008, 2017). Logging slash from post-fire logging may contribute to subsequent fire behavior (Donato *et al.* 2006, Thompson *et al.* 2007), as can the fuel array of densely planted even-aged trees (Odion *et al.* 2004).

On public lands, current fire policy promotes thinning over large landscapes (e.g., USDA Forest Service 2002, US Congress 2003, USDA Forest Service 2009, US Congress 2015), which is costly (Schoennagel and Nelson 2011), infeasible over large areas (Calkin *et al.* 2013, North *et al.* 2015*a*, Parks *et al.* 2015), and largely ineffective under extreme fire weather conditions (Lydersen *et al.* 2014, Cary *et al.* 2016). For instance, from 2001 to 2008, over 11 million hectares were thinned on national forests (mostly in the western USA) at a cost of more than $6 billion (Schoennagel and Nelson 2011). Mechanical vegetation treatments can cost over $3700 per hectare for each round of thinning (Kline 2004), which would need to be repeated at least every 15 to 20 years to keep flammable vegetation at low levels. Additionally, from 1985 to 2015, suppression costs were more than $25 billion to fight approximately 2 million fires on over 83 million hectares, mostly spent by the Forest Service (Ingalsbee and Raja 2015).

Thus, we concur with others that active management approaches could include more natural fire ignitions (Calkin 2013, Meyer 2015, North *et al.* 2015*b*) or resource objective wildfires (Meyer 2015) in which fire is put back on the landscape to hasten the process of forest restoration (Moritz *et al.* 2014, Moritz and Knowles 2016). This would also help to meet fire and fuels objectives and allow managers to better accommodate mixed-severity fire effects for ecosystem integrity (Meyer 2015, Dunn and Bailey 2016). We suggest that an ecosystem integrity approach is not inconsistent with current active fuel management on federal lands and may be a cost-effective way to achieve biodiversity goals (North *et al.* 2015*b*), while reducing some of the conflicts associated with extensive fuels-focused approaches—particularly impacts to imperiled species and at-risk ecosystems. We use the definition of ecosystem integrity common in the literature (e.g., Pimentel *et al.* 2000), also adopted by the USDA Forest Service (2012), as the ability of an ecological system to support and maintain a community of organisms that has a species composition, diversity, and functional organization comparable to those of natural habitats within a region.

Our focus is the Sierra Nevada region because of national attention given to so many recent fires therein. We include an example of a fire-adapted species (black-backed woodpecker) that uses high-severity patches, and an imperiled species (California spotted owl) known to decline within intensively managed post-fire landscapes. The Sierra Nevada is one of the most diverse temperate conifer forest regions on Earth and has exceptional levels of plant endemism (Ricketts *et al.* 1999). Approximately half of California's 7000 vascular plant species occur in this region, with 400 considered endemic and 200 rare. High levels of vertebrate richness and endemism also occur. Species composition varies across north-south, east-west, and elevational gradients, resulting in high levels of beta diversity.

Importantly, the 2012 forest planning rule (USDA Forest Service 2012) includes specific provisions for managing public resources to maintain or restore: (1) structure, function, composition, and landscape connectivity; (2) ecological conditions for recovery of imperiled and focal species; and (3) rare and unique habitat types (USDA Forest Service 2012). The National Cohesive Wildland Fire Management Strategy (USDI and USDA 2014) and Sierra national parks (e.g., Yosemite, Sequoia and Kings Canyon) also include multi-faceted approaches that promote greater wildfire ignitions. Though national forest lands compose most of the forested area in California, and are thus our focus herein, significant areas of federal forest in California are managed by the National Park Service (NPS), and a state agency, California Department of Forestry and Fire Protection (CAL FIRE), responsible for decisions and operations pertaining to fire suppression on private and state lands. NPS, like the Forest Service, is required to protect species listed under the Endangered Species Act (ESA), and CAL FIRE is subject to the California state ESA. Thus, our approach to wildfire management can be applied to these agencies and land ownerships regarding decisions about fire suppression and forest management that might impact imperiled or ESA-listed species associated with post-fire landscapes.

STUDY AREA

The Sierra Nevada management region is a 750 km long, north-south oriented mountain range in California composed of granitic rock, and distributed across three ecoregions: Sierra Nevada proper; portions of the Modoc Plateau; and the eastern portion of the southern Cascades (Bailey 1995; Figure 2). The regional climate is mediterranean with cool, wet winters, and warm, dry summers; precipitation generally decreases west to east and north to south (Millar1996).

There are 11 national forests totaling about 4.6 million hectares: Modoc, Lassen, Plumas, Tahoe, Eldorado, Stanislaus, Sequoia, Sierra, Inyo, Humboldt-Toiyabe (western portion), and Tahoe Lake Basin Management Unit. Forest planning is governed by the Sierra Nevada Framework (USDA Forest Service 2004), but the Forest Service is currently revising its forest plans for the Inyo, Sequoia, and Sierra national forests as "early adopters" (i.e., first national forests to test the planning rule) of the 2012 forest-planning rule (USDA Forest Service 2012). Three national parks—Lassen, Sequoia and Kings Canyon, and Yosemite—and several large wilderness and inventoried roadless areas >2000 ha also occur in the region.

Coarse-Filter and Fine-Filter Approaches to Ecosystem Integrity in Mixed-Severity Systems

Managers wishing to maintain ecosystem integrity via naturally ignited fires can do so using a combination of coarse- and fine-filter conservation approaches (Noon *et al.* 2003, USDA Forest Service 2012). Coarse filters invariably include relatively few indicators associated with the larger ecosystem of interest (e.g., major vegetation types or, in this case, different categories of burn severity). Their presence is meant to indicate that essential components of the whole system are intact, and they operate at broad spatial scales such as those associated with large fires (hundreds of square kilometers). Coarse filters are typically used to guide reserve design based on fundamental principles of conservation biology, including spatially redundant reserve complexes representative of the major forest types and fire severities interconnected across large landscapes. To achieve a pyrodiverse landscape, perhaps the best coarse filter would include high-severity fire patches interspersed with fire refugia (unburned areas) and low- to moderate-severity patches.

Fine-filter considerations complement coarse filters by adding site-specific or habitat elements associated with focal species, guilds, or other species groupings (USDA Forest Service 2012). Application of this kind of filter allows managers to evaluate whether habitat and special conservation needs are met through a given management plan, and ground-truth the utility of burn severity maps by linking mapped fire severities to habitat needs of target species. In addition, the approach allows managers to meet national forest planning requirements to monitor and evaluate a small suite of focal species selected to assess the degree to which ecological conditions are supporting the diversity of plant and animal communities within a given planning area (USDA Forest Service 2012). Focal species can, therefore, be used to monitor the integrity of the larger system to which they belong, and researchers (e.g., Seavy and Alexander 2014, Stephens *et al.* 2015, Siegel *et al.* 2016) have suggested using patterns of plant and animal distributions as a passive management strategy to accommodate mixed-severity systems. The Forest Service also now considers species of conservation concern as "a species, other than federally recognized threat-

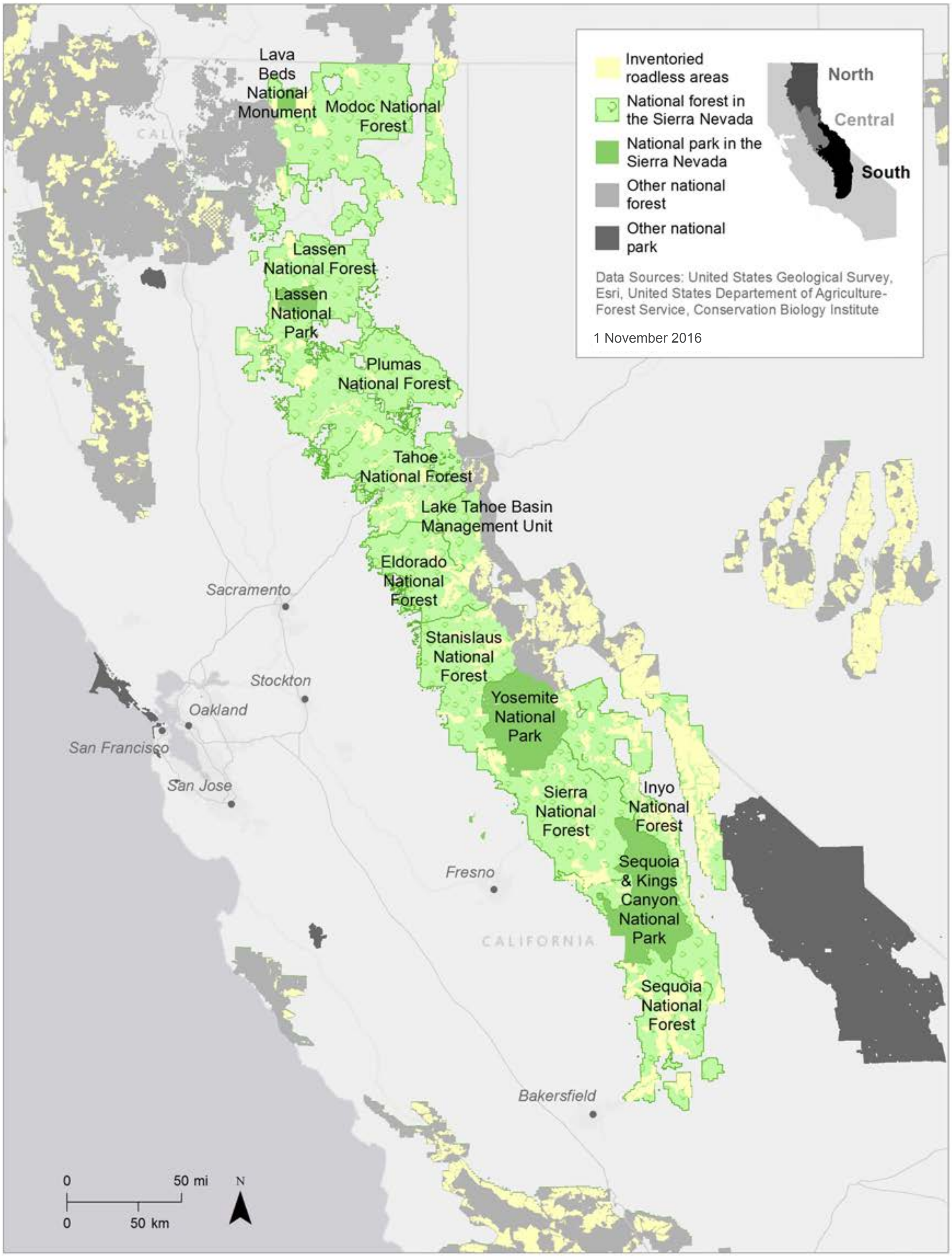

Figure 2. Sierra Nevada study region showing national forests, national parks, and inventoried roadless areas.

ened, endangered, proposed, or candidate species, that is known to occur in the plan area and for which the regional forester has determined that the best available scientific information indicates substantial concern about the species' capability to persist over the long-term in the plan area" (36 CFR 219.9(c); https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5359595.pdf, accessed 12 May 2017). The agency is required to maintain suitable habitat for these species to ensure viable populations are present in the planning area (USDA Forest Service 2012).

Comprehensive Wildland Fire Management

We recognize that land managers face many constraints (legal and social) and often competing regulatory and management objectives that limit wildfire management options. However, the Planning Rule and the National Cohesive Wildland Fire Management Strategy (USDI and USDA 2014) offer opportunities to put more fire back on the landscape whether through prescribed burning or managed wildfires. We provide some general concepts that managers might apply with pyrodiversity outcomes realized through mixed-severity fires that meet ecosystem integrity objectives.

Integrating Wildland Fire and Targeted Fire Suppression (Coarse Filter)

Mixed-severity fire effects for ecosystem benefits can be integrated with targeted suppression and fire-risk reduction efforts near towns using this coarse-filter approach. While we acknowledge that there was concern about the size and severity of the 2013 Rim Fire (Lydersen *et al.* 2014), the largest fire in recent Sierra Nevada history, we note that even this fire produced mostly low- to moderate-severity effects (i.e., ~20% of the burn was high severity based on Monitoring Trends in Burn Severity [MTBS]; http://mtbs.gov/MTBS_Uploads/data/2013/maps/ca3785712008620130817_map.pdf, accessed 23 April 2017), and a wide range of high-severity patch sizes, which contributed to significant heterogeneity at landscape scales. Thus, we concur with others (e.g., Moritz *et al.* 2014, Ingalsbee and Raja 2015, Dunn and Bailey 2016, Moritz and Knowles 2016, Schoennagel *et al.* 2017) that suppression could be focused narrowly to lands surrounding towns and used in combination with defensible space management nearest homes (Cohen 2000, 2004) so that more wildland fires can burn safely in the backcountry.

Notably, one way to safely modify fire suppression activity would be to restrict large fire crews and heavy equipment to protect homes and communities within the Wildland Urban Interface (WUI). The WUI is usually considered to extend to ~2 km from an at-risk community (US Congress 2003, USDA Forest Service 2004), even though most vegetation treatments are conducted farther from communities (Schoennagel *et al.* 2009). Beyond the WUI, point protection strategies would be used to keep fire away from isolated structures and infrastructures like cabins, communication towers, bridges, or other human assets that could be destroyed by fire. Relatively small, mobile fire crews would also use minimum impact suppression tactics (i.e., Minimum Impact Suppression Tactics [MIST]; https://www.nifc.gov/PUBLICATIONS/redbook/2003/AppendixU.pdf, accessed 12 May 2107) in backcountry areas, primarily monitoring fire spread but, when necessary, actively managing it (rather than containing and controlling wildfire as in traditional full-suppression strategies) by steering fire away from threatened social assets (Donovan and Brown 2005, 2008; Ingalsbee and Raja 2015). In municipal watersheds where fire management plans may want to avoid high-severity fires burning near water sources, more fires could be allowed to burn during moderate weather conditions. Wildfire management should be a useful tool for managing fuel loads in municipal watersheds where the use of chemicals or heavy equipment for either thinning or suppression would cause unacceptable impacts to water quality and soils. MIST could also be employed where fires in wilderness and roadless complexes, national parks, and even in roaded areas many kilometers from the nearest town pose low risk to residential areas. In sum, this approach would shift wildfire operations from limiting fire spread, size, or duration in backcountry areas to working with fire for ecosystem benefits while still effectively providing for community wildfire protection.

Sierra Nevada national forests and parks are large enough to accommodate most large fires over thousands or even tens of thousands of hectares (Appendix 1). For instance, many (>50 %) of the largest forest fires from 1984 to 2014 were primarily contained within an individual national forest or national park boundary. In general, federal lands offer unique opportunities in which the maintenance of pyrodiversity for biodiversity could be emphasized in large protected areas (wilderness and roadless area complexes; Appendix 2). Coordination among agencies with similar objectives may allow for more naturally ignited fires over mixed ownerships having similar objectives (e.g., wilderness or roadless areas, other remote forests, conservation areas juxtaposed with parks) using an all-lands approach. If reserves were too small to accommodate large fires or patches of different fire severities, then complexes of multiple reserves widely distributed across a region in redundant locations would collectively help maintain the full complement of post-fire stages using the coarse-filter approach.

In the Sierra Nevada, the draft revised forest plans for the three early-adopter national forests in the southern portion of the range have included a fire-management-zoning approach similar to what we suggest here, allowing more naturally ignited fire in remote areas and suppressing fires close to communities (USDA Forest Service 2016). However, the focus in the draft plans remains on mechanical thinning and post-fire logging (USDA Forest Service 2016). We submit that an approach that allows more natural fire ignitions is advisable and warranted from the standpoint of both ecosystem integrity and public safety, as discussed herein.

Focal Species and Species of Conservation Concern (Fine Filter)

By way of example, we consider two species that could be used to monitor mixed-severity effects. The black-backed woodpecker would be an ideal focal species given its very close association with high-severity fire patches, as would the California spotted owl, a species of conservation concern. Both species are complementary to mixed-severity fire management, given that the woodpecker is mainly associated with the high-severity component, and spotted owls use a broad gradient of fire severity patches. Moreover, while there is some overlap in geographic ranges, spotted owls generally occupy low- to mid-montane forests, while the black-backed woodpecker lives in mid- to high-elevation mixed-conifer forests up to subalpine forests.

Black-backed woodpecker as focal species of high-severity fire patches. In the Sierra Nevada, black-backed woodpeckers occur across mid- to upper-montane and subalpine conifer forests from ~1200 m to 2800 m, depending on latitude. While still uncommon even in burned areas, the greatest concentrations occur in severely burned, mixed-conifer and upper montane forests with high basal area of snags (Hanson and North 2008, Saracco *et al.* 2011) where wood-boring beetle larvae are abundant (Saab *et al.* 2007). Burned areas also typically harbor high densities of medium to large dead trees >30 cm dbh (Cahall and Hayes 2009, Saab *et al.* 2009, Tingley *et al.* 2014). Black-backed woodpeckers also occur (albeit much more rarely) in dense, mature unburned forests (Bonnot *et al.* 2009, Fogg *et al.* 2014) where they have relatively larger home ranges, presumably reflecting conditions that are less than optimal (Tingley *et al.* 2014). Nevertheless, unburned forests with high levels of dead trees from drought and native bark beetles might at least slow the rate of population decline during interludes between severe fires (Rota *et al.* 2014). Only a small fraction of fires burn suitable woodpecker habitat, due to the narrow convergence of conditions that include recent (generally ≤8 years post-fire) higher-severity fire effects in dense, mature, middle- to high-elevation conifer forest (Casas *et al.* 2016). Often a single pair of birds uses hundreds of hectares (Dudley and Saab 2007, Tingley *et al.* 2014).

Black-backed woodpeckers are vulnerable to even partial post-fire logging (Hutto and Gallo 2006, Koivula and Schmiegelow 2007, Saab *et al.* 2009, Rost *et al.* 2013). Radio-telemetry studies in the Lassen and Plumas national forests of California showed that home-range sizes were significantly larger in forests in which some post-fire logging occurred, and post-fire logged patches in the Sierra Nevada were avoided (Tingley *et al.* 2014). For example, even though post-fire logging was proposed for what seems like a minor portion of the King Fire, logging was especially concentrated within the highest quality woodpecker habitat (Figure 3), where a high density of medium to large snags occurred. Notably, on national forests of the Sierra Nevada, post-fire logging decisions have typically authorized removal of 40 % to 60 % of high-severity patches, displacing complex early seral forest with tree plantations (e.g., USDA Forest Service 2014, 2015, 2016). Retention of dead trees in logging units generally averages ~10 trees per hectare >38 cm dbh (USDA Forest Service 2004). By comparison, to maintain habitat for this focal species, generally hundreds of medium to large snags per hectare (>30 cm dbh to 40 cm dbh, and especially snags >50 cm dbh) are needed (Hanson and North 2008, Saab *et al.* 2009, Tingley *et al.* 2014) in patches consistent with home-range size, along with an ample supply of dense, mature or old conifer forest to facilitate conditions for high quality habitat when fires do occur (DellaSala *et al.* 2014).

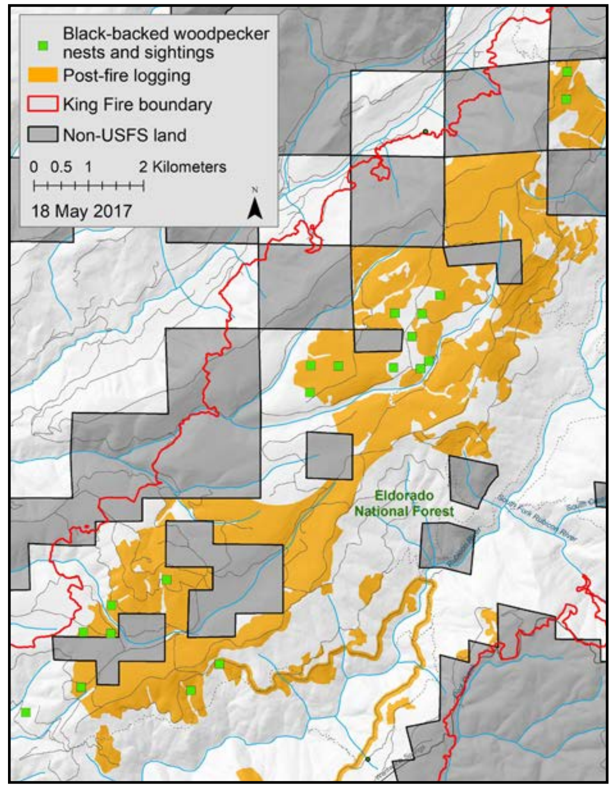

Figure 3. King Fire logging units on the Eldorado National Forest and black-backed woodpecker nests and sightings. After extensive surveys for black-backed woodpeckers were conducted for the US Forest Service throughout the fire area one year post fire, using playback recordings to detect the birds, all but one of the detections was in a relatively small area of dense, mature mid-montane conifer forest in a very large high-severity fire patch in the northern portion of the fire area (shown above). The Forest Service's decision authorized post-fire logging of ~80 % of these locations.

California spotted owl as species of conservation concern. Early studies on habitat associations and reproductive success of spotted owls in the Sierra Nevada were conducted in long-unburned forests, and "non-suitable" owl habitat was typically the result of logging (e.g., Moen and Gutiérrez 1997, Blakesley *et al.* 2005). Because spotted owls are usually associated with older, dense forests, it was assumed that effects of high-severity wildfires were similar to logging (Weatherspoon *et al.* 1992). However, recent studies have demonstrated that occupancy (Roberts *et al.* 2011, Lee *et al.* 2012, Lee and Bond 2015*a*) and reproductive success (Roberts 2008, Lee and Bond 2015*b*) were similar or higher in forests burned with a mixture of fire severities compared to long-unburned forests for up to at least 15 years post fire (longer-term studies have not been conducted). Lee and Bond (2015*a*) reported higher occupancy rates than any Sierra Nevada study area for historical owl breeding sites one year after the Rim Fire. The amount of high-severity fire within an owl pair's 120 ha protected activity center, as defined by the Forest Service, had no effect on occupancy, although occupancy by single owls declined slightly as the extent of severe-fire patches increased.

Thus, even though spotted owls are not considered a fire-dependent species, they do persist after mixed-severity fires when both unburned and severely burned patches occur within historical territories (Lee *et al.* 2012; Lee and Bond 2015*a, b*). Owls foraged preferentially in high-severity patches within mature forest in the southern Sierra Nevada (Bond *et al.* 2009) and used high- and moderate-severity patches in the San Bernardino Mountains in proportion to availability (Bond *et al.* 2016). Notably, structural complexity (including high density of dead trees) is important for spotted owl foraging habitat. Bond *et al.* (2009) found that dead tree basal area and shrub cover were highest in high-severity fire patches in which owls preferentially foraged. The owls found a rich food source, in the form of small mammal prey, in post-fire habitat (Bond *et al.* 2016). California spotted owls also selected high-severity patches for foraging more than any other fire severity condition or than long-unburned forests when within 1.5 km of the nest or roost (Figures 4 and 5). Although there are reports of California spotted owls nesting in moderate-severity patches, these raptors mostly nest and roost in long-unburned or lower-severity areas within a burned landscape (Bond *et al.* 2009), underscoring the importance of the mixed-severity mosaic. In contrast, Jones *et al.* (2016) found higher rates of territory extirpation and lower rates of colonization of owl sites that experienced >50 % high-severity fire in the King Fire on the Eldorado National Forest, and reported avoidance of high-severity patches for foraging. The circumstances of their study differed greatly from others (Lee and Bond 2015*a, b*), presumably due to pre- and post-fire logging within owl territories, as well as extensive high-severity fire in pre-fire clearcuts with young plantations.

Long-term occupancy monitoring without the confounding influence of post-fire logging is especially important to understanding fire effects on spotted owls. Hence, Bond *et al.* (2009) recommend that, if managers want to maintain spotted owl habitat after fire, they should prohibit post-fire logging and pesticide and herbicide applications within at least 1.5 km of historical spotted owl nest and roost sites. Even larger areas may be needed given that owl breeding-season home ranges can extend upwards of 700 ha (Bond *et al.* 2016), and some birds expand their range or migrate during the non-breeding season (Bond *et al.* 2010). Therefore, a reasonable protected area might be within 2.4 km of nest and roost sites, which corresponds to interim spotted owl management guidelines of the Forest Service's Pacific Southwest Research Station (http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd504726.pdf).

Restoration of Degraded Forests

Land-use stressors that degrade or impair ecosystem processes are fundamentally at odds with ecosystem integrity approaches (Pimentel *et al.* 2000, USDA Forest Service 2012). Thus, restoration treatments can be used to reverse the causative agents of ecosystem degradation. One example is to limit human-set fires via: (1) seasonal closure and decommissioning of roads, or convert roads not considered essential in firefighting within the

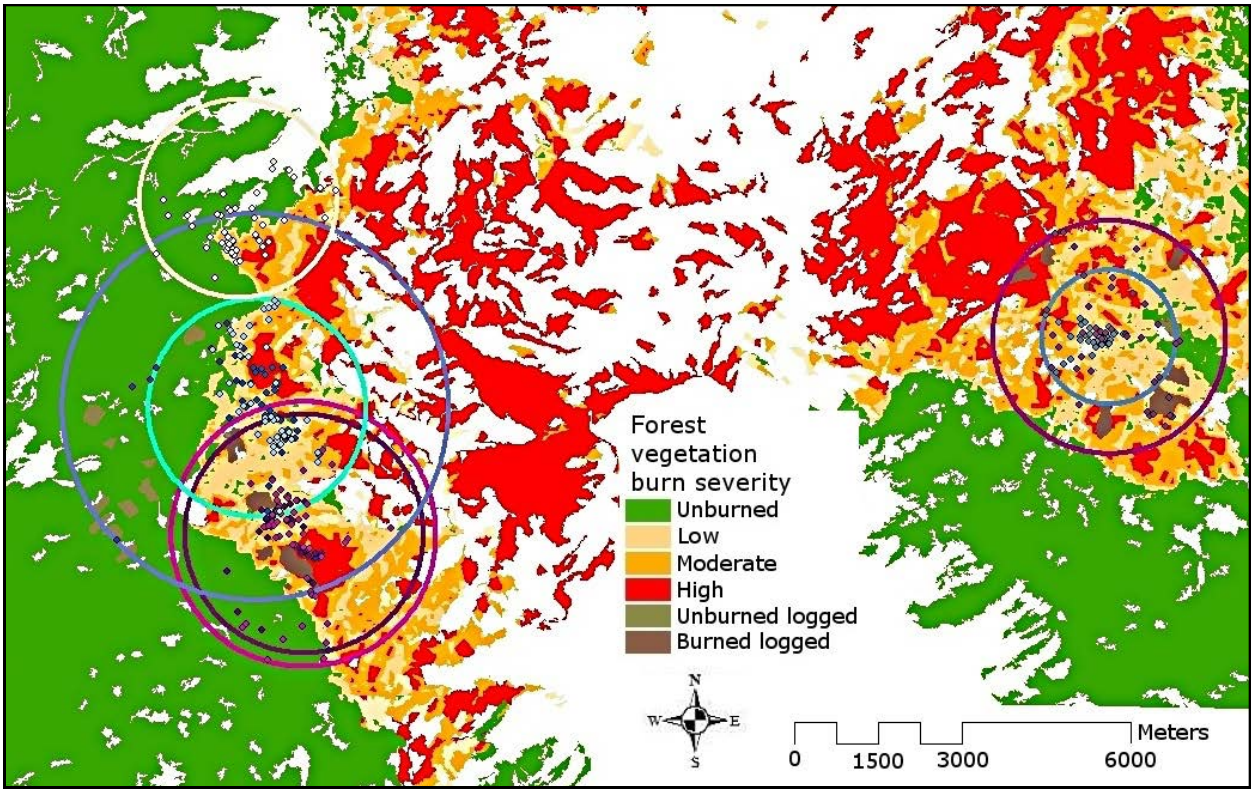

Figure 4. (A) Estimated foraging locations (obtained in 2006) of seven radio-marked California spotted owls in the 2002 McNally Fire, Sequoia National Forest, Sierra Nevada, USA. Different colored points represent each individual owl's estimated foraging location. Circles represent foraging ranges: each circle is centered on the nest with its radius extending to the farthest estimated foraging location for each individual owl. White areas are non-suitable for owls (e.g., foothill chaparral vegetation).

WUI to indefinitely closed; and (2) focused thinning and prescribed burning nearest homes, around campgrounds and other facilities, and along narrowly defined road prisms close to towns to avoid fire spread from anthropogenic ignitions. Managers could also concentrate thinning of small trees (shaded fuel breaks) along with prescribed burning nearest critical evacuation routes for communities with only one means of ingress or egress, redesign traveler stopping points along roads to avoid fire-prone settings, and concentrate visitation in fire-safe locations. Importantly, because tree plantations create unnaturally homogenized forests that lack complex structures, managers could integrate thinning with mixed-intensity prescribed burning, or naturally ignited fires, and create snags and downed logs to introduce structural complexity. Thinning small trees combined with prescribed fire (Kalies and Kent 2016) may reduce fire intensity in densely stocked tree plantations (Odion *et al.* 2004).

CONCLUSIONS

The 2012 Planning Rule provides the Forest Service with new direction for restoring and maintaining integrity and for managing focal species and species of conservation concern that can be integrated with fuels management approaches. The National Cohesive Wildland Fire Management Strategy (USDI and USDA 2014) allows managing wildfire for ecosystem benefits; hence, our findings can be applied to Department of Interior lands as well.

Figure 5. (A) General location of a California spotted owl nest territory in the 2002 McNally Fire (circle not to scale). Nest site was in a low-severity patch directly adjacent to high-severity patch (severity defined using Miller and Thode 2007). (B) Zoom-in (center snag) of general location of California spotted owl nest tree within McNally Fire burn patch shown in (A). Photos by M. Bond.

We suggest that managing for ecosystem integrity using both a coarse- and fine-filter approach centered on pyrodiverse fire effects can inform forest management in a biodiversity context. Our approach would have the added benefit of likely reducing suppression costs and some of the negative effects of mechanical vegetation removal over large areas (Dale 2006, Donovan and Brown 2008, Dunn and Bailey 2016). The complementary nature of conservation filters would allow managers to check burn severity maps with habitat associations of focal species to assess management efficacy.

Managers face substantial political and public pressure to suppress fires through the use of aggressive firefighting tactics, but such tactics do little to contain fires under extreme weather conditions (Lydersen *et al.* 2014, Moritz *et al.* 2014, Ingalsbee and Raja 2015, Carey *et al.* 2016). Instead, managers could be encouraged to use prescribed and naturally ignited fires that yield both cost savings and ecosystem benefits. Unfortunately, federal fire suppression budgets are dominated by suppression costs, causing siphoning of funds away from other essential programs (Ingalsbee and Raja 2015). To support managers in using more natural fire ignitions, conditions and certain trigger points could be more clearly defined and integrated with forest planning. This would allow flexibility to use several approaches to managing a fire, even on the same incident. Thus, in theory, a large fire could be managed in one area with general containment strategies that employ MIST (backcountry), while simultaneously in another area (near towns) with direct attack methods.

Accommodating mixed-severity fires for ecosystem benefits pertains to both ends of the fire continuum: large fires with high-severity effects that generate unique biological pulses (e.g., complex structures), and lower-severity systems that may have been homogenized through management and suppression. This suggests an important opportunity for expanding fire management beyond traditional kinds of prescribed burning to include prescriptions that benefit a broader suite of species associated with pyrodiverse landscapes (Moritz *et al.*

2014, DellaSala and Hanson 2015, Moritz and Knowles 2016). We note the conundrum of natural fire ignitions creating greater smoke emissions that may conflict with air quality objectives. Importantly, the Environmental Protection Agency (2016) recently revised policies to provide special regulatory exemptions and provisions that allow for more managed wildfires.

With proper planning and use of modern smoke management techniques, adverse effects of emissions on public health can be mitigated and fire restoration goals better accommodated. However, smoke emissions must be viewed as an unavoidable trade-off to be weighed against other potentially worse effects from attempted fire exclusion (that will eventually burn in a wildfire) or other chemical and mechanical methods for managing fuel loads that have ecosystem consequences.

There is clearly a need for research on whether natural fire ignitions can primarily provide desired mixed-severity fire effects. We suggest that studies are needed to determine the following.

(1) Specific locations and forest types best suited for mixed-severity fire effects, particularly in relation to ecological mechanisms by which pyrodiversity influences biodiversity.
(2) Current versus historical sizes and proportions of fire-severity patches and how those might be affected by climate change.
(3) Additional species that may be affected by suppression such as declining shrub-nesting birds associated with complex early-seral forests (Hanson 2014).
(4) Importance of other disturbance events (e.g., native insect outbreaks, drought) in maintaining ecosystem integrity.
(5) Effects of mechanical treatments before and after fire on the integrity and quality of mixed-severity patches including species of conservation concern and focal species.
(6) Kinds of education efforts required to implement this type of integrated disturbance ecology approach.
(7) Decision-support tools to help managers assess the costs and benefits of natural fire ignitions, along with conditions under which fires should be suppressed for human safety.

We argue that expanding natural fire ignitions for ecosystem benefits in combination with strategic use of defensible space, directed suppression, and active fuels management in appropriate areas provide untapped potential to enhance ecosystem integrity while protecting people and infrastructure with the potential for lower financial costs. Our approach is based on an ecological understanding of the importance of mixed-severity fires (DellaSala and Hanson 2015), and the need to reconsider "catastrophe" biases regarding natural disturbance processes (Lindenmayer *et al.* 2017).

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Appendix 1. Fires affecting national forests and parks within the Sierra Nevada region, California, USA, from 1984 to 2014 based on the Monitoring Trends in Burn Severity project (http://www.mtbs.gov, accessed 8 September 2015). SD = standard deviation.

Management unit	Unit area hectares	Cumulative burned area: Area (ha) (number of fires)	Cumulative burned area: (%)	Mean fire size hectares (SD)	Largest fire area (ha) (% of fire occurring within management unit)
Sequoia and Kings Canyon National Park	350 030	31 795 (34)	9.1	1 559 (1488)	3 806 (88.5)
Lassen Volcanic National Park	43 432	12 811 (9)	29.5	1 940 (3379)	6 383 (58.5)
Yosemite National Park	301 885	102 864 (49)	34.1	4 268 (15 174)	31 841 (30.6)
Eldorado National Forest	321 290.	63 458 (9)	19.7	7 882 (12 496)	40 005 (99.6)
Inyo National Forest	834 535	47 767 (26)	5.7	4 536 (11 391)	7 995 (13.5)
Lake Tahoe Basin National Forest	80 595	1 138 (2)	1.4	1 423 (285.5)	1 083 (88.7)
Lassen National Forest	602 442	145 393 (46)	24.1	7 607 (10 801)	18 632 (75.0)
Modoc National Forest	818 852	85 022 (37)	10.4	3 221 (6 348)	15 507 (41.8)
Plumas National Forest	579 996	141 396 (37)	24.4	5 111 (8 112)	26 371 (99.0)
Sequoia National Forest	470 505	163 731 (61)	34.8	3 801 (8 563)	51 284 (86.5)
Sierra National Forest	574 583	48 785 (30)	8.5	3 261 (4373)	9 538 (100)
Stanislaus National Forest	441 366	171 391 (35)	38.8	7 647 (17 908)	71 614 (68.8)
Tahoe National Forest	476 706	54 294 (19)	11.4	5 786 (9640)	8 394 (100)
Toiyabe National Forest	731 467	63 715 (33)	8.7	2 797 (3 692)	10 163 (100)

Appendix 2. Wilderness and adjacent inventoried roadless areas (IRA) in the Sierra Nevada region, California, USA, compared to largest fire sizes.

Wilderness/IRA complex	Complex size (ha)	Largest fire within associated forest unit[1] (1984 to 2014)
Eldorado	75 255	40 005
Inyo	601 756	7 995
Lassen	99 821	6 383
Modoc	109 725	15 507
Plumas	35 987	26 371
Sequoia	266 316	51 284
Sierra	293 314	9 538
Stanislaus	143 319	71 614
Tahoe	69 519	8 394
Toiyabe	348 597	10 163
Lake Tahoe Basin	28 345	1 083

[1] Fire sizes are for national forest units with wilderness/IRA complexes. Many fires extend beyond national forest and wilderness/IRA boundaries (see Appendix 1).

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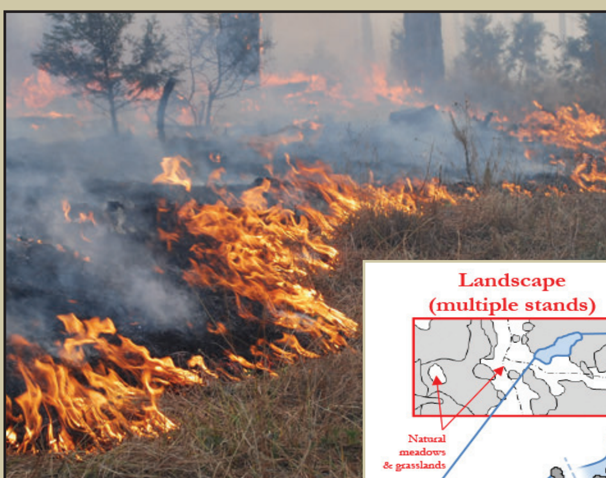
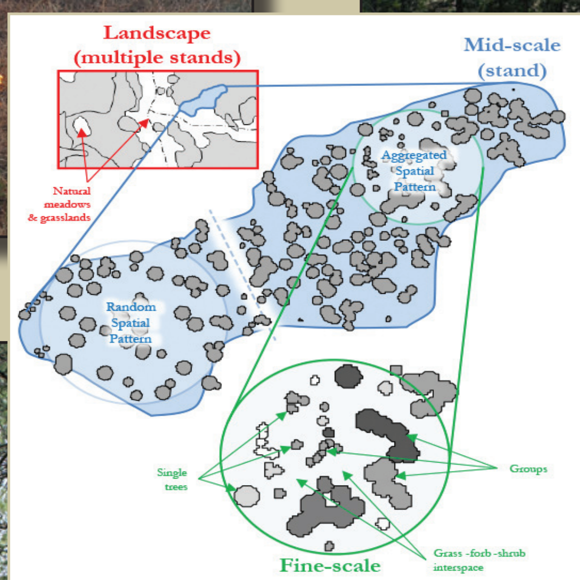
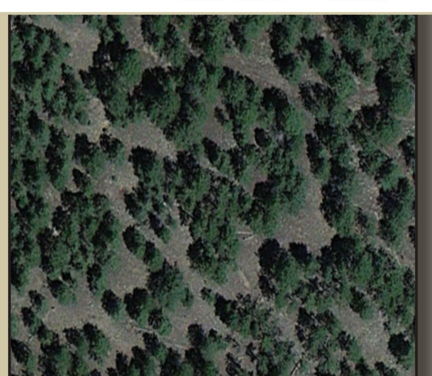
Restoring Composition and Structure in Southwestern Frequent-Fire Forests:

A science-based framework for improving ecosystem resiliency

Richard T. Reynolds, Andrew J. Sánchez Meador, James A. Youtz,
Tessa Nicolet, Megan S. Matonis, Patrick L. Jackson,
Donald G. DeLorenzo, Andrew D. Graves







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Abstract

Ponderosa pine and dry mixed-conifer forests in the Southwest United States are experiencing, or have become increasingly susceptible to, large-scale severe wildfire, insect, and disease episodes resulting in altered plant and animal demographics, reduced productivity and biodiversity, and impaired ecosystem processes and functions. We present a management framework based on a synthesis of science on forest ecology and management, reference conditions, and lessons learned during implementations of our restoration framework. Our framework focuses on the restoration of key elements similar to the historical composition and structure of vegetation in these forests: (1) species composition; (2) groups of trees; (3) scattered individual trees; (4) grass-forb-shrub interspaces; (5) snags, logs, and woody debris; and (6) variation in the arrangements of these elements in space and time. Our framework informs management strategies that can improve the resiliency of frequent-fire forests and facilitate the resumption of characteristic ecosystem processes and functions by restoring the composition, structure, and spatial patterns of vegetation. We believe restoration of key compositional and structural elements on a per-site basis will restore resiliency of frequent-fire forests in the Southwest, and thereby position them to better resist, and adapt to, future disturbances and climates.

Keywords: dry-mixed conifer, ecosystem services, ecosystem processes and functions, frequent-fire forests, forest structure, ponderosa pine, restoration, species composition, spatial patterns

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Executive Summary

Many forest landscapes in the Southwestern United States (Arizona, New Mexico, southwest Colorado, and southern Utah) have become increasingly susceptible to large-scale, severe wildfire, insect, and disease episodes. As a result, these areas are experiencing altered plant and animal demographics, reduced structural and spatial heterogeneity of vegetation, reduced productivity and biodiversity, and impaired ecosystem processes, functions, and services. Increased susceptibilities are most evident in frequent-fire forests—forests that historically experienced frequent, low-severity fire, which in the Southwest include ponderosa pine and dry mixed-conifer forests. Changes to these frequent-fire forests largely resulted from unregulated livestock grazing around the turn of the 20th Century, logging, and human activities such as fire suppression, resource use, and infrastructure development.

We present a management framework for improving the resistance and resiliency of frequent-fire forest ecosystems to severe disturbances. This is accomplished by restoring the characteristic vegetation composition and structure in these forests. Frequent-fire forests had a characteristic uneven-aged structure consisting of a temporally shifting mosaic of different aged tree groups and scattered individual trees in an open grass-forb-shrub matrix—a spatial and temporal pattern that provided and sustained plant and animal habitat adjacency, local biodiversity, and food webs. Hence, the key compositional and structural elements of our restoration framework are: (1) species composition (tree and understory vegetation); (2) groups of trees; (3) scattered individual trees; (4) open grass-forb-shrub interspaces between tree groups and individual trees; (5) snags, logs, and woody debris; and (6) variation in arrangements of these elements in space and time. Our framework is informed by:

- reference conditions (conditions of ecosystems before significant industrial human disturbance),
- natural ranges of variability (ranges of reference conditions for a specific ecosystem and time period),
- observed changes in disturbance regimes, and
- lessons learned during applications of our framework in frequent-fire forests in the Southwest.

The types, frequencies, and severities of disturbances (e.g., fires, insects, and diseases) played an important role in shaping the historical composition, structure, and function of frequent-fire forests. Therefore, where forest composition and its structure allow, the framework recommends that fire, the primary historical disturbance agent in these forests, play a prominent role in their restoration. The framework also emphasizes that mechanical treatments may be necessary to initiate suitable compositions and structures before reintroducing fire. Where use of fire is limited, mechanical treatments may be the only available tool to create and maintain restored forests. Conversely, fire may be the only tool in some areas. Restoration provides opportunities for the re-establishment of the characteristic disturbance regimes as well as the spatial and temporal links between pattern and process (e.g., the feedback relationship between forest structure and fire) that sustained the characteristic composition and structure of these forests. Implementation of our framework should improve overall ecosystem productivity and function and enhance ecosystem services such as soil productivity, biodiversity, wildlife habitat, clean air, water quality and quantity, wood products, and recreation.

Natural ranges of variability are considered a "best" estimate of a resilient and functioning ecosystem because they reflect the evolutionary and historical ecology of forests. Natural ranges of variability are thereby a powerful template for improving the resiliency of frequent-fire forests. Natural variability in the composition and structure across sites in these forests results from and drives spatial differences in fire effects, plant species compositions, tree establishment patterns and densities, and numbers and distribution of snags, logs, and woody debris. Managers are encouraged to recognize the natural variability in ponderosa pine and dry mixed-conifer forests and to use historical evidence, such as old trees, stumps, and logs, and biophysical site attributes (e.g., soils, slopes, aspects, and climate) to guide the restoration of variability in these forests. Studies of reference conditions in Southwestern ponderosa pine and dry mixed-conifer showed that trees occurred in a range of spatial patterns, most often aggregated but with a random distribution on certain soils. Tree groups were separated by open grass-forb-shrub interspaces of variable sizes and shapes that often contained scattered individual trees. In areas exhibiting strong tree aggregation, openness was typically higher, but on sites with less tree aggregation, openness may have been lower depending on the arrangement of trees, their sizes, and crown widths (Table 1). The distribution and abundance of snags and logs varied with site and likely coincided with the type, severity, and scale of historical disturbance (Table 1). While reference condition literature on the fine-scale (<10 acres) composition and structure in dry mixed-conifer is more limited than for ponderosa pine, studies showed many similarities—the consequence of their characteristic frequent, low-severity fire regimes. Nonetheless, ranges of reference conditions at small spatial scales showed that mean tree densities and basal areas were slightly greater in dry mixed-conifer forests than ponderosa pine, and snag and log abundances appeared similar to or slightly greater in dry mixed-conifer than in ponderosa pine forests. Compared to today's forests, characteristic dry mixed-conifer forests had higher proportions of fire-resistant/shade-intolerant tree species; lower tree densities; a more open structure comprised of higher proportions of large, old trees; and more spatial heterogeneity (groups and patches of trees).

To illustrate implementation of our framework, we describe a restoration treatment in a ponderosa pine stand in New Mexico that had experienced incidental tree cutting and no fire since the 1880s. While the stand had a characteristic component of old trees, there was a preponderance of mid-aged trees. Fire behavior modeling of pre-treatment conditions showed that 11 percent of the stand could support torching and active crown fire under dry conditions and moderate wind speeds. Our restoration treatment moved the composition and structure of the stand towards characteristic conditions—distinct tree groups, scattered single trees, and open interspaces between tree groups. Implementation of the framework resulted in predicted crown fire behavior on only 1 percent of the stand. Post-treatment abundance of logs and snags was lower than desired, but these elements will accumulate over time.

Our framework incorporates knowledge of the historical compositions, structures, functions, and processes in Southwest frequent-fire forests and how these operated through feedback mechanisms to sustain their characteristic compositions and structures. Current forest conditions are reviewed in light of historical conditions and how human-caused changes to these forests lowered their resistance and resilience to disturbance agents, which have become more intense and frequent. Our framework offers management recommendations for achieving the key compositional and structural elements for restoring frequent-fire forests. Once restored, these forests comprise a temporally shifting mosaic of groups of trees with interlocking crowns; scattered single trees; open grass-forb-shrub interspaces between tree groups; and dispersed snags, logs, and woody debris. It may not always be feasible or even desirable to restore exact reference compositions and structures. Instead, our framework's objective is to increase forest resiliency by managing forest composition and structure toward reference conditions. We believe restoration of key compositional and structural elements on a per-site basis will enhance the resiliency of frequent-fire forests in the Southwest, thereby positioning them to better adapt to future disturbances and climates. It is our intent that application of this framework be flexible and adaptive (i.e., learn-as-you-go), that it will evolve with accumulation of knowledge, and that its conceptual approach will provide a blueprint against which management plans and practices can be evaluated.

Table 1. Ranges of reference conditions for ponderosa pine and dry mixed-conifer forests in the Southwestern United States from studies detailed in Tables 3, 6, 7, and 9.

	Reference conditions by forest type	
Forest attribute	**Ponderosa pine**	**Dry mixed-conifer**
Trees / acre	11.7-124	20.9-99.4
Basal area (ft^2 / acre)	22.1-89.3	39.6-124
Openness (%)[a]	52-90	78.5-87.1
Openness on sites with strong tree aggregation (%)[a]	70-90	79-87
Spatial patterns	Grouped or random	Grouped or random
Number of trees / group	2-72	Insufficient data
Size of groups (acres)	0.003-0.72	Insufficient data
Number of groups / acre	6-7	Insufficient data
Snags / acre	1-10	≥ Ponderosa pine forests
Logs / acre	2-20	≥ Ponderosa pine forests

[a]Openness is the proportion of area not covered by tree crowns, estimated as the inverse of canopy cover. Openness data for dry mixed-conifer is limited; range of reference condition openness will likely change with additional studies.

Contents

Restoring Composition and Structure in Southwestern Frequent-Fire Forests: A Science-Based Framework for Improving Ecosystem Resiliency

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Table 2. Characteristic fire regimes of Southwestern forest types. Fire frequency refers to the mean number of years between fires, and fire severity relates to the effect of the fire on dominant overstory vegetation. Infrequent-fire forests (wet mixed-conifer and spruce-fir) are included for comparison to frequent-fire forests.

Forest type (subtype)	Fire regime[a]		Fire type[b]	Forest structure	Seral species[c]	Climax species
	Fire frequency	Fire severity				
Ponderosa pine (all subtypes)	Regime I					
	0-35 years	Low	Surface	Uneven-aged, grouped, open	**Dominant:** ponderosa pine	**Dominant:** ponderosa pine Shade-intolerant species.
Dry mixed-conifer	Regime I (common)				**Dominant:** ponderosa pine **Subdominant:** aspen, oak, Douglas-fir, Southwestern white pine, and limber pine	**Dominant:** ponderosa pine **Subdominant:** Douglas-fir and Southwestern white pine or limber pine Shade-intolerant species.
	0-35 years	Low	Surface	Uneven-aged, grouped, open		
	Regime III (rare)					
	35-100+ years	Mixed	Mixed	Uneven-aged, patched, open		
Wet mixed-conifer	Regime III (common)				**Dominant** (depending on plant association): aspen or Douglas-fir	**Dominant** (depending on plant association): white fir and/or blue spruce Shade-tolerant species.
	35-100+ years	Mixed	Mixed	Uneven-aged, patched, closed		
	Regime IV (rare)					
	35-100+ years	High	Stand-replacing	Even-aged, closed		
Spruce-fir (mixed, lower subalpine)	Regime III and/or IV					
	35-100+ years	Mixed / High	Mixed/ stand-replacing	Even-aged, closed	**Dominant** (depending on plant association): aspen or Douglas-fir	**Dominant** (depending on plant association): Engelmann spruce and/or white fir Shade-tolerant species.
Spruce-fir (upper subalpine)	Regime V					
	200+ years	High	Stand-replacing	Even-aged, closed	**Dominant** (depending on plant association): aspen, Douglas-fir, or Engelmann spruce	**Dominant:** Engelmann spruce and corkbark fir or subalpine fir Shade-tolerant species.

[a] Schmidt and others (2002)
[b] Smith (2006a, 2006b, 2006c)
[c] USDA Forest Service (1997)

Introduction

There is increasing recognition that frequent-fire forests, defined as forests with fire return intervals <35 years (Table 2), have become progressively more susceptible to large-scale, severe wildfire (Covington and Moore 1994b; Steele and others 1986; Westerling and others 2006). These forests, which in the Southwestern United States include ponderosa pine and dry mixed-conifer forests (see Appendix 1 for scientific names of species referred to herein), are also increasingly prone to insect and disease epidemics and altered plant and animal habitats, all leading to reduced biodiversity, ecological function, resilience, and sustainability (Allen and others 2002; Benayas and others 2009; Carey and others 1992; Carey and others 1999; Colgan and others 1999; Covington and Moore 1994a; Kalies and others 2012; Lynch and others 2010). Reduced ecosystem resilience to disturbances is more evident in frequent-fire forests where the composition, structure (age, size, density, and spatial patterns of vegetation), processes (e.g., disturbances), and functions (e.g., food webs) have changed to a greater degree due to reductions in fire frequency than in forest types where fire was historically less frequent (Agee 2003; Covington and Moore 1994a; Crist and others 2009; Hessburg and others 1999). This reduction in fire frequency is, in part, a result of more than a century of intensive human activities, including fire suppression, livestock grazing, and logging. Important compositional and structural changes in these forests resulting from human activities, especially those that changed historical fire regimes, include:

- increased tree densities,
- reduced structural and spatial heterogeneity of vegetation,
- declines in grass-forb-shrub vegetation,
- loss of old trees, and
- reductions in the diversity and quality of plant and animal habitats and food webs (Abella 2009; Arnold 1950; Covington and others 1997; Kalies and others 2012; Larson and Churchill 2012).

In addition to increasingly frequent and uncharacteristic disturbances such as large-scale severe fire events (Allen 2007; Covington and Moore 1994b; Fitzgerald 2005; Graham and others 2004; Swetnam and others 1999) and insect epidemics (Ferry and others 1995; Hessburg and others 2005; Kolb and others 1998; Negrón 1997), these changes resulted in environments that differed from those in which the native fauna and flora evolved (Carey 2003; Carey and others 1992, 1999; Colgan and others 1999; Covington and Moore 1994b; Kalies and others 2012; Reynolds and others 1992, 2006a). Furthermore, ecosystem services such as clean air and water, water yield, wood products, recreation, aesthetic and spiritual experiences, old-growth, nutrient cycling, pollination, and carbon sequestration have been altered and are now more vulnerable to rapid degradation by uncharacteristic fire and insect epidemics (Benayas and others 2009; Ferry and others 1995; Finkral and Evans 2008; Hessburg and others 2005; Kolb and others 1998; Negrón 1997; reviewed in Evans and others 2011 and Hunter and others 2007).

Prior to human-influenced changes to the characteristic fire regime, the composition, structure, and spatial pattern in frequent-fire forests were maintained by frequent, low-severity fire through a functional relationship between pattern and process; that is, frequent low-severity fires resulted in forest structures that facilitated continued low-severity fire (Fitzgerald 2005; Graham and others 2004; Hiers and others 2009; Mitchell and others 2009; Thaxton and Platt 2006). Over time, shifting mosaics of tree groups and individual trees of varying ages were maintained within a grass-forb-shrub matrix by relationships among the severity and frequency of fire, presence of surface fuels (fuels on or near the surface of the ground), and tree regeneration sites that escaped fire (Larson and Churchill 2012). Some dry mixed-conifer forests and ponderosa pine-shrub communities experienced mixed-severity fires, which included combinations of surface and crown fires (see Table 2), sometimes resulting in larger patches of tree aggregation (Agee 1993; Arno and others 1995; Kaufmann and others 2007; Larson and Churchill 2012).

Forest restoration guided by reference conditions (conditions that characterized the status of ecosystems before significant industrial human disturbance; *sensu* Kaufmann and others 1994) provides for the approximation of the historical (i.e., natural) effects of characteristic disturbances. Restoration is the process of assisting the recovery of degraded, damaged, or destroyed ecosystems (SER 2004). Restoration initiates or accelerates ecosystem recovery with respect to ecological health (productivity), integrity (species composition, community and ecosystem structure), and sustainability (resistance and resilience to disturbance)

(SER 2004). Ecosystem resiliency is the ability of an ecosystem to absorb and recover from disturbances without altering its inherent function (SER 2004). A functioning ecosystem provides opportunities for sustaining plant and animal habitats and populations, increased biodiversity, nutrient cycling, carbon sequestration, air quality, water quality and quantity, wood products, forage, recreation, and aesthetic and spiritual experiences (Aronson and others 2007; Benayas and others 2009). Restoring forest composition and structure improves ecosystem function and resiliency (Bradshaw 1984; Cortina and others 2006).

A holistic approach to forest restoration based on appropriate science can also help meet multiple management objectives, including fuels reduction; reintroduction of characteristic disturbances; and the return of wildlife habitats, native biodiversity, and food webs (Covington and Moore 1994b; Kalies and others 2012; Reynolds and others 1992, 2006a). Management informed by reference conditions and natural ranges of variability (the range of ecological and evolutionary conditions appropriate for an area; *sensu* Landres and others 1999) allow for the restoration of the characteristic composition, structure, spatial pattern, processes, and functions of ecosystems. Managing forests guided by historical conditions also restores the evolutionary environment (Kalies and others 2012; Moore and others 1999), enhancing the capacity of organisms in ecosystems to adapt to stressors such as fire, insects, disease, and climatic variability and change.

We describe a framework, including assumptions, principles, values, concepts, and procedures, for restoring the composition, structure, and spatial pattern of ponderosa pine and dry mixed-conifer forests in the Southwest. Our framework is a science-based approach to restoration that will prove useful for developing strategic plans and management applications. The framework emphasizes vegetation composition and structure, describes expected outcomes, and presents management recommendations for implementation. Expected outcomes include: increased biodiversity, plant and animal habitats, and ecosystem services; increased resilience to insects, disease, and climate change; and reduced fuel loads and fire hazards. Key compositional and structural elements of our restoration framework are:

(1) species composition (tree and understory vegetation);

(2) groups of trees;

(3) scattered individual trees;

(4) open grass-forb-shrub interspaces;

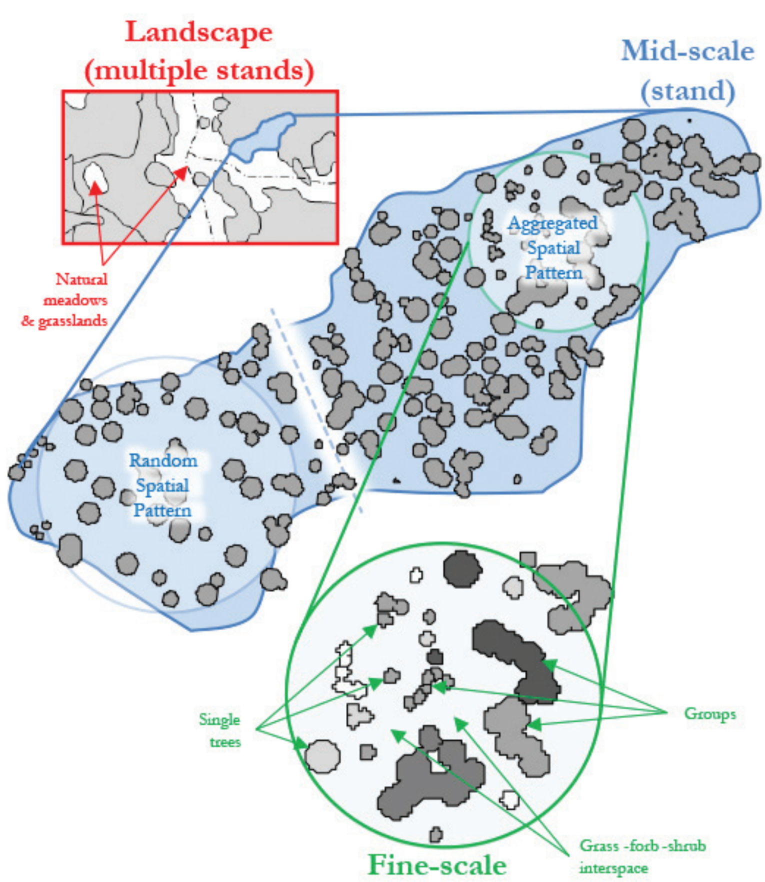

Figure 1. Characteristic vegetation patterns at three spatial scales for frequent-fire forests in the Southwest. The landscape-scale illustrates the importance of multiple stands (patches), meadows, and grasslands. The mid- and fine-scales illustrate grass-forb-shrub interspaces and uneven-aged stand conditions consisting of single, random, and grouped trees of different vegetation structural stages (from young to old) represented by different shades and sizes at the fine-scale. Also depicted are two different tree spatial patterns at the mid-scale (separated by the dashed line): trees are randomly spaced on the left side of the dashed line and are aggregated on the right (given the definition of stand as a homogenous area, both patterns could not actually be present).

(5) snags, logs, and woody debris; and

(6) variation in arrangements of these elements in space and time (Fig. 1).

Ecosystems are structured hierarchically and their composition, structure, processes, and functions are temporally and spatially dynamic. Therefore, we characterize the key compositional and structural elements at three spatial scales: fine (<10 acres), mid (10-1000 acres), and landscape (1000-10,000+ acres) (Fig. 1). These scales generally correspond with structural features in frequent-fire forests. The fine scale is an area in which the species composition—age, structure, and spatial distribution of trees (single and grouped)—and grass-forb-shrub interspaces are expressed. Aggregates of fine-scale units comprise mid-scale patches or stands, which are relatively homogeneous in vegetation composition and structure. The landscape scale is composed of aggregates of mid-scale units and usually has variable elevations, slopes, aspects, soil types, plant associations, disturbance processes, and land uses. Understanding and incorporating temporal scales (e.g., seasonal, annual, decadal, and centuries) in a restoration framework is required to sustain vegetation dynamics of forests that result from growth, succession, senescence, and the historical and anthropogenic disturbances that periodically reset the dynamics.

Management recommendations for implementing our framework are tempered by our management and research experience in frequent-fire forests, as well as by lessons learned during implementations of the framework in the Southwest. The intent of our framework is to inform management strategies that will facilitate the resumption of historical processes and functions. Managing for the framework's key elements should increase the resilience of the forests and facilitate opportunities for the resumption of characteristic function and disturbance regimes. The spatial and temporal aspects of these elements reflect the reciprocal interactions between pattern and process in these forests and are an ecological basis (Turner 1989) for incorporating spatial information in forest restoration (Larson and Churchill 2012).

Mechanisms Influencing Forest Composition

Plant species composition of a forest ecosystem is influenced by both deterministic and stochastic factors, including complex interactions among species' life histories, disturbance regimes, and chance events. The establishment, growth, and survival of under- and over-story species are affected by competition for space, light, nutrients, and moisture. For example, tree regeneration and growth is affected by species-specific shade tolerance (Fig. 2); open stand conditions favor the regeneration of shade-intolerant species while closed stands favor shade-tolerant species (Langsaeter 1944; Long 1985; USDA Forest Service 1990). Biophysical conditions, such as soils, temperature, and moisture regimes, also influence the establishment, development, and abundance of under- and over-story plant species. Disturbances (e.g., fire, insects, pathogens, drought, and wind) often interact with biophysical site characteristics to further influence composition and structure of forest ecosystems. Such disturbances have variable temporal and spatial effects on vegetation depending on their type, frequency, intensity, seasonality, and spatial scale, which collectively define a characteristic disturbance regime of an ecosystem. Species in a forest ecosystem evolved under its characteristic disturbance regime, resulting in a natural range of variability or the range of ecological and evolutionary conditions appropriate to an ecosystem (Landres and others 1999).

Fire is the primary disturbance agent in many Southwestern forests, and fire regimes are central to understanding an ecosystem's reference conditions and natural range of variability (Fig. 3; Table 2) (Fulé and others 2003). The species composition, as well as the structure and spatial pattern of vegetation in Southwestern frequent-fire forests developed in a feedback relationship with fire. Ponderosa pine and dry mixed-conifer forests are characterized by a frequent low-severity fire regime (Swetnam and Baisan 1996; Swetnam and Betancourt 1990) with historic mean fire return intervals ranging from 2-24 years (Brown and others 2001; Brown and Wu 2005; Evans and others 2011; Hunter and others 2007; Swetnam and Baisan 1996). Frequent low-severity fire favors shade intolerant and fire-resistant tree species (Fig. 2) and open forest conditions with discontinuous crowns and minimal fuels build-up, often with tree groups separated by open interspaces with grass-forb-shrub communities. In contrast, longer fire return intervals permit seedling development to larger, more fire-resistant tree sizes and favor survival of less fire-resistant species (Fig. 2) (Fulé and Laughlin 2007; Laacke 1990; Taylor and Skinner 2003).

Endemic forest insects and pathogens are important disturbance agents that do not threaten long-term

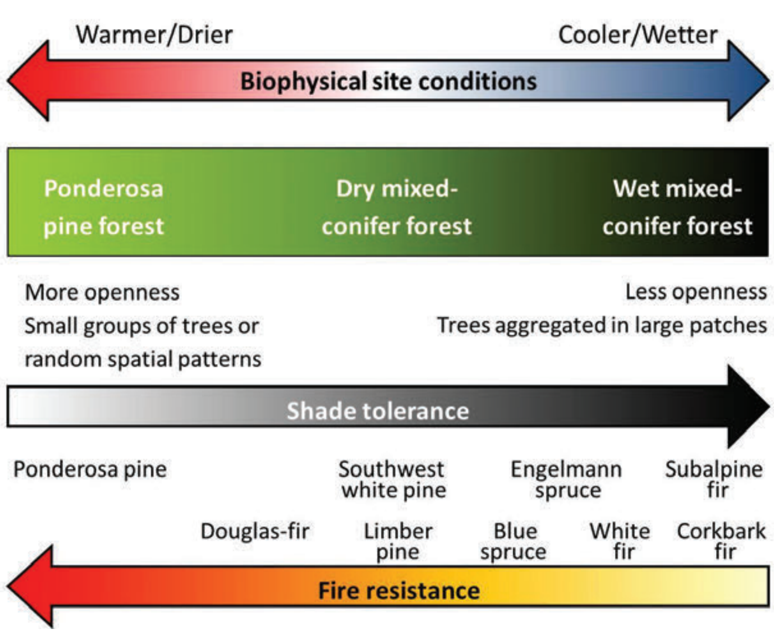

Figure 2. Dry mixed-conifer forests occupy the ecological gradient from warm/dry to cool/wet biophysical site conditions. Dry mixed-conifer is not a homogenous type, intergrading with ponderosa pine forest on warm/dry sites and wet mixed-conifer forests on cool/wet sites. Its structure and composition become more similar as it intergrades with adjacent forest. Common tree species in ponderosa pine and mixed-conifer forests also vary in their relative shade and fire tolerance.

Figure 3. Prescribed, low-severity surface fire carried by needles, cones, dried grass, and forbs on the Lincoln National Forest, 2010.

stability and productivity of forests under endemic conditions due to moderation by millions of years of evolution (Goheen and Hansen 1993). When large or uncharacteristic insect and disease outbreaks occur, profound changes to the composition, structure, processes, and functions of forests often take place. Insects and diseases affect nearly all aspects of forest stand dynamics, from seed viability to seedling survival, from bud, shoot, and leaf production to growth and maintenance, and, ultimately, the survival and distribution of mature trees (Castello and others 1995; Tainter and Baker 1996). Bark beetles, in particular, are considered primary sources of mortality in Southwestern ponderosa pine forests. In 2011 alone, bark beetles caused varying rates of ponderosa pine mortality on more than 144,000 acres in Arizona and New Mexico (USDA Forest Service 2012). Unlike bark beetles in ponderosa pine, the primary sources of mortality attributed to insects in mixed-conifer forests are typically defoliating insects. Damage from defoliators can range from large areas of widespread growth losses and infrequent mortality, as with the spruce budworm, to more localized, high levels of mortality caused by the Douglas-fir tussock moth (Wickman 1963).

While numerous species of dwarf mistletoe occur in frequent-fire forests, Southwestern (ponderosa pine) dwarf mistletoe and Douglas-fir dwarf mistletoe are the most prevalent. Dwarf mistletoes may be the most damaging of pathogens in Southwest forests with estimates of current infection being 30 percent or greater in ponderosa pine forests (Andrews and Daniels 1960; Maffei and Beatty 1988) and around 50 percent in mixed-conifer forests (Conklin and Fairweather 2010; Drummond 1982). Additionally, the presence and intensity of Southwestern dwarf mistletoe infection in ponderosa pine stands has been implicated as a source of mortality or as an exacerbating factor in bark beetle outbreaks (Negrón 1997; Stevens and Hawksworth 1984). Endemic soil fungi that cause root disease (e.g., armillaria and black-stain root diseases) also influence forest composition and structure (Rippy and others 2005). Root diseases are known to affect the ponderosa pine forests of the Southwest, with observations of mortality associated with root disease, mistletoe, and bark beetles as high as 25 percent (Wood 1983). In some locations, conifers killed by root disease are replaced by less susceptible conifers, hardwood species, or grass-forb-shrub interspaces. In the case of armillaria and related wood decay fungi, this shift in species composition can be maintained for decades due to remnant fungi in stumps and root systems (Roth and others 1980). In most situations, native root diseases do not cause irreplaceable loss of entire stands over large areas, nor do they threaten the existence of any host species. However, shifts in stand composition and other natural and human-caused disturbances have frequently resulted in increased damage from root diseases (Edmonds and others 2000).

Mechanisms Influencing Forest Structure

Frequent-fire forests typically comprise a mosaic pattern of groups of trees, scattered single trees, grass-forb-shrub interspaces, snags, logs, and woody debris (Cooper 1960; Larson and Churchill 2012; Pearson 1950; White 1985). Structural heterogeneity

Table 3. Historical spatial patterns and tree group characteristics in frequent-fire forests of the Southwest, arranged by forest type (PP: ponderosa pine, PO: pine-oak, DMC: dry mixed-conifer).

Location	Parent material	Elevation (ft)	Forest type	Reference date	Tree sizes (dbh in in.)	Group density (groups/ acre)	Group size (acres)	Trees per group[a]	Percent basal area in groups	Citation
Malay Gap, Arizona	Basalt	7200	PP	1952	≥ 4.0		0.16-0.32			Cooper 1960
Gus Pearson Natural Area, Arizona	Basalt	7398	PP	1875	Unknown		0.05-0.72	3-44		White 1985
Flagstaff, Arizona	Varying	7800	PP	1880	Unknown	1-33		2-25	28%-74%	Abella and Denton 2009
Woolsey Plots, Arizona	Basalt	7052	PP	1874	≥ 3.5	25-67	0.003-0.09	3-24	62%-75%	Sánchez Meador and others 2011
Coulter Ranch, Arizona	Basalt	7520	PO	1913	≥ 3.5		0.01-0.1			Sánchez Meador and Moore 2010
Uncompahgre Plateau, Colorado	Shale	8000	PP / DMC	1875	Unknown		0.1-0.25			Binkley and others 2008
Numerous national forests in Arizona and New Mexico	Varying	8650	PP / PO / DMC	1910	≥ 3.5	24-80	0.01-0.32	2-72	51%-85%	Sánchez Meador and others unpublished data[b]

[a]Values should be not interpreted as "strict" densities of trees within groups as authors used different definitions and methods to define and characterize "groups." We suspect that as the number of species and site productivity increase the metric of "tree group" becomes less useful than this metric at the mid- to landscape-scale. For example, when tree density is fixed and numbers of tree species varies (i.e., compare ponderosa pine vs. ponderosa-pine Gambel oak vs. dry-mixed conifer forests), the area available to a "tree group" will likely decrease.

[b]Data based on 2.47-acre plots reconstructed prior to Euro-American settlement (1876-1890) in Arizona (n = 17 plots) and New Mexico (n = 7 plots) using the same methods as Sánchez Meador and others (2010, 2011). Historical and contemporary field methods, as well as contemporary conditions, are detailed in Sánchez Meador and others (2010) and Moore and others (2004) who reported forest structural reference conditions (size distributions, tree density ranges, spatial patterns, etc.) on a subset of these same plots. In brief, all live and dead tree structures were measured, including stumps, snags, and wind-fallen trees, that grew to at least breast height (4.5 ft). All tree structures were located using historical stem-maps and measured spatial coordinates, and dendrochronological reconstructions were used to quantify structural and spatial reference conditions (Baker and others 2008; Sánchez Meador and others 2010). Spatial attributes (e.g., group size and density) were quantified using methods described in Sánchez Meador and others (2011).

Figure 4. A group of ponderosa pine trees comprised of two clumps of trees.

in these forests is a consequence of interactions among biophysical site conditions (e.g., topography, soils, climate); disturbance types, frequencies, intensities, and extent; levels of competition among species; and tree demographic rates. Variability in biophysical site conditions is a primary source of spatial and temporal variation in vegetation structure. Of studies that investigated the origin, distribution, and mortality of ponderosa pine forests, most reported uneven-aged reference conditions at the stand scale (Sánchez Meador and others 2010), but three different within-group age structures were identified. Cooper (1960) reported relatively even-aged tree groups, White (1985) and Abella (2008) reported groups of multi-aged trees, and Sánchez Meador and others (unpublished data; see Table 3 footnote) found mixtures of both types. Variation of tree ages within groups likely reflects the establishment and growth of a single, grouped cohort of trees and perhaps seedling establishment and growth of trees under, or adjacent to, tree groups (see Spatial Patterns: Formation and Maintenance) (Sánchez Meador and others 2009).

Heterogeneity of within-group tree sizes can generate from processes related to growth, competition, and disturbances and may result in a range of tree sizes irrespective of age (Mast and Veblen 1999; Pearson 1950; Sánchez Meador and others 2011; Taylor 2010; Woodall 2000). Trees on the perimeter of groups tend to have higher growth rates, attain larger sizes, lean away from the group center, and have asymmetrical crowns with larger lower limbs than interior trees (Pearson 1950). Heterogeneity in tree sizes and spacing within groups may decline over time due to mortality resulting in a gradual transition from dense to more uniform spacing of trees (Cooper 1961; Mast and Veblen 1999; Mast and Wolf 2004, 2006; Pielou 1960). However, tight clumps of trees sharing the same root ball often persist within groups (Fig. 4) (Larson and Churchill 2012). Mortality over time may also gradually reduce within-group tree density, resulting in increased variation in tree densities and ages within and among groups.

Like composition, the structure of forest vegetation is also affected by disturbances such as fire, insects, disease, wind, and drought (Brown and others 2001; Ehle and Baker 2003; Mast and others 1998, 1999). Numerous abiotic and biotic disturbances affect the composition, amount, arrangement, spatial continuity, and volatility of surface and canopy fuels (Franklin and others 2012), which in turn effects fire behavior (Van Wagner 1977). Dense forest structures can facilitate crown fire by providing a potential path for fire through tree crowns (Cruz and others 2003; Fulé and others 2001; Graham and others 2004; Stratton 2004; Van Wagner 1977, 1993). Forest density further influences surface and canopy fuels through interactions with insects and diseases. The effects of bark beetles in ponderosa pine stands are more pronounced during and following extended droughts and under dense stand conditions; both of which are conducive to the survival and reproduction of beetle populations. Negrón (1997) showed a link between roundheaded pine beetle attacks and higher densities of smaller, pole-sized trees in relatively homogenous stands of ponderosa pine in the Sacramento Mountains of New Mexico. Additionally, trees with heavy mistletoe infection are more susceptible to severe crown scorch and death from fires (Harrington and Hawksworth 1990; Hoffman and others 2007). Hawksworth and Wiens (1996) suggested that mistletoes have been important species in frequent-fire forests since fire first appeared on these landscapes.

The density and arrangement of forest canopies affects the penetration of sunlight, precipitation, humidity, and wind. In fact, dense forest structures can maintain relatively high fuel moistures and ameliorate wind effects. Forest canopies also influence the composition and abundance of surface fuels, which are essential to facilitate fire as a disturbance agent. Surface fuels also offer nutrients to soils, help reduce erosion, and influence understory vegetation productivity, density, and diversity (Kalies and others 2012; Kerns and others 2003; Moore and others 1999). In general, more fuel accumulates and persists in forests with longer fire return intervals than in those with more frequent surface fire (Brewer 2008; Minnich and others 2000). Fine fuels (grass, needles, cones, and woody material less than 0.25 inches in diameter) and small branches accumulate more rapidly under tree groups than in interspaces between tree groups (Fig. 5). This accumulation facilitates fire, in turn restricting the establishment and persistence of trees and shrubs under tree groups. The amount and composition of surface fuels interact with weather conditions to influence fire behavior. Herbaceous fuels respond quickly to relative humidity and thus carry fire less readily when humidity is high, whereas pine needles will readily carry fire under these conditions (see moisture of extinction in Anderson 1982; Scott and Burgan 2005). Furthermore, needle and twig litter will burn with higher intensity than herbaceous fuel under similar weather conditions.

Forest structure affects the distribution, density, and composition of surface and canopy fuels, which affects the behavior of fire and, ultimately, post-fire forest structure. Historically, seedling establishment was more frequent in fire-created areas of bare mineral soil where competition with other vegetation and the abundance of surface fuels were reduced (Agee 1993; Cooper 1960; Stephens and others 2008). However, regeneration is less affected by the availability of bare mineral soil in some plant associations and soil types (Hanks and others 1983; USDA Forest Service 1997). A study in the Southwest showed a high density of tree regeneration on sites with one or more of the following: fine-textured soils, understories where screwleaf muhly was the dominant graminoid, and sites with high annual precipitation (Puhlick and others 2012). Depending on seed availability, some individuals and small groups of seedlings may establish throughout the stand, including under

Figure 5. (a) Fine fuels (grasses, forbs, needles, branches, cones) beneath the crown of an individual tree and **(b)** under the canopy of a tree group.

Figure 6. A group of ponderosa pine saplings in a grass-forb interspace between mature tree groups that experienced faster growth and survived a prescribed fire. Shade-suppressed saplings in heavier fine fuel loadings under a mature group of pine did not survive the fire.

tree groups (Abella 2008; Sánchez Meador and others 2009; White 1985).

Tree seedlings that established in small forest openings are subsequently thinned by later fires and/or other sources of mortality (Fig. 6) (Cooper 1960, 1961; Sánchez Meador and others 2010; Stephens and Fry 2005; White 1985). Young tree groups in open areas reach fire-resistant sizes more rapidly than those beneath closed canopies (Fitzgerald 2005; Sackett and Hasse 1998; York and others 2004). Fire-caused thinning of young tree groups was more substantial if the group was overtopped by older trees due to suppressed seedling growth and increased litter accumulation (Agee 1993; Cooper 1960). Fire-spread through young tree groups may also be influenced by microclimate and fuel moisture in these groups (Harrington 1982). As trees grow, increasing needle and twig accumulations facilitate the spread of surface fire. Seedlings that establish some distance away from mature older trees are also more likely to survive fires due to less rapid accumulation of fine fuels and small branches from overstory trees (Fig. 5, 6), likely leading to less intense and severe fire (Cooper 1960) and variable spacing of tree groups. The seasonality and burning conditions of fire occurrence also result in variable outcomes.

Spatial Patterns: Formation and Maintenance

Spatial patterns of vegetation are a component of forest structure. The historical spatial mosaic of tree groups, scattered individual trees, and openings in frequent-fire forests was maintained by interactions among the locations and types of fuels, the frequency and severity of fire, and tree regeneration and mortality patterns. A landscape mosaic of tree groups and scattered individual trees within an open grass-forb-shrub matrix, along with snags, logs, and woody debris, provides for the predominance of surface fire mixed with small-scale, variable fire behavior (e.g., torching). An open or grouped spatial structure reduces canopy continuity, decreasing a stand's vulnerability to active crown fire (Fitzgerald 2005; Fulé and others 2004; Roccaforte and others 2008; Stephens and others 2009). These interactions were mediated by small-scale variability in fire behavior and effects and often resulted in sites with aggregated tree regeneration that were temporarily "free" or "safe" from fire (Larson and Churchill 2012). The location of some safe-sites for tree regeneration appeared to be related to local areas of previously more intense fire associated with accumulations of coarse woody debris (logs and other dead woody material greater than 3 inches in diameter) originating from the death of individual trees (Sánchez Meador and Moore 2010; West 1969; White 1985) or tree groups (Cooper 1960; Stephens and Fry 2005; Taylor 2010; West 1969). Death of individuals or groups of old trees create new snags and logs that, when consumed by fire, result in "safe" sites for tree regeneration. Extended fire-free periods may allow tree regeneration in areas not typically fire "safe" (Fig. 7) (Fulé and others 2009), resulting in temporal shifting of tree locations where new cohorts develop to fire-resistant sizes. The cyclic repetition of forest vegetation dynamics stemming from disturbances and tree regeneration perpetuates a shifting mosaic of tree groups and individual trees in different stages of development in a grass-forb-shrub matrix (Fig. 8).

Figure 7. Ponderosa pine regeneration under a group of snags. This site is not currently fire "safe" due to the accumulation of surface fuels over an extended fire-free period. In the absence of fire, these seedlings could grow to fire-resistant sizes. If fire occurs prior to the trees attaining fire-resistant size, the seedlings would likely not survive. However, the reduction of surface fuels post-fire may create a temporary fire-safe site for future regeneration.

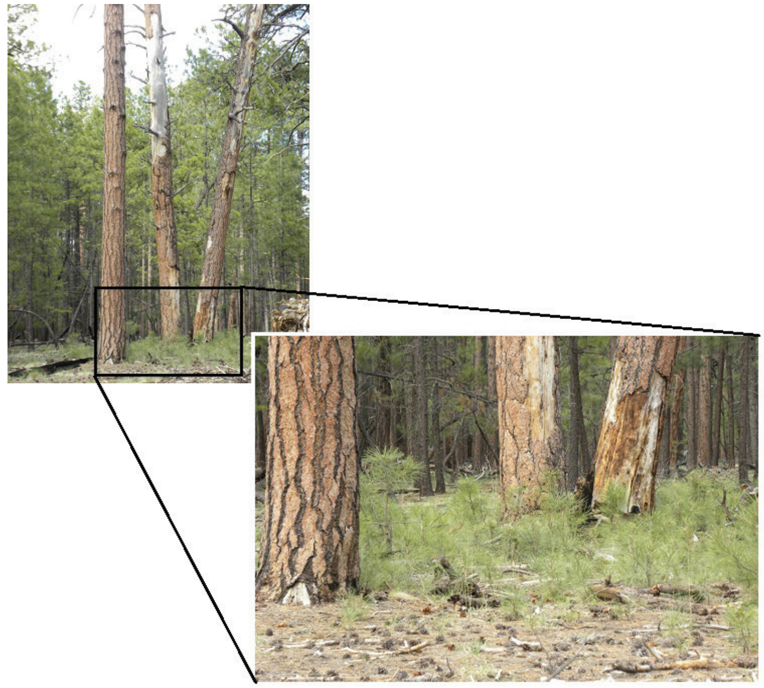

Figure 8. Tree groups and a single individual tree on the right in a grass-forb-shrub interspace.

Insects and diseases also shape spatial patterns of forested landscapes. Due to the slow spread of infection, it has been suggested that the current distribution of mistletoe throughout the Southwest is likely similar to its historical distribution, although spatial continuity and levels of infection may have changed (Conklin and Fairweather 2010). Under historical forest conditions, it is likely that large-scale, contiguous insect and disease outbreaks would have been rare. It is more likely that mistletoe would have thrived in denser multi-storied portions of stands that escaped fire pruning and thinning (see Conklin and Geils 2008 for additional discussion). In such portions, periodic tree deaths would have occurred directly from mistletoe, or infected trees would have had increased the likelyhood of succumbing to bark beetles or root disease. Localized mistletoe infections would have created pockets of tree death that could eventually serve as regeneration sites. In cases where regeneration occurred in larger openings between trees, trees may have escaped mistletoe infection altogether. Other scenarios can be envisioned. For instance, in cases of stands with relatively homogenous age and spacing, bark beetles may have had periodic population increases, causing high rates of local mortality. Localized beetle outbreaks likely occurred in stands with severe crown damage following fire (Breece and others 2008), and these infestations may have spilled over into undamaged trees nearby, creating larger openings. Root diseases also create scattered mortality, small openings, and increased volume of snags and downed large woody debris (Rippy and others 2005).

An understanding of forest processes and their effects at different spatial scales is important because landscapes are spatially dependent (Turner 1989). Inferences about patterns and processes in forests are contingent upon the scale at which they are investigated. For example, a fine-scale model for ponderosa pine regeneration showed that the majority of the variance (76 percent) in seedling density was explained by properties such as soil texture and pH, precipitation, seed tree proximity, and composition of the plant community (Puhlick and others 2012). However, at the mid- to landscape-scale, models including abiotic conditions and tree density at this broader scale accounted for less of the variability in observed seedling densities (only 13 percent) (Puhlick and others 2012). Fire further shapes tree spatial patterns at varying scales through its influence on seedling survival, with variability in the severity, seasonality, and frequency of fire (Cooper 1960; Pearson 1950; Stephens and Fry 2005; Taylor 2010; West 1969; White 1985). An overall aggregated (grouped) historical tree pattern separated by openings has been frequently reported in Southwestern frequent-fire forests (Fig. 8) (Larson and Churchill 2012). However, Abella (2008), Binkley and others (2008), and Sánchez Meador and others (unpublished data, see Table 3 footnote) observed grouped and random (no aggregation) historical tree spatial patterns (Fig. 9). Schneider (2012) observed only random historical tree spatial patterns in Southwestern ponderosa pine.

Spatial heterogeneity can exist at any scale, and the value of metrics used to assess forest conditions varies in usefulness with scale. At mid- and landscape scales, elements such as single tree and group density become less useful as a metric and elements such as patches, the grass-forb-shrub matrix, stand density, canopy cover, and basal area become more appropriate. Patches are roughly synonymous with stands, being defined as an area of relatively homogeneous vegetation composition and structure differing from its surroundings (Forman 1995). Patches are the basic unit of the landscape, and their sources of variability are attributed to scale-appropriate factors such as elevation, topography, climate, and land use. Our restoration framework describes forest composition, structure, and spatial patterns at fine-, mid-, and landscape-scales (Fig. 1).

Figure 9. Random (i.e., not aggregated) distribution of ponderosa pine trees in a patch of old trees. Also displayed are snags, logs, and coarse woody debris.

Southwestern Frequent-Fire Forests

The natural range of variability is a "best" estimate of a resilient and functioning ecosystem because it reflects the evolutionary ecology of these forests. Natural range of variability is therefore a powerful science-based foundation for developing a framework for restoring the composition and structure of forests (Kaufmann and others 1994; Keane and others 2009; Moore and others 1999). The natural range of variability can be estimated by pooling reference conditions across sites within a forest type. Reference conditions for a forest type typically vary from site to site due to differences in factors such as soil, elevation, slope, aspect, and micro-climate and manifests as differences in fire effects, tree densities, patterns of tree establishment and persistence, and numbers and dispersion of snags and logs. When pooled, these sources of variability comprise the natural range of variability of a site or forest type.

Our estimates of natural ranges of variability are derived from multiple lines of evidence based on historical ecology techniques (Egan and Howell 2001) such as written and oral historical records, historical photographs, early forest inventories, and dendrochronological studies (Table 4). While cultural accounts and early inventories provide a general context of historical conditions, they do not fully characterize forest structure by today's statistical standards. More recently, dendrochronological techniques for quantifying historical conditions, including spatial and temporal variation, have been developed (e.g., Covington and Moore 1994a; Covington and others 1997; Fulé and others 1997; Mast and others 1999; Sánchez Meador and others 2010; White 1985). Nonetheless, there is a clear need for additional reference condition data sets, including sites from a wider spectrum across environmental gradients (e.g., soils, moisture, elevations, slopes, aspects) occupied by frequent-fire forests in the Southwest, especially in dry mixed-conifer. While the quantity of reference data sets is increasing, existing data represent a largely unbalanced sampling across gradients (e.g., most data sets are from basaltic soils and on dry to typic plant associations), and there have been few studies quantitatively examining and reporting spatial patterns of trees and the sizes and shapes of grass-forb-shrub interspaces.

Ponderosa Pine

Woolsey (1911) described Southwestern ponderosa pine forests as having "…pure park-like stand(s) made up of scattered groups of 2-20 trees, usually connected by scattering individuals. Openings are frequent and vary in size. Because of the open character of the stand and the fire-resisting bark, often 3 inches thick, the actual loss in yellow (ponderosa) pine by fire is less than with other more gregarious species." Others also described historical ponderosa pine forests as having low density, open stands consisting of groups of pine trees interspersed with grassy or shrubby openings (Dutton 1882; Lang and Stewart 1910; Pearson 1923; White 1985).

Tree density, structure, spatial pattern, and ecological functions in today's ponderosa pine forests of the Southwest are greatly altered from their historical conditions. Most Southwest ponderosa pine forests are at much greater risk of high-intensity, severe fire than they were prior to Euro-American settlement (Covington 1993; Fulé and others 2004; Moore and others 1999; Roccaforte and others 2008). Historical ponderosa pine forests had widely spaced, large trees, typically occurring in small groups with scattered single trees, and open forest conditions with a productive grass-forb-shrub understory (Cooper 1960; Dutton 1882; Lang and Stewart 1910; Pearson 1923, 1950; Sánchez Meador and others 2009, 2011; White 1985). The grass-forb-shrub vegetation and other fine fuels and branches carried fires started by lightning and, to an uncertain extent, by Native Americans (Allen and others 2002; Kaye and Swetnam 1999). Forest composition, structure, and spatial patterns were maintained by low-severity surface fires that occurred every 2-26 years (Fig. 3), rarely killing large trees, thinning regeneration, and maintaining an open forest structure (Dieterich 1980; Fiedler and others 1996; Fitzgerald 2005; Pearson 1950; Swetnam and Dieterich 1985; Weaver 1951; Woolsey 1911). Fire chronologies in Western U.S. frequent-fire forests are

Table 4. Citations informing our restoration framework for frequent-fire forests arranged by information type.

Information type	Citations (arranged alphabetically)
Reference conditions from old-growth, natural areas, and other restoration studies	Abella (2008); Abella and Denton (2009); Abella and others (2011); Agee (2003); Binkley and others (2008); Biondi (1996); Biondi and others (1994); Boyden and others (2005); Cocke and others (2005); Cooper (1960, 1961); Covington and Moore (1994a, 1994b); Covington and Sackett (1986); Covington and others (1997); Fornwalt and others (2002); Friederici (2004); Fulé and others (1997, 2002a, 2003, 2009); Harrod and others (1999); Heinlein and others (1999, 2005); Hessburg and others (1999); Johnson (1994); Larson and Churchill (2012); Madany and West (1983); Mast and others (1999); Menzel and Covington (1997); Moore and others (2002, 2004); Pearson (1950); Roccaforte and others (2010); Romme and others (2009); Sánchez Meador and Moore (2010); Sánchez Meador and others (2009, 2010, 2011); Schneider (2012); Smith (2006a, 2006b, 2006c); Taylor (2010); Waltz and Fulé (1998); West (1969); White (1985); White and Vankat (1993); Williams and Baker (2011, 2012); Youngblood and others (2004)
Reference conditions from observations of early explorers, scientists, and managers	Beale (1858); Dutton (1882); Greenamyre (1913); Lang and Stewart (1910); Leopold (1924); Liebeg and others (1904); Meyer (1934); Pearson (1923); Plummer (1904); Rasmussen (1941); Wheeler (1875); Woolsey (1911)
Disturbance histories	Agee (1993); Allen (2007); Andrews and Daniels (1960); Brown and others (2001); Brown and Wu (2005); Dieterich (1980); Ehle and Baker (2003); Ferry and others (1995); Fulé and others (2003); Fulé and others (2004); Grissino-Mayer and others (1995, 2004); Hart and others (2005); Heinlein and others (2005); Hessburg and others (1994); Hessburg and others (2005); Kaye and Swetnam (1999); Korb and others (2013); Littell and others (2009); Lynch and others (2010); Maffei and Beatty (1988); Minnich and others (2000); Scholl and Taylor (2010); Stephens and others (2008); Swetnam and Baisan (1996); Swetnam and Bentacourt (1990); Swetnam and Dieterich (1985); Taylor (2010); Taylor and Skinner (2003); Touchan and others (1996); Weaver (1951); Williams and Baker (2012)

Table 4. *Continued.*

Information type	Citations (arranged alphabetically)
Disturbance effects (fires, insects, and diseases)	Arno and others (1995); Barton (2002); Bentz and others (2009); Conklin and Geils (2008); Castello and others (1995); DeLuca and Sala (2006); Dhillon and Anderson (1993); Drummond (1982); Edmonds and others (2000); Fettig (2012); Fitzgerald (2005); Franklin and others (2012); Fulé and Laughlin (2007); Goheen and Hansen (1993); Harrington and Hawksworth (1980); Hawksworth and Wiens (1996); Hessburg and others (1994); Hoffman and others (2007); Jenkins and others (2008); Lundquist (1995); Madany and West (1983); Miller and Keen (1960); Miller (2000); Moeck and others (1981); Naficy and others (2010); Negrón (1997); Negrón and others (2009); Parsons and DeBenedetti (1979); Rippy and others (2005); Savage and Mast (2005); Stevens and Hawksworth (1984); Tainter and Baker (1996); Von Schrenck (1903); Wickman (1963); Wood (1983)
Effects of forest management on ecosystem functions and processes	Arnold (1950); Baker (1986, 2003); Benayas and others (2009); Beier and others (2008); Boerner and others (2009); Breece and others (2008); Carey (2003); Carey and others (1999); Cocke and others (2005); Colgan and others (1999); Conklin and Geils (2008); Cortina and others (2006); Covington and others (1997); Covington and Sackett (1986, 1992); Cram and others (2007); Dodd and others (2006); Douglass (1983); Feeney and others (1998); Fettig and others (2007); Ffolliott and others (1989); Finkral and Evans (2008); Fulé and others (2001); Harr (1983); Honig and Fulé (2012); Kolb and others (1998); Koonce and Roth (1980); Korb and others (2003); Long and Smith (2000); Mitchell and others (2009); Moore and others (2006); Pilliod and others (2006); Roccaforte and others (2008); Stephens and others (2009); Stratton (2004); Strom and Fulé (2007); Troendle (1983); Waltz and Covington (2003); Wightman and Germaine (2006)
Climate change projections and impacts	Bentz and others (2010); Breshears and others (2005); Brown and others (2004); Harris and others (2006); Honig and Fulé (2012); Karl and others (2009); McKenzie and others (2004); Millar and others (2007); Miller and others (2009); Overpeck and others (2012); Parker and others (2000); Price and Neville (2003); Seager and others (2007); Shafer and others (2001); Smith and others (2008); Spittlehouse and Stewart (2004); Spracklen and others (2009); Westerling and others (2006)
Approaches to restoration and/or monitoring	Allen and others (2002); Aronson and others (2007); Block and others (2001); Bradshaw (1984); Busch and Trexler (2003); Clewell and others (2005); Covington (1993, 2003); Covington and others (1997); Crist and others (2009); Egan and Howell (2001); Falk (2006); Fiedler and others (1996); Fitzgerald (2005); Fulé and others (2002b); Graham and others (2004); Kaufmann and others (1994); Keane and others (2009); Landres and others (1999); Laughlin and others (2006); Lindenmayer and Likens (2010); Long and others (2004); Moore and others (1999); Morgan and others (1994); Mulder and others (1999); Murray and Marmorek (2003); Noon (2003); Palmer and Mulder (1999); Reynolds and others (1992, 2006a); Roccaforte and others (2010); SER (2004); Sitko and Hurteau (2010); Swetnam and others (1999); Wagner and others (2000); Walters (1986); Williams and others (2009)
Science syntheses and tools for forest management	Abella (2008); Abella and others (2006); Anderson (1982); Brewer (2008); Brown and others (2003); Clary (1975); Conklin and Fairweather (2010); Cruz and others (2003); Evans and others (2011); Graham and others (1994); Hunter and others (2007); Long (1985); Noss and others (2006); Patton and Severson (1989); Pearson (1950); Schmidt and others (2002); Schubert (1974); Scott and Burgan (2005); Triepke and others (2011); USDA Forest Service (1990)
Vegetation classifications	Comer and others (2003); DeVelice and others (1986); Hanks and others (1983); USDA Forest Service (1997); Winthers and others (2005)

reviewed in Evans and others (2011), Hunter and others (2007), Smith (2006b), and Swetnam and Baisan (1996).

Bark beetles also influenced pre-Euro-American ponderosa pine structure. Various sources indicate that bark beetle outbreaks occurred periodically in the Western United States since at least the 1750s (Bentz and others 2009) and likely much longer. Current forested landscapes are experiencing outbreaks that are larger and more frequent than previously recorded (Lynch and others 2010). For example, bark beetles caused variable amounts of mortality on more than 700,000 acres in Arizona and New Mexico in 2003 (Fettig and others 2007; USDA Forest Service 2004). Although there is no direct evidence linking the effects of bark beetles to the structure of pre-Euro-American frequent-fire forests, evidence from today's beetle population dynamics suggests that homogenous, dense, even-aged stands are highly susceptible to beetle outbreaks (Fettig and others 2007; Negrón 1997). However, historical observations suggest that high-density, even-aged stand structures were infrequent or rare in frequent-fire forests (Woolsey 1911; reviewed in Covington and Moore 1994a, 1994b). Alternatively, spatial heterogeneity would have been promoted and maintained at the fine scale by bark beetle attacks on single or small groups of trees, or perhaps in high density groups or patches, which would have created growing space for regeneration or surviving trees (Fettig 2012; Lundquist 1995; Von Schrenck 1903). During droughts, it was likely that many more trees would have succumbed to bark beetles (Bentz and others 2010; Negrón and others 2009). Bark beetles evolved under the range of natural variability where there would have been sufficient hosts (e.g., fire-stressed, lightning struck, and broken top trees) to maintain endemic beetle populations (reviewed in Jenkins and others 2008 and Moeck and others 1981).

Ponderosa Pine: Species Composition: Ponderosa pine is the dominant seral and climax tree species in Southwest ponderosa pine forests. Depending on locale, ponderosa pine forests may also have a mix of Gamble oak, evergreen oaks, junipers, pinyon pines (DeVelice and others 1986), with occasional presence of quaking aspen, New Mexico locust, Douglas-fir, or southwestern white pine. Ponderosa pine is one of the most fire-adapted conifer species in the West, and its resistance to surface fire increases as trees age (Miller 2000).

Composition of the grass-forb-shrub community in ponderosa pine forests is typically diverse, especially in open interspaces between trees (e.g., Fig. 8) (Abella and others 2011; Laughlin and others 2006; Moir 1966; Naumburg and DeWald 1999). Ponderosa pine plant associations (classified by understory plant assemblages, plant succession, and co-dominant tree species) are variable and are reflective of local biophysical site and climate conditions that both influence the type of disturbances and vegetation responses to disturbances (Table 5) (USDA Forest Service 1997). Southwestern ponderosa pine plant associations range from pure ponderosa pine to mixed tree species overstories with understories ranging from bunchgrass/forb to shrub-dominated types, and these can be broadly grouped into four forest subtypes: (1) ponderosa pine-bunchgrass, (2) ponderosa pine-Gambel oak, (3) ponderosa pine-evergreen oak, and (4) ponderosa pine-shrub (Appendix 2). The most mesic sites are the ponderosa pine-Gambel oak and some ponderosa pine-bunchgrass plant associations; the most xeric sites are the ponderosa pine-evergreen oak and some ponderosa pine-shrub plant associations. Bunchgrass plant associations generally occupy the mid-range of the moisture gradient for ponderosa pine forests in the Southwest.

Understory composition includes various combinations of grasses, forbs, shrubs, ferns, and cacti depending upon plant associations (Korb and Springer 2003; USDA Forest Service 1997), all of which contribute to the biodiversity found in frequent-fire forests (Laughlin and others 2006). The growth habit (e.g., bunchgrass, sod, or shrub) and spatial patterns of the understory influence the establishment and growth of trees (Biondi 1996; Boyden and others 2005; Sánchez Meador and others 2009; Youngblood and others 2004) and provide wildlife habitats (Dodd and others 2006; Reynolds and others 1992; Waltz and Covington 2003; Wightman and Germaine 2006; USDA Forest Service 1997). Variation in species composition among plant associations within forest subtypes influences forest dynamics. For example, within the ponderosa pine bunchgrass subtype, tree regeneration establishes rapidly following disturbance on sites with screwleaf muhly plant associations (the most mesic associations in the bunchgrass subtype), episodically on Arizona fescue plant associations (the typic associations in the bunchgrass subtype), and sparsely on blue grama plant associations (the most xeric associations in the bunchgrass subtype) (USDA Forest Service 1997). Tree establishment often occurs differently in shrub-dominated plant associations than in bunchgrass types, where rapid re-sprouting of shrub species (e.g., shrub live oak) following disturbances may inhibit pine regeneration. Other re-sprouting shrubs (e.g., New Mexico locust) are nitrogen-fixers and have been shown to facilitate pine seedling establishment (Fisher and

Table 5. Plant associations for the ponderosa pine series in the Southwestern United States sorted by ponderosa pine subtype, temperature-moisture gradient, dominant season of precipitation, and parent material type (USDA Forest Service 1997).

Plant association (common name)	Ponderosa pine subtype	Temperature-moisture gradient[a]	Climate[b]	Dominant season of precipitation[c]	Parent material type[d]
Ponderosa pine plant association series					
Ponderosa pine/screwleaf muhly-Arizona fescue	Bunchgrass	Cool-wet	Cold	Summer	Variable
Ponderosa pine/screwleaf muhly-Arizona fescue/blue grama	Bunchgrass	Cool-wet	Cold	Summer	Variable
Ponderosa pine/screwleaf muhly-Arizona fescue/Gambel oak	Bunchgrass	Cool-wet	Cold	Summer	Variable
Ponderosa pine/screwleaf muhly	Bunchgrass	Cool-wet	Cold	Winter	Sed./rhy./tuff
Ponderosa pine/screwleaf muhly/Gambel oak	Bunchgrass	Cool-wet	Cold	Winter	Variable
Ponderosa pine/Arizona fescue	Bunchgrass	Typic	Cold	Winter	Variable
Ponderosa pine/Arizona fescue/Parry's oatgrass	Bunchgrass	Typic	Cold	Winter	Volcanic
Ponderosa pine/Arizona fescue/blue grama	Bunchgrass	Typic	Cold	Winter	Variable
Ponderosa pine/Arizona fescue/Gambel oak	Bunchgrass	Typic	Cold	Winter	Variable
Ponderosa pine/blue grama	Bunchgrass	Warm-dry	Cold	Summer	Variable
Ponderosa pine/blue grama/gray oak	Bunchgrass	Warm-dry	Cold	Summer	Variable
Ponderosa pine/blue grama/Gambel oak	Bunchgrass	Warm-dry	Cold	Summer	Variable
Ponderosa pine/mountain muhly	Bunchgrass	Warm-dry	Cold	Summer	Variable
Ponderosa pine/Arizona walnut	Bunchgrass	Warm-dry	Cold	Summer	Alluvium
Ponderosa pine/Indian ricegrass	Bunchgrass	Warm-dry	Cold	Winter	Aeolian
Ponderosa pine/rockland	Bunchgrass	Variable	Variable	Variable	Variable
Ponderosa pine/Gambel oak	Gambel oak	Cool-wet	Cold	Winter	Variable
Ponderosa pine/Gambel oak/Arizona fescue	Gambel oak	Cool-wet	Cold	Winter	Variable
Ponderosa pine/Gambel oak/mountain muhly	Gambel oak	Cool-wet	Cold	Winter	Variable
Ponderosa pine/Gambel oak/New Mexico locust	Gambel oak	Cool-wet	Cold	Winter	Variable
Ponderosa pine/Gambel oak/longtongue muhly	Gambel oak	Warm-dry	Cold	Winter	Sed./gran.
Ponderosa pine/Gambel oak/two needle pinyon	Gambel oak	Warm-dry	Cold	Winter	Variable
Ponderosa pine/Gambel oak/blue grama	Gambel oak	Warm-dry	Cold	Winter	Variable

Table 5. *Continued.*

Plant association (common name)	Ponderosa pine subtype	Temperature-moisture gradient[a]	Climate[b]	Dominant season of precipitation[c]	Parent material type[d]
Ponderosa pine plant association series					
Ponderosa pine/netleaf oak	Evergreen oak	Cool-wet	Mild	Summer	Volcanic
Ponderosa pine/silverleaf oak	Evergreen oak	Typic	Mild	Summer	Volcanic
Ponderosa pine/gray oak/mountain muhly	Evergreen oak	Warm-dry	Mild	Summer	Variable
Ponderosa pine/gray oak/longtongue muhly	Evergreen oak	Warm-dry	Mild	Summer	Variable
Ponderosa pine/Arizona white oak	Evergreen oak	Warm-dry	Mild	Summer	Variable
Ponderosa pine/Arizona white oak/blue grama	Evergreen oak	Warm-dry	Mild	Summer	Variable
Ponderosa pine/Emory oak	Evergreen oak	Warm-dry	Mild	Winter	Alluvium
Ponderosa pine/blue grama/big sagebrush	Shrub	Cool-wet	Cold	Winter	Sed.
Ponderosa pine/kinnikinnik	Shrub	Typic	Cold	Winter	Rhy./tuff/gran.
Ponderosa pine/black sagebrush	Shrub	Warm-dry	Cold	Winter	Sed.
Ponderosa pine/pointleaf manzanita	Shrub	Warm-dry	Mild	Summer	Variable
Ponderosa pine/wavyleaf oak	Shrub	Warm-dry	Cold	Summer	Variable
Ponderosa pine/Stansbury cliffrose	Shrub	Variable	Cold	Winter	Limestone

[a]Typic refers to modal, mid-gradient temperature-moisture types.
[b]Climate refers to mean annual soil temperatures, with cold climates having frigid soils (mean annual soil temperatures <8 ˚C) and mild climates having mesic soils (mean annual soil temperatures >8 ˚C).
[c]Dominant season of precipitation refers to the 6-month period (winter = October-March, summer = April-September) that typically has higher average precipitation levels. Most ponderosa pine and dry mixed-conifer sites in the Southwestern United States receive bimodal precipitation, but the season listed in the table experiences higher average precipitation levels.
[d]Variable = multiple parent materials; sed. = sedimentary; rhy. = rhyolites; gran. = granites

Fulé 2004; USDA Forest Service 1997). Fire may also facilitate establishment of tree regeneration on sites with non-sprouting shrub species (e.g., black or big sagebrush species) by removing competition. Together, trees and the grass-forb-shrub community affect below- and aboveground microclimates (i.e., soil moisture, nutrients, etc.) as well as ecological processes and functions such as biodiversity, trophic interactions, food webs, disturbances, and hydrology (Abella 2009; Arnold 1950; Barth 1980; Covington and others 1997; Kalies and others 2012; Moir 1966; Parker and Muller 1982; Scholes and Archer 1997) (see Expected Outcomes of Framework Implementation). Environmental variables such as light intensity, soil pH, soil and litter depth, and percent litter cover are directly influenced by the presence of tree canopy cover (Evenson and others 1980). For example, Abella (2009) reported that understory species richness was greater and plant cover was up to eight times greater in openings than under tree canopies in a ponderosa pine/Gamble oak forest.

Mycorrhizal fungi are important species in ponderosa pine and play an important role in plant nutrition, nutrient cycling, soil structure, and food webs (Carey 2003; Johnson and others 1997). Two Arizona studies reported higher densities of mycorrhizal propagules in areas where grass cover was greater and tree cover was less, such as in areas following mechanical treatments and burning, and that increased light and soil moisture in restored stands likely increased photosynthesis and mycorrhizal infection (Korb and others 2003; Korb and Springer 2003). Other studies show that abundant arbuscular mycorrhizae can increase plant diversity and overall community structure (Klironomos and others 2000; van der Heijden and others 1998). Arbuscular mycorrhizae are particularly important in grass-dominated ecosystems (Dhillion and Anderson 1993; Koske and Gemma 1997), but little is known of their status in the grass-forb-shrub community in ponderosa forests (Korb and Springer 2003).

Ponderosa Pine: Forest Structure: Structure in ponderosa pine forests emanates from the vertical and horizontal arrangement of trees and grass-forb-shrub species. Specifically, the vertical and horizontal architecture of a forest arises from variations in tree and grass-forb-shrub species and their ages, heights, crown spreads, densities, and spatial heterogeneity. Human activities since the late 19th Century resulted in changes to forest structure due to a reduction in fire frequency causing tree density and surface fuel load increases (Moore and others 2004; Naficy and others 2010; Parsons and DeBenedetti 1979; Scholl and Taylor 2010). For example, Moore and others (2004) reported a mean tree density increase by a factor of almost 7 (32-208 trees per acre) between 1909 and the 1990s. Tree encroachment into grass-forb-shrub forest openings has resulted in a decline in percent cover, abundance, and biodiversity of open grass-forb-shrub communities (Abella 2009; Bogan and others 1998; Clary 1975; Covington and Moore 1994b; Moore and others 2006; Moore and Deiter 1992; Swetnam and others 1999).

Differences in reference conditions for tree densities have been reported for fine- versus coarse-textured soils (Abella and Denton 2009; Puhlick 2011). Average plot-level reference conditions in ponderosa pine on basalt soils ranged between 0-220 trees per acre and 33-83 square ft per acre of basal area while sites on coarse-textured soils (primarily limestone) ranged between 8 and 262 trees per acre and 22 and 89 square ft per acre of basal area (Table 6; Fig. 10). In general, ranges reported for reference tree densities on coarse-textured soils were higher than those reported on fine-textured soils (Table 6). The minimum diameters reported in Table 6 may also result in a source of error that can lead to small underestimates of historical tree densities reported in studies. Additional error may result from missing fully decomposed structures at time of measurement and reconstruction (Fulé and others 1997; Mast and others 1999; Moore and others 2004).

To date, only six studies report tree spatial reference conditions in the Southwestern ponderosa pine forests. Based on these studies, the historical conditions in ponderosa pine exhibited as many as 67 tree groups per acre. Tree groups ranged between 0.003 and 0.72 acres in size and were composed of 2-72 trees (Table 3; Fig. 4). Tree groups were separated by grass-forb-shrub openings of variable sizes and shapes that contained scattered individual trees (Fig. 8). The proportion of the stand or mid-scale area not covered by vertical projections of tree crowns (referred to as "openness") has received little attention. However, several studies have reported the inverse of openness—canopy cover (Table 7); White (1985), Covington and Sackett (1986), and Covington and others (1997) reported 21.9, 19.0, and 17.3 percent canopy cover for ponderosa pine stand reference conditions on the Fort Valley Experimental Forest, Arizona, respectively. A nearby study of a reconstructed ponderosa pine/Gambel oak site on the Coconino National Forest, Arizona, reported a range of 10.2-18.8 percent canopy cover (Sánchez Meador and others 2011). Fulé and others (2002) reported an average canopy cover of 48.3 percent for the Rainbow Plateau, an area in the Grand Canyon National Park-North Rim

Table 6. Historical forest structure of ponderosa pine (pine-oak shaded) forests of the Southwest, arranged by parent material and mean tree density.

Location	Parent material	Elevation (ft)	Size/age reported	Reference date	Trees per acre			Basal area (ft^2/acre)			Citation
					Range	Mean	Std Err	Range	Mean	Std Err	
Gus Pearson Natural Area, Arizona	Basalt	7398	Age	1875		15.0					White 1985
Coconino National Forest, Arizona (avg)[a]	Basalt	6907	Size	1910		16.0			38.1		Woolsey 1911
Gus Pearson Natural Area, Arizona[b]	Basalt	7400	Size	1925		21.8			56.6		Pearson 1950
Gus Pearson Natural Area, Arizona	Basalt	7300	No	1876		22.8			46.2		Covington and others 1997
Bar M Canyon, Arizona	Basalt	7000	No	1867	21-24	23.0			65.0		Covington and Moore 1994b
Flagstaff, Arizona[c]	Basalt	7355	No	1880	1-58	23.7	4.0				Abella and Denton 2009
Gus Pearson Natural Area, Arizona	Basalt	7300	Age	1876		24.0					Mast and others 1999
San Francisco Peaks, Arizona	Basalt	8594	Age	1876		24.8	2.6		33.0	4.9	Cocke and others 2005
Mt. Trumbull, Arizona	Basalt	7740	Age/Size	1870		25.2	3.5		38.8	6.1	Heinlein and others 1999
Coconino National Forest, Arizona (max)[a]	Basalt	6907	Size	1910		34.5			81.2		Woolsey 1911
Mt. Logan, Arizona[b]	Basalt	7483	Age/Size	1870		38.3	5.8		46.2	7.8	Waltz and Fulé 1998
Mt. Trumbull, Arizona	Basalt	6970	Size	1870	0-220	39.2	3.9	0-143	41.6	4.1	Roccaforte and others 2010
Chimney Spring, Arizona[a]	Basalt	7380	Size	1920		42.8					Biondi and others 1994
Coulter Ranch, Arizona[a]	Basalt	7520	Size	1913	30-66	51.5	10.8	67-120	83.0	19.5	Sánchez Meador and Moore 2010
Camp Navajo, Arizona	Basalt	7592	Age/Size	1883		59.9	5.8		56.2	6.1	Fulé and others 1997
Malay Gap, Arizona[b]	Basalt	7200	Age/Size	1952		124.0			70.1		Cooper 1960
Woolsey Plots, Arizona[a]	Basalt	7052	Size	1874	18-51	33.1	4.6	40-79	61.5	5.6	Sánchez Meador and others 2010
Flagstaff, Arizona[c]	Cinders	7355	No	1880	7-74	22.5	6.2				Abella and others 2011
Mt. Logan, Arizona[c]	Cinders	7115	Age/Size	1870	34-38	29.9	6.4	60-64	60.3	9.1	Waltz and Fulé 1998
Red Cinder, Arizona	Cinders	7631	Age/Size	1885		74.1			65.3		Abella 2008
Prescott National Forest, Arizona (avg)[a]	Granitic	5320	Size	1910		27.7			25.5		Woolsey 1911

Location	Parent material	Elevation (ft)	Size/age reported	Reference date	Trees per acre			Basal area (ft2/acre)			Citation
					Range	Mean	Std Err	Range	Mean	Std Err	
Tusayan, Arizona (avg)[a]	Limestone	7075	Size	1910		10.7			22.1		Woolsey 1911
Zion National Park, Utah	Limestone	7096	Age	1881	3-25	14.0					Madany and West 1983
Flagstaff, Arizona[a]	Limestone	7355	No	1880	14-34	22.0	2.2				Abella and others 2011
Walnut Canyon National Monument, Arizona[d]	Limestone	6808	Size	1876		29.1			39.2		Menzel and Covington 1997
North Kaibab, Arizona	Limestone	7300	No	1881		55.9					Covington and Moore 1994a
Kaibab Plateau, Arizona[c]	Limestone	7500	No	1929	40-55						Rasmussen 1941
Grandview, Arizona	Limestone	7422	Age	1887	4-247	64.6	10.4	19-281	74.1	12.6	Fulé and others 2002
Fire Point, Arizona	Limestone	7671	Age	1887	16-126	61.8	61.8	28-132	89.3	9.1	Fulé and others 2002
Rainbow Plateau, Arizona	Limestone	7612	Age	1879	8-228	56.7	5.7	1-99	39.6	2.6	Fulé and others 2002
Powell Plateau, Arizona	Limestone	7533	Age	1879	8-262	63.6	9.4	20-337	78.0	10.9	Fulé and others 2002
Zuni, New Mexico (max)[a]	Rhyolite	6557	Size	1910		22.6			52.8		Woolsey 1911
Cibola National Forest, New Mexico[a]	Rhyolite	8382	Age/Size	1890	47-61	54.2	6.9				Moore and others 2004
Carson, New Mexico (max)[a]	Shale	6983	Size	1910		38.4			79.9		Woolsey 1911
Mogollon Plateau, Arizonac	Mixed	Mixed	No	1890	33-76	57.3	30.7	48-79			Williams and Baker 2011, 2012
Uncompahgre Plateau, Colorado	Shale	7500	Size	1875	30-90	55		20-90	55		Binkley and others 2008

[a]Minimum tree DBH recorded = 3.5 inches
[b]Minimum tree DBH recorded = 6 inches
[c]Minimum tree DBH recorded = 4 inches
[d]Minimum tree DBH recorded = 10 inches

Table 7. Historical canopy cover and openness estimates of frequent-fire forests of the Southwest, arranged by forest type (PP: ponderosa pine, PO: pine-oak, DMC: dry mixed-conifer).

Location	Parent material	Forest type	Method	Reference date	Canopy cover (%)	Openness[a] (%)	Citation
Gus Pearson Natural Area, Arizona	Basalt	PP	Standing age class	1875	21.9	78.1	White 1985
Gus Pearson Natural Area, Arizona	Basalt	PP	Dendro-reconstruction	1876	19.0	81.0	Covington and others 1997
Chimney Springs, Arizona	Basalt	PP	Standing size class	1876	17.3	82.7	Covington and Sackett 1986
Woolsey Plots, Arizona	Basalt	PP/PO	Dendro-reconstruction	1874	10.2-18.8	81.2-89.8	Sánchez Meador and others 2011
Rainbow Plateau, Arizona	Limestone	PP/PO	Dendro-reconstruction	1879	48.3	51.7	Fulé and others 2002a
Cheesman Lake, Colorado	Granitic	PP/DMC	FVS[b]-reconstruction	1900	12.9-21.5	78.5-87.1	Fornwalt and others 2002

[a]Openness is the proportion of area not covered by tree crowns, estimated as the inverse of canopy cover.
[b]Forest Vegetation Simulator

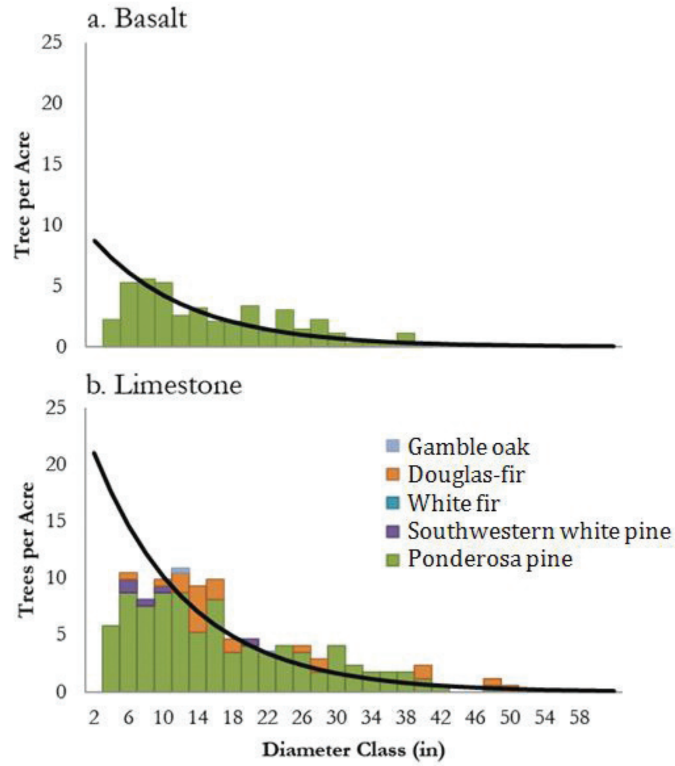

Figure 10. Theoretical diameter distributions representing reference conditions illustrating a superimposed basal area-diameter distribution (BDq) (where q = 1.2); (a) pure ponderosa pine present on basalt soils, (b) dry mixed-conifer on limestone soils. Seedling and sapling-sized tree distribution (i.e., trees in the 2-inch DBH class) on both sites may not be fully represented.

where the authors suggested that contemporary conditions were statistically similar to historical reference conditions as determined by basal area comparisons. A reference condition study conducted in ponderosa pine near Cheeseman Lake, Colorado, reported a range of 12.9-21.5 percent canopy cover (Fornwalt and others 2002). Overall, the range of canopy cover for ponderosa pine for these studies was about 10-50 percent, giving reference conditions for openness (i.e., inverse of canopy cover) of 50-90 percent. If areas with strong tree aggregation (i.e., with interlocking crowns; Fig. 11) exhibit lower mid-scale canopy cover (10.2-21.9 percent; Table 7), then it stands to reason that sites with less tree aggregation would have higher mid-scale canopy cover due to tree arrangement and reduced crown overlap (Christopher and Goodburn 2008).

Snags, logs, and woody debris are important structural and functional elements in frequent-fire forests (Figs. 12 and 13), yet little is known about volumes of coarse woody debris under historical fire regimes. Nonetheless, studies using extensive, historical stem-maps and/or locations of historical evidences (e.g., logs,

Figure 11. Interlocking or nearly interlocking crowns are components of groups of mid-aged to old trees.

Figure 12. Snags, logs, and woody debris are important components of frequent-fire forests. They provide structural diversity, nutrient cycling, and wildlife habitat.

stumps, and snags) reported a mean of 2.3 snags and 2.7 logs per acre (Moore and others 2004), 1-8 snags and 3-23 logs per acre (Sánchez Meador and others 2010), and 10 snags and 20 logs per acre as reference conditions for southwestern ponderosa pine (Abella 2008). These densities suggest that the distribution and

Figure 13. Litter, logs, and coarse woody debris contribute to fire spread and intensity. Old logs also provide local evidence of historical forest composition and structure. The excessive quantity of litter is a result of the lack of fire in this frequent-fire forest.

abundance of snags and logs varied with site and likely coincided with the type, severity, and scale of historical disturbance.

Dry Mixed-Conifer

Mixed conifer forests can be divided into two subtypes: a warm-dry (dry mixed-conifer) type and a cool-moist (wet mixed-conifer) type. Dry mixed-conifer forests are similar to ponderosa pine forests in general stand structure, but Douglas fir, white fir, white pines, and, occasionally, blue spruce are also important components of these forests (Fig. 14). Wet mixed-conifer forests typically lack ponderosa pine, have a greater abundance of Douglas-fir and white fir, and, on some sites, include other fire-intolerant and shade-tolerant species such as blue spruce, subalpine/corkbark fir, and Engelmann spruce (Fig. 2). Dry mixed-conifer forests typically occupy the lower, warmer, and drier end of the elevation zone occupied by mixed-conifer forests. They intergrade with the cool/moist ponderosa pine types on warmer/drier sites at the lower end of the mixed-conifer zone and with wet mixed-conifer forests on cooler/moister sites at the upper end of the zone (Korb and others 2013; Romme and others 2009; Smith and others 2008).

Dry mixed-conifer forests intergrade with or are adjacent to pure ponderosa pine forests and experience similar site conditions and ecological disturbances (types and frequencies) (Grissino-Mayer and others 1995). Romme and others (2009) suggested that the stand structure of dry mixed-conifer was maintained in part by recurrent fires of relatively low to moderate

Figure 14. Groups of dry mixed-conifer are similar to groups in ponderosa pine forests but often have more diverse assemblages of species and higher tree densities.

severity, although small areas of higher-severity crown fire were likely. While only a few studies report the extent of mixed-severity fires (Romme and others 2009), Fulé and others (2009) found no areas of high-severity fire larger than 158 acres as inferred by the current extent and presence of even-aged structures or early seral species.

Dry mixed-conifer forests occur on relatively warm sites at lower elevations or on southerly aspects at higher elevations and are characterized by historical frequent surface-fires synchronized by climate (approximately 9-30 years) (Brown and others 2001; Brown and Wu 2005; Fulé and others 2003, 2009; Grissino-Mayer and others 2004; Heinlein and others 2005). In contrast, wet mixed-conifer is typified by mixed-severity fire regime (Fulé and others 2003). Many studies based on fire-scarred trees show that dry mixed-conifer forests had frequent but variable fire return intervals. Some studies report fire return intervals that were similar to ponderosa pine, as frequently as every 4-14 years (Brown and others 2001; Touchan and others 1996; reviewed in Evans and others 2011), whereas other dry mixed-conifer forests experienced fires as infrequently as every 18-32 years (Fulé and others 2003; Korb and others 2013; Touchan and others 1996; reviewed in Evans and others 2011). A recent study in Southwestern Colorado warm/dry mixed conifer forests found a mean fire return interval ranging from 9-30 years on three different sites at similar latitude and elevation. Korb and others (2013) also showed significant influence of local site factors (e.g., topography, forest structure, and species composition) on fire frequency and severity. Departures from historical compositions, structures, and spatial patterns are likely greater on the warmer/drier than the cooler/wetter portion of the mixed-conifer environmental gradient due to a more severe disruption of the characteristic fire regime (Fulé and others 2002).

When direct evidence of historical fire regime is lacking (i.e., fire scars not present), plant associations that classify seral and climax species composition relative to the shade and fire tolerance of tree species and biophysical site conditions may assist in making inferences regarding fire regimes (see Tables 2 and 8). Openings in dry mixed-conifer include grasses, forbs, shrubs, ferns, and cacti (Korb and Springer 2003), but the specific assemblage of understory plants varies greatly by plant association, being broadly grouped as dominated by bunchgrasses or by forbs/shrubs (Table 8). Bunchgrass-dominated plant associations in dry mixed-conifer forests generally occur in warmer/drier conditions than sites dominated by forbs and shrub understories (e.g. white fir/Arizona fescue [warm/dry] compared to white fir/forest fleabane [cool/moist]; Table 8). For example, the U.S. Forest Service, Southwestern Region utilizes plant association classifications for mapping the spatial extent of dry and wet mixed-conifer forests on National Forest Lands.

Dry Mixed-Conifer: Species Composition: Due to a predominance of frequent, low-severity fire, historical species composition in dry mixed-conifer forests was dominated by fire-resistant, shade-intolerant conifers such as ponderosa pine, Southwestern white pine, and Douglas-fir (Fig. 2) (Evans and others 2011; Fulé and others 2003). Dry mixed-conifer forests occur in environments that are wet enough to support trees such as white fir and aspen. However, these species are also more susceptible to death from fire than fire-resistant pines and Douglas-fir (Fig. 2) (Evans and others 2011; Fulé and others 2003). Consequently, species composition in dry mixed-conifer forests was historically regulated by the balance between climate and disturbance agents such as fire. Periods of frequent fire in mixed-conifer gave fire-resistant species a competitive advantage, allowing them to establish dominance. During "fire-free" or less frequent-fire periods, ponderosa pine persisted

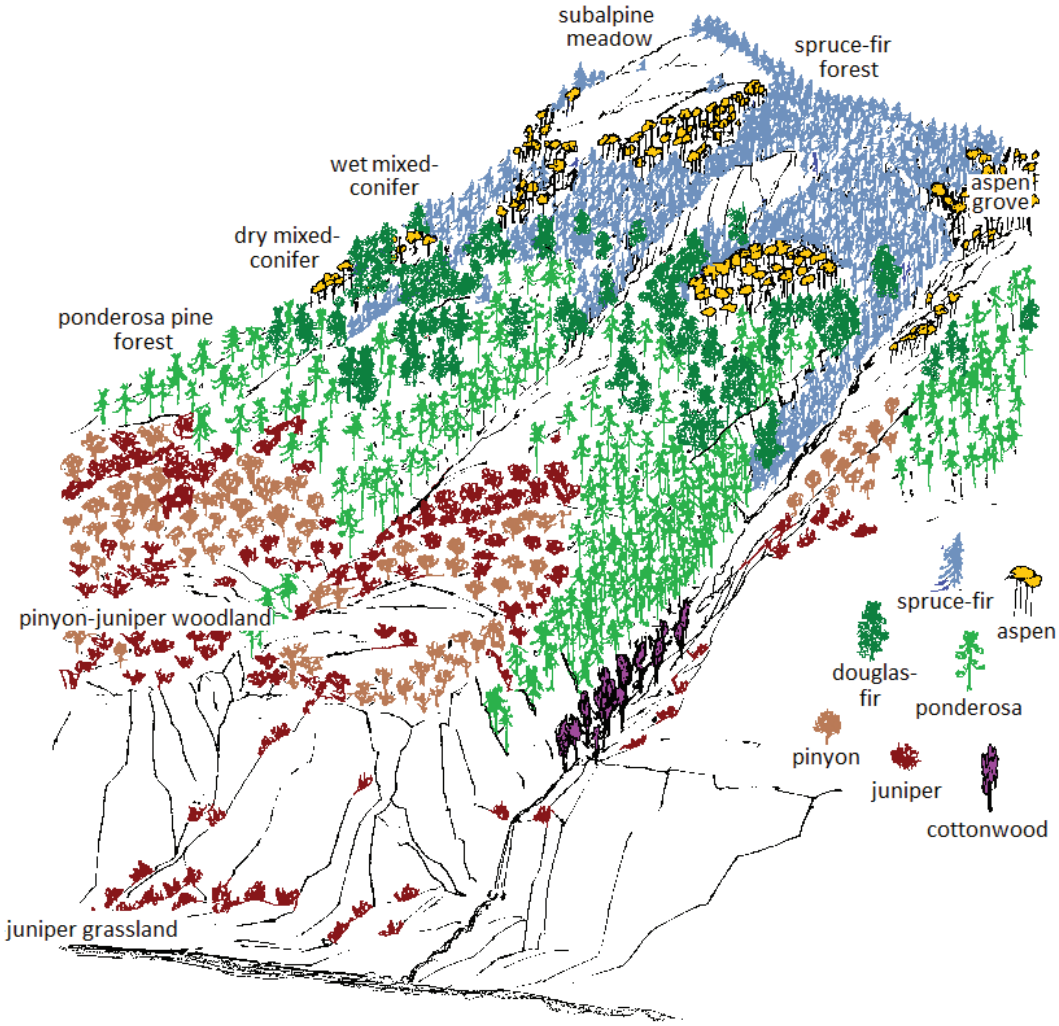

Figure 15. Illustration of changes in forest type by elevation and aspect (adapted from LANL 2011).

due to its dominant positions in the forest canopy (Fulé and others 2009). As a result, shade-tolerant, less fire-resistant species were historically minor components on drier sites, such as ridge tops and southwest-facing slopes, and likely more frequent on cooler and/or more mesic sites in frequent-fire forests, such as drainages and north-facing slopes (Fig. 15) (Romme and others 2009).

Compared to early 1900s Southwestern forest inventories, the current species composition of dry mixed-conifer forests has shifted toward more shade-tolerant, less fire-resistant species (Fulé and others 2009; Johnson 1994; Romme and others 2009). For example, one study in northern Arizona found that ponderosa pine represented an average 64 percent of basal area in the 1870 forest (range 54-69 percent) but only 36 percent in the same forest in 2003 (range 27-46 percent) (Fulé and others 2009). A recent study in Southwestern Colorado found that species composition prior to the last fire record on two different sites (1861 and 1878) was dominated by ponderosa pine, but white fir and Douglas-fir increased in dominance since the cessation of fire (Korb and others 2013). Other studies similarly concluded that extended fire exclusion in dry mixed-conifer forests resulted in substantial increases in stand-level tree density, especially by shade-tolerant white fir and Douglas-fir (Fulé and others 2004; Heinlein and others 2005). These increases resulted in forests with greater homogeneity in species composition across landscapes (Cocke and others 2005; White and Vankat 1993). Furthermore, early selective logging of ponderosa pine and intensive grazing exacerbated the compositional shift toward mesic species (Cocke and others 2005). The combination of fire exclusion, grazing, selective logging, and favorable climatic conditions for young tree establishment in the early 20th Century has created atypical stand compositions and structures in many of today's dry mixed-conifer forests

Table 8. Plant associations for dry mixed-conifer forests in the Southwestern United States sorted by plant association series, dry mixed-conifer subtype, temperature-moisture gradient, dominant season of precipitation, and parent material type (USDA Forest Service 1997).

Plant association (common name)	Dry mixed-conifer subtype	Temperature-moisture gradient[a]	Climate[b]	Dominant season of precipitation[c]	Parent material type[d]
Douglas-fir plant association series					
Douglas-fir/Arizona fescue	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/Arizona fescue/bristlecone pine	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/Arizona fescue/limber pine	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/Arizona fescue/quaking aspen	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/fringed brome	Bunchgrass	Cool-wet	Cold	Winter	Variable
Douglas-fir/Gambel oak/Arizona fescue	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/Gambel oak/screwleaf muhly	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/mountain muhly/limber pine	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/mountain muhly/two needle pinyon	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/screwleaf muhly	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir (scree)	Forb-shrub	Variable	Variable	Summer or winter	Igneous
Douglas-fir/creeping barberry	Forb-shrub	Warm-dry	Cold	Winter	Variable
Douglas-fir/Gambel oak	Forb-shrub	Warm-dry	Cold	Winter	Variable
Douglas-fir/Gambel oak/rockspirea	Forb-shrub	Warm-dry	Cold	Winter	Variable
Douglas-fir/kinnikinnik	Forb-shrub	Warm-dry	Cold	Winter	Rhy./tuff
Douglas-fir/mountain ninebark	Forb-shrub	Warm-dry	Cold	Winter	Variable
Douglas-fir/silverleaf oak/Chihuahua pine	Forb-shrub	Warm-dry	Mild	Summer	Variable
Douglas-fir/silverleaf oak/netleaf oak	Forb-shrub	Warm-dry	Mild	Summer	Rhy./tuff
Douglas-fir/silverleaf oak/ponderosa pine	Forb-shrub	Warm-dry	Mild	Summer	Variable
Douglas-fir/Arizona white oak	Forb-shrub	Warm-dry	Mild	Summer	Variable
Douglas-fir/bigtooth maple	Forb-shrub	Typic	Cold	Summer	Sed.
Douglas-fir/rockspirea	Forb-shrub	Warm-dry	Cold	Winter	Variable
Douglas-fir/wavyleak oak	Forb-shrub	Warm-dry	Cold	Summer	Sed.

Table 8. *Continued.*

Plant association (common name)	Dry mixed-conifer subtype	Temperature-moisture gradient[a]	Climate[b]	Dominant season of precipitation[c]	Parent material type[d]
White fir plant association series					
White fir/Arizona fescue	Bunchgrass	Cool-wet	Cold	Winter	Variable
White fir/Arizona fescue/Gambel oak	Bunchgrass	Cool-wet	Cold	Winter	Variable
White fir/Arizona fescue/muttongrass	Bunchgrass	Cool-wet	Cold	Winter	Variable
White fir/screwleaf muhly	Bunchgrass	Cool-wet	Cold	Summer	Variable
White fir/Arizona walnut	Forb-shrub	Cool-wet	Cold	Winter	Alluvium
White fir/creeping barberry	Forb-shrub	Typic	Cold	Winter	Variable
White fir/creeping barberry/common juniper	Forb-shrub	Typic	Cold	Winter	Variable
White fir/creeping barberry/New Mexico locust	Forb-shrub	Typic	Cold	Winter	Variable
White fir/Gambel oak	Forb-shrub	Typic	Cold	Winter	Variable
White fir/Gambel oak/Arizona fescue	Forb-shrub	Typic	Cold	Winter	Variable
White fir/Gambel oak/pine muhly	Forb-shrub	Typic	Cold	Winter	Cong./tuff/and.
White fir/Gambel oak/rockspirea	Forb-shrub	Typic	Cold	Winter	Variable
White fir/Gambel oak/screwleaf muhly	Forb-shrub	Typic	Cold	Winter	Variable
White fir/kinnikinnik	Forb-shrub	Typic	Cold	Winter	Rhyolite
White fir/mountain snowberry/limber pine	Forb-shrub	Cool-wet	Cold	Winter	Variable
White fir/mountain snowberry/ponderosa pine	Forb-shrub	Typic	Cold	Winter	Variable
White fir/Nevada pea	Forb-shrub	Typic	Cold	Winter	Variable
White fir/New Mexico locust	Forb-shrub	Typic	Cold	Winter	Variable
White fir/New Mexico locust/dryspike sedge	Forb-shrub	Typic	Cold	Winter	Variable
Blue spruce plant association series					
Blue spruce/Arizona fescue	Bunchgrass	Typic	Cold	Winter	Rhy./basalt
Blue spruce/dryspike sedge	Bunchgrass	Typic	Cold	Winter	Rhy./basalt

[a]Gradient based on all mixed conifer forests (dry and wet mixed-conifer types). Typic refers to modal, mid-gradient temperature-moisture types.
[b]Climate refers to mean annual soil temperatures, with cold climates having frigid soils (mean annual soil temperatures <8 ˚C) and mild climates having mesic soils (mean annual soil temperatures >8 ˚C).
[c]Dominant season of precipitation refers to the 6-month period (winter = October-March, summer = April-September) that typically has higher average precipitation levels. Most ponderosa pine and dry mixed-conifer sites in the Southwestern United States receive bimodal precipitation, but the season listed in the table experiences higher average precipitation levels.
[d]Variable = multiple parent materials; sed. = sedimentary; rhy.= rhyolites; cong. = conglomerate; and. = andesite

(Moore and others 2004). In many locations, large, dominant ponderosa pine and Douglas-fir trees have been reduced to few or none, leaving today's stands dominated by young ponderosa pine, Douglas-fir, and white fir (Fulé and others 2003).

Dry mixed-conifer plant associations are highly variable and reflective of local biophysical site conditions that influence the type of disturbances and vegetation responses to disturbances (Table 8) (USDA Forest Service 1997). These plant associations are classified by forest series representing the most shade-tolerant conifer species that can establish and grow on a given site, absent disturbance. However, ponderosa pine typically dominates the species mix in dry mixed-conifer forest series under the characteristic fire regime. Dry mixed-conifer forest series include: (1) Douglas-fir, (2) white fir, and (3) those blue spruce plant associations that do not classify as wet mixed-conifer. These series can be subdivided by understory plant composition into the general subtypes of bunchgrass and forb-shrub. The most mesic dry mixed-conifer sites are the forb-shrub plant associations, and the most xeric are the bunchgrass plant associations. These subtypes differ in their relative fire frequencies; bunchgrass understories support more frequent surface fire, while forb-shrub understories facilitate less frequent surface fire and greater fuel accumulation (Anderson 1982; LANDFIRE 2007; Scott and Burgan 2005; USDA Forest Service 1997).

Dry Mixed-Conifer: Forest Structure: Compared to ponderosa pine, there is considerably less literature on fine-scale forest structure and spatial pattern reference conditions in dry mixed-conifer forests. However, there are some historical references to similarities between structure and spatial pattern of these two forest types. Due to its frequent fire regime, the historical fine-scale structure and spatial pattern of dry mixed-conifer forests were similar to ponderosa pine in having a more open structure (Muldavin and Tonne 2003; Swetnam and Baisan 1996) and a similar aggregated arrangement of trees in some stands (Binkley and others 2008; Sánchez Meador and others unpublished data, see Table 3 footnote). Lang and Stewart (1910; p. 19) noted that "evidence indicates light ground fires over practically the whole forest, some of the finest stands of yellow pine show only slight charring of the bark and very little damage to poles and undergrowth." Dutton (1882) observed that within both the ponderosa pine and mixed ponderosa pine/Douglas-fir forest types "the trees are large and noble in aspect and stand widely apart, except in the highest parts of the [Kaibab] plateau where the spruces predominate. Instead of dense thickets where we are shut in by impenetrable foliage, we can look far beyond and see the tree trunks vanishing away like an infinite colonnade." These observations are consistent with statements that "pure ponderosa pine forests and warm-dry mixed conifer forests were affected primarily by frequent, low-severity fires that maintained an open stand structure with a broad range of tree sizes and ages" (Romme and others 2009).

Empirical evidence also indicates that historical dry mixed-conifer forests had lower tree densities and a more open structure comprised of a higher proportion of old and large trees, were more spatially heterogeneous (having groups and patches of trees), and were more uneven-aged compared to current conditions (Fig.16) (Binkley and others 2008; Fulé and others 2002a, 2003, 2009; Heinlein and others 2005; Moore and others 2004). However, as mixed conifer forests transition toward cooler and wetter site conditions, less frequent and more severe fires result in mixtures of even- and uneven-aged forest structures. At the landscape scale, wet mixed-conifer forests were historically more spatially heterogeneous than ponderosa pine forests because of a mixed-severity fire regime affected by topography, soils, land use, and vegetation (Binkley and others 2008; Fulé and others 2002a, 2003, 2009; Muldavin and Tonne 2003; Smith 2006a; Romme and others 2009; Touchan and others 1996). Variable forest structures and spatial patterns across landscapes

Figure 16. Aerial photo of a dry mixed-conifer forest on a north-facing slope in the Cibola National Forest. In this stand, about 60-70 percent of the area is under mid- to old-age tree cover and 30-40 percent is in grass-forb-shrub interspaces.

resulted, in part, from variation among sites on the temperature/moisture continuum and their species compositions, successional status, and disturbance regimes. Warm, dry mixed-conifer sites likely experienced more frequent and less severe surface fire, resulting in more open forests with a mixture of small tree groups and areas with random tree spatial patterns. In contrast, cool, moist sites experienced mixed or high-severity fires at longer fire return intervals, resulting in relatively closed forests with tree cohorts distributed in larger patches (Fig. 14) (Fulé and others 2003; Romme and others 2009).

Studies of reference conditions for dry mixed-conifer forests reported mean tree densities and basal areas similar to those in ponderosa pine stands but with slight increases at the fine scale (Table 9; Fig. 17). For example, pre-Euro-American settlement dry mixed-conifer forests on limestone soils ranged between 36 and 100 trees per acre and 34 and 124 square ft of basal area per acre on sites in Arizona and New Mexico, respectively (Table 9; Fig. 10). For dry mixed-conifer forests on the Uncompahgre Plateau in Colorado, Binkley and others (2008) reported reference conditions for canopy cover ranging from 12.0-21.5 percent in areas that exhibit fine-scale aggregation; openness was therefore 78.5-88.0 percent in these areas. Fornwalt and others (2002) modeled reference canopy cover conditions of 13-22 percent (78-87 percent openness) for forests with fine-scale tree aggregation on the Colorado Front Range (Table 7). Based on reported studies, historical dry mixed-conifer forests were structurally similar to ponderosa pine with respect to tree groups with small meadows between them (Binkley and others 2008).

Abundance of snags, logs, and woody debris in dry mixed-conifer was likely similar to or slightly greater than that of ponderosa pine. Moore and others (2004) reported 4.9-34.9 snags per acre for dry mixed-conifer reference conditions as determined from extensive, historical stem-maps and relocation of historical evidences (e.g., logs, stumps, and snags). While the historical amount of these structural elements in dry mixed-conifer has received little attention, contemporary studies suggest that more productive dry mixed-conifer sites had higher fuel loads than ponderosa pine sites (Brown and others 2003; Graham and others 1994).

Despite the above similarities, dry mixed-conifer forests occur on a diverse range of sites and have more diversity in species composition, structure (Fig. 17), spatial pattern, processes (i.e., fire regimes and other disturbances), and functions than ponderosa pine forests. While studies demonstrate considerable similarity between dry mixed-conifer and ponderosa pine disturbance processes and forest structures, we point again to the limited numbers and geographical locations of studies of historic structural conditions in dry mixed-conifer and call for additional research to increase our understanding of historical ranges of conditions for these forests (see Monitoring, Adaptive Management, and Research Needs).

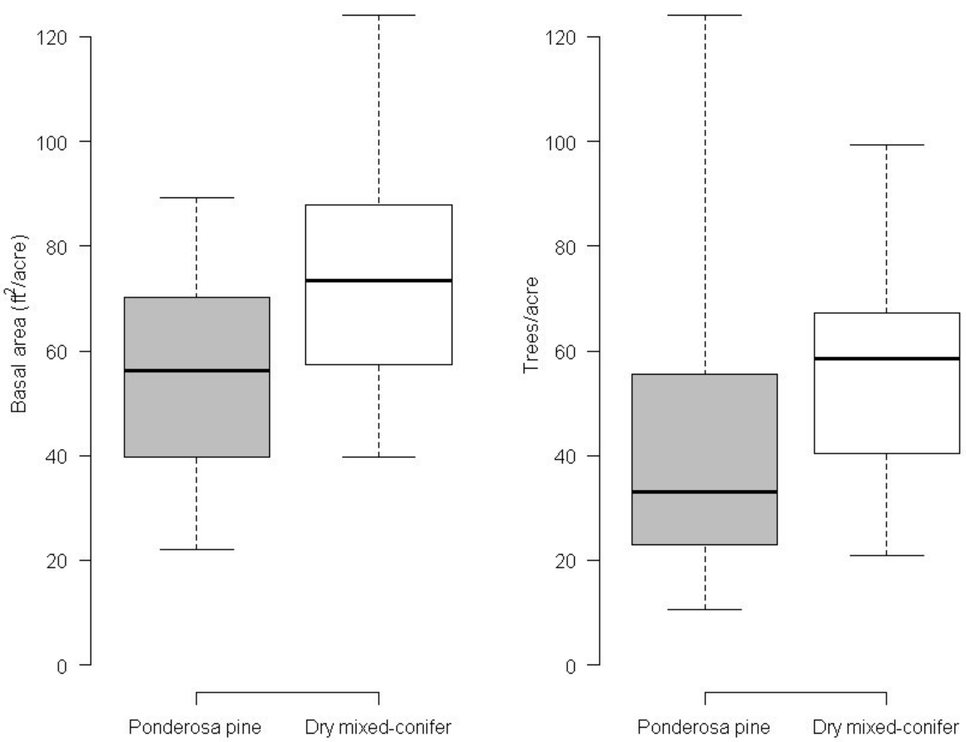

Figure 17. Distribution of reference conditions reported in Tables 6 and 9 for basal area and trees per acre in ponderosa pine and dry mixed-conifer forests. Lines bisecting boxes represent median values; lower and upper borders of boxes represent first and third quartile values; and whiskers (i.e., endpoints of dashed lines) represent maximum and minimum reported values.

Table 9. Historical forest structure of dry mixed-conifer forests of the Southwest, arranged by parent material and average tree density.

Location	Parent material	Elevation (ft)	Size/age reported	Reference date	Trees per acre			Basal area (ft²/acre)			Citation
					Range	Mean	Std Err	Range	Mean	Std Err	
San Francisco Peaks-East, Arizona	Basalt	8318	Age	1892		20.9	3.4		39.6	3.9	Heinlein and others 2005
San Francisco Peaks-West, Arizona	Basalt	8318	Age	1876		21.0	1.7		54.0	6.1	Heinlein and others 2005
Sitgreaves National Forest, Arizona (max)[a]	Basalt	6300	Size	1910		31.0			66.9		Woolsey 1911
San Francisco Peaks, Arizona	Basalt	9200	Age	1876		65.1	6.8		77.9	12.8	Cocke and others 2005
Blue and White Mountains, Arizona[b]	Basalt	8950	Size	1912		68.7			84.4		Greenamyre 1913
Middle Mountain, Colorado	Granitic	8520	Size	1870	51-59	57.3	4.0	43-60	47.9	4.6	Fulé and others 2009
Jemez, New Mexico (max)[a]	Limestone	7013	Size	1910		35.6			91.2		Woolsey 1911
Kaibab Plateau, New Mexico[c]	Limestone	[illegible]	Size	1909		45.3			60.7		Lang and Stewart 1910
Alamo, New Mexico (max)[a]	Limestone	[illegible]	Size	1910		46.5			97.9		Woolsey 1911
Gila, New Mexico[a]	Limestone	9055	Age/Size	1890		65.6					Moore and others 2004
Jemez, New Mexico[a]	Limestone	[illegible]	Age/Size	1890	66-112	88.8	23.2				Moore and others 2004
Little Park, Arizona	Limestone	8640	Age	1880		98.3	5.8		76.7	9.1	Fulé and others 2003
Swamp Ridge, Arizona	Limestone	8143	Age	1879	36-151	99.4	5.2	65-235	124.1	7.8	Fulé and others 2002a
Black Mesa, Arizona[c]	Sedimentary	Mixed	No	1890		58.4	27.3				Williams and Baker 2011; 2012
Uncompahgre Plateau, Colorado	Shale	8000	Size	1875	30-110	60		25-130	70		Binkley and others 2008

[a]Minimum tree DBH recorded = 3.5in.
[b]Minimum tree DBH recorded = 4 in.
[c]Minimum tree DBH recorded = 6 in

The Restoration Framework

Here, we describe our framework for restoring resiliency to frequent-fire forests in the Southwest. We first provide an overview of our framework, including its ecological foundation, its key elements, and the sources of its science base. We then discuss the spatial and temporal scales at which forest structures are described, and follow this with a description of the desired key compositional and structural elements of a restored forest at those scales for ponderosa pine and then dry mixed-conifer forests. Finally, we provide recommendations for implementing the framework in these forests and finish with brief before and after descriptions of the composition and structure in a ponderosa pine area in New Mexico where we implemented our framework.

The framework is organized around key compositional and structural elements at three spatial scales and is based on a synthesis of reference conditions, literature on the ecology of frequent-fire forests (Table 4) (see Science Review: Forest Ecology), our understanding of the ecology of these forests, decades of collective experience of forest managers and researchers (e.g., Schubert 1974), and lessons learned during applications of our framework in Southwestern frequent-fire forests. Our framework is informed by the ranges of mean forest characteristics from reference conditions research plots, which were typically <10 acres and therefore best describe variability at the fine scale (Tables 3, 6, 7, and 9). Means across plots are more representative of mid-scale conditions than means reported for individual sample plots. Therefore, we point out that any point estimates with a range of mean values may not be appropriate for a given site and we recommend using local, site-specific biophysical conditions and historical evidences to inform specific treatments.

Forest ecology, historical (reference) conditions, and the natural range of variability are frequently used to define restoration goals, to estimate the restoration potential of sites, and to evaluate the success of restoration efforts. Natural range of variability is useful for understanding the natural variability in composition, structure, processes, and functions among sites and for understanding the dynamic nature of ecosystems. It is also a useful reference for establishing limits of acceptable change for ecosystem components and processes (Morgan and others 1994). Our framework is not intended to re-create specific reference conditions. Rather, the framework identifies key elements that characterized frequent-fire forests before industrial logging and the disruption of historical disturbance regimes. These key compositional and structural elements are:

(1) species composition (tree and understory vegetation);

(2) groups of trees;

(3) scattered individual trees;

(4) open grass-forb-shrub interspaces;

(5) snags, logs, and woody debris; and

(6) variation in arrangements of these elements in space and time.

The key elements provide inferences about species compositions, structural conditions, and the cumulative effects of disturbances on processes and functions that provide frequent-fire forests with resistance and resilience to disturbance.

Citations supporting our restoration framework appear mostly in the Science Review: Forest Ecology section but in other sections as needed. We recognize the limited number and geographic extent of scientific studies of reference conditions for ponderosa pine and especially for dry mixed-conifer, not only in the Southwest but throughout the western United States. Nonetheless, our framework is timely because of the growth in knowledge over the past decades regarding current and historical ecology of these forests. It is also timely because of increased frequencies, intensities, and extents of uncharacteristic disturbances, which may worsen under climate change (Littell and others 2009; Millar and others 2007; Miller and others 2009; Westerling and others 2006). We believe that moving current forest conditions toward their characteristic compositions, structures, and spatial patterns will increase their resistance and resilience to future disturbances and will result in outcomes as varied as fire fuels reduction, restoration of wildlife habitats, biodiversity, diverse food webs, and increased ability of these forests to provide ecosystem services.

Spatial and Temporal Scales

Ecosystems are structured hierarchically and their composition, structure, process, and function are temporally and spatially dynamic. Therefore, we characterize the key compositional and structural elements at three spatial scales: the fine-scale (<10 acres), mid-scale (10-1000 acres), and landscape-scale (1000-10,000+ acres) (Fig. 1). These scales generally correspond with structural features in frequent-fire forests. For example, the fine scale is an area in which the species composition, age, structure, and spatial distribution of trees (single and grouped), and open grass-forb-shrub interspaces are expressed. Aggregates of fine-scale units comprise mid-scale units, which are referred to as patches (i.e., stands) and are relatively homogeneous in vegetation composition and structure that differ from their surroundings. The landscape scale is composed of aggregates of mid-scale units and usually has variable elevations, slopes, aspects, soil types, plant associations, disturbance processes, and land uses. Understanding and incorporating temporal scales (seasonal, annual, decadal, and centuries) in a restoration framework is required to sustain vegetation dynamics of a forest that result from growth, succession, senescence, and the natural and anthropogenic disturbances that periodically reset the dynamics.

Key Elements by Forest Type: Ponderosa Pine

Southwest ponderosa pine forests occur at elevations ranging from approximately 5000-9000 ft and typically intergrade with woodland types on warm/dry sites (typically at lower elevations) and mixed-conifer types on cool/moist sites (typically at higher elevations). The characteristic fire regime for ponderosa pine is frequent, low-severity fires (Fire Regime 1; Table 2). Surface fuels (fine fuels, branches, and coarse woody debris) and small trees facilitate this fire regime. Fires burn primarily on the forest floor and rarely spread to tree crowns and canopies. Individual trees or tree groups may occasionally torch during fires. Based on plant associations, a system for classifying plant communities on their potential climax species compositions (Table 5) (USDA Forest Service 1997), we differentiated four Southwestern ponderosa pine forests subtypes: (1) ponderosa pine-bunchgrass, (2) ponderosa pine-Gambel oak, (3) ponderosa pine-evergreen oak, and (4) ponderosa pine-shrub (Appendix 2).

Ponderosa Pine: Fine-Scale Elements (<10 acres):

Species composition: Overstories are dominated by ponderosa pine but may occasionally contain other conifer or hardwood species. Herbaceous understories are typically grasses and forbs at the mid-point within the temperature/moisture gradient over which ponderosa pine occurs. At the warm/dry end of the gradient, ponderosa pine forest intergrades with pinyon-juniper or evergreen oak woodlands (e.g., juniper, pinyon, Emory oak, Arizona white oak, silverleaf oak, and grey oak) with a shrub component (e.g., manzanita, shrub live oak, sumac, or mountain mahogany). In the cool/moist portion of the gradient, Gambel oak is often a component of ponderosa pine forests, and grass and forb understories may include shrubs (e.g., ceanothus, and currants) (Table 5). At the cool/moist end of the gradient, ponderosa pine intergrades with dry mixed-conifer forests where there may be a minor presence of quaking aspen, Douglas-fir, Southwestern white pine, white fir, and blue spruce. Variation in overstory species composition influences forest structure, disturbance types and intensities, tree mortality rates, and the composition and structure of the grass-forb-shrub community.

- Trees typically occur in irregularly shaped, small groups with interlocking or nearly interlocking crowns when in the mid- to old-aged structures (Fig. 11), range in size from 2-72 trees, and occupy between 0.003 and 0.72 acres each (Table 3; Fig. 4). Groups can be even- or uneven-aged. Size, shape, number of trees per group, and number of groups per area are variable (see Science Review: Forest Ecology). If trees are aggregated (i.e., grouped), more productive sites will have more trees per group, and if not aggregated, will support more individual trees per acre. Where groups are even-aged, a high level of interspersion of groups of differing ages constitutes the desired uneven-aged structure at the fine- and mid-scale. Trees within groups are variably spaced with some tight clumps.
- Where reference conditions show the presence of scattered individual trees, their ages are variable (young to old) and they can comprise 15-70 percent of total stand basal area, with the remaining stand basal area being trees in groups (Table 3). Variability in number of individual trees is associated with various factors, such as soils, plant associations, climate, and disturbances.
- Grass-forb-shrub interspaces surround tree groups and individual trees (Fig. 8) and are variably shaped and sized.
- Snags, top-killed, lightning- and fire-scarred trees, and coarse woody debris (logs and other dead woody material greater than 3 inches in diameter) are generally large in diameter and height, scattered throughout the mid-scale, and concentrated in past disturbance sites in abundances of 1-10 snags and 3-10 tons per acre (Figs. 12 and 13). Overall, snags, logs, and coarse woody debris are spatially and temporally variable.

Figure 18. Uneven-aged forest comprised of an interspersion of tree groups of different ages.

Ponderosa Pine: Mid-Scale Elements (10-1000 acres):

The mid-scale is an aggregate of fine-scale units (i.e., tree groups, scattered individual trees, and grass-forb-shrub interspaces) and is collectively referred to as a patch or stand. Mid-scale patches are relatively homogeneous in vegetation composition and structure and differ from surrounding patches.

- Tree species composition is relatively homogenous within patches and is a function of disturbance, time since disturbance, tree density and size/age structure, topography, soils, local climate, site history, ecological legacy, and stochasticity (e.g., mass seeding and weather events).
- Average total tree densities and basal areas generally range from 11-124 trees per acre and 22-90 square ft of basal area per acre (Table 6).
- More productive sites may have more trees per area. Aggregates of many randomly distributed trees (areas >10 acres) function as patches.
- For sustainability and biodiversity purposes, it is desirable that patches comprise uneven-aged forests with an approximate balance of age classes ranging from young to old (Fig. 18). Infrequently, patches of even-aged forest structure may be present.
- All age classes of appropriate hardwood species (e.g., Gambel and evergreen oaks and other hardwoods) are present depending on a site's plant association (Table 5).
- "Openness" (estimated as the inverse of canopy cover) ranges from 52-90 percent. In areas exhibiting fine-scale aggregation of trees, mid-scale openness is typically high (78-90 percent; Table 7), and on more productive sites, especially where tree arrangement is random, openness may be less (see discussion of openness in Science Review: Forest Ecology).

Ponderosa Pine: Landscape-Scale Elements (1000-10,000+ acres):

- The landscape scale is an aggregate of mid-scale units and includes areas with variable topography (i.e., elevation, slope, and aspect), soils, plant associations, disturbance types, and land use legacies. The landscape is a functioning ecosystem that contains all components, processes, and functions that result from characteristic disturbances, including snags, downed logs, and old trees.
- Old-growth structural features occur throughout the landscape as tree groups or single trees within uneven-aged patches (stands) or occassionally as small even-aged patches. Old-growth structural features include old trees, snags, downed wood (coarse woody debris), and horizontal and vertical structural diversity in a grass-forb-shrub matrix (Table 10; Fig. 9). The location of old-growth structural features may shift on the landscape over time as a result of succession and disturbance.

Table 10. Essential structural features of old growth in frequent-fire forests. Note that whether or not a feature is essential may depend on scale—fine-, mid-, and landscape-scale. For example, age variability is possible at all scales, but snags and large dead and downed fuels may not exist in some groups and patches (adapted from Kaufmann and others 2007).

Structural feature	Essential structural feature?	Comment
Large trees	No	Tree size depends on species and site characteristics (moisture, soils, and competition). Young trees may be large, and old trees may be small.
Old trees	Yes	Trees develop unique structural characteristics when old (e.g., dead tops, flattened crowns, branching characteristics, bark color and texture).
Age variability	No	An important feature in some old-growth forest types. Some forests regenerate episodically (even-aged) with most trees establishing in a few years to a decade, probably in conjunction with wet years and large seed crops and in concurrence with relatively long intervals between fires. Others may regenerate over decades (uneven-aged).
Snags and large dead and downed fuels	Yes	Snags and large logs are essential for old growth, but forests with more frequent fires may have fewer logs. Densities and sizes of snags and logs vary depending on forest type, precipitation, and other factors. Snags, logs, and woody debris typically distributed unevenly in landscapes.
Between-patch structural variability	Yes	High variability is a critical feature. Within-patch variability may be low, but variation among patches may be high. Proportions of patches with different developmental stages vary depending on forest type, climate, etc.

- Plant associations vary across environmental gradients (e.g., changes in slope, aspect, climate, and soil type) and reflect their historical species composition, structure, and spatial aggregations.
- Denser tree conditions may exist as patches in locations such as north-facing slopes and canyon bottoms.
- Natural and anthropogenic disturbances such as fire or tree thinning treatments are sufficient to maintain desired overall species composition, tree density, age structures, snags, coarse woody debris, and nutrient cycling.

Key Elements by Forest Type: Dry Mixed-Conifer

Southwest dry mixed-conifer forests generally occur at elevations ranging from 5500-9500 ft. At lower elevations within this range, dry mixed-conifer forests commonly occur on north-facing slopes or canyon bottoms and ponderosa pine forests on south-facing slopes and ridgetops. At the upper elevation range, dry mixed-conifer forests typically occupy south and west slopes, with wetter forest types (e.g., wet mixed-conifer) on north aspects. Dry mixed-conifer forests are dominated by shade-intolerant trees such as ponderosa pine, Douglas-fir, Southwestern white pine, limber pine, quaking aspen, and other hardwoods, with a lesser presence of shade-tolerant species such as white fir and blue spruce depending on biophysical site conditions and the frequency of low-severity fire. Aspen may occur individually or in groups of variable size. While less is known about historical conditions in dry mixed-conifer than in ponderosa pine, available information shows a similarity in the structure and spatial pattern of these two forest types.

Characteristic fire regimes for Southwestern dry mixed-conifer are frequent low-severity fires (Fire Regime 1) with infrequent mixed-severity fires (Fire Regime 3; Table 2) operating at all spatial scales. Surface fuels and small trees facilitate this fire regime. While fires burn primarily on the forest floor, occasionally individual trees or tree groups may torch. Crown fires rarely spread from tree group to tree group.

Dry Mixed-Conifer: Fine-Scale Elements (<10 acres)

- Species composition: Overstories are dominated by fire-resistant, shade-intolerant trees such as ponderosa pine, Douglas-fir, Southwestern white pine, and limber pine, with occasional inclusion of aspen and other hardwoods. Shade-tolerant conifers, such as white fir and blue spruce, may be present but are subdominant in abundance. At the warm/dry end of the temperature/moisture gradient occupied by dry mixed-conifer types, this forest type intergrades with ponderosa pine-bunchgrass and ponderosa pine-Gambel oak subtypes. At the cool/moist end of the gradient, dry mixed-conifer intergrades with the wet mixed-conifer type typified by a mixed-severity fire regime. Differences in overstory species composition influences structure (tree density, tree group size, number of individuals, regeneration), disturbance events (species-specific insect and diseases, fuel type and quantity), distribution of snags and coarse woody debris, and species composition of the grass-forb-shrub community.

- Where dry mixed-conifer forests occur at the warmer/drier end of the environmental gradient (Fig. 2), trees typically occur in irregularly shaped groups, trees within groups are variably spaced, and group sizes generally range from a few trees up to an acre (Fig. 14), similar to ponderosa pine forest types. Reference conditions show tree group sizes ranging from 0.01-0.33 acres (Table 3) (see Science Review: Forest Ecology). Trees within groups are of similar or variable ages and groups are composed of one or more species. Crowns of trees within the mid-aged to old groups are interlocking or nearly interlocking (Fig. 11). Size, shape, number of trees per group, and numbers of groups per area are variable (see Science Review: Forest Ecology). If aggregated, more productive sites will have more trees per group, or if not aggregated will support more trees per acre.
- No data are available on the proportion of stand basal area in individual trees verses tree groups. More research is needed (see Monitoring, Adaptive Management, and Research Needs).
- Grass-forb-shrub interspaces surround tree groups and individual trees (Figs. 14 and 16) and are variably shaped and sized.
- Snags, top-killed, lightning- and fire-scarred trees, logs, and coarse woody debris (>3 inches diameter) are generally large in height and diameter, scattered throughout, and concentrated at past disturbance events in abundances of 5-35 snags and 8-16 tons per acre (see Science Review: Forest Ecology). Overall, snags, logs, and coarse woody debris are spatially and temporally variable.

Dry Mixed-Conifer: Mid-Scale Elements (10-1000 acres)

- The mid-scale is an aggregate of fine-scale units (i.e., tree groups, scattered individual trees, and grass-forb-shrub interspaces) and is collectively referred to as a patch or stand. At the mid-scale, patches can be relatively homogeneous in vegetation composition and structure and differ from surrounding patches. Vegetation is typically characterized by variation in the sizes and numbers of tree groups and the density and extent of patches of trees, each typically varying by elevation, soil type, aspect, and site productivity. Occasionally, patches may be composed of randomly arranged trees.
- In general, tree densities range from 20-100 trees per acre and 40-125 square ft basal area per acre (Table 9) (see Science Review: Forest Ecology). Stand density is likely to increase as site conditions transition toward the cooler/moister end of the environmental gradient for dry mixed-conifer forests and on more productive soil types.
- For sustainability and biodiversity purposes, it is desirable that patches have an uneven-aged forest structure with an approximate balance of age classes ranging from young to old. Infrequently, patches of even-aged forest structure may be present.
- Species composition may be variable within patches and is a function of disturbance, tree density, tree size and age structure, topography, soil, local climate, site history, ecological legacy, and stochasticity (e.g., weather events, mass seeding).
- It is desirable that all age classes of appropriate hardwood species (e.g., aspen, Gambel oak, and maple) are present depending on a site's plant association (Table 8).
- "Openness" is similar to ponderosa pine at the warmer/drier end of the environmental gradient occupied by dry mixed-conifer forests (Table 7) but is likely to decrease from the warmer/drier site conditions to the cooler/wetter end of the environmental gradient due to moister conditions, higher productivity, and less frequent low-severity fire.

Dry Mixed-Conifer: Landscape-Scale Elements (1000-10,000+ acres)

- The landscape scale is an aggregate of mid-scale units and includes areas with variable topography, soils, plant associations, disturbance types, and land use legacies. The landscape is a functioning ecosystem that contains all its components, processes, and functions that result from characteristic disturbances, including snags, downed logs, and old trees.
- Old-growth structural features occur throughout the landscape as tree groups or single trees within uneven-aged patches (stands) or occassionally as small even-aged patches. Old-growth structural features

include old trees, dead trees (snags), downed wood (coarse woody debris), and horizontal and vertical structural diversity in a grass-forb-shrub matrix (Table 10). The location of old-growth may shift on the landscape over time as a result of succession and disturbance (tree growth and mortality).

- Plant associations vary across environmental gradients (e.g., changes in slope, aspect, climate, and soil type) and reflect their historical species composition, structure, and spatial aggregations.
- Denser tree conditions may exist as patches in some locations such as north-facing slopes and canyon bottoms.
- Natural and anthropogenic disturbances such as fire or tree thinning treatments are sufficient to maintain desired overall species composition, tree density, age structures, snags, coarse woody debris, and nutrient cycling.

Implementation Recommendations

Here, we offer recommendations for implementing our framework. These were developed from our understanding of the body of forest ecology and management literature (see Science Review: Forest Ecology), our research and management experience, and lessons learned during implementations of our restoration framework. At the end of this section we present an overview of a case study on the implementation of our framework that illustrates its success in moving current forest conditions toward uneven-aged forest mosaics comprised mostly of fire-adapted species; tree groups; scattered individual trees; grass-forb-shrub interspaces; snags, logs, woody debris; and the spatial arrangement of these elements.

Classification of Site Variability

Ecological classification of a site indicates its biological capabilities regarding species composition, structure, processes, and functions. Ecological classification is useful for implementing our restoration framework because classification depends on variability of local climate, soil, vegetation, geology and geomorphology, and a site's characteristic disturbances and vegetation responses (USDA Forest Service 1997). The variability within and among sites across landscapes is the basis for describing the range of variation in forest conditions in our restoration framework. Recognition of within- and among-site variability is paramount for developing localized restoration objectives. Example classification systems include the U.S. Forest Service Terrestrial Ecosystem Unit Inventory (Winthers and others 2005), which classifies land units by soil, climate, slope, geology, geomorphology, and plant associations, and NatureServe's Ecological Systems (Comer and others 2003). The biotic and abiotic variables used in these classification systems describe a site's biophysical characteristics.

Recommendations by Key Elements

Species Composition

- Manage for percent species composition as indicated by local historical evidence (live trees and snags and logs from trees that originated prior to 1880), biophysical site conditions, and other management objectives (e.g., favoring scarce species; preserving genetic diversity; enhancing wildlife habitat; resilience to climate change; or achieving other resource objectives, social values, and regulatory requirements).

Tree Groups and Individual Trees

- Use a site's historical spatial patterns to inform restoration targets and treatments. Where information on reference conditions is not available, fine-scale spatial patterns may be informed by reference data in Table 3, 6, 7 and 9 and combined with local historical evidence (see Friederici 2004) such as grouped and individual old trees, large logs, and stumps, and a site's biophysical conditions.
- Evaluate current conditions in relation to desired conditions to develop management prescriptions. Avoid arbitrary constraints such as diameter limits for tree cutting (see Abella and others 2006; Triepke and others 2011).
- Where spatial heterogeneity is desired, consider combinations of burns, intermediate and free thinning, and individual tree or small group selection cutting methods to create a heterogeneous structure of groups, single trees, and grass-forb-shrub interspaces. Once heterogeneity is established, consider maintaining the desired structure and spatial pattern with fire and/or single tree and small group selection.
- Where trees are spatially aggregated, maintain interlocking or nearly interlocking crowns in mature and old groups and provide for variable tree spacing within groups; avoid thinning old tree groups.
- Manage young tree groups to create future variable tree spacing and interlocking crowns. Thin young tree-groups to facilitate development of desired within-group characteristics (e.g., variable tree spacing and interlocking or nearly interlocking crowns) in mid- to old-aged tree groups.
- Tree groups generally are small (2-72 trees per group, see Science Review: Forest Ecology) (Fig. 4). Use historical evidence and biophysical capabilities to determine a site's mean and range (minimum, maximum) of trees per group and numbers and spacing of tree groups per area.
- Mid-scale patches (stands) of less-aggregated or randomly arranged trees may be appropriate where

historical evidences do not exhibit spatial aggregation or for achieving other resource objectives.

- Where appropriate, retain or regenerate scattered individual trees between groups.
- Use historical evidence, biophysical site conditions, plant associations, and current conditions (e.g., competition from brush on certain plant association types, degree of disease or insect infestation) to inform regeneration treatments.
- Where management objectives are to maintain conifer dominance and where post-treatment dominance by shrub understories is undesired (e.g., in some ponderosa pine-evergreen oak, ponderosa pine-shrub, and dry mixed conifer-forb/shrub forest types), consider smaller interspaces to avoid excessive shrub response and increased ladder fuel accumulation.
- Consider temporary deviations from uneven-aged management to even-aged cutting methods to initiate recovery on sites damaged by epidemic (severe) insect or disease infestation or other disturbances.
- Manage fire (wildfire or prescribed) frequency and severity towards achieve desired forest structures, spatial arrangements, regeneration patterns, and fuel consumption objectives.
- Design and place regeneration treatments to favor recruitment of shade-intolerant, fire-resistant species.
- Vary treatment prescriptions (cutting and/or fire) to create a mosaic of groups of trees, scattered single trees, and grass-forb-shrub interspaces.

Grass-Forb-Shrub Interspaces

- The grass-forb-shrub community is the matrix in which tree groups and scattered individual trees are arranged (Fig. 8).
- The size and arrangement of grass-forb-shrub interspaces reflect local site conditions and historical evidence. Where trees are grouped, interspaces may be as wide as 1-2 mature tree heights from nearest drip lines of adjacent tree groups. Binkley and others (2008) reported approximately 150 ft between historic groups of trees in dry mixed-conifer in Southwest Colorado; Pearson (1923) reported 100-150 ft diameter openings (interspaces) between historic tree groups in ponderosa pine forests in northern Arizona.
- Sizes of grass-forb-shrub interspaces are a less useful metric for tree spacing in areas where trees are more randomly spaced (i.e., not aggregated). Use a site's historical vegetation spatial patterns as a guide for restoration.
- Grass-forb-shrub interspaces are generally larger on dry sites. Interspaces provide rooting space to support grouped trees.
- Meadows, grasslands, and other non-forested areas may be present as inclusions in forested landscapes; these areas are not considered interspaces.

Snags, Logs, and Woody Debris

- Manage for the continuous presence of snags, logs, and woody debris, especially large snags in various stages of decay throughout the landscape (Figs. 12 and 13). Frequent fires both recruit and consume these elements.

Arrangement of Key Elements in Space and Time

- Recognize the importance of spatial and temporal heterogeneity in forest composition and structure to ecological processes and functions.
- Where objectives include sustainability of wildlife habitat, biodiversity, and wood products, manage for a balance of age classes from cohort establishment (seedling/saplings) to old forest structure, and for grass-forb-shrub interspaces (Figs. 18 and 19).
- Where threatened, endangered, or other rare species are a concern, alternative composition and structures may be needed.

Management Feasibility

Our key elements focus on the compositional and structural features of frequent-fire forests with the goal of creating opportunities for the resumption of characteristic ecological processes and functions and to re-establish the pattern-process link. In some cases, fire can be used to develop the desired composition and structure, while in other cases, it may be more effective when it follows the restoration of forest composition and structure through mechanical treatments. Some of the recent wildfire events in the Southwest may present opportunities to initiate the post-fire "reset" of composition and structure toward desired conditions through broad-scale application of managed fires. In many Southwestern areas, restoration of frequent-fire forests will be labor intensive and costly. In other areas, implementation, or certain implementation tools, may be constrained by logistic, economic, social considerations, and special land designations (e.g., wilderness and protected areas). For example, degraded conditions in current forests may limit the use of fire. In such areas, mechanical treatments may be necessary before introducing fire. In areas where silvicultural treatments

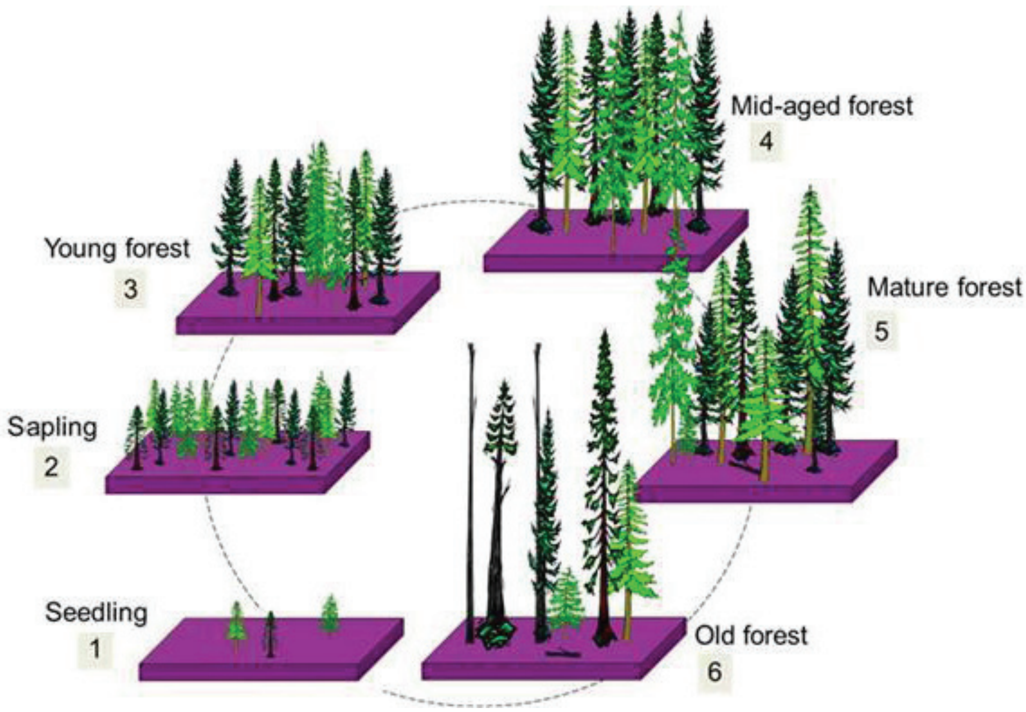

Figure 19. Illustration of the development of tree groups from seedlings to old forest at the fine scale.

are constrained by operational feasibility (e.g., access, slope, or economics) or in wilderness areas, fire may be the only management tool.

- It may not be feasible for management to approximate historical composition and structure patterns and/or fully restore characteristic ecological processes and functions everywhere.
    - o Socio-economic considerations (e.g., smoke, operational capacity, and public safety) may limit the use of fire and prescribed cutting. Some areas may require combinations of treatments to create and maintain desired compositions, structures, processes, and functions.
- Existing conditions influence treatment prescription and choice of tools.
    - o Fire alone can be used where there may be less need for precise outcomes. Fire may result in more variable forest density, numbers, and sizes of groups, and greater distribution of age classes.
    - o Where sustained production of ecosystem services is desired, managing at the extremes of the natural range of variability may be desired. For example:
        - ▪ Higher forest density and a balance of forest structural stages may be desirable to ensure economic sustainability (i.e., to maintain some level of sustained wood products) and for maintaining denser tree habitat conditions for some wildlife species.
        - ▪ Lower forest density and open forest structure may be desirable to facilitate additional reductions in fire hazard and for maintenance of more open habitat for some wildlife species.
    - o Depending on existing conditions, achieving the key elements may require multiple treatments (e.g., prescribed cutting and fire) over long time periods.
- Past disturbances, such as those resulting from fire and insects, may provide early management opportunities (i.e., reforestation and fire management) to put recovering forests on trajectories toward development of key compositional and structural elements.
- Consider strategic placement of restoration treatments to capitalize on the use of wildfire, under appropriate conditions, across broad landscapes.

Implementation of the Framework: Bluewater Demonstration Site

One of several implementations of our restoration framework was on the Cibola National Forest (Bluewater demonstration project) in New Mexico in 2010. Objectives of this project were to:

(1) create resilient forest composition and structure;

(2) move a predominately mid-aged forest toward uneven-aged conditions with an approximate balance of tree age classes;

(3) restore grass-forb-shrub interspaces;

(4) reduce fuels and fire hazard; and

(5) promote wildlife habitat, biodiversity, and wood products.

Our attempt to achieve the key compositional and structural elements in one treatment on the Bluewater site was limited by existing conditions; a portion of the mature and old trees had been harvested in prior treatments, there was little existing regeneration, and the site had a preponderance of mid-aged ponderosa pine trees. A comparison of pre- and post-treatment conditions (Figs. 20 and 21; unit 5A) attests to on-the-ground feasibility and utility of our framework recommendations for restoring the key elements in Southwestern ponderosa pine forests. Details for this project are available from the Forestry Staff with the USDA Forest Service Southwestern Region in Albuquerque, New Mexico.

Pre-Treatment Conditions

The Bluewater demonstration site is a 73-acre ponderosa pine stand (Fig. 22) that contained three different plant associations: ponderosa pine/mountain muhly, ponderosa pine/Arizona fescue, and ponderosa pine/blue grama, all of which are characterized as bunchgrass plant associations. The ponderosa pine site index is 72 for a base age of 100 years (Minor 1964). Soils are moderately productive and variable throughout the unit, comprised of alluvium and residuum from granite, and residuum derived from sandstone and claystone. The climate is temperate, with an average 180-day frost-free growing season from mid-May through mid-September and annual precipitation ranging from 17-25 inches, with greater than half occurring during the growing season.

Sanitation and improvement harvests occurred in the stand in the mid-1980s to remove diseased, dying, and poorly formed trees and, with the exception of piled slash burning in that treatment, the site had not experienced fire since the early 1900s. Prior to treatment, stand density averaged 216 trees and 125 square ft of basal area per acre. The stand was uneven-aged but had a predominance of mid-aged trees (Table 11). Fire behavior modeling demonstrated that 11 percent of the area had potential to support torching and active crown fire under dry conditions (i.e., completely dried fuel) and 15-mile/hr unobstructed wind speed.

Prescription Description

Tree marking occurred in spring 2010, tree cutting occurred in summer 2010, and prescribed burning is scheduled for fall and winter 2013. Treatment prescriptions were developed to produce the composition, structure, and spatial pattern identified in our framework for ponderosa pine: a predominant composition of ponderosa pine; re-establishment of a grass-forb-shrub community in interspaces between trees; groups of trees with interlocking or nearly interlocking crowns in the older age-classes; scattered individual trees; and retention of snags, logs, and woody debris.

The objective was to adjust stocking and spatial arrangement of residual trees (i.e., leave trees) to create or move the forest toward an uneven-aged and aggregated stand structure with a balance of age classes. Treatment prescriptions allowed within-site flexibility in numbers of trees per group and numbers and dispersions of groups per area as informed by historical evidence (i.e., old trees, logs, stumps with establishment date <1880) and existing forest structure. Treatment prescriptions used group selection to create grass-forb-shrub interspaces and regeneration sites and free thinning in immature leave tree groups to develop/retain interspersion of tree groups of different age classes and group sizes. Tree marking crews were instructed not to thin mature and old groups of trees except to remove young trees within these groups to reduce ladder fuel. Our intent was to have about 40 percent of the forested area occupied by mature-to-old tree groups, both of which meet old-growth objectives.

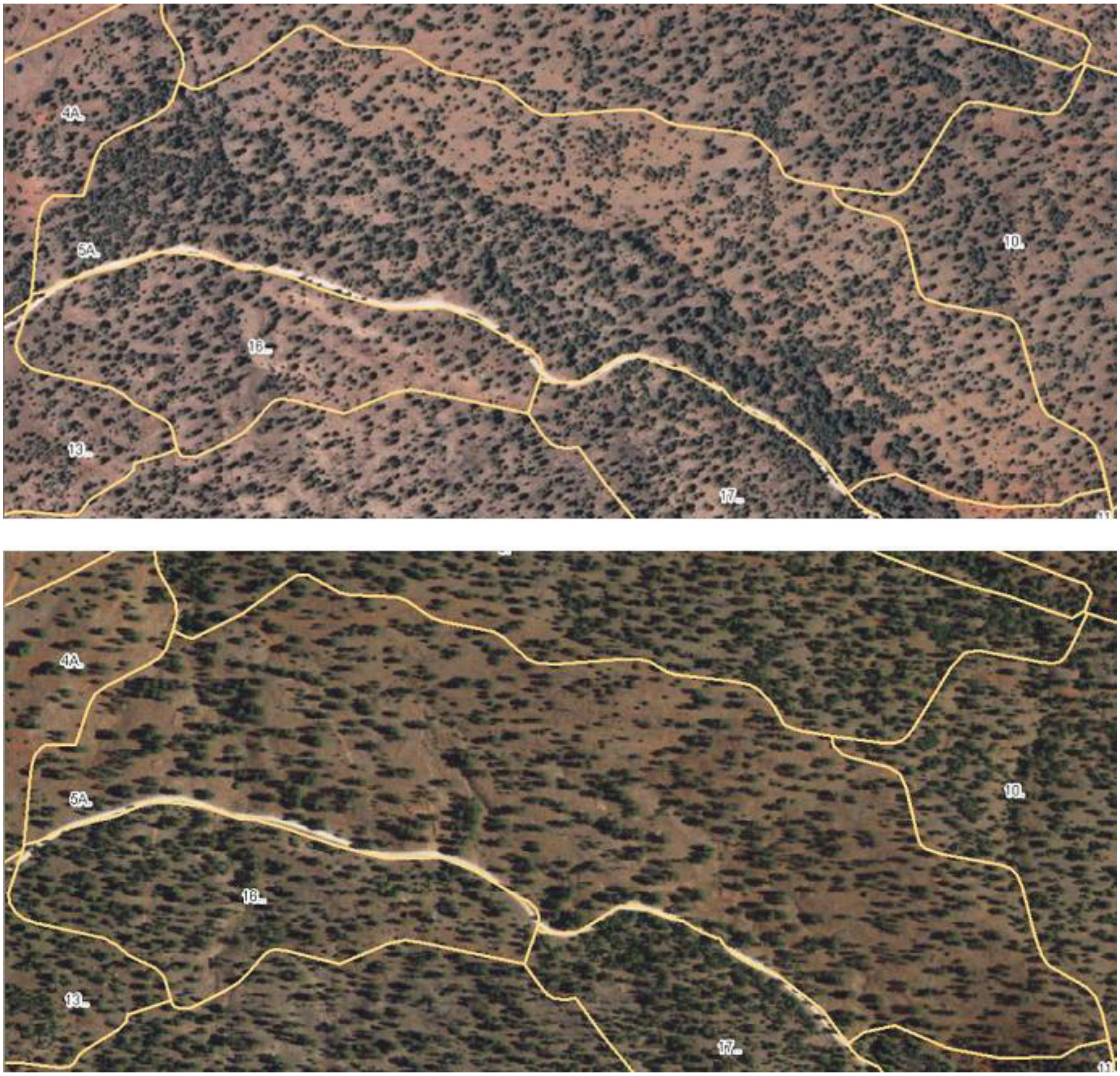

Figure 20. Aerial views of unit 5a on the Bluewater demonstration site in the Cibola National Forest, New Mexico. Prior to treatment (top image), forest density was substantially greater and more spatially homogenous than after the 2010 restoration treatment (bottom image) that applied the principles of our restoration framework.

Objectives were to favor retention of Southwestern white pine, ponderosa pine, and Douglas-fir; maintain minor components of pinyon pine and some juniper species; and favor Gambel oak and Rocky Mountain juniper trees for wildlife habitat. Leave-tree marking identified tree groups and single trees for retention. Leave trees were selected based on tree vigor and ages, with the objective of retaining an approximate balance of age classes. Special emphasis was also placed on within-group structure, including the retention of subdominant, dead-topped, and lightning-struck trees for wildlife habitat. Because no snags were present on the site, trees with declining vigor were retained for snag recruitment. Leave tree groups were either a single size or a blend of variably-sized trees. Trees within young groups were selected to encourage the development of future interlocking crowns. Overly dense young tree groups were thinned to facilitate vigor and future crown growth. Leave tree groups were generally 0.25-0.75 acres, but groups as small as a few trees and as large as 2 acres were also desired. After an initial training period, the marking crew successfully created the desired pattern of groups, scattered single trees, and grass-forb-shrub interspaces. However, they tended to mark numerous small-sized groups instead of a range of group sizes. To establish group size variability, we revisited the treatment area and added trees to some groups.

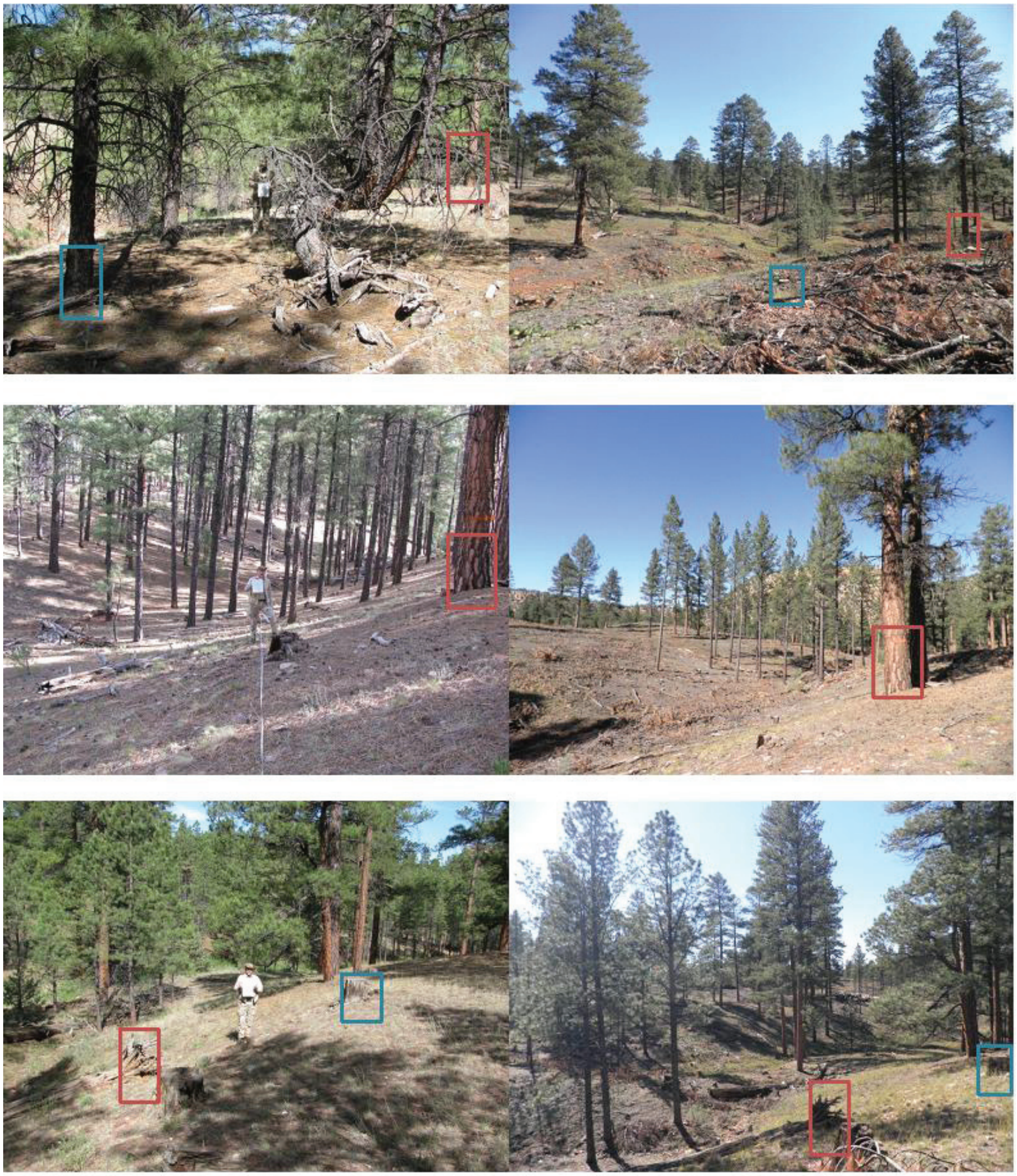

Figure 21. Paired photos from the same point before (left) and after (right) treatment in the Bluewater demonstration site, Cibola National Forest, New Mexico, USA. Colored boxes identify the same trees, cut stumps, or logs in before and after photos.

Interspaces between tree groups were created to provide for grass-forb-shrub vegetation and areas for root development. Desired interspace distances between leave groups ranged from 20-100 ft (drip line to drip line), with most distances ranging from 50-70 ft. To remedy a deficit of seedlings and saplings, regeneration sites ranging from 0.33-1.0 acre were created.

Treatment prescriptions specified the desired abundance of snags, logs, and woody debris: averages of 2 snags per acre with diameter at breast height (dbh) >12 inches and 3 downed logs per acre with dbh >12 inches. Where existing snag density was less than 2-3 per acre, live trees with broken tops or defects or fading green trees were retained for future snag and log recruitment.

The northern goshawk, tassel-eared squirrel, and Merriam's turkey were given special consideration. The treatment prescription was consistent with the restoration of habitats of plants and animals in the northern goshawk's food web (Reynolds and others

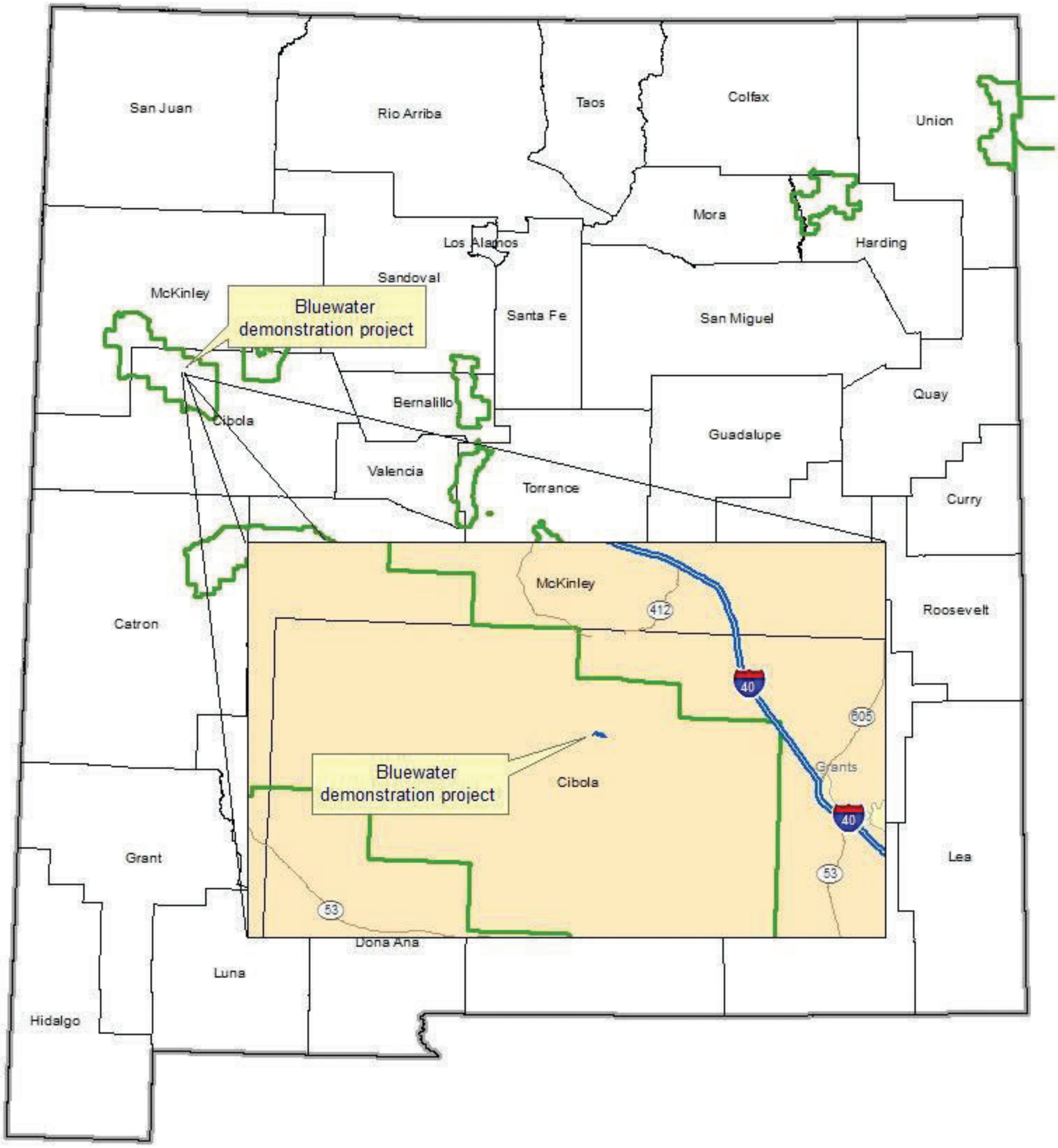

Figure 22. Location of the Bluewater demonstration project (108.45555° W, 38.45461° N) on the Cibola National Forest (green outline) in New Mexico, USA.

Table 11. Estimated proportion of stand area represented by different tree ages and sizes pre- and post-treatment on the Bluewater demonstration site.

Tree structural classes		Proportion of stand area under tree canopy	
Tree age[a]	dbh[b] range (inches)	Pre-treatment conditions	Post-treatment conditions
Seedling/sapling	0-4.9	5%	22%
Young	5-11.9	35%	26%
Mid-aged	12-17.9	40%	32%
Mature	18-23.9	10%	10%
Old	>24	10%	10%

[a]Tree ages are assumed to be related to sizes of dominant /co-dominant trees
[b]dbh = diameter at breast height

1992, 2006a, 2006b), including older tree groups with interlocking crowns for tassel-eared squirrels (Dodd and others 2003, 2006; Reynolds and others 1992) and high interspersion of grass-forb-shrub interspaces (foraging and brood habitat), closed-canopied tree groups (nesting and hiding cover), and large, old trees (roosting habitat) for Merriam's turkey (Hoffman and others 1993; Porter 1992).

Post-Treatment Conditions

This restoration treatment succeeded in creating the key compositional and structural elements identified in our framework (Figs. 21 and 23). The treatment retained the uneven-aged structure in the stand, increased the degree of interspersion of age classes, and is on a trajectory toward an approximate balance of age classes. The stand still had fewer seedling-saplings and mature and old trees than desired due to deficits in pre-treatment conditions (Tables 11 and 12). Approximately 28 percent of the area in the post-treatment stand was under the crowns of mid- to old-aged trees and 72 percent was open with no tree cover (Fig. 20). Approximately 20 percent of the post-treatment open area is designated for future tree recruitment, which will result in a desired 52 percent openness and 48 percent under tree cover. Open interspaces between tree groups were created for grass-forb-shrub communities and fire-safe sites were created for tree regeneration (Fig. 23). Post-treatment stand densities averaged 57 trees and about 40-80 square ft of basal area per acre. Most leave trees were arranged in groups with interlocking crowns, but scattered individual trees were retained across the site. Tree group sizes ranged from a few trees to 0.47 acres based on the area covered by tree crowns estimated from aerial photographs.

The post-treatment composition, structure, and spatial pattern of the stand reduced the risk of crown fire from pre-treatment conditions. Post-treatment FlamMap simulations predicted surface fires across 99 percent of the area and passive crown fire on 1 percent. Post-treatment abundance of small diameter woody debris was higher than intended, but prescribed burning will consume much of this material. Post-treatment abundance of logs and snags was lower than desired; however, these key structural features are expected to accumulate over time and with maintenance treatments. Mechanical treatments moved this forest stand toward restored conditions, but many years and multiple follow-up treatments (fire, mechanical, or combinations of these) will be needed to produce and maintain the desired key elements.

Future Management

Future plans are to broadcast burn the Bluewater site in the fall and winter of 2013 in order to initiate nutrient cycling and maintain fuels at desired levels. Subsequent entries will involve either tree felling, fire, or combinations of these to maintain or enhance the restoration treatment and manage for the desired mix and balance of tree age structures. Post-treatment conditions are being monitored at fixed photo-plots (Fig. 21) to determine whether compositional and structural objectives are being met and to inform future management.

Figure 23. Implementation of our framework in a ponderosa pine forest on the Bluewater demonstration site created groups of trees of a variety of vegetation structural stages (Table 11). The mechanical treatment also created open areas that will support grass-forb-shrub communities and tree regeneration.

Table 12. Post-treatment stocking level for the Bluewater demonstration site. All tree species are included in these estimates.

dbh[a] range (inches)	Trees/acre	Basal area (ft^2/acre)
1-4.9	3	0.4
5-8.9	17	4.6
9-12.9	23	16.2
13-16.9	5	6.1
17-20.9	5	10.2
21-24.9	2	4.3
>25	2	6.1
	Total 57	47.9

[a]dbh = diameter at breast height

Expected Outcomes of Framework Implementation

Our restoration framework is intended to promote ecosystem resilience by using fire and prescribed cutting treatments to restore the species compositions, structures, and spatial patterns of Southwestern frequent-fire forests. Restoring these features should allow re-establishment of characteristic processes such as disturbance regimes, nutrient cycling, food webs, hydrologic function, and ecosystem services such as biodiversity, old-growth, wood products, aesthetics, and recreation. Restoring characteristic compositions, structures, processes, and functions should also re-establish the evolutionary environment to which plants and animals native to these forests were adapted. Having intact, self-regulating, productive, and adaptive ecosystems is a compelling strategy for allowing species in the ecosystem to adapt to changing environments and facilitate their migration in the face of uncertain climate changes and disturbances. The following description of expected outcomes from restoring forest composition, structure, and spatial pattern in Southwestern frequent-fire forests is intended as an overview of some important outcomes from the restoration of these forests; this overview is not a comprehensive review of the literature. Improved understandings of these and other outcomes will require additional research (see Monitoring, Adaptive Management, and Research Needs).

Ecosystem Resilience to Climate Change

Restoring ecosystem resilience based on historical conditions has been a central concept in ecosystem management (Covington 2003; Folke and others 2004; Scheffer and others 2001). However, the relevance of historical conditions as reference points and targets for restoration has been questioned on the basis of uncertainty of future ecological conditions due to global climate change (Harris and others 2006; Millar and others 2007; Wagner and others 2000). Specific challenges for restoring and sustaining frequent-fire forests in the face of climate change are uncharacteristically rapid alterations of environments and combinations of disturbances and non-native biotic factors producing conditions never before documented in evolutionary time—conditions that may overwhelm characteristic ecological processes (Fulé 2008). In light of these challenges, we review the evolutionary history of these forests.

Over the past several million years, forests and woodlands in the Southwest, including their associated microbial, plant, and animal communities, have tracked favorable habitats and climates whose migrations across geographical and elevational ranges were driven by major climate fluctuations (Bonnicksen 2000; Covington 2003; Delcourt and Delcourt 1988). Since the end of the last major glacial period (14,000 years ago), ponderosa pine returned to the high elevation plateaus and mountains of Arizona about 10,000 years ago and to the central Rocky Mountains only about 5000 years ago (Baker 1986; Covington 2003; Latta and Milton 1999; Millar 1998). In the last 50 million years, frequent-fire forests survived wide swings in environmental conditions (Moore and others 1999). Component species of frequent-fire forests adapted over evolutionary time to arid environments that have been characterized by variable wet and dry periods, including prolonged droughts, and disturbances such as fire, insects, and diseases. These disturbances varied in frequency, intensity, and extent (Covington and Moore 1994b); served as checks on the demographic rates of component species; and resulted in self-regulating processes of nutrient cycling, productivity, and regeneration (Allen and others 2002; Cooper 1960; Covington and others 1997; Covington and Moore 1994b; Falk 2006).

The highest confidence in future climates is associated with projections that are consistent among climate change models and observed climate changes. Surface temperatures in the Southwest are predicted to increase substantially, with more warming in the summer and fall than in winter and spring; summer heat waves will become longer and hotter, with reductions of late winter/spring mountain snowpack due mostly to warmer temperatures (Overpeck and others 2012). Observed Southwest droughts have been exacerbated by warmer summer temperatures and are projected to become hotter, more severe, and more frequent, suggesting an increased drying in the Southwestern United States and that historical drought levels may become the norm (Overpeck and others 2012; Seager and others 2007). Such droughts will directly increase tree mortality and vulnerability to pathogen attacks (Breshears and others 2005) and enhance the size and severity of wildfires (Fulé 2008). Thus, current conditions in frequent-fire forests (i.e., high stand densities, accumulations of fuels on the forest floor, and encroachment of fire-susceptible species; Cocke and others 2005; Cooper 1960) will increase the susceptibility to stand-killing fire (Fulé 2008). It is also likely that on some sites, fire-caused changes in vegetation (e.g., forest to grasslands or shrublands) may not at all resemble those of historical forests (Barton 2002; Savage and Mast 2005; Strom and Fulé 2007). Predicted changes to warmer climates in the American Southwest are expected to affect forests via geographical shifts in suitable environments for the dominate forest species. Shifts are expected to be to higher elevations and northward (Fulé 2008; Shafer and others 2001).

Uncertainties associated with future climate changes make the development of restoration strategies increasingly complex and challenging. The scenario of future hotter, more severe, and more frequent droughts in the Southwest (see Karl and others 2009) includes increased competition for water and increased frequency and extent of high-severity fire, insect, and disease disturbances. Restoring the characteristic composition, structure, and spatial pattern in frequent-fire forests would thereby:

- reduce tree densities and canopy continuity;
- recreate grass-forb-shrub plant communities;
- reduce competition for space, water, and nutrients (Covington and others 1997); and
- provide for the re-establishment of characteristic disturbance regimes (Covington and others 1997; Fulé and others 2002b; Kolb and others 1998).

Nonetheless, restoration strategies should account for an ecosystem's current condition as they may influence an ecosystem's development under future climate. Alternative successional pathways under future climactic variability may invalidate reference conditions as baselines for restoration (Clewell and others 2005; Pilliod and others 2006).

While climate forecasting remains imperfect, fire predictions for Western North America suggest substantial increases in occurrences, spread, and intensity (Brown and others 2004; Honig and Fulé 2012; McKenzie and others 2004; Spracklen and others 2009). Thus, managing frequent-fire forests toward the historical composition, structure, and spatial pattern is consistent with a reduced vulnerability to catastrophic loss (Allen and others 2002; Falk 2006; Honig and Fulé 2012). While we recognize that uncertainties in how species and communities can and will respond to rapid climate change, we agree with Fulé (2008) that it makes sense to restore fire and fire-related composition, structures, and spatial patterns to enhance resistance to catastrophic loss. Restoring the composition, structure, and spatial pattern of these forests should increase their resistance and resilience to climate changes, thereby providing opportunities for species to migrate or develop local adaptations. In fact, Fulé (2008) suggested a restoration strategy that focuses on mesic areas at higher latitudes and elevations (i.e., upper portions of the ponderosa zone and the transitional dry mixed-conifer zone) where forests are more likely to survive climate change. Fulé (2008) recommended using reference conditions from low and southerly areas to guide management in higher-elevation ecosystems to provide for the migration of species as climate warms.

In summary, both reference conditions and natural range of variability are useful guides for management because Southwest frequent-fire forests were historically resilient to drought, insect pathogens, and severe wildfire. Our restoration framework should therefore increase the resistance (by forestalling impacts), resilience (through improved recovery after disturbance), and response (allowing transitions or migrations to new conditions) of frequent-fire forests to climate change (Millar and others 2007; Parker and others 2000; Price and Neville 2003; Spittlehouse and Stewart 2003).

Disturbance Regimes

Restoring the composition, structure, and spatial patterns of frequent-fire forests will provide for the re-establishment of feedback relationships between pattern and disturbance processes in these forests (Larson and Churchill 2012). Disturbances are temporary changes in environmental conditions that cause changes in ecosystem composition and structure. Restoring the composition and structure of frequent-fire forests will result in a more open forest structure and decrease the potential for epidemic outbreaks of insects and diseases and stand-replacing fire (Fitzgerald 2005; Fulé and others 2002, 2004; Graham and others 2004; Roccaforte and others 2008; Strom and Fulé 2007). The restoration of grass-forb-shrub interspaces and resultant separation of tree canopies will increase herbaceous plant development and provide fuels to carry frequent surface fires. In turn, restoration of characteristic fire regimes should sustain forest composition, structure, processes, and functions. Reduced tree densities result in reduced competition for resources, increased tree vigor, and reduced insect and disease infestations (Hessburg and others 1994; Kolb and others 1998).

The intent of our framework is not to eliminate insects and diseases but to return populations and their effects to an endemic, low background level of tree mortality (Miller and Keen 1960). In areas with higher tree densities that may have escaped repeated surface fire, bark beetles can be a significant agent for shaping forest structure and fine-scale spatial heterogeneity. Increasing the spacing between groups of trees can reduce the continuity of mistletoe occurrence across the landscape and reduce mistletoe spread between groups, creating the opportunity for groups of trees that are free of mistletoe (Hawksworth 1961). Frequent surface fires can elevate tree crown bases and increase tree spatial heterogeneity, both of which can slow mistletoe spread (Conklin and Geils 2008). Frequent surface fire can also reduce the severity of mistletoe infection by killing heavily infected trees (Conklin and Geils 2008; Koonce and Roth 1980).

Nutrient Cycling

A restored fire regime can also improve soil nutrient conditions. Intense heat from fire volatilizes nitrogen from the soil and surface fuels, often causing the total nitrogen concentration of forest soils to decline (Boerner and others 2009; DeLuca and Sala 2006). However, nitrogen concentrations tend to recover and even increase two to four years following fire as soil microbes decompose ash and plant litter (Boerner and others 2009). Fire can also cause an immediate pulse of inorganic nitrogen due to the combustion of organic matter and mortality of soil microbes (DeLuca and Sala 2006). Soil ammonium concentrations in ponderosa pine forests may increase as much as 20-fold following fire followed by dramatic increases in nitrate levels after the first year (Covington and Sackett 1992). Frequent burning can maintain elevated levels of inorganic nitrogen in forest soils by depositing charcoal, which binds to inorganic nitrogen and slows its leaching, and by promoting the establishment of grasses and herbaceous vegetation (DeLuca and Sala 2006; Hart and others 2005). Grasses and herbaceous vegetation produce litter with higher nitrogen-to-carbon ratios than conifer vegetation; thus, the presence of herbaceous vegetation may stimulate decomposition and enhance the availability of inorganic nitrogen in forest soils (Hart and others 2005). Fires also kill large trees, creating snags that ultimately become coarse woody debris that plays an important role in nutrient cycling (Brown and others 2003; Cram and others 2007; Graham and others 1994; Harvey and others 1988; Lowe 2006).

Biodiversity and Food Webs

Many ecosystem processes influence plant productivity, soil fertility, water availability, and other local and global environmental conditions. These processes are often controlled by the diversity and composition of plant, animal, and microbial species native to an ecosystem, and recent studies suggest that losses in biodiversity can alter the magnitude and stability of ecosystem processes (Naeem and others 1999). As a dominant species in frequent-fire forests, ponderosa pine influences the understory vegetation, soils, and plant and animal habitats and communities (Moore and others 1999). A community is a group of organisms that interact and share an environment. Organisms in a community may compete for resources, profit from presence of other organisms, or use other organisms as a food source. In the Southwest, ponderosa pine forests are occupied by over 250 species of vertebrates, invertebrates, soil organisms, and plant species (Allen and others 2002; Patton and Severson 1989), many of which adapted to high levels of the spatial heterogeneity and biodiversity that characterized historical frequent-fire forests. A compositionally and structurally diverse understory provides food and cover for many species of vertebrates and invertebrates, each contributing to ecological functioning and food webs. For example, the dispersion of mycorrhizal fungi, a root symbiont critical to the growth and health of trees, is likely reliant on small mammal transfer via feces (Johnson 1996).

Current frequent-fire forests are uncharacteristically homogeneous in composition and structure with reduced plant and animal habitats and lowered biodiversity (Allen and others 2002; Kalies and others 2012; Laughlin and others 2006; Patton and Severson 1989; Villa-Castillo and Wagner 2002; Waltz and Covington 2003). Achieving our restoration framework's key elements restores habitats at multiple spatial scales, especially through the re-establishment of species-rich grass-forb-shrub communities and the productivity, biodiversity, and trophic interactions they support (Abella 2009; Clary 1975; Kalies and others 2012; Oliver and others 1998; Reynolds and others 1992, 2006a; Rieman and Clayton 1997). Dense tree conditions in current frequent-fire forests favor plants and animals that do better in more close-canopied forests. Restoration to more open forest conditions may result in the decline of these species but should increase abundance of more open forest species (Kalies and others 2012). Nonetheless, because our framework creates a variety of forest age and structural stages, including groups and patches with dense forest structures, declines of denser-forest obligates may be minimized (e.g., tassel-eared squirrel; Dodd and others 2003, 2006; Kalies and others 2012), resulting in higher overall species diversity (Noss and others 2006).

Another concern is that the fine-scale structural heterogeneity of forests resulting from restoration of frequent-fire forests may lower the abundance and viability of large-area-dependent species (e.g., spotted owl; Holthausen and others 1999; Prather and others 2008). These concerns might be ameliorated by developing specific desired conditions for breeding sites (e.g., on denser north slopes) and feeding sites with prey habitats (Prather and others 2008; Reynolds and others 1992). It is worth noting that breeding sites or entire refugia for imperiled species may receive protection from loss by encircling them with restored forests, lowering risk of catastrophic loss through fire or insects (Prather and others 2008). This indicates that restoration of these forests and the habitats they contain may provide for the historical distribution and abundance of plants and animals in Southwestern frequent-fire forests.

Restoration of frequent-fire forests should lead to more robust food webs by re-creating diverse habitats across landscapes. Species diverse and productive grass-forb-shrub communities in interspaces between tree groups support broad-based food webs that many invertebrates, birds, mammals, and their predators depend upon (Abella 2009; Dodd and others 2003; Ganey and others 1992; Kalies and others 2012; Linkhart and others 1998; Reynolds and others 1992, 2006a; Rosenstock 1998). The importance of diverse tree and grass-forb-shrub habitats and robust food webs at multiple spatial scales was demonstrated by temporal variations in the vital rates of northern goshawk (Reynolds and others 1992, 2005, 2006a, 2006b), a sensitive species that has been the subject of extensive research in the Southwest (Beier and Drennan 1997; Beier and others 2008; Boal and Mannan 1994; Ingraldi 2005). In the Southwest, goshawk reproduction typically varied extensively year-to-year and was strongly associated with the abundance and availability of food; in years when prey numbers were low, goshawk population reproduction was a fraction of reproduction in years when prey was abundant (Beier and others 2008; Reynolds and others 2005; Salafsky and others 2005, 2007). Goshawks typically feed on a broad suite of prey—from robins, jays, woodpeckers, doves, and grouse to tree squirrels, ground squirrels, rabbits, and hares, each occupying different habitats (Reynolds and others 1992, 2006a). Annual population highs and lows of each prey species are not always in phase; a year's population low of one or more prey is often compensated by higher abundances of other species (Salafsky and others 2005). Due to compensation, forest management strategies that provide a fine- to mid-scale interspersion of habitats are more likely to successfully maintain an entire suite of prey at higher total abundance through both good and poor prey years in individual goshawk home ranges (Reynolds and others 1992, 2006a). For the goshawk and the many other avian and mammalian predators (e.g., raptors, weasels, bobcats, and coyotes) in Southwestern frequent-fire forests, the grass-forb-shrub prey community is particularly important because it is occupied by a large proportion of the birds and mammals native to these forests as well as many important prey species, including rabbits, grouse, ground squirrels, mice, and voles. Prey species in this vegetation layer had larger body masses than most other species occurring in frequent-fire forests (Reynolds and others 1994; Salafsky and others 2005). Furthermore, several of these species are known to attain high population abundance in response to grass-forb-shrub productivity and biodiversity (Ernest and others 2000; Gross and others 1974; Hernández and others 2011; Hostetler and others 2012; McKay 1974). Others of our framework's key elements also create important habitats in Southwestern frequent-fire forests, including:

- dense groups and patches of older-aged trees with interlocking crowns for tree squirrels and species requiring denser forest conditions;
- snags for woodpecker foraging and nesting;
- snags for secondary-cavity nesters, bark gleaning birds, and hunting and sallying perches;
- logs for many invertebrate species (spiders, ants), woodpeckers, mice, rabbits, ground squirrels, grouse, and wild turkey; and
- woody debris for many small mammals.

Old-Growth

The key elements described in the restoration framework provide and sustain old-growth tree components at all spatial scales. Old-growth components provide a number of ecosystem services—plant and animal habitat, biodiversity, carbon sequestration, hydrologic function, high-quality wood products, aesthetics, and spiritual values. Old-growth structure includes old trees, dead trees (snags), downed wood (coarse woody debris), and structural diversity (Figs. 9, 12, and 13) (Franklin and Spies 1991; Helms 1998; Kaufmann and others 2007). The concept of old-growth includes multiple spatial and temporal scales, ranging from individual trees to tree groups and patches to landscapes and their development overtime. Definitive characteristics of old growth in the Southwest vary by forest type as a consequence of differences in species composition, tree longevities and sizes, and the characteristic types, frequencies, and severities of disturbances (Harmon and others 1986). Old-growth forests in the Southwest have been partitioned into three groups based on different fire regimes and resultant compositional and structural features (Table 10): frequent, low-severity fire; mixed-severity fire; and infrequent, high-severity fire (Table 2).

Old-growth in frequent-fire forests occurs as old trees in groups and as scattered individuals within uneven-aged forests. These forests are less dense and have fewer logs and woody debris than high-severity infrequent-fire forests. Old-growth structural features typically occur at the fine scale (Meyer 1934; Weaver 1951) and are composed of small, old tree groups interspersed with similarly sized groups of younger trees, seedlings to mid-aged (Table 10) (Cooper 1961; Harrod and others 1999; Morgan and others 2002; Pearson 1950; Woolsey 1911). The fine-scale age diversity through growth and development sustained the old-growth tree components. Our framework's key restoration elements in frequent-fire forests include all the essential structural features of old growth distributed throughout the uneven-aged forest (Kaufmann and others 2007).

In contrast to frequent-fire forests, old-growth in forests with a mixed-severity fire regime (Table 2) is characterized by adjacent forest patches burned by either low- or high-severity fire (Fulé and others 2003; Grissino-Mayer and others 1995). This results in landscapes with patches of old-growth intermixed with patches of different forest ages. Under an infrequent, high-severity regime (Table 2), old-growth forests are driven by mid- to landscape-scale, high-severity fire followed by vegetation recovery and succession occurring over long periods between fires. Infrequent, high-severity fire regimes typically have large (>100 acres) patches of forests dominated by large, old trees with multiple canopy layers with similar times since disturbance and vegetation origin dates.

Hydrologic Function

We found no published studies that evaluated the long-term effects of restoration on hydrologic function and water yield in Southwestern frequent-fire forests (see Monitoring, Adaptive Management, and Research Needs). However, studies on the effects of different tree harvest prescriptions on hydrologic function and water yield offer insights into the probable effects of reducing current high tree densities through restoration of frequent-fire forests in the Southwest. Hydrologic function and water yield in forests are greatly influenced by the amount and distribution of vegetation, precipitation, snow melt, basin physiography, and soil type. In dense (92-140 ft^2/acre) ponderosa pine forests, reduction of residual basal area to less than 100 ft^2 per acre resulted in increased water yield, although large variations in yield are typical. In addition, initial mean increases in water yield of 15-45 percent can be realized in ponderosa pine forests on basalt-derived soils when high basal area in current forests is reduced. However, increases can be expected to decline with time as vegetation establishes and develops (Baker 1986; Douglas 1983; Harr 1983; MacDonald and Stednick 2003; Troendle 1983). Removal or reduction of forest cover can increase soil water storage, which then becomes available for groundwater recharge (Baker and others 2003). Soil water content was reported to be higher in thinned and thinned-and-burned areas than in untreated-control areas on basalt soils in northern Arizona. However, observed annual variation in water yield showed that the amount and timing of precipitation had a greater overall effect on water yield than did the removal of trees (Feeney and others 1998).

From the above it seems reasonable that restoring our framework's key elements will benefit hydrologic function by reducing stand density and creating open grass-forb-shrub interspaces, decreasing canopy transpiration and interception losses, concentrating snow in interspaces, and increasing soil infiltration, water storage, and stream and spring flow (Baker 1986; Ffolliott and others 1989). While an objective of increasing water yield may not be a sufficient justification for forest restoration, increases in water yield are a significant incidental benefit (Baker 2003).

Wood Products

The re-establishment of frequent, low-severity fire is critical to the success of our restoration framework. However, because of limitations such as proximity to human developments, air quality restrictions, and workforce capacity, the use of fire will probably continue to be limited. Therefore, mechanical-only treatments, or perhaps combinations of fire and mechanical treatments, are likely to be the restoration tools of choice in much of the Southwestern landscape. Another limitation to restoration is the economic viability of treatments; can treatments generate revenue to fund restoration or must they be subsidized? In the initial stages of forest restoration, an abundant supply of lower-valued wood products could help create local products, industries, and enterprises and generate some revenue. Establishment of small-diameter tree markets, followed by shifts to markets targeting the use of restoration by-products (e.g. traditional and emerging products utilizing a wider range of tree sizes), will be essential to long-term restoration and stable local industries. Yields between 400 and 700 cubic ft per acre seem reasonable from a cutting cycle of 25 to 30 years once restoration achieves an approximate balance of structural stages in frequent-fire forests (Youtz and Vandendrieshe 2012). Such yields would help offset costs of achieving multiple objectives.

Aesthetics and Recreation

The public often judges the ecological health of a forest by appearance. Hill and Daniel (2008) found that acceptance of restoration activities may be contingent on public perceptions of aesthetics and knowledge of ecological benefits. People prefer landscapes with large trees, openings, and varied spatial distribution of vegetation that provide views through the site and into the landscape (Brush 1979). Recreational campers preferred camp-sites that were about 60 percent shaded (James and Cordell 1970), while others preferred uneven-aged forest landscapes over even-aged, dense stands (Brown and Daniel 1984, 1986, 1987; Ryan 2005). Restored forests meet these scenery preferences, suggesting greater public acceptance and support.

Monitoring, Adaptive Management, and Research Needs

Frequent-fire forests in the Southwest are complex and dynamic, and our understanding of how they function and respond to disturbances is limited by available data. Knowledge gaps and unexpected events inevitably make forest management and restoration inherently challenging. Key to meeting restoration challenges are the conduct of ecological monitoring, adaptive management, and additional research. This framework and its application are intended to be dynamic and adaptive and will evolve with accumulations of new monitoring and research information.

Ecological monitoring is the means by which managers evaluate whether the current conditions of an ecological system match, or are on a trajectory to match, some desired condition (Noon 2003). Monitoring provides feedback on the impacts of management treatments (Lindenmayer and Likens 2010; Palmer and Mulder 1999) and is typically divided into three categories: implementation, effectiveness, and validation (Busch and Trexler 2003). Implementation monitoring occurs during implementation and determines whether treatments were carried out as intended. Effectiveness monitoring determines the extent to which treatments achieved their ultimate objectives. Validation monitoring assesses the degree to which underlying assumptions about ecosystem relationships are supported (Block and others 2001; Busch and Trexler 2003) and functions to identify knowledge gaps or research needs.

Adaptive management requires feedback obtained from monitoring regarding the success or failure of treatments (Walters 1986). Adaptive management is the "rigorous approach for learning through deliberately designing and applying management actions as experiments" (Murray and Marmorek 2003). In contrast to simply measuring treatment effects and making slight adjustments to future treatments, adaptive management depends on structured, adaptive decision making (Williams and others 2009). It is most useful when managers and scientists identify threshold values for triggering management actions (Noon 2003). A clear description in a plan of how monitoring will be used in decision-making is essential (Noon 2003; Williams and others 2009). This could be achieved administratively (Mulder and others 1999; Sitko and Hurteau 2010), legally via the National Environmental Policy Act process (Buckley and others 2001), or through collaborative agreements (Gori and Schussman 2005). Monitoring data should be compiled, analyzed, and reported in a timely manner so that managers are provided information to improve decision-making (Mulder and others 1999) and to identify knowledge gaps.

Although much is known about historical forest composition, structure, and disturbance in frequent-fire forests, our knowledge of the mechanisms of spatial pattern formation and maintenance is limited, indicating a research need (Larson and Churchill 2012). A limited understanding of reference conditions on different parent material, especially in dry mixed-conifer, is an important data limitation for designing and implementing appropriate resource management. While the number of reference data sets is increasing, existing data have focused largely on tree density. There is a clear need for studies on spatial patterns and the sizes and shapes of grass-forb-shrub interspaces, as well as the mechanisms for the formation and maintenance of spatial patterns. Additional research needs are:

- Increased understanding of reference conditions and the natural range of variation across ecological gradients such as latitude and longitude, soils, topography, and climate in Southwest frequent-fire forests, especially in dry mixed-conifer.
- Increased understanding of differences between ponderosa pine and dry mixed-conifer forests in reference conditions and the historical types, frequencies, severities of disturbances, and responses of vegetation. Of particular need are:
  (1) A greater understanding of variation of reference conditions (composition, structure, and spatial pattern) in forest subtypes and different plant associations.
  (2) How reference conditions influenced the effects of fire on tree regeneration and mortality in forest subtypes and in the transition zones between subtypes.
  (3) The effectiveness of restoration treatments at achieving desired objectives, especially on avoiding the conversion of these subtypes to alternative plant associations.
- Increased understanding of ecosystem processes and functions as they respond to restoration of the composition and structure of frequent-fire forests.

- Increased understanding of the mechanisms of spatial pattern formation (e.g., aggregated and random tree distributions) within- and among-groups, including the presence, abundance, and dispersion of individual trees.
- An understanding of historical roles of insect and disease in shaping forest composition, structure, and spatial pattern, and the effects of restoration on the frequency and severity of insect and disease disturbances at all scales.
- An understanding of the effects of exotic insect, disease, plant, and animal species, and how these may alter forest composition, structure, processes, and functions.
- Increased understanding of the efficacy of fire versus tree cutting only and cutting combined with fire at achieving the desired composition, structure, processes, and functions in frequent-fire forests at all scales.
- Identification of management strategies for restoring composition and structure in transitional zones between forest types and future directions given climate change.
- Development and refinement of new and existing tools and metrics for measuring spatial heterogeneity at ecologically meaningful scales.
- Improved understanding of wildlife habitat and wildlife uses of restored composition and structure of frequent-fire forests.
- Improved understanding of long-term effects of restoration and maintenance treatments (mechanical, fire, and a combination of the two) on water yield and quality.
- Assessment of ecological, economic, and social benefits and costs (e.g., invasive species) of different restoration methodologies and implementation practices, such as methods for treating slash, tree marking approaches, spatial scales of treatment, and frequency of maintenance treatments.
- Exploration of management applications to implement our framework on broad landscapes in an economically efficient manner.

Summary

Our forest restoration framework provides managers and researchers a review of existing knowledge regarding the historical compositions and structures in Southwest frequent-fire forests and how these operated through feedback mechanisms that sustained their characteristic compositions, structures, and functions. Current forest conditions, the cumulative consequences of various human activities that altered historical conditions, are reviewed in light of historical conditions with a focus on how human-caused changes lowered the resistance and resilience of these forests to historical disturbance agents that themselves have become more intense and frequent. Guided by our understanding of how the composition, structure, and spatial pattern of historical frequent-fire forests affected their resistance, resilience, and responses to disturbances, our restoration framework identifies desired key compositional and structural elements of these forests and provides management recommendations for restoring those key elements. We believe implementation of our framework provides opportunities for re-establishing characteristic processes such as frequent, low-severity fire and ecological functions such as habitat, biodiversity, and food webs.

The key compositional and structural elements of historical frequent-fire ponderosa pine and dry mixed-conifer forests in the Southwest can be envisioned over time as a shifting mosaic of groups of trees with interlocking crowns; single trees; open grass-forb-shrub interspaces; and dispersed snags, logs, woody debris (Larson and Churchill 2012; Long and Smith 2000; Reynolds and others 1992). Research shows that the degrees of tree aggregation; sizes and numbers of tree groups; numbers and dispersion of single trees; sizes and shapes of grass-forb-shrub interspaces; and numbers, sizes, and dispersions of snags, logs, and woody debris in reference conditions varied among sites by soil, topography, climate, disturbance regime, and past stochastic events. Our restoration framework recognizes this site-to-site variability and articulates the importance of restoring that variability by using existing evidence (e.g., old trees, snags, stumps, and logs) and biophysical site indicators as guides for restoring local variability. In our view, restoration of spatial and non-spatial elements of forest structure on a per-site basis is the most practical, science-based strategy to return frequent-fire forest ecosystems in the Southwest to resistant, resilient, and responsive conditions that will best position them to adapt to future disturbance regimes and climates (Larson and Churchill 2012; Millar and others 2007). We intend this framework and its application to be flexible and adaptive (i.e., learn-as-you-go) and to evolve with accumulation of knowledge, and for its conceptual approach to provide a blueprint against which management plans and practices can be evaluated.

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Glossary

Age class is defined as trees that originated within a relatively distinct range of years. Typically, the range of years is considered to fall within 20 percent of the average maturity (e.g., if 100 years is required to reach maturity, then there would be five 20-year age classes) (Helms 1998).

Basal area is the cross-sectional area of all stems of a species or all stems in a stand measured at breast height (4.5 ft above the ground) and expressed per unit of land area.

Biodiversity is the variety and abundance of life forms, processes, functions, and structures of plants, animals, and other living organisms, including the relative complexity of species in communities, gene pools, and ecosystems at spatial scales from local to regional to global (Helms 1998).

Canopy cover (*see* forest canopy cover)

Canopy fuels are all burnable materials, including live and dead foliage, lichen, stems, and branch wood located in the forest canopy.

Characteristic (natural) conditions (e.g., vegetation composition and structure), processes (e.g., disturbance regimes), and functions (e.g., habitat, biodiversity, and food webs) of a forest type that are present under the natural range of variability.

Clump refers to (1) the aggregate of stems issuing from the same root, rhizome system, or stool; or (2) an isolated generally dense group of trees (Helms 1998). A clump is relatively isolated from other clumps or trees within a group of trees, but a stand-alone clump of trees can function as a tree group or a single structure (Fig. 4).

Coarse woody debris is dead woody material on the ground greater than 3 inches in diameter, including logs (Figs. 12 and 13).

Composition is the array of species present in an ecosystem. In forestry, this term often refers to the proportion of each tree species in a stand expressed as a percentage of the total number, basal area, or volume of all tree species in the stand (Helms 1998).

Diameter at breast height (DBH) is the diameter of a tree typically measured at 4.5 ft above ground level.

Disturbance (characteristic and uncharacteristic): Any relatively discrete event in time that disrupts ecosystems, communities, or population structure and changes resources, substrate availability, or the physical environment (Helms 1998). Characteristic disturbances are those whose extent, frequency, and severity fall within the natural range of variability. Uncharacteristic disturbances are outside the natural range of variability and interrupt characteristic processes and functions.

Dry mixed-conifer forests occupy the warmer and drier sites between elevations of 5000 and 10,000 ft and are characterized by a relatively frequent historic fire regime (<35 years fire return interval), resulting in surface fire and infrequently, mixed-severity fire effects. This forest type is typically dominated by shade-intolerant species such as ponderosa pine, with minor association of aspen, Douglas-fir, and Southwestern white pine during early seral stages. More shade-tolerant conifers such as Douglas-fir, white fir, and blue spruce are dominant at climax stages. In the Southwestern United States, this type is primarily described by the Society of American Foresters cover types interior Douglas-fir and white fir.

Ecological (ecosystem) health (*see* forest health)

Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. Restoration initiates or accelerates ecosystem recovery with respect to its health (productivity), processes, and functions (biodiversity, food webs, and sustainability) (adapted from SER 2004).

Ecosystem integrity is the state or condition of an ecosystem that displays the biodiversity characteristic of the reference, such as species composition and community structure, and is fully capable of sustaining normal ecosystem functioning (SER 2004).

Ecosystem resiliency is the ability of an ecosystem to absorb and recover from disturbances without altering its inherent functions (SER 2004).

Ecosystem services are the benefits people obtain from ecosystems, including provisioning services such as food and water; regulating services such as flood and disease control; cultural services such as spiritual, recreational, and cultural benefits; and supporting services such as nutrient cycling, that maintain the conditions for life on Earth (Millennium Ecosystem Assessment 2005).

Ecosystem stability is the ability of an ecosystem to maintain its given trajectory (SER 2004).

Ecosystem sustainability is the capacity of ecosystems to maintain ecosystem services in perpetuity without degradation of its productivity and function at all scales. For example, in the context of our restoration framework, sustainability results in maintaining the key elements in space and time.

Even-aged forests are forests that are comprised of one or two distinct age classes of trees.

Evolutionary environment refers to the range of abiotic and biotic conditions that have exerted selection pressure on and are critical to the survival of species or groups of species (Kalies and others 2012; Moore and others 1999).

Fine fuels are fast-drying dead or live fuels, generally characterized by a comparatively high surface area-to-volume ratio, that are less than 0.25 inch in diameter and have a time-lag of one hour or less. These fuels (grass, leaves, needles, etc.) ignite readily and are consumed rapidly by fire when dry (NWCG 2012).

Fire regime refers to the patterns of fire occurrences, frequency, size, severity, and sometimes vegetation and fire effects in a given area or ecosystem. A fire regime is a generalization based on fire histories at individual sites (McPherson and others 1990).

Fire return interval is the number of years between two successive fires in a specified area (McPherson and others 1990).

Forest canopy cover is the proportion of ground or water covered by a vertical projection of the outermost perimeter of tree canopies, regardless of tree spatial arrangement.

Forest health is the state or condition of forest ecosystems in which its attributes (i.e., productivity) are expressed within "normal" ranges of activity relative to its ecological stage of development. A restored ecosystem expresses health if it functions normally relative to its reference ecosystem (adapted from SER 2004).

Frameworks provide a set of assumptions, concepts, values, and practices that constitute a way of viewing reality (American Heritage Dictionary 2011).

Free thinning is the removal of trees to control stand spacing and favor desired trees using a combination of thinning criteria without regard to crown position (Helms 1998).

Frequent-fire forests are forests with fire regime 1, those forests with fire frequency <35 years (Schmidt and others 2002).

Functions (ecological functions) are the outcomes of ecosystem components and processes (e.g., interactions within and among species). Examples include primary and secondary production and mutualistic relationships. Ecosystem functions are broadly categorized as regulation functions, habitat functions, production functions (e.g., genetic and medicinal resources), and information functions (e.g., spiritual and historic information) (De Groot and others 2002).

Group refers to a cluster of two or more trees with interlocking or nearly interlocking crowns (Fig. 4 and 12) at maturity surrounded by grass-forb-shrub interspaces (Fig. 8). Size of tree groups is typically variable depending on forest type and site conditions and can range from fractions of an acre (i.e., a two-tree group), such as in ponderosa pine or dry mixed-conifer forests, to many acres, as is common in wet mixed-conifer and spruce fir forests. Trees within groups are typically non-uniformly spaced, some of which may be tightly clumped.

Group cutting (selection) is the removed of small groups of trees to establish of new age classes (Helms 1998).

Improvement harvests involve the removal of poorly formed or low-vigor trees to improve stand productivity and/or quality (Helms 1998).

Interspaces are areas not currently under the vertical projection of the outermost perimeter of tree canopies (Fig. 8). They are generally composed of grass-forb-shrub communities but could also be areas with scattered rock or exposed mineral soil. Interspaces do not include meadows, grasslands, rock outcroppings, and wetlands (i.e., exclusions adjacent to and sometimes within forested landscapes).

Leave trees or snags (*see* residual (leave) trees or snags)

Matrix refers to the background cover type of an area. In frequent-fire forests, grass-forb-shrub communities form the background matrix upon which tree groups and individual trees are spatially arranged. It is the most extensive and connected landscape element that plays the dominant role in landscape functioning. The expression of this matrix between tree groups and individual trees is referred to as interspace. The location of tree groups and individual trees on the matrix and the proportion of patches represented by the matrix will change over time due to disturbance.

Mixed-severity fire regimes are characterized by closely juxtaposed forest patches affected by low- and high-severity burning (Fulé and others 2003).

Natural (historical, characteristic) range of variation describes the variability of ecological conditions (e.g., reference compositional and structural conditions) and the spatial and temporal variation in these conditions during a period of time specified to represent characteristic conditions (i.e., conditions relatively unaffected by people) for an ecosystem in a specific geographical area (Kaufmann and others 1994; Landres and others 1999).

Old growth in Southwestern forested ecosystems is defined differently than the traditional definition based on Northwestern infrequent-fire forests. Due to large differences among Southwest forest types and their characteristic disturbances, old growth forests vary extensively in tree size, age classes, presence and abundance of structural elements, stability, and presence of understory. Important structural features of old growth in frequent-fire forests are large trees, old trees, age variability, snags, large dead and downed fuels, and between-patch structural variability (Fig. 9 and Table 10) (Kaufmann and others 2007).

Openness is estimated as the inverse of forest canopy cover for a given area. For example, a forest with 70 percent canopy cover would have openness of 30 percent.

Patches are areas larger than tree groups in which the vegetation composition and structure are relatively homogeneous (sensu Forman 1995). Patches can be composed of randomly arranged trees or multiple tree groups, and they can be even-aged or uneven-aged. Patches comprise the mid-scale, ranging in size from 10-1000 acres. Patches and stands are roughly synonymous.

Pattern (*see* spatial pattern)

Plant associations are plant community types based on land management potential, successional patterns, and species composition (Helms 1998).

Ponderosa pine forests are widespread in the Southwest occurring at elevations ranging from 6000-7500 ft and occupying warmer and drier sites within the montane forest life zone. These forests are characterized by a relatively frequent historic fire regime resulting in surface fire effects. Ponderosa pine is the dominant tree species in this forest type, but other tree species may be present, including Gambel oak, pinyon pine, and juniper species. This forest type often has a shrubby understory mixed with grasses and forbs but sometimes occurs as savannah with extensive grasslands interspersed between widely spaced clumps or individual trees. The ponderosa pine type is distinguished from dry mixed-conifer types by the plant community successional stages. The ponderosa pine forest type is dominated at all successional stages from seral to climax by ponderosa pine. Ponderosa pine often dominates early seral stages of dry mixed-conifer forests also, but these types are not considered to be ponderosa pine forest types because the climax species composition is dominated by other conifer species or ponderosa pine in mixtures with other conifer species.

Processes (ecological processes) are the dynamic attributes of ecosystems in terms of matter and energy, including interactions among organisms and interactions between organisms and their environment (De Groot and others 2002; SER 2004). Examples of processes are: evolution, fire and insect disturbances, photosynthesis, seed dispersal, decomposition, and soil formation.

Reference conditions are conditions existing prior to the suppression or exclusion of the primary processes and mechanisms influencing a system along a natural trajectory (sensu Kaufmann and others 1994). The reference can consist of one or several specified locations that contain model ecosystems, a written description, or a combination of both. Information collected on the reference includes both biotic and abiotic components (SER 2004)

Regeneration sites are tree-free areas created by group cutting for the purpose of establishing tree regeneration.

Residual (leave) trees or snags are those remaining after an intermediate or partial cutting of a stand (Helms 1998).

Resilience (*see* ecological resiliency)

Resiliency (*see* ecological resiliency)

Restoration (*see* ecological restoration)

Sanitation harvests involve the removal of trees to improve stand health by stopping or reducing the actual or anticipated spread of insects and disease (Helms 1998).

Safe zones (fire-free zones) are microsites where seedlings can establish and grow above the lethal flaming zone. Safe zones can be created by fire, such as the ash bed of a consumed log.

Single tree selection cutting is removal of individual trees of all size classes more or less uniformly throughout the stand to promote growth of remaining trees and to provide space for regeneration (Helms 1998).

Site index is an indicator of site quality expressed in terms of the average height of trees (defined as a certain number of dominants, codominants, or the largest and tallest trees per unit area) of a given species at a specified index or base age (Helms 1998).

Snags are standing dead or partially dead trees (snag-topped), often missing many or all limbs. They provide essential wildlife habitat for many species and are important for forest ecosystem function (Fig. 12).

Spatial pattern is the spatial arrangement of elements at the fine-, mid-, and landscape-scales that determine the function of a landscape as an ecological system (adapted from Helms 1998).

Stand density index is a widely used measure that expresses relative stand density based on some standard condition such as the relationship of number of trees to the stand quadratic mean diameter (Helms 1998) or the biological maximum density for a specific species (Long 1985).

Stands are areas in which the biophysical site conditions and the vegetation composition and structure are relatively homogeneous. Stands comprise the mid-scale, thus ranging in size from 100-1000 acres. Stands and patches are roughly synonymous

Structure is the physiognomy or architecture of an ecosystem with respect to the density, horizontal stratification, spatial pattern, and frequency distribution of vegetation (i.e., overstory, understory, etc.) size, age, and/or life form (adapted from SER 2004).

Surface fuel includes all fuels lying on or near the surface of the ground, consisting of leaf and needle litter, dead branch material, downed logs, bark, tree cones, and low stature living and dead plants (adapted from NWCG 2012).

Sustainability (*see* ecosystem sustainability)

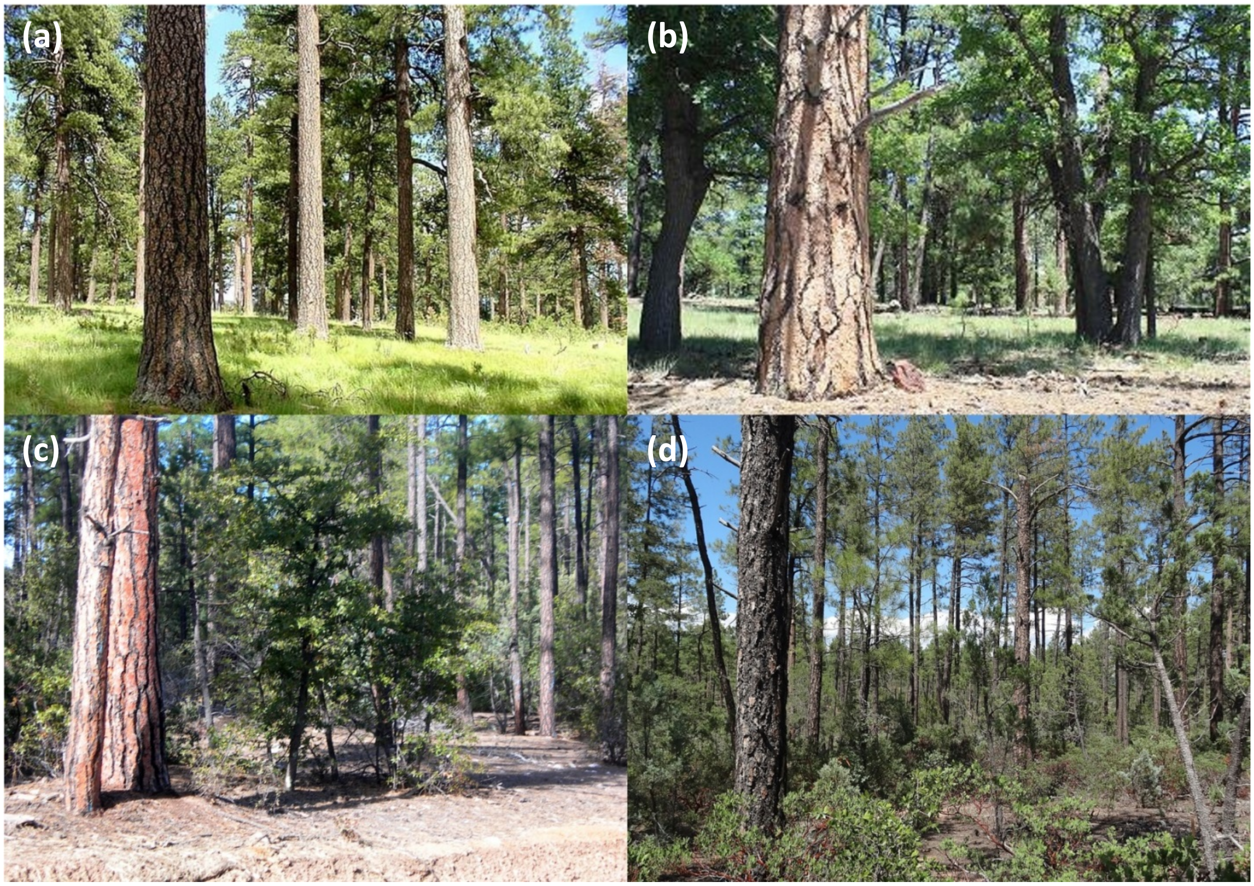
Uneven-aged forests are forests that are comprised of three or more distinct age classes of trees, either intimately mixed or in small groups (Fig. 18) (Helms 1998).

Appendix 1. Common and Scientific Names for Species Referenced in This Document.

Common name	Scientific name
Tree species	
Arizona walnut	*Juglans major*
Arizona white oak	*Quercus arizonica*
Bigtooth maple	*Acer grandidentatum*
Blue spruce	*Picea pungens*
Bristlecone pine	*Pinus aristata*
Chihuahua pine	*Pinus leiophylla*
Corkbark fir	*Abies lasiocarpa* var. *arizonica*
Douglas-fir	*Pseudotsuga menziesii* var. *glauca*
Emory oak	*Quercus emoryi*
Evergreen oaks	*Quercus* spp.
Gamble oak	*Quercus gambelii*
Grey oak	*Quercus grisea*
Junipers	*Juniperus* spp.
Limber pine	*Pinus flexilis*
Pinyon pines	*Pinus* spp.
Ponderosa pine	*Pinus ponderosa*
Quaking aspen	*Populus tremuloides*
Silverleaf oak	*Quercus hypoleucoides*
Southwest white pine	*Pinus strobiformis*
Subalpine fir	*Abies lasiocarpa*
Two-needle pinyon	*Pinus edulis*
White fir	*Abies concolor*
Shrub species	
Big sagebrush	*Artemisia tridentata*
Black sagebrush	*Artemisia nova*
Ceanothus	*Ceanothus* spp.
Common juniper	*Juniperus communis*
Creeping barberry	*Mahonia repens*
Currant	*Ribes* spp.
Kinnikinnik	*Arctostaphylos uva-ursi*
Manzanita	*Arctostaphylos* spp.
Mountain mahogany	*Cercocarpus montanus*
Mountain ninebark	*Physocarpus monogynus*
Mountain snowberry	*Symphoricarpos oreophilus*
Netleaf oak	*Quercus rugosa*
New Mexico locust	*Robinia neomexicana*
Pointleaf manzanita	*Arctostaphylos pungens*
Rockspirea	*Holodiscus dumosus*
Shrub live oak	*Quercus turbinella*

Stansbury cliffrose	*Purshia stansburiana*
Sumac	*Rhus* spp.
Wavyleaf oak	*Quercus undulata*
Grass and sedge species	
Arizona fescue	*Festuca arizonica*
Blue grama	*Bouteloua gracilis*
Dryspike sedge	*Carex siccata*
Fringed brome	*Bromus ciliatus*
Indian ricegrass	*Achnatherum hymenoides*
Longtongue muhly	*Muhlenbergia longiligula*
Mountain muhly	*Muhlenbergia montana*
Muttongrass	*Poa fendleriana*
Parry's oatgrass	*Danthonia parryi*
Screwleaf muhly	*Muhlenbergia virescens*
Forb species	
Forest fleabane	*Erigeron eximius*
Nevada pea	*Lathyrus lanszwertii*
Parasitic plant species	
Douglas-fir dwarf mistletoe	*Arceuthobium douglasii*
Southwestern (Ponderosa pine) dwarf mistletoe	*Arceuthobium vaginatum* subsp. *cryptopodum*
Fungus species	
Armillaria root disease	*Armillaria* spp.
Black stain root disease	*Leptographium* spp.
Insect species	
Bark beetles	*Dendroctonus* spp. and *Ips* spp.
Douglas-fir tussock moth	*Orgyia pseudotsugata*
Roundheaded pine beetle	*Dendroctonus adjunctus*
Spruce budworm	*Choristoneura occidentalis*
Mammal species	
Ground squirrels	*Callospermophilus* spp.
Coyote	*Canis latrans*
Tassel-eared squirrel	*Sciurus aberti*
Hares	*Lepus* spp.
Bobcat	*Lynx rufus*
Rabbits	*Sylvilagus* spp.
Bird species	
Northern goshawk	*Accipiter gentilis*
Merriam's turkey	*Meleagris gallopavo* var. *merriami*

Appendix 2. Major Ponderosa Pine Forest Subtypes: (a) Ponderosa Pine/Bunchgrass, (b) Ponderosa Pine/Gambel Oak, (c) Ponderosa Pine/Evergreen Oak, and (d) Ponderosa Pine/Evergreen Shrub.



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Planning for Connectivity

A guide to connecting and conserving wildlife within and beyond America's national forests

Defenders of Wildlife

The Center for Large Landscape Conservation

Wildlands Network — Reconnecting Nature in North America

Yellowstone to Yukon Conservation Initiative

ACKNOWLEDGEMENTS

Planning for Connectivity is a product of The Center for Large Landscape Conservation, Defenders of Wildlife, Wildlands Network and Yellowstone to Yukon Conservation Initiative. This guide focuses on requirements established under the National Forest System land management planning rule to manage for ecological connectivity on national forest lands and facilitate connectivity on planning acrosss land ownerships. The purpose of the guide and its parent publication, *Planning for Diversity*, is to help people inside and outside of the Forest Service who are working on forest plan revisions navigate these complex diversity and connectivity requirements.

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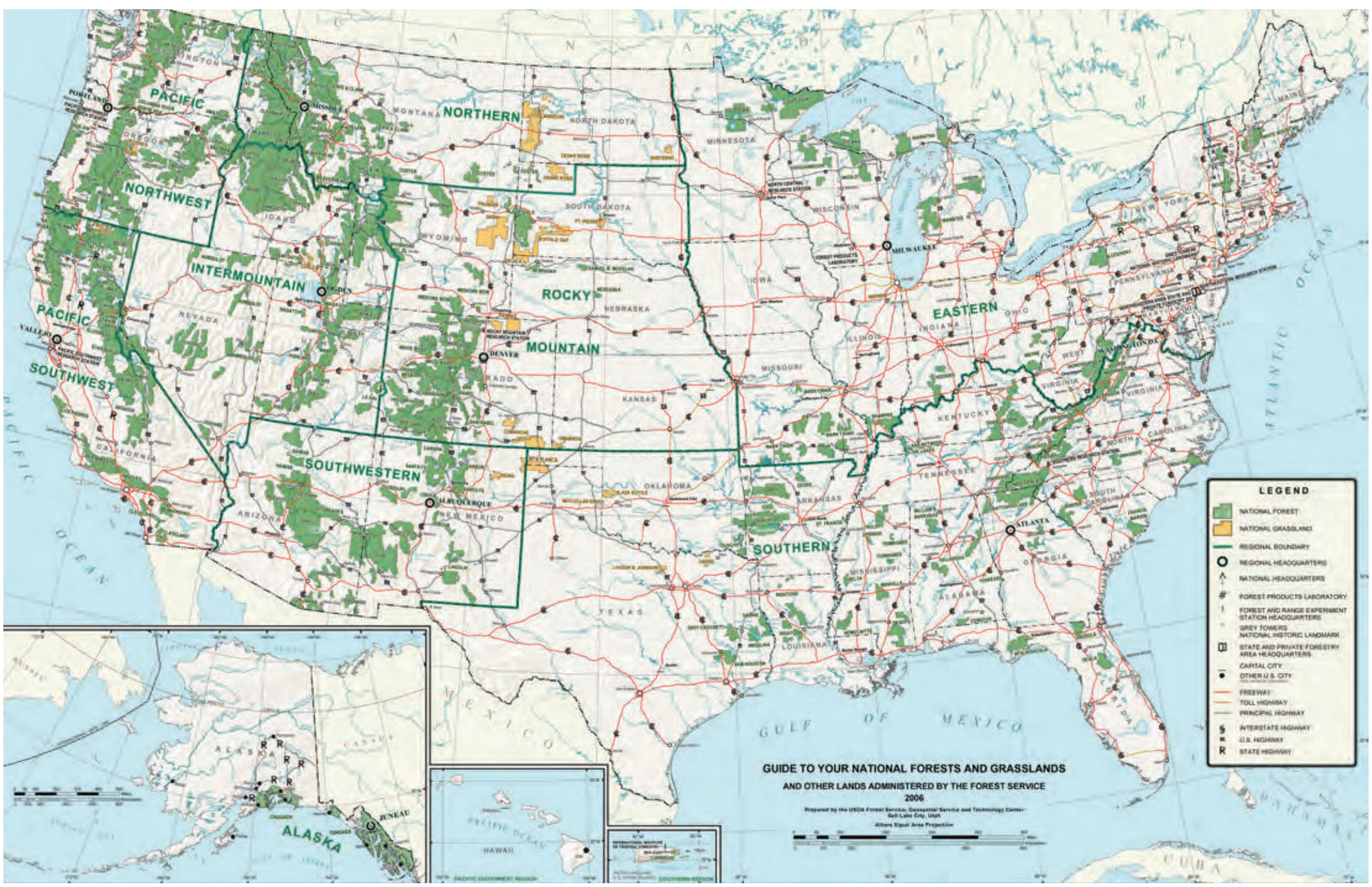
INTRODUCTION

The United States Forest Service manages more than 193 million acres—over 8 percent of all U.S. lands—an area about the size of Texas and twice the size of the National Park System. The National Forest System comprises 154 national forests and 20 national grasslands and one national prairie (collectively referred to as "national forests" in this guide). Located in 42 states, Puerto Rico and the U.S. Virgin Islands, these public lands are essential to the conservation of wildlife habitat and diversity. National forests encompass three-quarters of the major U.S. terrestrial and wetland habitat types—including alpine tundra, tropical rainforest, deciduous and evergreen forests, native grasslands, wetlands, streams, lakes and marshes. This variety of ecosystems supports more than 420 animals and plants listed under the Endangered Species Act (ESA) and an additional 3,250 other at-risk species.

To guide the management of each national forest, the Forest Service is required by law to prepare a land management plan (forest plan). Forest plans detail strategies to protect habitat and balance multiple uses to ensure the persistence of wildlife, including at-risk and federally protected species.

In April 2012, the Forest Service finalized regulations implementing the National Forest Management Act (NFMA). These regulations, commonly referred to as the "2012 Planning Rule" established a process for developing and updating forest plans and set conservation requirements that forest plans must meet to sustain and restore

The National Forest System



the diversity of ecosystems, plant and animal communities and at-risk species found on these public lands (36 C.F.R. §§ 219.1-219.19, abbreviated throughout this report by omitting "36 C.F.R. §").

The forest planning rule includes explicit requirements for managing for ecological connectivity on national forest lands and facilitating connectivity planning across land ownerships—the first such requirements in the history of U. S. public land management. The pending revisions of most forest plans provide a significant opportunity to protect and enhance the diversity of habitat and wildlife on national forest lands by developing forest plans that promote the conservation and restoration of ecological connectivity.

This guide is designed to help people, working within and outside of the Forest Service, develop effective connectivity conservation strategies in forest plans developed under the 2012 Planning Rule. It summarizes the role of connectivity within the conservation framework of the rule and offers guidance and examples of how to conduct connectivity planning in the land management planning process.

The guide is a collaboration of Defenders of Wildlife, The Center for Large Landscape Conservation, Wildlands Network and Yellowstone to Yukon Conservation Initiative and is our collective interpretation of the connectivity requirements of the 2012 Planning Rule. The guide is intended to add value to official agency policies developed to support implementation of the rule. In January 2015, the Forest Service published Final Agency Directives for Implementation of the 2012 Planning Rule (FSM 1900 Planning, FSH 1909.12). While this guide and those directives may in some cases describe different approaches to implementing the connectivity requirements of the planning rule, we believe our interpretations are consistent with the planning rule and NFMA and hope the guide is viewed as a useful companion set of recommendations from the perspective of conservation organizations experienced in national forest planning, connectivity science and policy.

The guide covers the unique connectivity aspects of the planning rule, a rule that addresses complex ecosystem and species conservation processes and has many specific requirements.

How to Use This Guide

Planning for Connectivity presents guidance and best practices for connectivity planning, including examples from case studies in forest planning. Resources associated with the case studies are listed in the references section. We suggest using this guide in tandem with *Planning for Diversity,* a companion publication that addresses the overarching conservation framework of the 2012 Planning Rule. *Planning for Diversity,* additional resources on diversity and connectivity science and planning and a collection of forest planning case studies are available online at www.defenders.org/forestplanning.

THE IMPORTANCE OF CONNECTIVITY

It is useful to think of connectivity contributing to both the structure and function of ecosystems and landscapes. Structural connectivity is the physical relationship between patches of habitat or other ecological units; functional connectivity is the degree to which landscapes actually facilitate or impede the movement of organisms and processes of ecosystems (Ament et al. 2014).

The structure or pattern of an ecosystem or landscape can be defined as the arrangement, connectivity, composition, size and relative abundance of patches that occur within an area of land at a given time. Patches are surface areas that differ from their surroundings in nature or appearance (Turner et al. 2001). They can be characterized by vegetation type, seral stage, habitat type or other features relevant to a species and also by the condition of surrounding lands, which can significantly affect the biological character of a habitat patch.

Fragmentation, the breaking up of habitat or cover type into smaller disconnected patches (Turner et al. 2001), may result from natural or anthropogenic disturbances that introduce barriers to connectivity. In natural landscapes, patches that differ from the surrounding area would likely be areas disturbed by fire, flood, blowdown or other natural processes. In managed landscapes, habitat or cover can be fragmented by human caused disturbances such as road-building or removal of vegetation. In natural and managed fragmented landscapes, patches can be thought of as the remaining undisturbed areas. The greatest conservation needs are usually associated with maintaining or restoring connectivity among patches.

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The arrangement of patches of vegetation defines the pattern of a landscape like this one in Medicine Bow National Forest.

Other terms related to connectivity and wildlife movements include (Ament et al 2014):

- **Corridor.** A distinct component of the landscape that provides connectivity (think of it as a linear patch).
- **Linkage area or zone**. Broader regions of connectivity important to maintain ecological processes and facilitate the movement of multiple species.
- **Permeability.** The degree to which landscapes are conducive to wildlife movement and sustain ecological processes.

The 2012 Planning Rule defines connectivity as:

> *Ecological conditions that exist at several spatial and temporal scales that provide landscape linkages that permit the exchange of flow, sediments, and nutrients; the daily and seasonal movements of animals within home ranges; the dispersal and genetic interchange between populations; and the long distance range shifts of species, such as in response to climate change* (219.19).

The planning rule definition reflects both structural and functional aspects of connectivity. The rule's reference to spatial scales and "landscape linkages" suggests a structure of connected patches and ecosystems. Functional connectivity is also part of the definition: water flows, sediment exchange, nutrient cycling, animal movement/dispersal, species climate adaptation and genetic interchange are all ecological processes that are sustained by connectivity.

Any comprehensive strategy for conserving biological diversity requires maintaining habitat across a variety of spatial scales and includes the maintenance of connectivity, landscape heterogeneity and structural complexity (Lindenmayer and Franklin 2002). Connectivity is especially important for enabling adaptation to changing stressors, including climate change. The challenge of climate change was a driving factor in the development of the 2012 Planning Rule (77 Fed. Reg. 21163). A review of 22 years of recommendations for managing biodiversity in the face of climate change found improving landscape connectivity is the most frequently recommended strategy for allowing biodiversity to adapt to new conditions (Heller and Zaveleta 2009).

Wildlife species are becoming increasingly isolated in patches of habitat surrounded by a human-dominated landscape. Exacerbating this fragmentation is the effect of exurban development that continues to encroach on Forest Service lands (Hansen et al. 2005; Stein et al. 2007). The distribution of many wildlife populations continues to shrink as a result. Aquatic and terrestrial landscape patterns have been substantially altered, reducing or eliminating ecological connectivity for many wildlife populations. Physical barriers with human development further reduce connectivity. Changes in habitat, such as the simplification of complex forest vegetation, can also make critical areas for movement less permeable to some species. Scientists recognize that preserving or enhancing connectivity can be a practical tool for conserving biodiversity in such circumstances (Worboys et al. 2010).

THE 2012 FOREST PLANNING RULE

The 2012 Planning Rule is a federal regulation implementing NFMA (1600 U.S.C. § 1600 et seq.). NFMA was enacted in 1976 in large part to elevate the value of ecosystems, habitat and wildlife on our national forests to the same level as timber harvest and other uses. NFMA codified an important national priority: forest plans must provide for the diversity of habitat and animals found on national forests.

NFMA established a process for integrating the needs of wildlife with other multiple uses in forest plans. Most importantly, the law set a substantive threshold Forest Service actions must comply with for sustaining the diversity of ecosystems, habitats, plants and animals on national forests. However, the law gave discretion to the Forest Service, through the development of forest planning regulations and forest plans, to define that threshold.

THE PLANNING PROCESS

According to NFMA, forest plans are to be revised on a 15-year cycle. The planning rule provides a process for developing, revising or amending plans that is adaptive and science-based, engages the public and is designed to be efficient, effective and within the agency's ability to implement (77 Fed. Reg. 21162).

The planning rule establishes a three-phase process:

1. **Assessment**. The assessment identifies and evaluates information relevant to the development of a forest plan. The assessment is used during plan revision to evaluate what needs to change in the current plan, including what is needed to meet the requirements of the planning rule.
2. **Development.** During the plan development stage, the Forest Service develops and finalizes the forest plan and plan monitoring program. A draft proposal is developed and management alternatives are evaluated through the process established by the National Environmental Policy Act (42 U.S.C. § 4321 et seq.).
3. **Implementation/monitoring.** After finalizing the forest plan, the agency begins to implement the plan, including the development and implementation of management projects. Projects must be consistent with the forest plan and implementation of the plan must be evaluated through a monitoring program. Monitoring information is then evaluated to determine if aspects of the forest plan should be changed.

In addition, the Forest Service must use the best available scientific information to inform the planning process (219.3) throughout all three phases.

The planning rule describes these phases as iterative, complementary and sometimes overlapping. The intent is to provide a planning framework that is responsive to new information and changing conditions.

FOREST PLAN COMPONENTS

Forest plans guide subsequent project and activity decisions, which must be consistent with the forest plan. Forest plans do this through the use of plan components, the basic building blocks of forest plans. Plan components (Table 1) shape implementation of the forest plan and are the means of meeting the requirements of the 2012 Planning Rule.

Two fundamental types of plan components are associated with the diversity requirements of the rule: landscape components and project components.

Landscape components relate to the vision and priorities for the plan area, a landscape larger than individual project areas. These components are outcome-oriented, describe how the Forest Service would like the plan area to look and function and include desired conditions and objectives. Projects to be initiated under the forest plan are designed to contribute to achieving one or more of these outcomes. It is important that desired conditions and objectives be specific enough to establish a purpose and need for the projects designed to help achieve them.

Project components pertain to how individual projects are designed and implemented under the forest plan. They include standards, guidelines and suitability determinations that prohibit specific uses. They can preclude or regulate particular management options, dictate the outcome specifications for project areas or establish procedures

Table 1. Plan components under the 2012 Planning Rule

Plan Component	Description (219.7(e))
Desired Conditions (Landscale-level)	A description of specific social, economic and/or ecological characteristics of the plan area (or a portion of the plan area) toward which management of the land and resources should be directed. Desired conditions must be described in terms specific enough to allow progress toward their achievement to be determined, but do not include completion dates.
Objectives (Landscape-level)	A concise, measurable and time-specific statement of a desired rate of progress toward a desired condition or conditions. Objectives should be based on reasonably foreseeable budgets.
Standards (Project-level)	A mandatory constraint on project and activity decision-making established to help achieve or maintain the desired condition or conditions, to avoid or mitigate undesirable effects or to meet applicable legal requirements.
Guidelines (Project-level)	A constraint on project and activity decision-making that allows for departure from its terms as long as the purpose of the guideline is met. Guidelines are established to help achieve or maintain a desired condition or conditions, to avoid or mitigate undesirable effects or to meet applicable legal requirements.
Suitability of Lands (Project-level)	Specific lands within a plan area are identified as suitable for various multiple uses or activities based on the desired conditions applicable to those lands. The plan also identifies lands within the plan area as not suitable for uses that are not compatible with desired conditions for those lands.

that must be followed in preparing projects. It is very important to note that project plan components—especially standards—are most useful when greater certainty is important, such as in meeting diversity requirements necessary to protect at-risk species. Under the planning rule, every action proposed on Forest Service lands must comply with standards and guidelines and may not occur on lands unsuitable for that action.

DIVERSITY

NFMA requires that the Forest Service's planning regulations "provide for diversity of plant and animal communities based on the suitability and capability of the specific land area in order to meet overall multiple-use objectives" (16 U.S.C. § 1604(g)(3)(B)). This diversity requirement has been interpreted by the agency in the NFMA planning regulations and by the courts.

The Forest Service has interpreted the diversity requirement in NFMA through the development of the 2012 Planning Rule, which offers an approach to meeting the diversity requirement described in more detail in the following section on the ecosystem-species approach. A pivotal piece of the diversity interpretation is the persistence of individual species on national forest lands. Maintaining viable populations of native species is the scientifically accepted method of achieving the conceptual goal of maintaining species diversity. According to a 1999 Committee of Scientists report commissioned for the purposes of forest planning, "[d]iversity is sustained only when individual species persist; the goals of ensuring species viability and providing for diversity are inseparable" (Committee of Scientists 1999: 38).

The federal judiciary's interpretation of the diversity requirement in the rule include a ruling that the NFMA diversity mandate not only imposes a substantive standard on the Forest Service, it "confirms the Forest Service's duty to protect [all] wildlife" (*Seattle Audubon Society v. Moseley*, 1489). Courts have also recognized that the Forest Service's "statutory duty clearly requires protection of the entire biological community" (*Sierra Club v. Espy*, 364).

THE ECOSYSTEM-SPECIES APPROACH

Three overarching substantive requirements (Table 2) in the planning rule pertain to NFMA's diversity requirement:

1. Maintain or restore the ecological integrity of terrestrial and aquatic ecosystems (219.9(a)).
2. Maintain or restore the diversity of ecosystems and habitat types (219.9(a)).
3. Provide the ecological conditions necessary for at-risk species (219.9(b)).

The fundamental premise of the planning rule for meeting the NFMA diversity requirement is that plan components for ecosystem integrity and diversity will provide the ecological conditions to both maintain the diversity of plant and animal communities and support the persistence of most (but not all) native species in a

Table 2. Ecological concepts and requirements of the 2012 Planning Rule[1]

Ecological Concept	Definition and Requirement from the Planning Rule (219.9, if applicable)
Ecosystem	*Definition:* A spatially explicit, relatively homogeneous unit of the Earth that includes all interacting organisms and elements of the abiotic environment within its boundaries. An ecosystem is commonly described in terms of its composition, structure, function and **connectivity**. *Requirement:* The plan must include plan components, including standards or guidelines, to maintain or restore the diversity of ecosystems and habitat types throughout the plan area. In doing so, the plan must include plan components to maintain or restore key characteristics associated with terrestrial and aquatic ecosystem types, rare aquatic and terrestrial plant and animal communities, and the diversity of native tree species similar to that existing in the plan area.
Ecological Integrity	*Definition:* The quality or condition of an ecosystem when its dominant ecological characteristics (e.g., composition, structure, function, **connectivity**, species composition and diversity) occur within the natural range of variation and can withstand and recover from most perturbations imposed by natural environmental dynamics or human influence. *Requirement:* The plan must include plan components, including standards or guidelines, to maintain or restore the ecological integrity of terrestrial and aquatic ecosystems and watersheds in the plan area, including plan components to maintain or restore their structure, function, composition and **connectivity**.
At-risk Species ▪ Threatened and Endangered ▪ Candidate and Proposed ▪ Species of Conservation Concern	*Definition:* Threatened and endangered species are federally listed under the ESA; proposed and candidate species have been either formally proposed or are being formally considered for listing under the ESA. Species of conservation concern are species for which the regional forester has determined that the best available science indicates substantial concern over the species' capability to persist over the long-term in the plan area. *Requirement:* The responsible official shall determine whether or not the (ecosystem) plan components provide the ecological conditions necessary to contribute to the recovery of federally listed threatened and endangered species, conserve proposed and candidate species, and maintain a viable population of each species of conservation concern within the plan area. If the responsible official determines that the (ecosystem) plan components are insufficient to provide such ecological conditions, then additional, species-specific plan components, including standards or guidelines, must be included in the plan to provide such ecological conditions in the plan area.
Ecological Conditions	*Definition:* The biological and physical environment that can affect the diversity of plant and animal communities, the persistence of native species and the productive capacity of ecological systems. Ecological conditions include habitat and other influences on species and the environment, e.g., the abundance and distribution of aquatic and terrestrial habitats, **connectivity**, roads and other structural developments, human uses and invasive species.
Viable Population	*Definition:* A population of a species that continues to persist over the long term with sufficient distribution to be resilient and adaptable to stressors and likely future environments.
Focal Species	*Definition:* A small subset of species whose status permits inference to the integrity of the larger ecological system to which it belongs and provides meaningful information regarding the effectiveness of the plan in maintaining or restoring the ecological conditions to maintain the diversity of plant and animal communities in the plan area. Focal species would be commonly selected on the basis of their functional role in ecosystems.

plan area (219.9). To meet the rule's requirements for at-risk species (which include federally listed threatened and endangered species, proposed and candidate species, and species of concern (SCC)), additional "species-specific" plan components may be necessary. The rule's two-tiered conservation approach (alternatively called the "ecosystem-species" or "coarse-fine filter" planning method) relies on the use of surrogate measures, or key characteristics,

1. Ecological "conditions" are defined broadly to include human structures and uses, while "ecological integrity" stresses dominant "characteristics" that suggest natural conditions and should not include human structures and uses. The term "key ecosystem characteristics" is commonly used in discussions of ecological integrity, but should not be understood to apply to human structures and uses in that context. Human structures and uses are nevertheless relevant to species viability and persistence, and therefore to diversity.

Connectivity is an ecological condition that pronghorn and other species need to persist within and beyond the boundaries of national forests and grasslands.

to represent the condition of ecosystems, as well as the identification of at-risk species and evaluation of whether those species will be sustained through ecosystem-level plan components, or whether they require specific management attention in the form of species-level plan components.

At the ecosystem scale, the rule requires forest plans to have plan components to maintain or restore the integrity of terrestrial and aquatic ecosystems in the plan area (219.9(a)(1)) and the diversity of ecosystems and habitat types (219.9(a)(2)). Essentially this requires forest plans to maintain or restore the variety of ecosystems and habitat types found on the forests (e.g., conifer forests, wetlands, grasslands), as well as the condition of the ecosystems themselves. If the ecosystem-scale plan components are not sufficient to provide ecological conditions (i.e., meet the conservation needs) for at-risk species, additional plan components to do so are required (219.9(b)(1)). In some cases, the Forest Service may determine that it is beyond its authority or "not within the inherent capability of the plan area" to provide those conservation conditions and thus other requirements apply (219.9(b)(2)).

Connectivity plays a key role in the rule's conservation approach (see Table 2). As a key characteristic of ecosystems, connectivity should be addressed through ecosystem-scale plan components in order to maintain or restore "ecological integrity." Connectivity may also be an "ecological condition" needed by individual species, and so forest plans may need to address connectivity at the species level. For example, a recent amendment to forest plans in Wyoming protects migration corridors between seasonal habitats for pronghorn (Ament et al. 2014).

The rule's approach to conservation planning relies on the use of key characteristics in assessments, planning and monitoring to represent the condition of ecosystems, as well as the identification of at-risk species, some of which may require connectivity conditions to persist. It will be necessary for forest plans to identify key characteristics of ecosystem connectivity, as well as structure, function and composition (Table 3).

The concept of ecological integrity is used to represent the status of an ecosystem. An ecosystem is considered to have integrity when its key ecosystem characteristics occur within the natural range of variation (NRV) (219.19). NRV can be thought of as a reference condition reflecting "natural" conditions. Those conditions can be estimated using information from historical reference ecosystems or by other science-based methods. For example, many current forest ecosystems exhibit landscape connectivity patterns that differ from historical or reference conditions. For the purpose of sustaining ecosystems and wildlife, the 2012 Planning Rule directs the Forest Service to manage key characteristics of ecosystems, including their connectivity characteristics, in light of these reference conditions.

It is therefore important that forest plans have plan components, including desired conditions, to move landscapes toward a more natural range of connectedness.

ISSUES OF SCALE

The definition of connectivity in the planning rule intends for it to be provided at appropriate ecological scales. Strategies for managing connectivity in forest plans will vary based on the relevant species and their particular requirements for connectivity. The planning process must consider the habitat needs of target species and the nature of their movements. Forest plans should provide for habitat connectivity to address localized movements, as well as landscape-scale linkages between larger blocks of habitat.

Land managers must look at the broader landscape context when addressing connectivity in forest plans (219.8(a)(1)). They should consider what they are connecting and be alert to connecting specific watersheds or other geographic areas identified as being relatively more important for a particular species. Aquatic species provide a good example of large-scale connectivity needs because the existence of a connected network of aquatic ecosystems is known to be critically important to migratory aquatic species, especially when disturbances occur.

SCOTT MCREYNOLDS/CALIFORNIA DEPARTMENT OF WATER RESOURCES

Chinook salmon and other migratory fishes need a connected network of aquatic ecosystems to survive. Forest plans must consider the large-scale connectivity needs of these species.

For many species, persistence within a national forest depends on connectivity that extends beyond forest boundaries. While the Forest Service has no authority to regulate land uses outside national forests, it can influence conservation on adjacent lands by how it chooses to manage its own lands. A forest plan should consider

Table 3. The use of key characteristics in forest planning

Ecosystem Character	Definition (219.19)	Examples of Key Characteristics
Connectivity	Ecological conditions that exist at several spatial and temporal scales that provide landscape linkages that permit the exchange of flow, sediments and nutrients; the daily and seasonal movements of animals within home ranges; the dispersal and genetic interchange between populations; and the long-distance range shifts of species, such as in response to climate change.	Structural: size, number and spatial relationship between habitat patches, mapped landscape linkages and corridors. Functional: measure of ability of native species to move throughout the planning area and cross into adjacent areas.
Composition	The biological elements within the different levels of biological organization, from genes and species to communities and ecosystems.	A description of major vegetation types, patches, habitat types, soil types, landforms and wildlife populations.
Structure	The organization and physical arrangement of biological elements such as snags and down woody debris, vertical and horizontal distribution of vegetation, stream habitat complexity, landscape pattern and connectivity.	Arrangement of patches within a landscape, habitat types within a forest, trees within a forest stand, wildlife within a planning area.
Function	Ecological processes that sustain composition and structure such as energy flow, nutrient cycling and retention; soil development and retention; predation and herbivory; and natural disturbances such as wind, fire and floods.	Types, frequencies, severities, patch sizes, extent and spatial pattern of disturbances such as fires, landslides, floods and insect and disease outbreaks.

connectivity when prioritizing lands for acquisition or conservation easements on adjacent ownerships. At a finer scale, a forest plan's requirements for size and arrangement of patch characteristics may be sufficient to produce an appropriately structured landscape for connectivity.

CONNECTIVITY INFORMATION

The scientific literature includes many connectivity and corridor studies and analyses. Peer-reviewed connectivity information pertaining to all regions of the country is readily available to inform national forest planning. In recent years, the Forest Service Research and Development Branch itself has produced numerous materials on various aspects of connectivity that can be used to support analyses of conditions, trends and sustainability. The available literature includes general publications about the science of connectivity and research on specific locations and/or species.[2] Examples include Cushman and others' analysis of corridors (2012) and McKelvey and others' (2011) identification of wolverine corridors.

Independent analyses of connectivity are also now available for many areas. The nationwide system of Landscape Conservation Cooperatives (LCC) has prioritized managing for connectivity across the country. For example, the South Atlantic LCC is completing a project titled "Identifying and Prioritizing Key Habitat Connectivity Areas for the South Atlantic Region." The Western Governors Association spearheaded the development of databases and mapping systems in the western states to identify important habitat and corridors region-wide.

The planning rule also cites other governmental management plans as sources of information to consider in assessing and planning for connectivity (219.6(a)(1)). It is critical that forest plans take into account land uses on adjacent lands and the importance of such lands to connectivity. The Forest Service should engage with highway departments, state wildlife agencies, tribal governments and county planning organizations that might affect connectivity on adjacent or intervening landscapes. These entities may have identified potential corridors that should be recognized in the forest planning process.

CONNECTIVITY COORDINATION

There is an additional requirement in NFMA that is particularly important to developing plan components for connectivity. It is a procedural requirement that the planning process be "coordinated with the land and resource management planning processes of State and local governments and other Federal agencies" (16 USC § 1604(a)). One of the purposes of the planning rule was to "[e]nsure planning takes place in the context of the larger landscape by taking an 'all-lands approach'" (77 Fed. Reg. 21164).[3] To accomplish this, forest plans should consider how habitat is connected across ownership boundaries.

The planning rule accounts for this type of "all lands" connectivity by:

- Requiring assessments to evaluate conditions, trends and sustainability "in the context of the broader landscape" (219.5(a)(1)).
- Recognizing that sustainability depends in part on how the plan area influences, and is influenced by, "the broader landscape" (219.8(a)(1)(ii), (iii)).
- Requiring coordination with other land managers with authority over lands relevant to populations of species of conservation concern (219.9(b)(2)(ii)).
- Requiring coordination with plans and land-use policies of other jurisdictions (219.4(b)).
- Requiring consideration of opportunities to coordinate with neighboring landowners to link open spaces and take joint management objectives into account (219.10(a)(4)).

Achieving the broader scale "all-lands" goals of the planning rule requires partnerships and compatible management across landscapes among multiple landowners and jurisdictions. In particular, there is a need for a landscape-scale strategic approach to conserving connectivity. NFMA has established that the way to communicate a long-term and reliable management commitment for National Forest System lands is through forest plan decisions for specific areas.

There is a significant commitment to connectivity conservation within Forest Service policy and from many agency partners. Examples of coordinated multi-agency planning efforts that specifically address connectivity and can guide the Forest Service as it seeks to implement the new rule are summarized in Appendix A.

2. Forest Service research publications on the topic may be found by entering the search term "connectivity" at www.treesearch.fs.fed.us/.

3. The planning rule defines landscape as "[a] defined area irrespective of ownership or other artificial boundaries, such as a spatial mosaic of terrestrial and aquatic ecosystems, landforms, and plant communities, repeated in similar form throughout such a defined area" (219.19).

BEST PRACTICES FOR CONNECTIVITY PLANNING

The following sections present guidance and best practices for connectivity planning, including examples from case studies in forest planning. Resources associated with the case studies are listed in the references at the end of the guide. Additional forest planning case studies are available online at www.defenders.org/forestplanning.

ASSESSING CONNECTIVITY

The planning rule requires that assessments be conducted prior to plan revisions to determine what needs to be changed in the existing forest plan, to serve as the basis for developing plan components and to inform a monitoring program. The Forest Service must review all relevant existing information and then determine the best available scientific information about conditions, trends and sustainability for connectivity in relationship to the forest plan within the context of the broader landscape (219.5(a)(1)). The Forest Service must document in the assessment report "how the best available scientific information was used to inform the assessment" (219.6(b)).

For connectivity, the assessment should address both ecosystem and species-level connectivity issues. At the ecosystem-scale, the assessment needs to identify the ecosystems and habitat types within the planning area, and then evaluate the diversity and integrity of those based on information related to their structure, function, composition and connectivity.

We recommend including the following in an assessment of connectivity at the ecosystem level:

- The selection of key characteristics for connectivity (see Table 3, page 10).
- A discussion of the NRV or "reference conditions" for the characteristics (e.g., historical pattern and connectivity).
- An evaluation of system drivers (e.g., climate change) and stressors (e.g., barriers to connectivity) on the characteristics.
- A discussion of the future status of the characteristic under current management and the current plan.

The end result should be a connectivity assessment that can be used to determine:

- How the current plan needs to change to maintain or restore connectivity.
- What plan components may be necessary to achieve the ecosystem-based connectivity requirements in the rule.

Connectivity must also be assessed as a potential condition necessary to sustain individual species. In the assessment, the Forest Service will present information on the ecological needs of species so that plan components can be developed to meet the rule's requirements for species. Particular attention should be paid to the connectivity needs of all at-risk species. To demonstrate that plan components will be effective in maintaining a "viable population" in the plan area, the assessment must provide a means of determining a "sufficient distribution" (see Table 2, page 8). The assessment should describe the relationship between connectivity and the distribution of species necessary for persistence, especially with regard to stressors like climate change. It is important that the assessment evaluate how species move, what barriers to those movements may exist and how the Forest Service can reduce the impact of those barriers within the context of recovery, conservation and viability.

The Flathead National Forest plan revision (assessment, 2014), which is being conducted under the 2012 Planning Rule, offers an example of assessing connectivity needs. The Flathead assessment includes a significant discussion of connectivity for terrestrial habitat, views connectivity from both an ecosystem and species perspective and considers both shorter term vegetation barriers on the forest and longer term human barriers between national forest lands. The example below shows how the Flathead National Forest presented a key ecosystem characteristic, description and data source for connectivity (adapted from Flathead 2014: 103, Table 26):

Key Ecosystem Characteristic: Horizontal Patterns and Landscape Connectivity

Description: The horizontal pattern of forest size/structure classes across the landscape and the spatial linkages between them, which is influenced both by human activities, such as harvesting and development, and natural processes, such as wildland fire.

Data Source for Current Condition: Montana Natural Heritage Program databases; Flathead National Forest VMap; Flathead National Forest NRV analysis.

The assessment provides a description of current and reference (NRV) conditions and expected trends for this key characteristic, as well as an evaluation of the impact of stressors (e.g., from timber harvest and developments) on habitat. The following is a key finding from the assessment:

> *Significant departures from historical conditions in patch sizes and density was noted in the NRV analysis for nearly all forest structural classes forest-wide. This trend mirrored that occurring at the larger Northern Rocky Mountain ecoregion, where drastically increased forest fragmentation was noted. The analysis found a decrease in patch size and corresponding increase in patch density, resulting in a trend of increasing forest fragmentation. The changes were most dramatic for the early successional forest patches and found to be outside the range of historical variability, which is of particular concern to ecological integrity* (Flathead 2014: 137, internal citations omitted).

JOHN JACOBSON/WASHINGTON DEPARTMENT OF FISH AND WILDLIFE

The Flathead National Forest connectivity assessment for the fisher specifies that this at-risk species requires mature forests arranged in connected, complex shapes with few isolated patches.

The Flathead assessment also presented connectivity information for an at-risk species, the fisher. This information can be used to determine the effectiveness of the current plan in providing for habitat connectivity for the species or to develop new plan components:

> *At the scale of 50–100 km2 (12,355–24,710 acre) landscapes, fishers in northern Idaho and west-central Montana selected for home ranges with greater than 50 percent mature forest arranged in connected, complex shapes with few isolated patches, and open areas comprising <5 percent of the landscape. Jones and Garton (1994) stated that preferred habitat patches should be linked by travel corridors of closed canopy forest and that riparian areas make excellent corridors provided they are large enough to enable fishers to avoid predation* (Flathead 2014: 197).

CONNECTIVITY MANAGEMENT AREAS

For connectivity, it is especially important to determine where plan components will apply. While it may be relatively easy to state desired forest-wide conditions related to connectivity, this approach by itself fails to focus efforts on areas with known connectivity values (e.g., roadless areas) and may not effectively promote integration with other uses that can lead to recognition of conflicts.

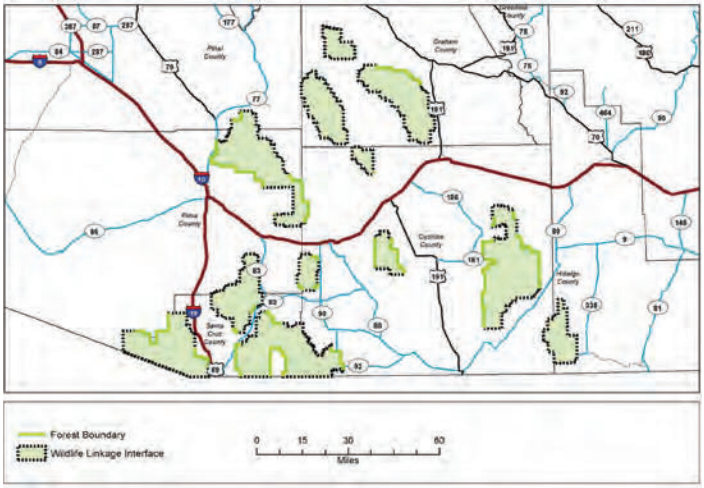
The planning rule states that the plan must indicate to which part of the plan area each plan component applies (219.7(e)). It defines "management areas" as parts of the plan area that have "the same set of applicable plan components" (219.19). Desired conditions and other plan components should be specified for particular linkage areas or corridors where they can be identified and the assessment finds them to be important to the persistence of target species in the plan area. Where connectivity is constrained, it may be necessary to identify specific areas to be managed as patches and their connecting corridors. Identifying specific management area(s) for connectivity provides clear forest plan direction on the importance of these areas and clarity for future projects.

The following case studies are examples of spatially recognizing connectivity in forest planning. An additional example is provided in the section on "Barriers to Connectivity" on page 18.

CASE STUDY: Wildlife Linkages in the Sky Islands

The mountainous "sky islands" of the Coronado National Forest in Arizona are made up of forested ranges separated by valleys of desert and grassland plains. They are among the most diverse ecosystems in the world because of their topographic complexity and location at the convergence

Figure 1. Wildlife linkages on the Coronado National Forest



Source: Coronado 2013: 64, Figure 3

U.S. FISH AND WILDLIFE SERVICE

A remote camera captured this image of an ocelot in the Huachuca Mountains of Arizona, an area where the proliferation of highways has affected connectivity among ocelot populations. To address the problem, the Coronado National Forest plan designated linkage areas on the boundary of the forest to coordinate connectivity management with other jurisdictions.

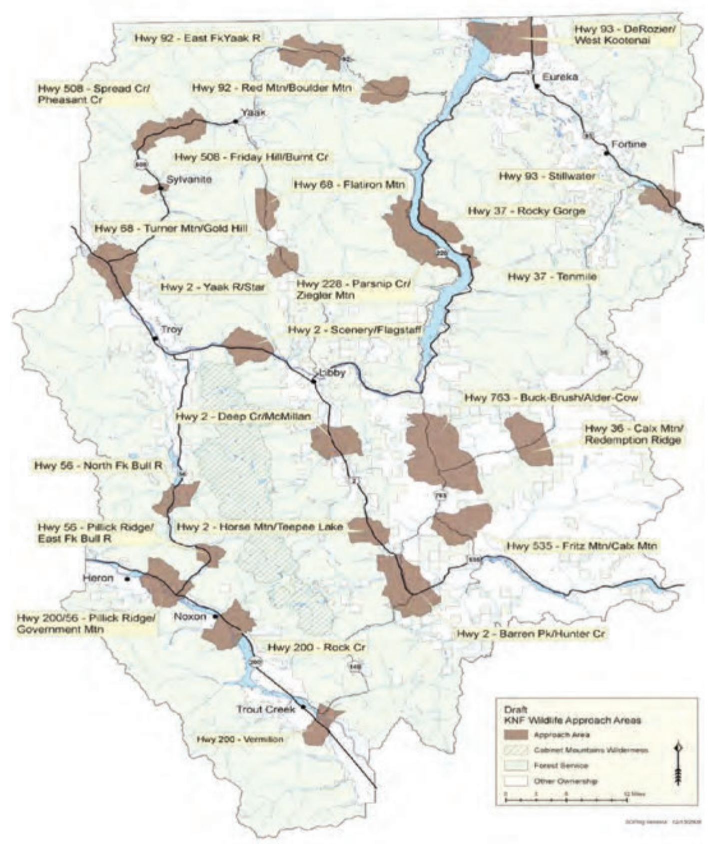
of several major desert and forest biological provinces. The valleys act as barriers to the movement of certain woodland and forest species. Species such as mountain lions and black bears depend on movement corridors between mountain islands to maintain genetic diversity and population size. Ocelots and jaguars at the northern end of their range here depend on connectivity to source populations in Mexico. The proliferation of highways and resulting increase in the number of road deaths among dispersing ocelots has affected connectivity among ocelot populations and colonization of new habitats. Movement corridors for jaguars in the American Southwest and northern Mexico are not well known but probably include a variety of upland habitats that connect some of the isolated, rugged mountains, foothills and ridges in this region.

The revised plan for the Coronado (draft, 2013) designates "wildlife linkages interface" areas, based on a statewide interagency effort that produced *Arizona's Wildlife Linkages Assessment* (Arizona Wildlife Linkages Workgroup 2006). The forest plan recognized that land management outside of the national forest boundaries affects biological resources on the national forest. Using data from the interagency group, the plan designates linkage areas on the boundary of the national forest (see Figure 1). These designated areas have management direction to maintain and reduce connectivity barriers and to coordinate connectivity management with other jurisdictions.

CASE STUDY: Grizzly Bear Approach Areas

The Kootenai National Forest in Idaho and Montana provided an excellent example of how to plan strategically for connectivity that has been confined to identifiable corridors and linkage areas. In 2008, the Kootenai identified and mapped locations of 24 approach areas important for grizzly bear connectivity using the best available scientific information from existing government and nongovernmental organizations, criteria for barriers (land ownership, topography, forest cover, land development) and wildlife use (Figure 2). Approach areas were defined as places where corridors or linkage zones cross what are termed "fracture zones" (e.g., valley bottoms

Figure 2. Grizzly bear approach areas on the Kootenai National Forest[4]



Source: Brundin and Johnson 2008: 3, Figure 1

4. The approach areas were not carried forward into the final, revised forest plan.

NATIONAL PARK SERVICE

The Kootenai National Forest plan identified "approach areas"—places where roads and other barriers to connectivity may hinder grizzly bear movement.

with highways and railways) where animal movements may be hindered and mortality risk elevated. The Kootenai also identified conservation measures that could be included in the forest plan as plan components for the approach areas and identified private lands where land exchanges, conservation easements or direct acquisition may be appropriate to improve management for one or more wildlife species (IGBC Public Lands Wildlife Linkage Taskforce 2004).

CASE STUDY: Blue Mountains Wildlife Corridor Management Area

The draft Blue Mountains National Forests plan (proposed plan, 2014), which covers the Malheur, Umatilla and Wallowa-Whitman national forests (the three forests span the states of Oregon, Washington and Idaho), establishes a management area identified as a "wildlife corridor" to connect wilderness areas and provide for landscape connectivity and defined as follows:

> *Wildlife corridors are areas designed to maintain habitat linkages between wilderness areas. Although disagreement exists regarding the utility of corridors, this management area emphasizes management for landscape connectivity, which is "the degree to which the landscape facilitates or impedes movement among resource patches," [sic] (Taylor et al. 1993) or "the functional relationship among habitat patches, owing to the spatial contagion of habitat and the movement responses of organisms to landscape structure," [sic] (With et al. 1997). A wide variety of vegetation structure and composition is present, with some showing evidence of past human disturbance and others showing affects primarily from natural disturbances, such as wildfires. Both summer and winter motor vehicle travel is restricted to designated routes. Recreation users can expect to find evidence of human activity in the form of vegetation management, mining, and road building. However, many of the roads that are closed to motor vehicle travel occur in these areas* (Blue Mountains 2014: 90).

The plan also provides a "strategy" for each management area. While the draft forest plan has drawn some criticism over unrelated issues, establishing a management area for corridors based on landscape function and structure allows for the design of habitat linkages in a variety of forms other than just simple linear connection between habitat patches.

LANDSCAPE PLAN COMPONENTS FOR CONNECTIVITY

Forest plan connectivity assessments should indicate if plan components are necessary to maintain or restore connectivity, either as an important contribution to ecological integrity or to provide conditions necessary for an at-risk species. An early consideration in forest plan connectivity planning should be the desired structure and pattern of the planning area landscape and the development of landscape plan components—desired conditions and objectives, where the desired condition describes how the connected landscape should look, and the objectives describe the timeframe and steps for achieving the desired condition.

Forest plans should include desired conditions and objectives for the sizes and distribution of habitat patches and other key characteristics of connectivity. It is also important to show the general areas where connectivity will be emphasized on a map and that the identification and management of these areas take into account the role and contribution of national forest lands to connectivity across other land ownerships.

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The Canada lynx, a species listed as threatened under the Endangered Species Act, requires connected habitat across wide areas. Forest plan standards are in place to ensure that the connectivity and other habitat needs of lynx are met on national forests.

Table 4. Examples of landscape connectivity plan components in forest plans

Landscape Plan Components	Case Study and Comments
▪ Forest boundaries are permeable to animals of all sizes and offer consistent, safe access for ingress and egress of wildlife. In particular, segments of the national forest boundary identified in [the wildlife linkages interface] remain critical interfaces that link wildlife habitat on both sides of the boundary. Fences, roads, recreational sites and other man-made features do not impede animal movement or contribute to habitat fragmentation.	The Coronado National Forest consists of isolated mountain ranges, leading the draft plan to explicitly recognize the importance of connectivity and the value of coordinated planning with adjacent jurisdictions. This is especially important to ocelots and jaguars, which occur here at the northern end of their range and depend on connectivity to source populations in Mexico (Coronado 2013). This is direction for a specific management area.
▪ Retain natural areas as a core for a regional network while limiting the built environment to the minimum land area needed to support growing public needs. ▪ Reduce habitat loss and fragmentation by conserving and managing habitat linkages within and, where possible, between the national forests and other public and privately conserved lands. ▪ Preserve wildlife and threatened, endangered, proposed, candidate and sensitive species habitat and connecting links between the San Diego River Watershed and San Dieguito/Black Mountain.	The forest plan for the Cleveland National Forest was revised in conjunction with three other California national forests. The forests face a common management challenge of collaborating in nontraditional formats with local communities and governments to maintain and restore habitat linkages between the national forests and other open space reserves. This is forest-wide direction, but also refers to specific locations.
Landscape patterns are spatially and temporally diverse and have a positive influence on overall ecological function and scenic integrity. Landscape patterns provide connectivity, allowing animals to move across landscapes. Landscape patterns are resilient and sustainable, considering the range of possible climate change scenarios. The plans include a forest-wide desired condition that mentions “the ability of species and individuals to interact, disperse, and find security within habitats in the planning area” (Blue Mountains 2014: 30).	The Blue Mountains National Forests provide an important wildlife corridor connecting habitats and wildlife migration routes between the Rocky Mountains and central Oregon (Blue Mountains 2014). This is forest-wide direction about landscape patterns, in addition to the specific management area direction described above.
Federal ownership is consolidated when opportunities arise to improve habitat connectivity and facilitate wildlife movement.	This is forest-wide direction in the proposed action for the Nez Perce-Clearwater plan revision for use in subsequent land adjustment planning. Identifying priority locations in the plan would be more helpful (Nez Perce-Clearwater 2014).

Table 4 presents examples of landscape connectivity plan components in forest planning. (The language of the plan components is either verbatim or summarized. See the “References” section for source materials.) It should be noted that these examples (drawn from older forest plans) would need to be worded more explicitly under the 2012 Planning Rule, which requires desired conditions to be “specific enough to allow progress toward their achievement to be determined” (219.7(e)(1)(i)).

PROJECT PLAN COMPONENTS FOR CONNECTIVITY

Project components pertain to how projects are designed and implemented under the forest plan. Standards and guidelines, and suitability determinations for connectivity should be designed to promote achievement of the desired conditions and objectives for connectivity. Connectivity standards should be developed when greater certainty is important, such as in meeting diversity requirements necessary to protect at-risk species.

Table 5 provides examples of standards and guidelines for connectivity in forest planning. (The language of the plan components may be verbatim or summarized. See the “References” section for source materials.)

AQUATIC ECOSYSTEM CONNECTIVITY

Forest Service lands are most often found in the higher elevations of watersheds where streams provide clear, high-quality water. Management of aquatic ecosystems often centers on providing habitat that will support important fisheries.

Plan components for ecosystem integrity (including connectivity) must take into account the interdependence of terrestrial and aquatic ecosystems (219.8(a)(1)). There

Table 5. Examples of connectivity standards and guidelines in forest plans

Project Connectivity Plan Component	Case Study and Comments
▪ Retain connections of at least 400 feet in width to at least two other [late-successional/old growth] stands. ▪ Connections should occur where medium diameter or larger trees are common, and canopy closures are within the top one-third of site potential. ▪ The length of connecting corridors should be as short as possible. ▪ Understory should be left in patches or scattered to assist in supporting stand density and cover.	The Eastside Screens are rules for logging adopted as amendments to forest plans east of the Cascade crest in Washington and Oregon in 1996. They are intended to protect remaining late-successional and old-growth forests and to retain "connectivity corridors" between them (USFS 1995).
▪ When highway or forest highway construction or reconstruction is proposed in linkage areas, identify potential highway crossings. ▪ [National forest] lands in lynx linkage areas shall be retained in public ownership. ▪ New permanent roads should not be built on ridge-tops or saddles, or in lynx linkage areas.	The Canada lynx was listed as a threatened species in March 2000, largely due to a lack of adequate regulatory mechanisms in existing land management plans for federal lands. Lynx are known to disperse over wide areas, therefore it was important to add conservation measures to forest plans for lynx connectivity, which the Forest Service did in 2007 (USFS 2007) .

is an additional requirement in the planning rule to maintain or restore the ecological integrity of riparian areas, "including plan components to maintain or restore structure, function, composition, and connectivity …" (219.8(a)). This must be done by establishing "riparian management zones" and applying plan components to them that address riparian management issues. In particular, plan components for riparian management areas must specifically address ecological connectivity, blockages of watercourses, and aquatic and terrestrial habitats (219.8(a)(3)).

Many connectivity issues become intertwined in riparian areas, and plans can address them in conjunction with either terrestrial or aquatic connectivity or both. At a broad scale, management of riparian zones contributes to overall ecological integrity by providing connectivity between watersheds for both terrestrial and aquatic species. Riparian zones also provide connectivity that contributes to the terrestrial and aquatic integrity of individual watersheds. At a fine scale, the integrity of riparian areas themselves depends on the quality of aquatic and terrestrial habitat and often requires connectivity within and from riparian areas to other systems, including the hydrologic connectivity of a water body to floodplains or groundwater (floodplain connectivity can be a limiting factor for fish).

Sophisticated conservation strategies for salmonid species have been included in forest plans in the inland Pacific Northwest for two decades. The "PACFISH" and "INFISH" conservation strategies (1995) developed by the Forest Service and the Bureau of Land Management address connectivity in two primary ways. At the broader scale, they designate watersheds where management will emphasize water quality and fish habitat. This includes existing stronghold populations of fish and, importantly, additional watersheds that can be connected to those strongholds and restored. This will create a network of connected high-quality habitat that allows recolonization after a disturbance event such as a wildfire, flood or drought has rendered an area temporarily unsuitable.

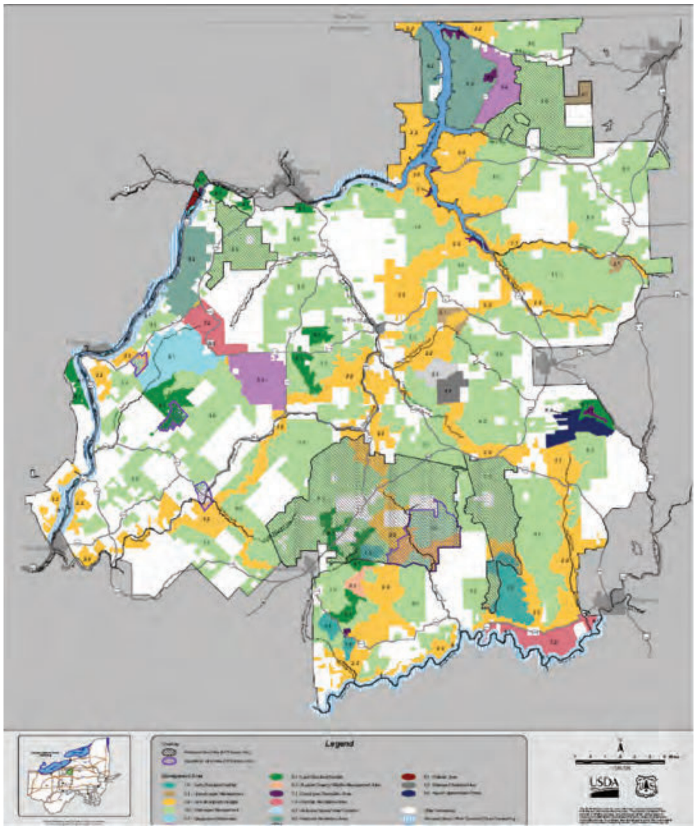
The Eastern Brook Trout Joint Venture, a partnership of state and federal agencies, nongovernmental organizations, and academic institutions, used a similar approach with the eastern brook trout in its native habitat (Maine to Georgia).According to its publication, *Conserving the Eastern Brook Trout: Action Strategies*, restoration should focus on habitat supporting populations that are doing relatively well, and then extend to adjacent habitats. An important part of this strategy is to "[i]dentify barriers to fish passage and re-establish habitat connectivity where possible" (Eastern Brook Trout Joint Venture 2008: 26).

The combination of designating watersheds and identifying connectivity barriers should lead to objectives that prioritize locations for restoration, such as the following connectivity objectives:

- Increase aquatic habitat connectivity through replacement of 90 culverts.
- Restore stronghold watersheds connectivity for aquatic species in four to six subwatersheds or on 80 to 120 stream miles.
- Establish self-sustaining brook trout populations in 10 percent of known extirpated key watersheds by 2025.

Existing forest plans also define riparian management areas, where standards and guidelines to protect aquatic resources apply to various management activities. While

Figure 3. Old forest connectivity management



Source: Allegheny National Forest Management Area Map (2007)

these plan components are primarily for the purpose of protecting resident fish, they also facilitate migration. The following type of standard would specifically address this connectivity issue: Construction or reconstruction of roads shall provide for passage of fish at all stream crossings.

BARRIERS TO CONNECTIVITY

National forest lands encompass a variety of permanent developments such as roads, railways, energy and mineral development infrastructure, recreation infrastructure and fencing. Evaluation and management of connectivity require determining the nature and effect of barriers on permeability and providing direction to reduce the effects of existing barriers and to avoid the creation of new ones. The more confined and unique the corridors or linkage zones are, the more attention should be paid to how barriers are managed**.** Forest plans must address barriers to connectivity that are relevant to ecological diversity and the persistence of species in a plan area.[5]

One key aspect of barriers that must be considered in relation to national forest management is their cause and degree of permanence. If barriers to wildlife movement and connectivity are due to natural disturbance (e.g., a forest opening caused by a fire or landslide), they can be viewed as transitory barriers that can be expected to "move" from place to place as new openings are created and then closed by natural succession. However, if the movement barrier for a particular species of wildlife is a lack of habitat that is difficult to restore, such as old-growth forest, the connectivity problem may be longer term and the need to protect existing patches using project plan components may be greater.

The Allegheny National Forest in Pennsylvania provides an example of old forest connectivity management, where habitat diversity was one of the key issues identified at the beginning of the plan revision process. The forest plan paid specific attention to "providing late structural and old growth forests and habitat connectivity across the landscape" (ROD, 2007: B-3). The revised plan established a management area for "late structural linkages" based on

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Forest plans should recognize the value of rare habitats, such as old-growth forest like this in the Siuslaw National Forest, in providing for connectivity.

5. While the effectiveness of habitat corridors providing connectivity is no longer disputed (Gilbert-Norton et al. 2010), potential negative consequences may result from movement of invasive, exotic, and otherwise harmful species or diseases, especially in aquatic habitats. This has been noted especially for inland trout species, where enhancing connectivity could do more harm than good by promoting competition or hybridization with non-native species, or introducing diseases. These kinds of risks should be identified and mitigated to the extent possible when designing landscape connections. Moreover, efforts to connect landscapes that have not historically been connected should be avoided.

existing core blocks of wilderness areas, research natural areas, national recreation areas and other protected areas. It was also designed to specifically include areas of known goshawk nest sites and rattlesnake dens, thus affording additional protection for these species (see Figure 3).

ROADS AND CONNECTIVITY

Roads and their associated human uses are one of the most common, persistent and obstructive barriers to terrestrial and aquatic wildlife connectivity. The National Forest System has approximately 375,000 miles of roads.[6] Decisions to build, decommission, open or close roads can affect connectivity in significant ways. Recognition of the role of unroaded (i.e., roadless) areas for the purposes of connectivity planning is equally important. Forest plans provide the overall guidance for how many roads there will be on a forest and how they are to be used.

Use of roads by the public is also governed by the Forest Service "Travel Management Rule," regulations published in 2005 to establish a nationally consistent approach to local determinations of where excluding motorized use is necessary to protect other resources or, conversely, where such use is desirable and ecologically acceptable. The regulations require each national forest to identify and designate roads, trails, and areas that are open to motor vehicle use. Motorized use is prohibited anywhere that is not so designated. These decisions are part of travel management plans, and these plans must be consistent with forest plans.

Clearly, decisions to have a road or to allow motorized use should take into account the effect of that particular road on connectivity. To fully understand the effects, it is necessary to know what role an area or corridor is expected to play in providing connectivity and what else is likely to happen there that will affect its connectivity value. The forest plan is the place to provide answers to those questions.

Where motorized use is inconsistent with the desired condition for an area, including desired connectivity conditions, a forest plan can identify the area as one that is not suitable for motorized use. This precludes the establishment of motorized routes in the area. It should also lead to eliminating any existing motorized use through road or area closures.

Site-specific desired conditions for connectivity are helpful in deciding where to manage for motorized use. The Gallatin National Forest Travel Plan Final


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Roads and their associated human uses are one of the most common, persistent and obstructive barriers to connectivity on national forest lands. The National Forest System has about 375,000 miles of roads.

6. See www.fs.fed.us/eng/transp/.

Environmental Impact Statement (2006) includes a site-specific goal for identified "wildlife corridors," which provides a good example of a desired condition that should be included in a forest plan:

> *Provide for wildlife movement and genetic interaction (particularly grizzly bear and lynx) between mountain ranges at Bozeman Pass (linking the Gallatin Range to the Bridger/Bangtails); across highway 191 from Big Sky to its junction with highway 287 (linking the Gallatin and Madison Mountain Ranges); the Lionhead area (linking the Henry's Lake Mountains to the Gravelly Mountains and areas west); Yankee Jim Canyon (linking the Absoroka Mountains to the Gallatin Range); and at Cooke Pass (linking the Absoroka/Beartooth Range to areas south)* (Gallatin 2006: 3-88 – 3-89).

A connectivity characteristic commonly used in forest plans to protect wildlife and fish habitat is road density. Road density limits are especially useful for protecting big game hunting opportunities. The presence and use of roads have also been found to create risks to movement of large carnivores such as grizzly bears, a federally listed threatened species. To comply with the ESA, forest plans in grizzly bear range include restrictions on road density.

The Flathead National Forest provides some of the most important grizzly bear habitat in the National Forest System. As a result of ESA consultation on the forest plan, the Forest Service adopted Amendment #19 in 1995 that applied objectives and standards for each of 70 grizzly bear management subunits across the Flathead (where national forest ownership is greater than 75 percent) (Flathead 1995). For example, an objective was developed stating that within five years total road density of greater than two miles per square mile would occur on less than 24 percent of the grizzly bear management unit and in 10 years that would be further reduced to less than 19 percent. Similarly, standards were used to ensure there would be no net increases in road densities above a certain threshold and to maintain the security of core grizzly bear habitat areas. These types of connectivity and security plan components have been successful in reducing the number of roads forest-wide by approximately 700 miles and increasing secure core area from 63 percent to 70 percent (Flathead 2012: unpaginated, Tables 16b-9 and 16b-10).

For terrestrial species, it is often the use of the road that is more of a barrier to connectivity than the physical presence of the road. Many current plans address the need to limit motorized access during big game hunting season or to protect sensitive big game habitat such as winter range.

CONCLUSION

The connectivity planning direction found in the 2012 Planning Rule provides a significant opportunity to develop and implement landscape- and project-scale connectivity strategies on Forest Service lands and to coordinate connectivity planning across land ownerships. To be successful, forest planning stakeholders—including Forest Service planners, conservation advocates, scientists and other agencies and governments—must collaborate to devise innovative approaches.

Connectivity planning also requires forward thinking to execute the vision of a connected landscape. There is no one way to develop and implement connectivity strategies within forest plans. We hope this guide stimulates innovative ideas and is a starting point for developing effective approaches to connectivity planning within forest plans.

Share Your Experiences

Please share your forest planning experiences with us and let us know if this guide was useful. Your input will help us build our list of case studies and improve the effectiveness of this planning tool. Send your feedback to Pete Nelson (pnelson@defenders.org).

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APPENDIX:
EXAMPLES OF COORDINATED CONNECTIVITY PLANNING

Multi-Organization Initiatives, including the Forest Service

America's Great Outdoors Initiative
www.doi.gov/americasgreatoutdoors/index.cfm

One of the goals of the President's America's Great Outdoors Initiative is "the conservation of land, water, wildlife, historic, and cultural resources, creating corridors and connectivity across these outdoor spaces, and for enhancing neighborhood parks." The "Large Landscapes Initiative" seeks to "improve collaboration across federal agencies and with state and local partners, especially given the inherent cross-jurisdictional nature of restoring large landscapes." It currently includes a study of specific wildlife linkage locations across major highways in the "Crown of the Continent" ecosystem in Montana.

Department of the Interior, Landscape Conservation Cooperatives
www.fws.gov/landscape-conservation/lcc.html

LCCs provide a forum for federal agencies (including the Forest Service), states, Tribes, non-governmental organizations, universities and others to work together to coordinate management response to climate change at the landscape level. "New wildlife corridors" was one of the specific needs identified nationally. The Great Northern LCC partners, for example, agreed to conservation goals that prominently feature connectivity as an important element of ecosystem integrity, and they also identified "target species" that depend on connectivity. Land management plans would be the vehicle for the Forest Service to incorporate broader landscape conservation goals.

Western Governors' Association Wildlife Corridors and Crucial Habitat Initiative
www.westgov.org/wildlife-corridors-and-crucial-habitat

The Western Governors' Association's initial policy stated that federal land management agencies should identify key wildlife migration corridors in their land management plans. The Forest Service is participating in implementing this connectivity guidance. In November 2012, the Forest Service encouraged forest supervisors conducting forest planning to consider information compiled by states for this initiative as part of implementing the 2012 Planning Rule.

Grizzly Bear Recovery Planning
www.igbconline.org/index.php/population-recovery/grizzly-bear-linkage-zones

The Recovery Plan for Grizzly Bear identifies the need to evaluate potential linkage areas within and between recovery areas. The Interagency Grizzly Bear Committee (IGBC, which includes the Forest Service) determined that "… linkage zone identification and the maintenance of existing linkage opportunities for wildlife between large blocks of public lands in the range of the grizzly bear are fundamental to healthy wildlife." Maps of linkage areas have been developed by the U.S. Fish and Wildlife Service and sanctioned by the IGBC.

Forest Service Initiatives

Properly addressing connectivity in land management plans will also promote coordination and integration within the Forest Service and advance other agency prerogatives.

The Forest Service Strategic Framework for Responding to Climate Change includes "development of wildlife corridors to facilitate migration" as a strategy to address climate change effects (www.fs.fed.us/climatechange/pdf/Roadmapfinal.pdf). One of the "immediate initiatives" in the roadmap is connecting habitats to improve adaptive capacity by:

- Collaborating with partners to develop strategies that identify priority locations for maintaining and restoring habitat connectivity. Seeking partnerships with private landowners to provide migration corridors across private lands.
- Removing or modifying physical impediments to species movement most likely to be affected by climate change.
- Managing forest and grassland ecosystems to reduce habitat fragmentation.
- Continuing to develop and restore important habitat corridors for fish and wildlife.

The Forest Service Open Space Conservation Strategy states that "[o]ur vision for the 21st century is an interconnected network of open space across the landscape that supports healthy ecosystems and a high quality of life for Americans" (www.fs.fed.us/openspace/national_strategy.html).

DEFENDERS OF WILDLIFE

THE CENTER FOR LARGE LANDSCAPE CONSERVATION

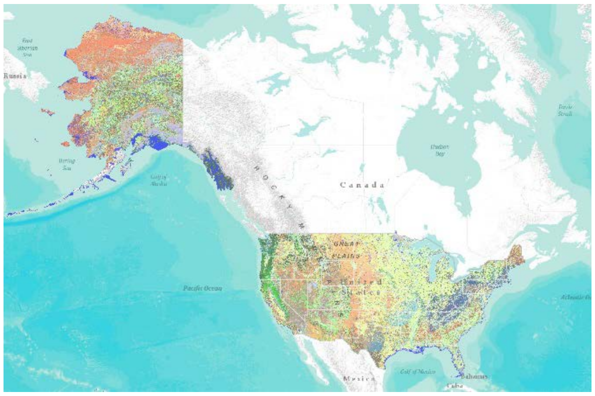
WILDLANDS NETWORK
RECONNECTING NATURE IN NORTH AMERICA

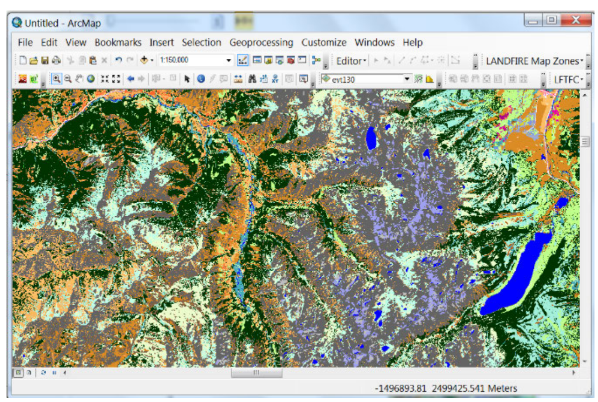
Yellowstone to Yukon
Conservation Initiative

Modifying LANDFIRE Geospatial Data for Local Applications

Version 1

September 2016





TEAMS

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Wildland Fire Management RD&A

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LANDFIRE

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Introduction

The LANDFIRE Program provides "wall-to-wall" geospatial data of vegetation, wildland fuel, fire regime, disturbance, and topographic characteristics for the United States (Rollins 2009). LANDFIRE data are often an excellent choice for wildland fire and land management planning applications due to their consistent mapping methodologies across land ownership boundaries and relevancy to common conservation and land management questions. LANDFIRE data are distributed free of charge through the Program's website at www.landfire.gov.

This guide will focus on LANDFIRE data for the conterminous United States, Alaska, and Hawaii. A subset of LANDFIRE data products is available for the Pacific and Caribbean U.S. insular areas; however, the mapping methodologies for these areas vary substantially enough from those for the conterminous U.S., Alaska, and Hawaii that we do not include discussion of these data in this version of the guide. We also focus primarily on LANDFIRE versions 1.0.5 (LANDFIRE 2001) through 1.3.0 (LANDFIRE 2012) as some major changes to mapping methodology occurred between version 1.0.0 (LANDFIRE National) and LANDFIRE 2001.

Although developed for sub-regional to national-scale planning applications, the utility of LANDFIRE data at finer scales has been demonstrated. The data are commonly applied on active wildland fire incidents (Noonan-Wright et al. 2011) and in landscape-level land management planning (Helmbrecht et al. 2012, Price et al. 2012, Scott et al. 2012, Tuhy et al. 2010). However, the applicability of LANDFIRE data at finer scales varies by the data product in question, its intended use, and location of interest. The LANDFIRE Program states that:

> *"Managers and planners must evaluate LANDFIRE data according to the scale and requirements specific to their needs (for example, habitat requirements for the species being considered or requirements by community leaders and interagency partners).… It is the responsibility of the user to be familiar with the value, assumptions, and limitations of LANDFIRE products"* (USFS 2015).

It is within this context that we present this guide, with the purpose of providing direction on the critique and modification of LANDFIRE geospatial data products for local applications. This guide builds upon previous work on this topic by others (Stratton 2006, 2009; The Nature Conservancy 2011a; The Nature Conservancy 2011b; The Nature Conservancy 2013). It is not so much a "cookbook" or "how-to" guide, as the specifics vary greatly by data product, intended use, scale, and location. Rather, we present primary considerations for using and modifying the data for use in local applications and provide examples and demonstrations of available tools and methods for completing common critique and modification tasks.

This guide is presented in seven chapters:

Chapter 1 provides a brief overview of LANDFIRE data products; discusses general considerations of scale, accuracy, and resolution in the critique of LANDFIRE geospatial data; and provides examples of common reasons for modifying LANDFIRE geospatial data.

Chapter 2 presents a conceptual framework for critiquing and modifying geospatial data, emphasizing the importance of framing analysis objectives and an iterative approach.

Chapter 3 describes the LANDFIRE disturbance data mapping process and discusses considerations specific to data currency, disturbance causality, and modifying data to reflect changes due to new disturbances.

Chapter 4 defines LANDIFRE potential and existing vegetation data products; describes the LANDFIRE vegetation mapping process; and discusses considerations specific to application of the NatureServe Ecological Systems classification, map zone boundaries, and succession class mapping rules.

Chapter 5 defines LANDFIRE fuel data products; describes the LANDFIRE fuel mapping process; and discusses considerations specific to map zone boundaries, application scale, disturbance updates, and modeling.

Chapter 6 describes the LANDFIRE vegetation dynamics models and their role in developing fire regime and vegetation departure products and discusses considerations specific to the integrated nature of LANDFIRE vegetation products, knowledge uncertainty, map zone boundaries, differences between data versions, and conducting local vegetation departure analysis.

Chapter 7 presents two interpreted examples of critiquing and modifying LANDFIRE data for local applications. The first example focuses on using LANDFIRE data for wildfire hazard analysis in the Rogue Basin of southwest Oregon. The second example focuses on using LANDFIRE data for vegetation departure analysis in the southern Sierra Nevada Mountains.

Chapter 1: Background

LANDFIRE Product Overview

LANDFIRE produces more than 20 geospatial data layers, a suite of vegetation dynamics models representing pre-Euro-American settlement vegetation conditions, and databases with vegetation plot and management activities information. The geospatial data, which are the focus of this guide, cover all lands in the United States and are developed using methods that utilize Landsat imagery, plot data, and biophysical gradient modeling (Rollins 2009). The mapping methodology is generally consistent by version across all regions of the country. LANDFIRE periodically updates its data products to incorporate changes over time (Nelson et al. 2013, Table 1).

Table 1: Comparison of LANDFIRE versions 1.0.0 (LANDFIRE National) through 1.3.0 (LANDFIRE 2012).

LANDFIRE Version	Currency	Distribution Date	Version Information
National (1.0.0)	Circa 2001	2008	The first full iteration of LANDFIRE data based on Landsat imagery from 1999-2001.
2001 “Refresh” (1.0.5)	Circa 2001	2011	Enhanced National by improving biophysical setting and existing vegetation type, cover and height mapping.
2008 “Refresh” (1.1.0)	Circa 2008	2011	Updated 2001 products for disturbance and succession. Landsat 1984-2008 imagery analyzed for change.
2010 (1.2.0)	Circa 2010	2013-14	Products updated for disturbance and succession. Landsat 2007-2011 imagery analyzed for change. Refined urban, agriculture, and wetlands mapping.
2012 (1.3.0)	Circa 2012	2014-15	Products updated for disturbance and succession. Landsat 2010-2013 imagery analyzed for change.

LANDFIRE geospatial data can be divided into five primary categories: vegetation, wildland fuels, fire regime, disturbance, and topography (Table 2). The vegetation data layers characterize both existing and potential vegetation type, vegetation structure, and vegetation development stage, and are primary inputs for developing other data layers. The wildland fuel data layers depict surface and canopy fuel characteristics that are inputs to various geospatial fire modeling systems. The fire regime data layers estimate the fire frequency and severity prior to European-American settlement as well as the current condition of the vegetation within the context of the historical disturbance regime. The disturbance data layers depict landscape changes that result from natural disturbances (e.g. wildfires and hurricanes) and management activities (e.g. prescribed fire and timber harvest), and are used to update the vegetation and fuel data layers over time (Nelson et al. 2013). Finally, the topographic data layers are required inputs to common geospatial fire behavior modeling systems and are used as base data for developing other LANDFIRE data layers. Modification of topographic data (elevation, slope, and aspect) is uncommon and therefore not discussed in this guide.

Table 2: LANDFIRE data products organized by data category.

Data Category	Abbreviation	Data Products
Vegetation	EVT	Existing Vegetation Type
	EVC	Existing Vegetation Cover
	EVH	Existing Vegetation Height
	SCLASS	Succession Class[a]
	ESP	Environmental Site Potential
	BpS	Biophysical Setting
	--	Vegetation Dynamics Models[b]
	LFRDB	LANDFIRE Reference Database[c]
Fuel	FBFM13	13 Fire Behavior Fuel Models
	FBFM40	40 Fire Behavior Fuel Models
	CFFDRS	Canadian Forest Fire Danger Rating System (AK only)
	FCCS	Fuel Characteristics Classification System Fuelbeds
	FLM	Fuel Loading Models
	CC	Forest Canopy Cover
	CH	Forest Canopy Height
	CBD	Forest Canopy Bulk Density
	CBH	Forest Canopy Base Height
Fire Regime	FRG	Fire Regime Groups
	MFRI	Mean Fire Return Interval
	PLS	Percent Low-severity Fire
	PMS	Percent Mixed-severity Fire
	PRS	Percent Replacement-severity Fire
	VCC	Vegetation Condition Class[d]
	VDEP	Vegetation Departure[e]
Disturbance	DYEAR	Disturbance 1999-2012
	FdistYEAR	Fuel Disturbance
	VdistYEAR	Vegetation Disturbance
	Events	Public Events Geodatabase[f]
Topography	ASP	Aspect
	DEM	Elevation
	SLP	Slope

[a]LANDFIRE groups succession class with its fire regime datasets because it is used to assess current vegetation condition, but in this guide it is grouped with the vegetation datasets because it is created from a compilation of existing vegetation datasets.
[b]The Vegetation Dynamics Models are non-spatial products used as primary inputs for mapping the fire regime datasets, to provide rulesets for mapping succession classes and as an ancillary data source for mapping existing vegetation type, biophysical settings and fire behavior fuel models.
[c]A database with geo-reference plot information used for mapping the vegetation datasets.
[d, e]Vegetation condition class and vegetation departure were not created for LANDFIRE 2010.
[f]A geo-referenced collection of disturbance and management information used to create the disturbance datasets.

The remainder of this chapter presents general considerations about the critique and modification of LANDFIRE geospatial data. Subsequent chapters will provide greater detail about specific considerations in the vegetation, fuels, fire regime, and disturbance categories.

Considerations of Scale

A primary consideration when evaluating a geospatial data layer is its scale. Traditionally map, or cartographic, scale is defined as the mathematical relationship between a given feature on a map and that same feature on the ground. For example, a typical topographic map from the U.S. Geological Survey's 7.5-minute quadrangle series has a map scale of 1 to 24,000 (1:24,000) meaning that one map unit is equivalent to 24,000 of the same units on the ground. Geospatial data do not have a map scale in the traditional sense. A geographic information system (GIS) stores the exact coordinates of every feature in a geospatial data layer, allowing users to zoom in and out on the monitor to view data at nearly any map scale, regardless of the precision of the underlying source data. This does not mean that geospatial data do not have a scale; rather, the scale of geospatial data may be difficult to discern.

In a more general sense scale may be defined as the spatial (or temporal) dimension of an object or a process, and is characterized by grain and extent (Turner et al. 2001). Grain is the finest level of resolution in geospatial data. For raster data, grain refers to cell size and for polygon data (i.e., vector data), grain refers to the minimum mapping unit. LANDFIRE raster data have a 30m x 30m cell size—that is, each data cell, or pixel, represents a $900m^2$ (approximately 0.22 acre) area on the ground. LANDFIRE data therefore have a 30-meter spatial resolution, or grain size. However, LANDFIRE data are not intended to be accurate or useful at the extent of an individual pixel or small group of pixels. The scale at which LANDFIRE data are applicable varies by product, intended use, and the location of interest.

With geospatial data there are no concrete rules that specify the required scale for a given application. Different analyses require data at different scales. For example, the scale needed to identify threatened and endangered species habitat is different than the scale needed to distinguish forests from grasslands. The critical question is, are the data good enough to meet the analysis needs? Evaluating the data accuracy and resolution requirements of your analysis will help answer this critical question.

Considerations of Accuracy and Resolution

Evaluating the accuracy and resolution of LANDFIRE data will help determine its suitability for a given use. Two types of accuracy to evaluate are **positional accuracy**, or the ability of the data to reflect the true or accepted geographic location of features in space, and **content accuracy**, or the agreement between mapped units and the true or accepted value of those units. Likewise, evaluation of resolution includes **spatial resolution**, or the amount of ground area represented by a pixel, and **thematic resolution**, or the level of detail in the classification of map units. Issues with accuracy and resolution are not mutually exclusive—problems with one may result in problems with the other. Ultimately, the goal of understanding these issues is to evaluate the strengths and weaknesses of a geospatial data layer to determine its suitability for a particular analysis. Next we discuss considerations of accuracy and resolution relevant to LANDFIRE data.

Positional Accuracy

Positional accuracy refers to the ability of the data to reflect the true or accepted geographic location of features. There is little the end user of LANDFIRE data can do to improve issues of positional accuracy. Boundaries or distinctions between feature types (e.g., vegetation types) may not be precisely located solely due to the raster format of LANDFIRE data. The spatial resolution of raster data has a direct effect on positional accuracy: the larger the cell size the less accurate the location (Figure 1). However, these should be minor issues if applying LANDFIRE data at an appropriate scale, one in which the data meet the analysis needs. It is also worth noting that vector data, such as the LANDFIRE event polygons, and plot data from the LANDFIRE Reference Database, are not immune to error in the location of features. Issues of positional accuracy may arise due to errors in source data, precision of field measurements, or errors in data entry.

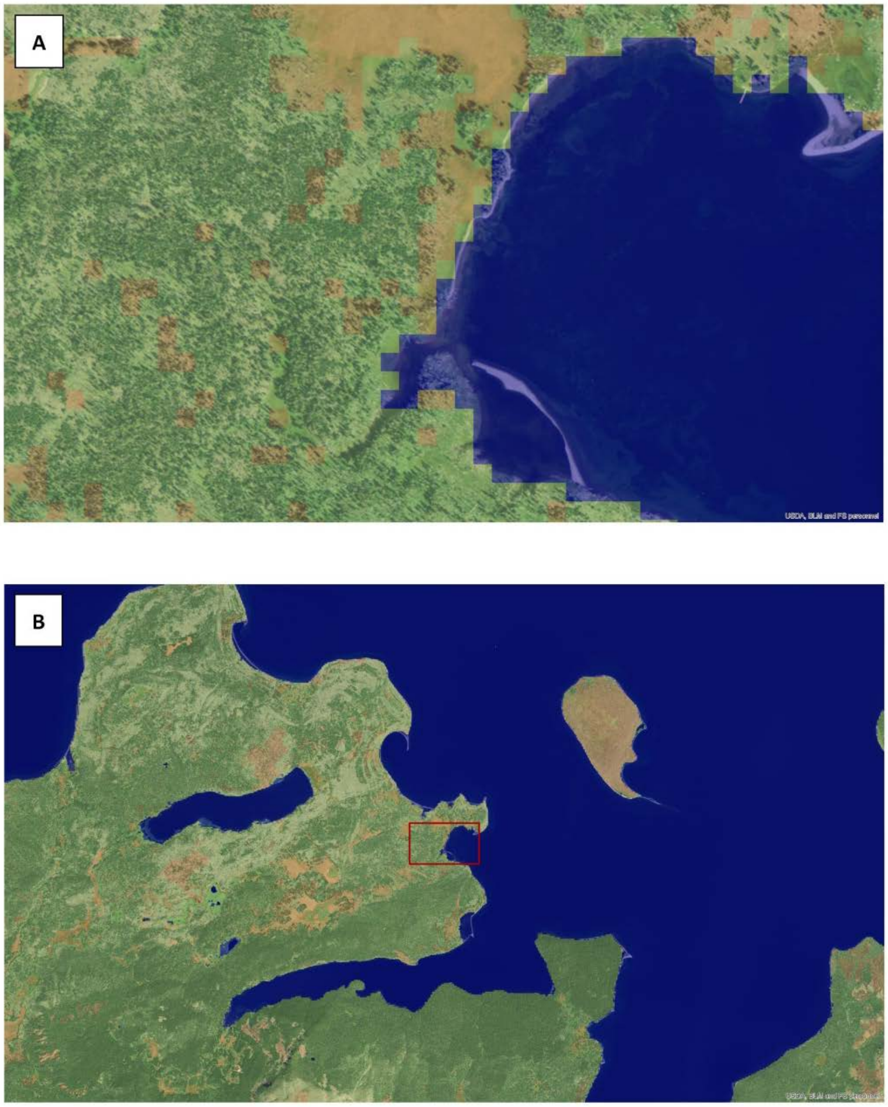


Figure 1. An example of how the spatial resolution of raster data has a direct effect on positional accuracy. LANDFIRE 30-meter resolution data does not precisely depict the shoreline or the boundary between grass (yellow shade) and forest (green shade) when viewed at a small extent (A), but at a broader extent (B), these differences are less apparent and less significant. The red rectangle in panel B shows the extent of panel A.

Content Accuracy

Content accuracy refers to the agreement between mapped units and the true or accepted value of those units. In other words, are the pixel values correct? There is much that can be done by end users of LANDFIRE data to improve issues of content accuracy based on local knowledge, additional data sources, and an understanding of the LANDFIRE data development process. This is the primary focus of this guide. Different types of errors that may affect content accuracy are described next.

Map Unit Errors

Errors in map unit assignments in LANDFIRE data may arise through errors or limitations in remote sensing data, field plots, statistical modeling, processing logic, or a combination of these and other factors. Due to variation in data sources, this error is typically not systematic geographically. For example, the abundance and quality of plot data is inconsistent across the U.S., and cloud-free imagery is more difficult to acquire in certain areas of the country (e.g., Alaska) than others.

Data Currency Errors

One of the most obvious sources of error in vegetation and fuel data is the currency, or age, of the data. Vegetation and fuels change over time due to disturbance and vegetation growth. Disturbance may include the development of previously undeveloped land, natural disturbances, such as windthrow or wildfire, and management activities such as forest thinning or prescribed fire. Vegetation growth over time may result in changes to species composition, structure, and associated dead wood and surface matter. LANDFIRE updates its products accounting for both disturbance and vegetation growth every two years (Nelson et al. 2013), but by the time the data are delivered to the user, they are typically three or more years out-of-date. For example, LANDFIRE 2010 existing vegetation and fuel data were not available for the northwest and southwest United States geographical areas until May 2013.

The importance of updating for these temporal changes should ultimately be determined by the analysis objectives, but the need will also be influenced by the geography and vegetation dynamics of the analysis area. In areas where disturbance is uncommon or where vegetation growth is slow, less frequent updating will be required than in areas that experience frequent disturbances or rapid vegetation growth. For example, vegetation and fuel maps likely need more frequent updating in the south-eastern United States where vegetation growth is more rapid and human and natural disturbances are more frequent than in the desert portions of the southwest. In more mesic life zones, such as mid-elevation forest, the geospatial data layers likely need more frequent updating than in drier low-elevation shrub or grassland zones. Even within local areas there are typically management areas with higher wildland fire or other disturbance activities that require updating as compared to adjacent areas with low activity. Other factors to consider when assessing data currency are the type and size of disturbances that need to be reflected in the data to meet analysis objectives.

Processing or Logic Errors

In some cases, content accuracy issues are introduced during data processing. Unintentional or accidental errors may be difficult to find and correct, but a common source of content error in LANDFIRE products is the result of applying generalized mapping rule sets—a pixel's value is determined by a combination of values from other data as specified in a rule. For example, a rule may assign fire behavior fuel model TU5 (Very High Load, Dry Climate Timber-Shrub), when vegetation type equals mesic mixed conifer and canopy cover is less than 60%. Rule sets are developed and applied at the map zone level (Figure 2). While these rules may be appropriate at the scale of an entire map zone (LANDFIRE map zones range between 12 and 60 million acres in size in the conterminous U.S. and Alaska; Hawaii is a single map zone of 4 million acres), they may need to be refined for application at finer scales. In other words, the "best fit" for an entire map zone may be a compromise between different parts of the map zone. There are also often inconsistencies in mapping rules between adjacent map zones resulting in an "artificial edge" in the data. Many analysis areas often extend across two or even three map zone boundaries. On the ground these changes would start gradually near the boundary between map zones displaying continuous change across the boundary. However, accurately mapping this type of gradual transition is very difficult to achieve in a large national mapping program such as LANDFIRE.

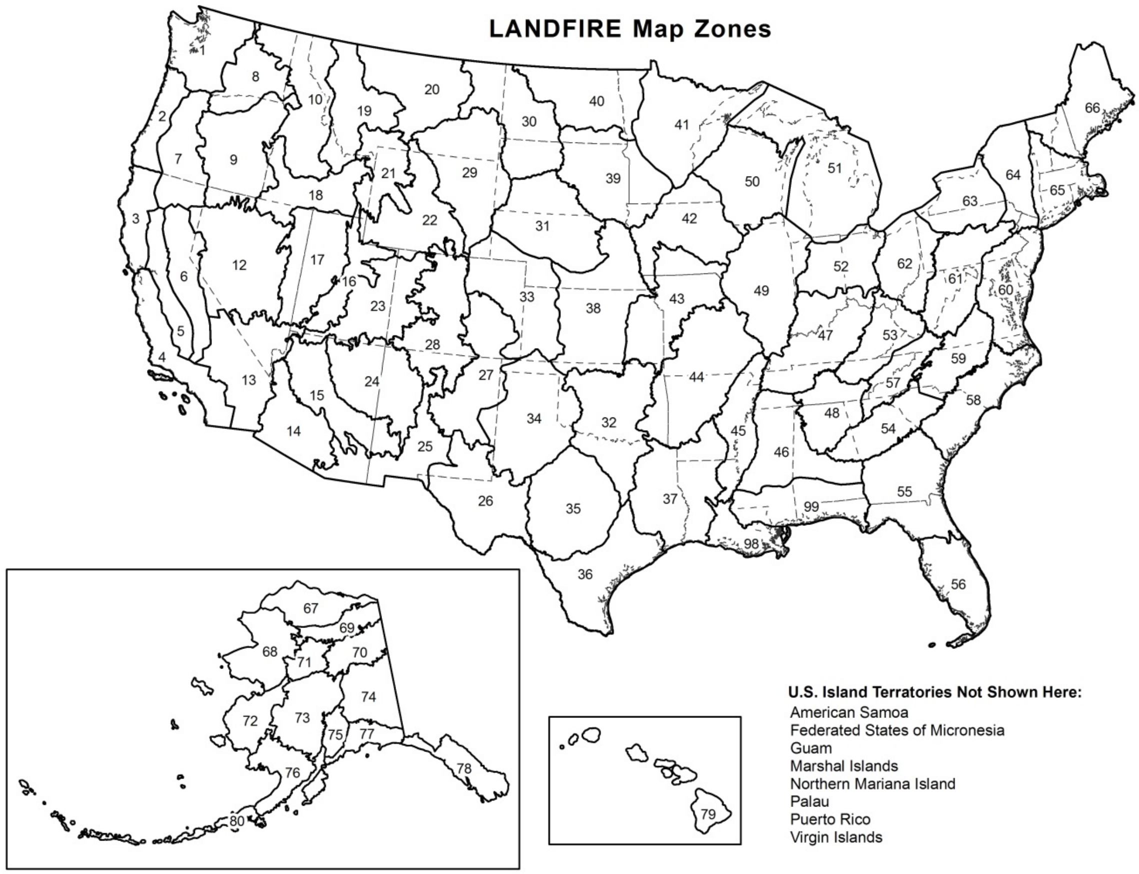


Figure 2. LANDFIRE map zones. There are 80 LANDFIRE map zones across the continental U.S., Alaska, and Hawaii, ranging in size from 4 to 60 million acres.

Content error may also arise due to incomplete knowledge and uncertainty. For example, LANDFIRE's pre-Euro-American fire frequency and severity data are created using a lookup table that links a biophysical setting with the results of a model used to simulate vegetation dynamics and estimate the mean fire frequency and fire severity distribution. The models are created using the best available literature and expert knowledge, but for many biophysical settings, the available information is far from complete. For example, there is considerably more information available to create vegetation dynamics models for biophysical settings that have economic value (e.g. productive forests) than biophysical settings that are rare or traditionally have had little economic value (e.g. arid shrublands; Blankenship et al. 2012). Greater uncertainty about historical fire regime characteristics is also associated with biophysical settings where evidence of historical fires is sparse, non-existent, or just harder to acquire, such as in stand-replacement or very long-interval fire regimes.

Spatial Resolution

As mentioned above, the spatial resolution of raster data is defined as the amount of ground area represented by a pixel. LANDFIRE data are based on Landsat satellite imagery, which have a 30m x 30m pixel size. In other words, each individual pixel represents an area of 30m x 30m, or 900m^2 (about .22 acres), on the ground.

Spatial resolution can be adjusted if necessary to meet analysis objectives. Decreasing spatial resolution by increasing pixel size (e.g., resampling 30m resolution data to 270m resolution) is sometimes done to: reduce processing time for computationally intensive analyses; decrease file storage space requirements; speed up display time; and/or, reduce the "pixelated" look of a map by absorbing isolated cells into larger patches. While it is possible to adjust resolution the other way, that is to change from coarser to finer resolution, greater map detail can only be achieved if finer resolution geospatial data are incorporated into the resampling process. That is, resampling to a finer resolution without additional finer-scale information gives a false sense of accuracy (see sidebar).

Resampling Raster Data Layers

Resampling is the process of changing the resolution of a dataset. Raster data may be made coarser by aggregating adjacent pixels. Some users of LANDFIRE data who perform national summaries of the data have resampled LANDFIRE grids from 30m to 270m. At this broad extent, resampling may have little impact on the results but can greatly increase computer processing efficiency.

Resampling to a finer resolution is sometimes referred to as downscaling and is often associated with the process of obtaining local level climate data from global climate models. Resampling to a finer resolution is possible using the resample techniques available in ArcGIS, but these techniques will not change the accuracy of the underlying data.

There are several resampling methods available in ArcGIS software, and the resampled raster values will differ depending on the method used.

Thematic Resolution

Thematic resolution refers to the level of detail in the map units. The thematic resolution of LANDFIRE data varies by data product. The most common reason that an end user of LANDFIRE data might change thematic resolution is to ensure that the level of detail in the map units aligns with the level of detail needed to achieve the analysis objectives.

Thematic resolution can be changed to achieve either coarser or finer map units by grouping or splitting map units respectively. Grouping map units is accomplished by aggregating similar map units or by choosing a higher or coarser level within a classification hierarchy (Table 3). One advantage of grouping map units is that it may improve the content accuracy because fewer and more broadly defined units can be mapped, thus minimizing potential error. Splitting map units to achieve higher thematic resolution requires more detailed ancillary data such as maps, plots, higher resolution imagery, or other geospatial data that can be used to distinguish units at a finer level than the original geospatial data layer.

Table 3. Hierarchy of LANDFIRE biophysical setting and existing vegetation type map units. Users can choose the level that best fits their needs or create a hybrid classification by aggregating units. Note that the Society of Americana Foresters and Society of Range Management map units that are provided in the existing vegetation type data layer attribute table is for reference only. This "cover type based" map unit classification is not equivalent to the NatureServe ecological systems classification used by LANDFIRE (see Chapter 4).

Data Layer	Map Unit Level	Example
Biophysical Settings	BpS Name	Central Mixed Grass Prairie
	Group Name	Bluebunch Wheatgrass-Big Bluestem-Little Bluestem-2
	Group Vegetation	Grassland
Existing Vegetation Type	EVT Name	Laurentian-Acadian Northern Hardwoods Forest
	System Group Physiognomy	Hardwood
	System Group Name	Yellow Birch-Sugar Maple Forest
	Society of American Foresters & Society of Range Management Cover Type	SAF 27: Sugar Maple
	National Vegetation Classification System Physiognomic Order	Tree-dominated
	National Vegetation Classification System Physiognomic Class	Closed tree canopy
	National Vegetation Classification System Physiognomic Subclass	Deciduous closed tree canopy

Reasons for Modifying LANDFIRE Data

The above considerations should be helpful in determining whether LANDFIRE data are appropriate for specific objectives and whether modifications are necessary. LANDFIRE geospatial data are commonly modified for the following reasons:

1. update for landscape changes that have occurred since the LANDFIRE version,
2. calibrate to local data and knowledge,
3. improve the thematic agreement (accuracy),
4. change the spatial or thematic resolution (e.g. lump or split map units),
5. modify the map unit classification,
6. create additional data versions that reflect temporal variability (e.g. peat soils being available for burning in drought situations, or exotic annual grasses being present in wet years but not dry years),
7. facilitate comparative analysis by creating data versions (e.g. analyzing pre- and post-treatment effects or comparing treatment alternatives),
8. analyze future conditions (e.g. modifying data to represent future conditions under a climate change scenario).

Conclusion

This chapter provided an overview of LANDFIRE data products, general considerations for critiquing LANDFIRE geospatial data products, and a list of common reasons why these geospatial data are modified for local applications. LANDFIRE's suite of products provides a rich set of data that have proven useful for addressing sub-regional, regional, and national level land management issues and research questions (e.g. Aycrigg et al. 2013, Cochrane et al. 2012, Reeves and Mitchell 2011, Swaty et al. 2011, Zhu et al. 2010). Through proper critique and modification by local natural resource and geospatial professionals, LANDFIRE data may also be appropriately applied to finer-scale, local applications. (e.g., Helmbrecht et al. 2012, Price et al. 2012, Scott et al. 2012, Tuhy et al. 2010). The importance of issues and the time and effort spent addressing them should be determined by the analysis objectives.

Chapter 2: Framework for Data Critique and Modification

This chapter presents a five-step conceptual framework for data critique and modification (Figure 3). The framework begins with defining objectives. Having a clear understanding of objectives will provide a foundation for the remaining steps of the framework. The process is iterative, as findings in one step of the framework may require reevaluation of a previous step. The framework is meant to be flexible and some steps may be combined, depending on the analysis objectives, processes being performed, and experience of the analyst. Certain tools may facilitate the integration of steps. For example, the LANDFIRE Total Fuel Change tool (LFTFCT 2011) allows the analyst to critique, modify, and analyze certain aspects of fuel mapping simultaneously. The framework is typically applied by a team, wherein specialists with expertise in various disciplines (e.g., fire/fuels, silviculture, ecology, and GIS) are involved in the process.

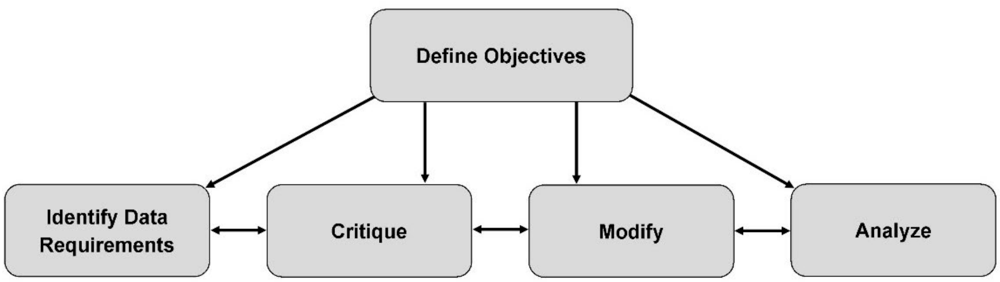


Figure 3. A conceptual framework for data critique and modification. The five-step framework begins with defining clear project objectives. The objectives will dictate the data requirements, influence the type of critique performed, dictate the types of modifications that are needed and determine the analysis performed. The framework is meant to be flexible and in some cases the process may be iterative.

Define objectives

The first step in the data critique and modification process is to define the team's objectives. Clear objectives will be a guiding principle for every other step in this process. For a given analysis determine what is needed from the data (and why), and its intended use. Defining objectives will help determine the data used, the landscape extent, the type of critique to do, and the type and level of modifications necessary.

Identify data requirements

With clear objectives in mind, the next step is to identify the data required to achieve those objectives. For example, if the objective is to assess vegetation departure from a historical reference condition, data is required that characterizes both the historical and current vegetation condition. If the objective is to assess potential wildland fire behavior, data is required that characterizes the fuels and topography of the area of interest.

As will be discussed in subsequent chapters, it is important to understand the linkages among LANDFIRE datasets, as well as the dataset creation method. Resolving issues with data that are mapped using a rule-based methodology, such as fire behavior fuel model or succession class, may require critiquing the data from which those data are derived, such as vegetation type, cover, and height or biophysical setting, thereby increasing the data requirements.

Critique

After identifying data requirements, the critical question is: are the data are good enough to meet the analysis objectives? Data need not be perfect to be useful. Ask what the important characteristics of the data are, and answer this question being mindful of the considerations discussed in Chapter 1. For the given objective: is the scale appropriate, are the data current, are the map units appropriate, and is the spatial resolution (pixel size) too coarse, or too fine? This is an iterative step; the critique may identify the need for additional data and that data will also need to be critiqued. For example, if the data are obsolete due to a recent disturbance, and that disturbance needs to be represented in the data to meet the objectives, then acquire and critique the disturbance data as it will be used to update the original data set.

Modify

Modification of data is the technical step and where GIS skills are mandatory. Subsequent chapters will provide examples of methods for conducting common modification tasks. This is also an iterative step. After modifying the data, critique it once again to be sure the desired result is achieved.

Analyze

The type of analysis performed is determined by the analysis objectives. Common analyses with LANDFIRE data include fire behavior modeling, vegetation departure assessment, and comparative analysis between land management alternatives. It is not uncommon for the results of a particular analysis to reveal data issues or requirements overlooked the first time through the framework. This step may be integrated with the previous step depending on both the analysis type and the experience of the analyst (Chapter 7).

Conclusion

This chapter presented a conceptual framework for critiquing LANDFIRE data for use in local applications. The following chapters discuss specific considerations for critiquing and modifying data from four of the five LANDFIRE data categories: disturbance, vegetation, fuels, and fire regime. Modification of topographic data (elevation, slope, and aspect) is uncommon and therefore not discussed in this guide; however, know that errors may still exist in these data. Having a thorough understanding of the assumptions and limitations of the data is of primary importance in data critique. Therefore, each of the following chapters begins with an overview of how LANDFIRE develops the data products of each category. Next are primary considerations for critiquing the data in each category and examples of why these considerations are important to local applications. Chapter 7 introduces common tools and techniques used for critiquing and modifying LANDFIRE data through interpreted examples.

Chapter 3: Disturbance

Landscape change due to planned and unplanned disturbances is continuously occurring across the United States. Updating LANDFIRE geospatial data for recent disturbances to vegetation and fuels is therefore a common modification task users of LANDFIRE data will encounter: this discussion of data critique and modification considerations thus begins with the disturbance data category. Additional considerations about updating for disturbance as it pertains specifically to vegetation, fuels, and fire regime data will be discussed in subsequent chapters.

LANDFIRE Disturbance Mapping Process

LANDFIRE maps the location, extent, type, and severity of both planned and unplanned disturbances. These data are used for determining vegetation transitions over time, and subsequently updating vegetation and fuel data products. As of LANDFIRE version 1.3.0 (LANDFIRE 2012), yearly geospatial disturbance data are available from 1999 through 2012. The yearly disturbance data are also compiled into two composite disturbance data layers—vegetation disturbance and fuel disturbance—representing disturbances occurring over the previous ten year time period. A time-since-disturbance attribute is recorded in the composite disturbance layers (Figure 4).

Yearly Disturbance Value Attribute Table

Rowid	VALUE *	COUNT	DIST_YEAR	DIST_TYPE	TYPE_CONFI	SEVERITY	SEV_CONFID	SOURCE1	SOURCE2	SOURCE3	SOURCE4	
3	13	13953	2009	Wildfire	High	Medium	High	MTBS				MTBS mapped wildfire.
4	14	6430	2009	Wildfire	High	High	High	MTBS				MTBS mapped wildfire.
5	15	156	2009	Wildfire	High	Increased Green	High	MTBS				MTBS mapped wildfire.
6	21	711	2009	Wildfire	High	Unburned/Low	High	BARC				BARC mapped wildfire.
7	22	94	2009	Wildfire	High	Low	High	BARC				BARC mapped wildfire.
8	23	32	2009	Wildfire	High	Medium	High	BARC				BARC mapped wildfire.
9	31	37	2009	Wildfire	High	Unburned/Low	High	RAVG				RAVG mapped wildfire.
10	32	2	2009	Wildfire	High	Low	High	RAVG				RAVG mapped wildfire.
11	33	1	2009	Wildfire	High	Medium	High	RAVG				RAVG mapped wildfire.
12	413	1	2009	Development	High	High	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Development Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
13	421	2834	2009	Clearcut	High	Low	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Clearcut Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
14	422	424	2009	Clearcut	High	Medium	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Clearcut Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
15	423	1749	2009	Clearcut	High	High	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Clearcut Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
16	431	6951	2009	Harvest	High	Low	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Harvest Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
17	432	1118	2009	Harvest	High	Medium	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Harvest Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
18	433	1793	2009	Harvest	High	High	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Harvest Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
19	441	19638	2009	Thinning	High	Low	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Thinning Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
20	442	1535	2009	Thinning	High	Medium	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Thinning Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.

1 (0 out of 101 Selected)

Composite Disturbance Value Attribute Table

Rowid	VALUE *	COUNT	D_TYPE	D_SEVERITY	D_TIME	R	G	B	RED	GREEN	BLUE
0	0	2877793	No Disturbance	NA	NA	0	0	0	0	0	0
1	111	483654	Fire	Low	One Year	25	0	0	1	0	0
2	112	290323	Fire	Low	Two to Five Years	25	0	0	1	0	0
3	113	352163	Fire	Low	Six to Ten Years	25	0	0	1	0	0
4	121	186765	Fire	Moderate	One Year	25	0	0	1	0	0
5	122	105493	Fire	Moderate	Two to Five Years	25	0	0	1	0	0
6	123	121960	Fire	Moderate	Six to Ten Years	25	0	0	1	0	0
7	131	51118	Fire	High	One Year	25	0	0	1	0	0
8	132	64489	Fire	High	Two to Five Years	25	0	0	1	0	0
9	133	49453	Fire	High	Six to Ten Years	25	0	0	1	0	0
10	211	14359	Mechanical Add	Low	One Year	25	10	0	1	0.4	0
11	212	31130	Mechanical Add	Low	Two to Five Years	25	10	0	1	0.4	0
12	213	38477	Mechanical Add	Low	Six to Ten Years	25	10	0	1	0.4	0
13	221	263	Mechanical Add	Moderate	One Year	25	10	0	1	0.4	0
14	222	840	Mechanical Add	Moderate	Two to Five Years	25	10	0	1	0.4	0
15	223	4543	Mechanical Add	Moderate	Six to Ten Years	25	10	0	1	0.4	0

1 (0 out of 37 Selected)

Figure 4. Yearly and composite disturbance data attribute tables. The yearly disturbance data layers are attributed with the year, type, and severity of the disturbance as well as up to four input data sources, type and severity confidence levels, and a synopsis of the data and reasoning used to determine the map unit classification. The yearly disturbance data are compiled into a composite disturbance data layer. The

disturbance year (dist_year) is classified into a time-since-disturbance category (d_time) in the composite layer.

LANDFIRE disturbance data are developed through a multistep process that incorporates Landsat satellite imagery, disturbance polygons derived from local agencies, and other ancillary data. The remainder of this section provides a general overview of the LANDFIRE disturbance mapping process (Figure 5). More detailed information is available on the LANDFIRE website and in the literature cited below.

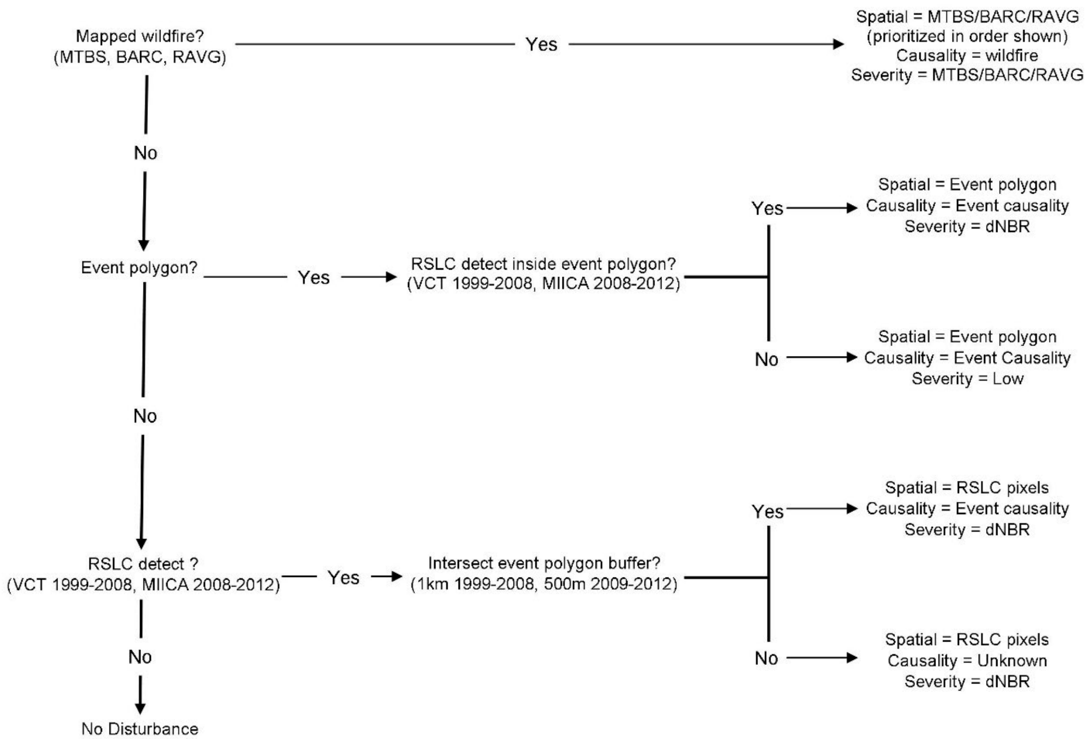


Figure 5. The LANDFIRE disturbance mapping process. LANDFIRE disturbance data are developed through a multistep process that incorporates Landsat satellite imagery, local agency derived disturbance polygons, and other ancillary data.

The first step in this process is to detect when and where disturbances have occurred. Three sources of information are used to accomplish this task: wildfire severity data from the Forest Service Remote Sensing Applications Center (RSAC), event polygons from the LANDFIRE events geodatabase, and change detection data derived from Landsat satellite imagery.

Wildfire Severity Data

RSAC manages three wildland fire severity mapping programs: Monitoring Trends in Burn Severity, Burned Area Emergency Response, and Rapid Assessment of Vegetation Condition after Wildfire. The data from these programs differ in the date of post-fire imagery and/or the severity mapping methodology used to create them. LANDFIRE uses all three datasets to map the extent and severity of wildfires.

LANDFIRE Events Data

Polygon data of vegetation and fuel management activities comprise the LANDFIRE events geodatabase. These data are obtained from federal, state, local, and private organizations and are compiled by LANDFIRE analysts. Events on national forest system lands rely heavily on data from the USDA Forest Service, Forest Activities Tracking System (FACTS). Regardless of the source, all events are crosswalked to one of 22 (including the exotic plants map unit) LANDFIRE event types (USFS 2013).

Change Detection

Lastly, LANDFIRE has incorporated two landscape change detection methodologies that apply Landsat satellite imagery in the development of the disturbance data. In the LANDFIRE 2008 mapping effort, a vegetation change and tracking process referred to as the Vegetation Change Tracker (VCT; Huang et al. 2010) was used. Beginning with the LANDFIRE 2010 mapping effort, the program adopted a new process called Multi-Index Integrated Change Analysis (MIICA; Jin et al. 2013). The MIICA process improves detection of disturbances in non-forest vegetation types, whereas VCT primarily identified disturbances in forested vegetation (D. Long, personal communication, July 23, 2013). MIICA was used to detect 2008 disturbances not identified through the VCT process, and all disturbances in 2009 through 2012. MIICA was not retroactively applied to the individual year disturbance data prior to 2008.

The second step in the disturbance mapping process is to assign causality, or disturbance type, to an identified disturbance. If the causality is know, that is, it is a mapped wildfire or LANDFIRE event, the causality is recorded in the yearly disturbance data attribute table. If the disturbance is identified through the change detection process, two additional sources of information are used to assign the likely causality: the National Gap Analysis Program's Protected Area Database and the USDA Forest Service, Pacific Northwest Research Station's SmartFire information system. Yearly disturbance layers are attributed with up to 19 of the 22 LANDFIRE event types plus an "unknown" class. This class indicates a disturbance occurred but the causality is uncertain (Table 4).

Table 4: Comparison of disturbance type attributes between LANDFIRE individual year and composite disturbance data layers. The composite vegetation disturbance (VDist) information is used to inform updates to the existing vegetation type data layer. The composite fuel disturbance (FDist) information is a subset of the VDist used to inform updates to fuel data layers.

LANDFIRE Event Type	Yearly Disturbance	VDist	Description	FDist	Applicable FDist Lifeforms
Wildfire	Wildfire	Fire	A catch-all term used to describe any non-structure fire that occurs in the wildland.	Fire	Herbaceous, Shrub, Tree
Wildland Fire Use	Wildland Fire Use				
Prescribed Fire	Prescribed Fire				
Wildland Fire	Wildland Fire				
Mastication	Mastication	Mechanical Add	A mechanical activity by which fuel is added to the natural fuelbed or in which the natural fuel structure is changed from a vertical to horizontal arrangement (e.g., mastication).	Mechanical Add	Shrub, Tree
Other Mechanical	Other Mechanical				
Clearcut	Clearcut	Mechanical Remove	A mechanical activity in which fuel is not added to the natural fuelbed (e.g., whole-tree harvesting) or in which natural fuels are removed.	Mechanical Remove	Shrub, Tree
Harvest	Harvest				
Thinning	Thinning				
Weather	Weather	Windthrow	Weather related event that results in loss of vegetation such as blowdown, hurricane, or tornado.	Windthrow	Tree
Insects	Insects	Insects-Disease	Infestations of insects and/or disease that can affect vegetative health and structure.	Insects-Disease	Shrub, Tree
Disease	Disease				
Insects/Disease	Insects/Disease				
Insecticide	Insecticide	Chemical	Application of a chemical substance such as herbicide.	NA	NA
Herbicide	Herbicide				

LANDFIRE Event Type	Yearly Disturbance	VDist	Description	FDist	Applicable FDist Lifeforms
Chemical	Chemical				
Biological	Biological	Biological	The use of living organisms, such as predators, parasites, and pathogens, to control weeds, pest insects, or diseases.	NA	NA
Development	Development	Development	Conversion of natural lands into housing, commercial, or industrial building sites. Involves permanent land clearing.	NA	NA
Exotic Plants	Exotics	Exotics	The presence of non-native species.	Exotics	Herbaceous, shrub
Planting	NA	NA	NA	NA	NA
Reforestation	NA	NA	NA	NA	NA
Seeding	NA	NA	NA	NA	NA
NA	Unknown	NA	Sources indicate that a disturbance occurred but causality is uncertain.	NA	NA

The final step in the disturbance mapping process is to map the disturbance severity. Information for determining disturbance severity may come from any one of the three data sources described above: RSAC wildfire severity, LANDFIRE events geodatabase, or remotely sensed change detection methods. Disturbance severity is assigned to one of three classes: low, moderate, or high (Table 5).

Table 5: LANDFIRE disturbance severity classes.

Severity	Description
Low	Less than 25% above-ground biomass removed.
Moderate	25 – 75% above-ground biomass removed.
High	Greater than 75% above-ground biomass removed.

The flow chart shown in Figure 5 may be used as an aid to understand this process. Where a wildfire has been mapped by one or more of the RSAC wildfire severity mapping programs, the information is used to determine the extent, year, causality (i.e., wildfire), and severity of the fire. In areas where a wildfire has not been mapped by one of the RSAC programs, but a LANDFIRE event has been mapped using other methods, the extent and causality of the event polygon are used. If the change detection process also detected the disturbance, severity is derived from the remote sensing data using the differenced Normalized Burn Ratio methodology (Key and Benson 2005). If no disturbance was detected via change detection, the year attributed to the event polygon is used and severity is set to low. Finally, if neither a wildfire or event is mapped to an area but a change is detected via remote sensing, the extent, year, and severity are determined by inference. This is done through analysis of the change detection data and assignment of causality based on proximity to event polygons and other ancillary data such as the National Gap Analysis Program's Protected Area Database and buffered SmartFire points. In addition to year, type, and severity, the yearly disturbance data layers are attributed with input data sources, type and severity confidence levels, and a synopsis of the data and reasoning used to determine the map unit classification (Figure 4).

The yearly disturbance data layers are then integrated into two composite data layers representing disturbances occurring over the previous ten year time period. In instances where multiple disturbances from different years overlap, the type and severity of the most recent disturbance is used in the composite data layer. An exception to this rule is in the case of a fire disturbance type (prescribed or wildfire) which overrides other disturbance types and is assigned to the composite layers regardless of when the fire occurred in the series of events. The disturbance type attribute of the yearly disturbance layers is reclassified into one of nine disturbance type map units in the final composite *vegetation* disturbance layer (Table 4). The year of the disturbance is classified into one of three time-since-disturbance classes: one year, two to five years, or six to ten years.

The composite *vegetation* disturbance layer is used to inform updates to the existing vegetation type, cover, and height data layers (Chapter 4). Both the yearly disturbance and composite vegetation disturbance layers are compiled from "raw" disturbance data. As such, direct comparison with existing vegetation data may reveal illogical combinations (e.g., fire and water mapped to the same pixel). When vegetation transition rules are applied to update the vegetation data layers, illogical combinations are filtered out.

The composite *fuel* disturbance layer is used to inform updates to fuel data layers (Chapter 5). The composite *fuel* disturbance layer is a subset of the composite *vegetation* disturbance layer and does not include chemical, biological, or development map units (see comparison in Table 4). The reasoning for this is that the composite fuel disturbance data layer is only applied in cases where both the post-disturbance vegetation characteristics *and* the disturbance that created those characteristics influence the post-disturbance fuels. For example, an herbicide application may cause a transition in vegetation type, cover, and/or height; and a fire behavior fuel model would be assigned based on these post-disturbance vegetation characteristics. The fact that the change was caused by the application of an herbicide does not factor into the assignment of the fuel model. This is in contrast to what would occur in a forested vegetation type after a wildfire, for example, where the post-disturbance vegetation characteristics *and* the fact that fire consumed dead wood and surface organic matter would both need to be taken into consideration in assigning the post-disturbance fuel model (Chapter 5). The composite fuel disturbance layer undergoes additional filtering to remove inconsistent disturbance/lifeform combinations (e.g., windthrow in herbaceous- or shrub-dominated landscapes, Table 4).

Considerations

Time-Since-Disturbance

LANDFIRE periodically updates the geospatial data products it develops to represent change due to disturbances; however, the update process itself takes two to three years to complete. Under the current update schedule, LANDFIRE data are typically 3–5 years out of date for any given year. In regards to the vegetation and fuel disturbance data layers, the time-since-disturbance attribute may therefore be out of date. For example, LANDFIRE 2012 data reflects conditions through the end of 2012. Thus, a disturbance that occurred in 2012 would be assigned to the one year time-since-disturbance class in the LANDFIRE 2012 data. However, the LANDFIRE 2012 data were released in the later months of 2014. For application in 2015, the original 2012 disturbance is 3 years old putting it in the 2–5 year time-since-disturbance class (Table 6). Likewise, disturbances that occurred in 2008 and 2009 should be shifted from the 2–5 year time-since-disturbance class to the 6–10 year class in 2014.

Table 6: Comparison of time-since-disturbance (TSD) between currency and release dates. For application in 2015, LANDFIRE 2012 disturbance data in the one year TSD class should be updated to the two to five year class. Likewise, disturbances that occurred in 2008 and 2009 should be updated to the six to 10 year TSD class.

Disturbance Year:	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
Data TSD (Years)	10	9	8	7	6	5	4	3	2	1	--	--
Current (TSD) Years	12	11	10	9	8	7	6	5	4	3	2	1

Whether this is of concern or not depends on the particular data products, their intended use, and the location of your assessment. For example, the assignment of fire behavior fuel model for use in fire behavior simulation is sensitive to the time-since-disturbance attribute. This is especially true in areas of the country where vegetation growth and fuel accumulation are rapid.

Disturbance Type

Disturbance type, or causality, is assigned to the vegetation disturbance and fuel disturbance data layers by pairing remote sensing data with information from the LANDFIRE Events Geodatabase. Individual disturbances are first classified into one of 22 LANDFIRE event types. Nineteen of these event types, plus an "unknown" class, are used to attribute the yearly disturbance data layers. The event types of the yearly layers are then reclassified into one of nine disturbance types in the composite vegetation disturbance layer and six types in the composite fuel disturbance layer (Table 4). Two disturbance types in particular—mechanical add and mechanical remove—can be especially challenging to assign from the information available in the events data but are very important for determining post-disturbance fuel. Whether the surface fuels (e.g., branches, needles, bark) generated from a forest management activity are added, rearranged, or removed from a site is highly dependent on factors such as site characteristics, management techniques, and management objectives. The management techniques and objectives are strongly influenced by law, regulations, and policies (local through national). These factors are highly variable in both location and time. For example, in more humid areas of the United States where downed wood decomposes quickly, activity fuel may be left on site to decompose and provide valuable nutrients to the soil. Conversely, in drier climates where this fuel takes years to decades to decompose, local, regional and/or national regulations or policy may dictate that activity fuel be removed from the site.

The information in the events data is typically not specific enough to discern these differences and LANDFIRE updates must therefore resort to the broad definitions of mechanical add/remove shown in Table 4. For local applications however, local resource professionals often have the institutional knowledge and/or ancillary information to critically critique, and update if necessary, disturbance type attributes.

Most Recent Disturbance Rule

As discussed above, in instances where multiple disturbances from different years overlap, the composite disturbance data layer is assigned the attributes of the most recent disturbance. The only exception to this rule is if fire is one of the disturbances, in which case the severity and time-since-disturbance of the fire is assigned to the composite layer regardless of when it occurred in the series of events. Multiple entries in the same treatment unit are quite common (e.g., a thinning treatment followed by treatment of activity fuels). In areas where timber harvesting is common, four or more entries may be found in short succession (e.g., a pre-treatment, one or more harvest entries, a fuel treatment, and site-preparation for planting or natural regeneration). A harvest treatment is also common, as timber salvage, after a fire.

In these situations the "most recent disturbance" rule, or "fire overrides other disturbances" rule, can lead to issues of content accuracy in the composite disturbance layers. For example, consider a high-severity harvest, such as a clearcut or shelterwood cut, followed by a low-severity disturbance, such as site-preparation or piling activity fuels. If these subsequent activities are at least a year apart, the composite data layer will be assigned "low-severity," even though all or most of the overstory vegetation was removed.

New Disturbances

The above considerations about time-since-disturbance and disturbance type attributes were presented in the context of critiquing disturbances that were already mapped and included in the LANDFIRE disturbance data products. As discussed previously, the composite disturbance data may be 3–5 years out of date upon time of version release. Updates are therefore often necessary, especially in actively managed landscapes or landscapes in which natural disturbances have occurred after the currency date of the latest LANDFIRE version. New disturbances may be added to the vegetation and fuel disturbance data layers using a variety of geospatial techniques and tools. The most appropriate technique may be influenced by the availability of recent disturbance data, the thematic and spatial detail of the data, and the experience of the analyst. For example, recent disturbance data may be in the form of a polygon shapefile depicting the location, extent, and type of disturbance without information on severity (e.g., locally developed prescribed fire burn unit map), or in the form of a raster data layer representing multiple classes of severity (Figure 6 e.g., RSAC wildfire severity data). Regardless of the techniques applied, new disturbances must be attributed with type, severity, and time-since-disturbance to be added to the vegetation disturbance and fuel disturbance data layers.

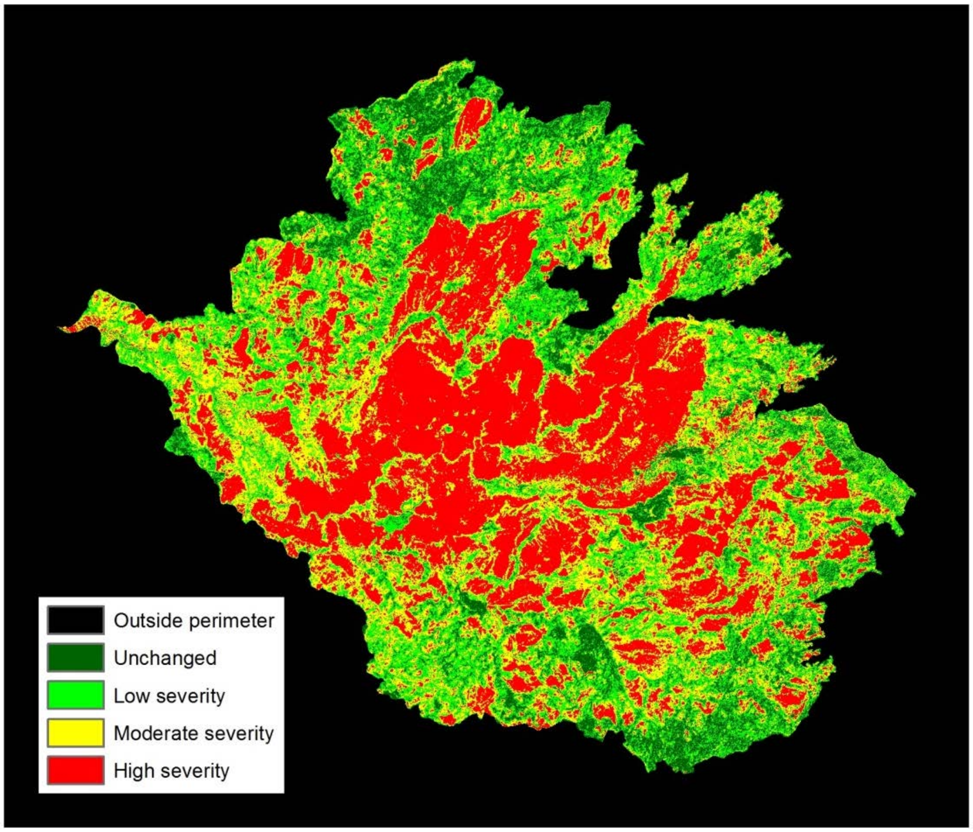


Figure 6. Four class severity classification of the 2013 Rim fire in California. Data were acquired from the U.S. Forest Service Remote Sensing Applications Center, Rapid Assessment of Vegetation Condition after Wildfire program.

Chapter 4: Vegetation

LANDFIRE develops geospatial data of potential and existing vegetation. The potential vegetation products include environmental site potential and biophysical setting. In contrast to the environmental site potential, the biophysical setting reflects potential for the historically dominant vegetation. The existing vegetation products include existing vegetation type, existing vegetation cover, existing vegetation height, and succession class. These six vegetation products are foundational to the development of other LANDFIRE geospatial data depicting fuel and fire regime characteristics.

This chapter presents an overview of the LANDFIRE vegetation mapping process, common considerations for critiquing LANDFIRE vegetation data, and examples of common pitfalls.

Vegetation Mapping Process

Potential Vegetation

Potential vegetation refers to the vegetation that could be supported at a given site based on the site's biophysical environment. LANDFIRE maps two representations of potential vegetation: environmental site potential and biophysical setting. Environmental site potential represents the late successional vegetation community that may become established at a site in the absence of disturbance. Biophysical setting represents the vegetation community that may have been dominant at a site prior to Euro-American settlement based on both the biophysical environment and an approximation of the historical disturbance regime.

Potential vegetation is mapped by LANDFIRE using a predictive modeling approach referred to generally, as *classification and regression tree* (CART; Figure 7) analysis, in conjunction with rule-based mapping techniques. First, field plot data (available in the LANDFIRE Reference Database; LFRDB [n.d.]), are keyed to environmental site potential classes based on the presence and abundance of indicator plant species that identify the biophysical conditions of the site. These plots are then intersected and attributed with information from biophysical gradient data layers (e.g., soil depth, average temperature, and average daily precipitation). The gradient layers are modeled from climate, soil, and topographic data and indirect topographic gradients such as elevation, slope, and indices of slope position. The information gathered from plot locations is then used as training data to develop the CART model—a statistical model used to predict a dependent variable (environmental site potential class) based on correlation with the independent variables (biophysical gradients). The CART model is then applied spatially to create a draft map of the environmental site potential of every pixel across the landscape based on combinations of the biophysical gradient data. The draft product is then refined using rule sets derived from the Nature Serve

Ecological Systems map unit descriptions and expert review.

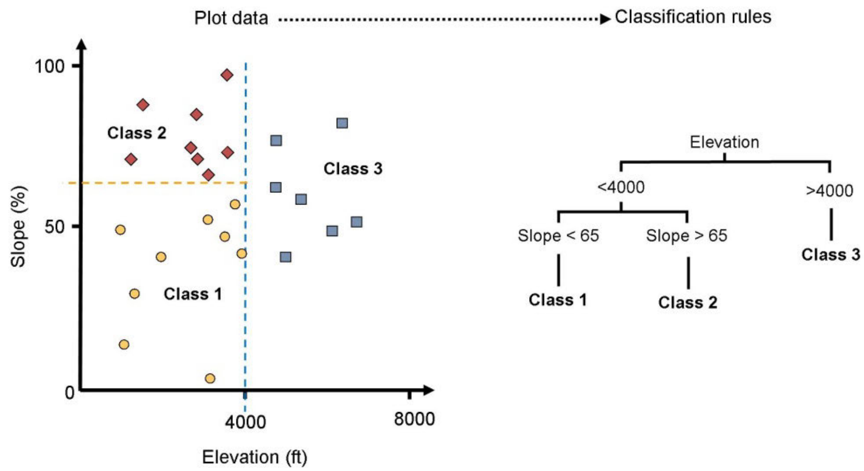


Figure 7. Classification tree conceptual diagram. In this simplified example, three classes of vegetation are plotted in respect to two biophysical gradients: elevation and slope (left side of figure). The relationship between the three vegetation classes and two biophysical gradients are then translated into classification rules (right side of figure), which are then in turn used to build spatial data layers. Approximately 40 biophysical gradients are used in the creation of the LANDFIRE potential vegetation data layers**.**

The environmental site potential data layer becomes the starting point for mapping Biophysical Settings. Environmental site potential units are associated with biophysical setting units using rule sets based on assumptions pertaining to vegetation dynamics and disturbance regimes. For example, an environmental site potential that is dominated by shade-tolerant species such as Douglas-fir or grand fir in the absence of disturbance may be mapped as a ponderosa pine- or western larch-dominated biophysical setting in an area with a frequent low-severity fire regime that would favor species that are less shade-tolerant and more fire-adapted. In other cases, alternate CART models were built to map biophysical settings from General Land Office survey data and Natural Resource Conservation Service Ecological Site Descriptions.

Existing Vegetation

Existing vegetation refers to the vegetation that is currently present on a given site. LANDFIRE maps four characteristics of existing vegetation: type, cover, height, and succession class. Existing vegetation is mapped using a predictive modeling approach similar to that used for potential vegetation; the primary difference is the input data. Like potential vegetation, methods for mapping existing vegetation type apply geospatial data of biophysical gradients and information from field plots. Because plot data can sometimes be many years old and vegetation characteristics may change rapidly, an additional filtering process is applied to ensure that current data are being used to develop the CART models. The existing vegetation type mapping process also includes data derived from Landsat satellite imagery as input. The base Landsat imagery used by LANDFIRE to derive existing vegetation products was acquired in the years 1999–2003, with newer imagery brought in to detect changes over time due to disturbance during the disturbance update process (Chapter 3).

Existing vegetation cover represents the area of the ground covered by a vertical projection of the canopy: in other words, the area of the ground covered if one were to look straight down from above (Figure 8). This is not to be confused with canopy closure, which is the proportion of the sky hemisphere obscured by vegetation when viewed from a single point (Jennings et al. 1999). Cover is mapped separately for herbaceous, shrub, and tree lifeforms using a predictive modeling approach based on plot data, satellite imagery, and biophysical gradient data layers. The canopy cover of each lifeform is binned into ten-percent classes[1] and then merged into a composite data layer in which the upper-layer lifeform's cover is assigned to the pixel. The training data for each lifeform are based on plot-level, ground assessments. However for the tree lifeform, plot canopy cover is modeled using a stem-mapping approach developed by Toney et al. (2009). This method was applied to the LANDFIRE 2001 data and is being applied to subsequent versions as an improvement over the canopy cover mapping in LANDFIRE National, which tended to over-predict tree canopy cover (Nelson et al. 2013, USFS [n.d.], Forest Canopy Cover…).

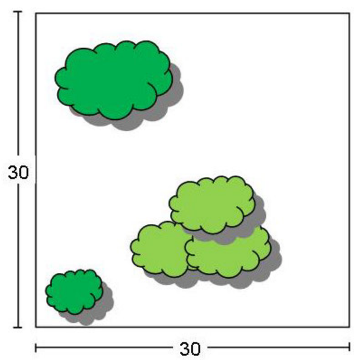


Figure 8. Vertically projected canopy cover. Existing vegetation cover represents the vertically projected canopy cover of the dominant lifeform for a pixel. In this example, the canopy cover within a 30-by-30 meter pixel is approximately 25%.

The existing vegetation height product represents the average height of the dominant lifeform. Like canopy cover, canopy height is mapped separately for herbaceous, shrub, and tree lifeforms using plot data, satellite imagery, and biophysical gradient data layers in a predictive modeling approach. The height of each lifeform is binned into classes and then merged into a composite data layer in which the upper-layer lifeform's height is assigned to the pixel (Table 7). For forests, a Shuttle Radar Topography Mission (SRTM) derived vegetation height product (Kellendorfer et al. 2004) is added to the other data sources for predictive modeling in LANDFIRE 2001 (Toney et al. 2012). The addition of the SRTM data provides a vertical structure measurement unavailable from the two-dimensional Landsat imagery which improved forest height mapping (Nelson et al. 2013, LANDFIRE 2008). Existing vegetation height for forests represents the average height of the dominant and co-dominant trees (weighted by basal area) for the pixel (Toney et al. 2012). In other words, the height value does not represent the average height of all individual trees, nor does it represent the average height of only the dominant trees. For non-forest areas, existing vegetation height represents the average height of the dominant lifeform. This is determined from species height weighted by species cover composition.

[1] For Alaska, tree and shrub cover is binned into three classes: 10%-25%, 26%-60%, and > 60%; herbaceous cover is binned into two classes: 10%-60% and > 60%.

Table 7: LANDFIRE height classes by lifeform and geographic area.

Lifeform:	Height Class (m) CONUS and HI	Height Class (m) Alaska
Herbaceous	0 - 0.5	0 - 0.5
	0.5 - 1	> 0.5
	> 1	
Shrub	0 - 0.5	0 - 0.5
	0.5 - 1	0.5 – 1.5
	1 – 3	> 1.5
	> 3	
Tree	0 - 5	No difference
	5 - 10	
	10 - 25	
	25 - 50	
	> 50	

The final characteristic of existing vegetation mapped by LANDFIRE is succession class. Succession class represents the current stage of vegetation development within an individual biophysical setting. It is very important to understand that the succession class should not be used independent of its biophysical setting. **Without its biophysical setting the succession class has no definition**. LANDFIRE maps up to seven succession classes using a rule-based approach—for each biophysical setting, a succession class is assigned based on rules that define specific combinations of existing vegetation type (primarily lifeform), existing canopy cover, and existing canopy height (Figure 9). Up to five of the seven succession classes are used to represent development stages characteristic of those found under the historical disturbance regime. Two classes are used to represent uncharacteristic conditions. Uncharacteristic native identifies native vegetation conditions that would be unlikely to occur under the historical disturbance regime, such as excessive canopy cover for a biophysical setting succession class with a frequent low-severity fire regime. Uncharacteristic exotic identifies areas in which exotic species have partially or completely replaced the native species.

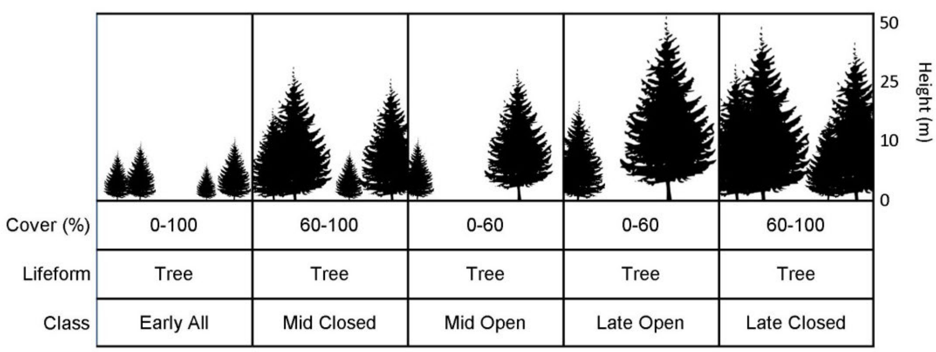


Figure 9. Typical five-class model for a forested biophysical setting, demonstrating succession class assignment based on vegetation characteristics. LANDFIRE maps up to seven succession classes using a rule-based approach—for each biophysical setting, a succession class is assigned based on rules that define specific combinations of existing vegetation type (primarily lifeform), existing canopy cover, and existing canopy height.

Updating Existing Vegetation

The existing vegetation products are periodically updated for changes due to disturbances and growth. The disturbance updating process was discussed in Chapter 3. Changes due to growth are incorporated in the mapping process after the disturbance update through a series of transition rules. Rules for updating the non-forest vegetation type for growth are developed based on the vegetation dynamics development models and the judgment of LANDFIRE analysts and other regional experts. Transition rules for forest vegetation type, cover, and height were developed based on forest growth simulations for Forest Inventory and Analysis plots modeled in the Forest Vegetation Simulator (FVS, Dixon 2002; Nelson et al. 2013). In Hawaii and Alaska (except southeast AK), where Forest Inventory and Analysis data are not available, forest transitions were developed by LANDFIRE analysts and other regional experts (Nelson et al. 2013). In LANDFIRE 2008 and 2010 the vegetation products were updated for both disturbances and growth. In LANDFIRE 2012, the vegetation products were updated for disturbance only. The transition rules are documented in databases available from the LANDFIRE Program website.

Considerations

One Classification, Three Interpretations

LANDFIRE uses the same map unit classification and naming system for the environmental site potential, biophysical setting, and existing vegetation type data layers. However, each of these layers must be interpreted differently, since they have different definitions and processing methods. A first step in identifying and mitigating possible vegetation type mapping issues (existing or potential) is to have a thorough understanding of this map unit classification and naming system and how it is used in the LANDFIRE existing vegetation type, environmental site potential, and biophysical setting data layers.

LANDFIRE uses NatureServe's Ecological Systems classification (Comer et al. 2003) as the primary map units and naming system for its existing vegetation type, environmental site potential, and biophysical setting products. The Ecological Systems classification units are intended to represent *existing* vegetation communities that persist for anywhere from 50 to hundreds of years, but LANDFIRE applies this concept in three different ways. In the existing vegetation type data layer, Ecological Systems are used as they were designed—to classify existing vegetation communities. For the environmental site potential data layer, LANDFIRE uses Ecological Systems to classify potential vegetation communities that could exist on a site given its biophysical characteristics in the absence of disturbance. Environmental site potential classes are modified to map LANDFIRE's biophysical setting concept which represents vegetation communities that may have been present prior to European-American settlement based on the biophysical environment and the historical disturbance regime. These are major differences and can have substantial effects on interpreting the data. For example, the same pixel classified as a Douglas-fir/grand fir forest environmental site potential based on the physical environment could be classified as a ponderosa pine forest biophysical setting, because of its historical fire regime, and a grass- or shrub-existing vegetation type due to a recent high-severity fire event. In rangeland, the same pixel classified as pinyon-juniper environmental site potential could be classified as a grassland biophysical setting, because of its historical fire regime, and a shrub existing vegetation type due to reduction in fire frequency.

Another important consideration specific to LANDFIRE existing vegetation type is that the NatureServe Ecological Systems map units represent vegetation communities that are typically comprised of groups of species. Most existing vegetation map users are more familiar with the concept of cover types. Cover types, in contrast to Nature Serve Ecological Systems map units, represent one or more dominant species at a single point in time. NatureServe Ecological Systems map units are not equivalent to cover types. The LANDFIRE existing vegetation type attribute table provides a crosswalk to the Society of American Foresters (SAF) and Society for Range Management (SRM) cover types classes as a guide to help users better understand LANDFIRE's map units. However, because SAF and SRM map units represent cover types and LANDFIRE's units represent systems, the crosswalk should not be interpreted as an exact match.

Potential vs. Existing Vegetation Type Rectification

The LANDFIRE potential vegetation data layers (environmental site potential and biophysical setting) were mapped using a predictive modeling approach based on plot data and biophysical gradient data layers, but did not incorporate imagery or use the existing vegetation type to modify the mapping process. This results in the potential vegetation data layers being inherently coarser in concept than the existing vegetation type data layer, which integrates Landsat satellite imagery. However, due to time and budgetary constraints, the LANDFIRE program has not been able to rectify either environmental site potential or biophysical setting with existing vegetation as to the inclusion or exclusion of specific existing vegetation types that would better depict site potential, thus improving content accuracy. Therefore, the user may find illogical combinations of these data layers and existing vegetation type for the same pixel, such as an existing vegetation type mapped to the same pixels as a biophysical setting that would not support the vegetation type due to moisture or topo-edaphic (i.e., soil) constraints. An example of this would be a riparian existing vegetation type, such as upper montane riparian, mapped to a non-riparian biophysical setting, such as sagebrush steppe. In the vegetation departure data products (Chapter 6) this situation may falsely indicate ecological departure. In these situations it can be difficult to determine which data layer is correct, but it is typically assumed that the existing vegetation type data layer is more likely to accurately depict the site because it integrates satellite imagery, and plot data go through additional filtering in its development.

Map Zone Boundaries

Because LANDFIRE vegetation data were mapped independently by map zone (Figure 2), differences or abrupt changes are sometimes found along map zone boundaries. For example, where map zone boundaries coincide with ecological division boundaries (Comer et al. 2003), there may be a change in the existing vegetation type map unit for similar vegetation types, such as Colorado Plateau pinyon-juniper woodland (Intermountain Basins ecological division) to Southern Rocky Mountain pinyon-juniper woodland (Rocky Mountain ecological division) (Figure 10). This does not necessarily indicate a mapping issue; however, secondary data products for which existing vegetation type is a variable in their mapping methodology—such as succession class (below), fire behavior fuel model (Chapter 5), and the fire regime and vegetation departure products (Chapter 6)—may be influenced by the difference in vegetation type map unit.

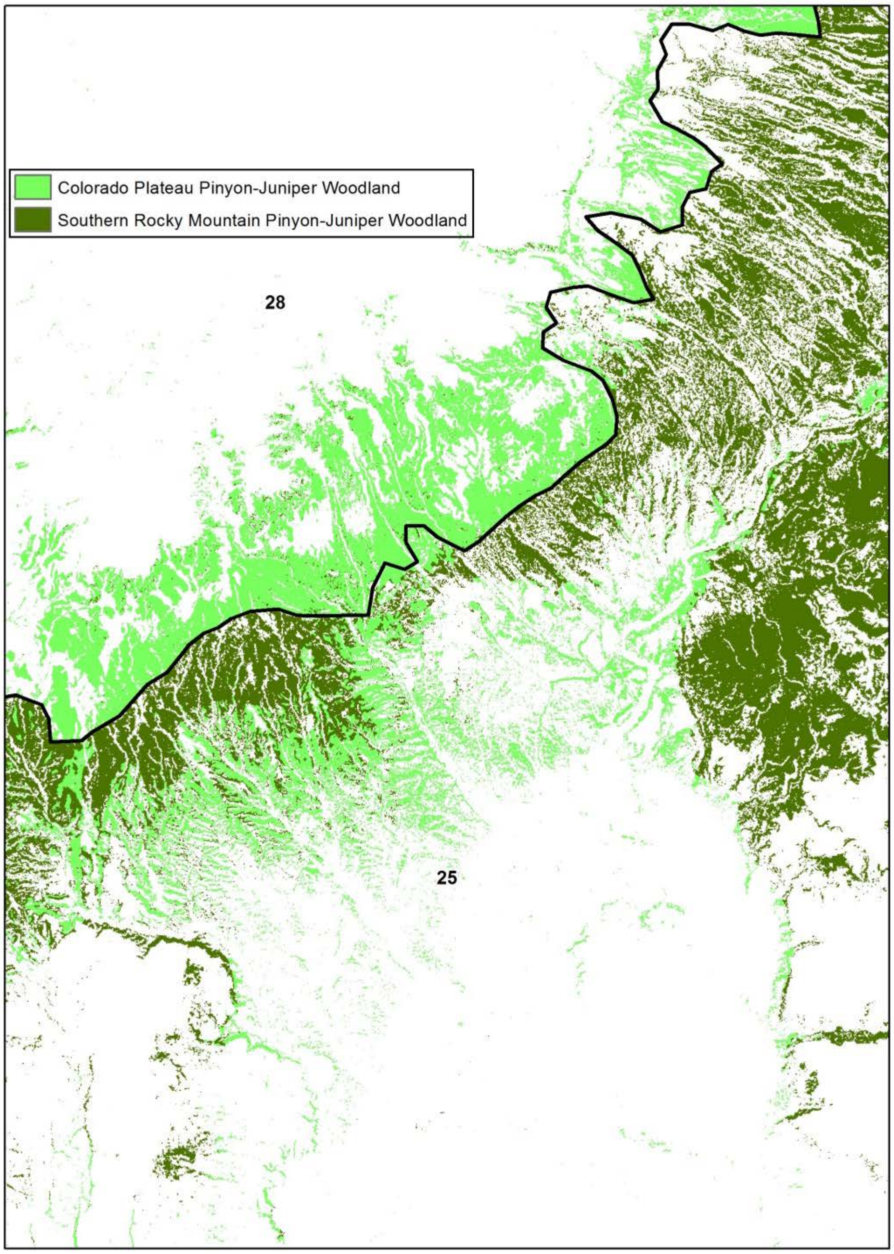


Figure 10. Comparison between the Colorado Plateau pinyon-juniper woodland existing vegetation type (Intermountain Basins ecological division) and the Southern Rocky Mountain pinyon-juniper woodland existing vegetation type (Rocky Mountain ecological division) at the map zone 25 and 28 map zone boundary. Secondary data products for which existing vegetation type is a variable in their mapping methodology may be influenced by the difference in vegetation map units at the boundary.

Existing vegetation cover is a primary variable in mapping secondary data products (i.e., succession class, fire behavior fuel model, and vegetation departure products). An abrupt change in vegetation cover within the same existing vegetation type is sometimes found at map zone boundaries (Figure 11). This may occur if the satellite imagery used for the adjacent zones was collected in different years and those years received significantly different amounts of precipitation, especially in dry southwestern ecosystems, or if different configurations of plot data were used between the zones (D. Long, personal communication, July 6, 2015). This may lead to an artificial demarcation in secondary data products and subsequently the results of analyses that use these products such as fire behavior and vegetation departure.

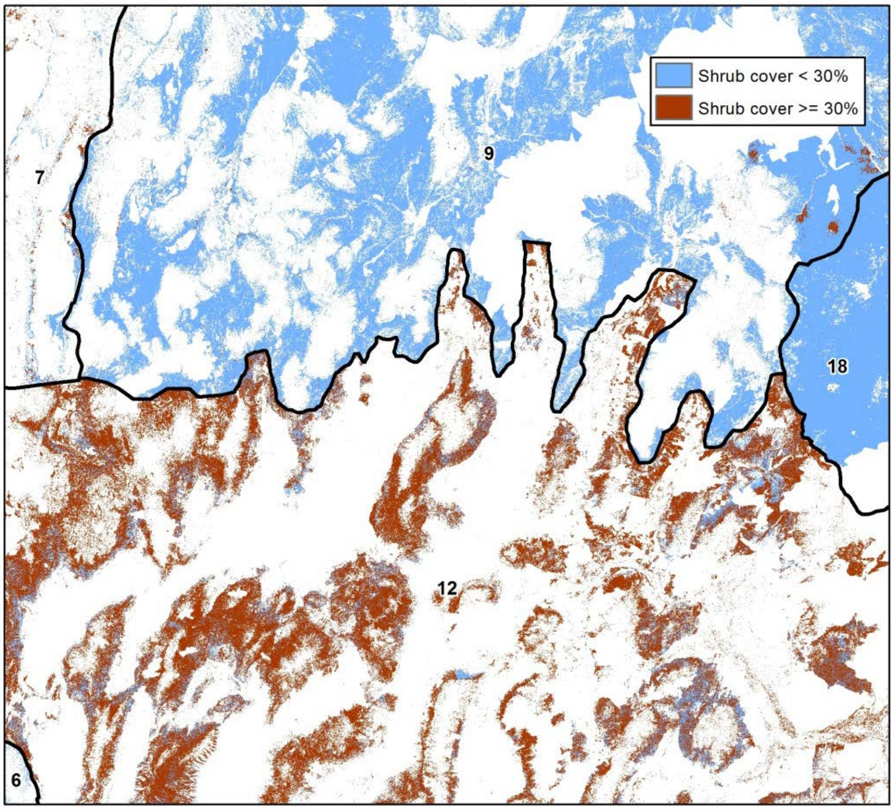


Figure 11. Abrupt change in canopy cover in the Inter-Mountain Basins Big Sagebrush Shrubland existing vegetation type at the boundary between map zones 9 and 12. This can have a profound effect on secondary data layers that use existing canopy cover as a mapping variable.

Succession Class Mapping Rules

LANDFIRE succession class is mapped using a rules-based approach. The mapping rules are based on unique combinations of biophysical setting and existing vegetation type, existing vegetation cover, and existing vegetation height. The rules were developed through a series of workshops by regional experts in vegetation dynamics and fire ecology (Rollins 2009) and are described in both the LANDFIRE vegetation dynamics model descriptions and vegetation dynamics model tracker database available for download from the LANDFIRE website.

One primary consideration in critiquing succession class mapping rules is that the modelers who developed the vegetation dynamics models sometimes emphasized species composition and structure in the definition of classes, while the mappers relied primarily on lifeform and structure to map the classes. As a result, in cases where species composition differentiates between classes of the same structure (Figure 12), LANDFIRE may not have mapped the succession classes appropriately. Post-processing in a GIS may be required to refine the succession class map based on species composition.

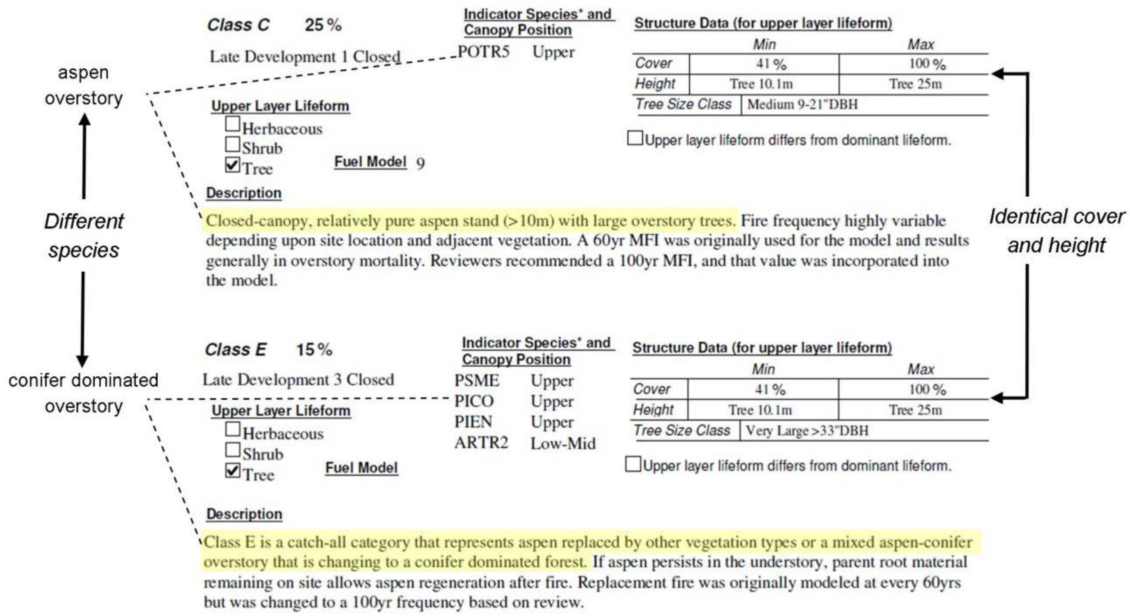


Figure 12. Example of a biophysical setting where species composition (not lifeform) is the primary variable for differentiating between succession classes. As of version 1.2.0 (LANDFIRE 2010) succession class E was not mapped for this biophysical setting in map zone 21 thus requiring GIS post-processing to map it.

Another consideration is that structure, as defined in the vegetation dynamics models, may be difficult to map using remote-sensing based techniques (as is done in mapping LANDFIRE existing vegetation). For example, although a rule may differentiate between succession classes based on whether herbaceous vegetation height is less than or greater than 0.5m, this level of precision is difficult to map accurately using the satellite-based predictive modeling approach described above (Riano et al. 2002). Conversely, the existing vegetation height classes in forested vegetation (Table 7) may be too coarse to accurately differentiate between succession classes (e.g., 10m to 25m and 25m to 50m) or a poor surrogate for vegetative development stage altogether. Chapter 6 contains additional considerations for using the LANDFIRE succession class data layer in vegetation departure analyses.

Chapter 5: Fuels

LANDFIRE produces geospatial data depicting surface and canopy fuel characteristics. For surface fuel data we will focus on the 40 Scott and Burgan fire behavior fuel models data layer (Scott and Burgan 2005), as it is the most commonly used LANDFIRE surface fuel data product. However, the concepts presented in the Considerations section of this chapter are applicable to the other LANDFIRE surface fuel products as well—13 Anderson fire behavior fuel models (Anderson 1982), Canadian forest fire danger rating system fuel types, fuel characteristic classification system fuelbeds, and fuel loading models.

In combination with forest canopy cover, forest canopy height, and topographic data (slope, aspect and elevation), LANDFIRE fire behavior fuel model and canopy fuel data (canopy base height and bulk density) can be used to create a "landscape" file (LCP) required by common geospatial fire behavior modeling systems used in the United States, such as FlamMap (Finney 2006), FARSITE (Finney 1998), and FSPro (USDAFS 2009). Although an LCP file may be downloaded directly from the LANDFIRE data distribution website, we do not discuss the critique or modification of these data in the LCP file format.

This chapter presents an overview of the LANDFIRE fuel mapping process and common considerations for critiquing LANDFIRE fuel data with examples relevant to local applications.

Fuel Mapping Process

Surface Fuels

Technically, a fire behavior fuel model—Anderson (1982) or Scott and Burgan (2005)—is a set of fuelbed inputs required by fire behavior modeling systems that use the Rothermel (1972) fire spread model. More practically speaking, a fire behavior fuel model represents a range of fuelbed conditions (e.g., load, depth, surface-area-to-volume ratios) in which fire behavior may be expected to respond similarly to changes in fuel moisture, slope, and wind speed (Figure 13). In this sense, a fire behavior fuel model is not so much a model of fuels as it is a model of fire behavior.

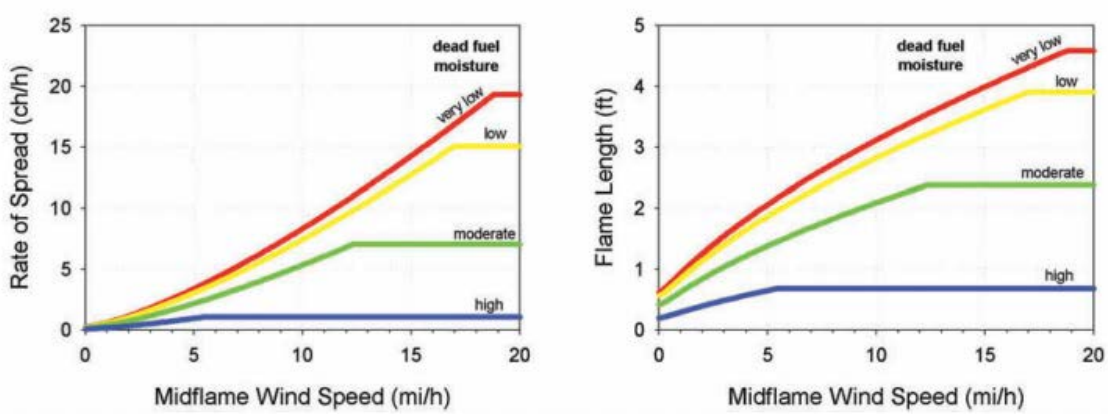


Figure 13. Differences in rate of spread and flame length by dead fuel moisture content and wind speed for fuel model Timber-Understory 1 (Low Load Dry Climate Timber-Grass-Shrub; Scott and Burgan 2005).

Like succession class (Chapter 4), all LANDFIRE surface fuel data products are mapped using an expert-opinion, rule-based approach, where mapping rules are based on unique combinations of: existing vegetation type, cover, and height; biophysical setting; and disturbance (Chapters 3 & 4). Fire behavior fuel model mapping rules were developed by fire and fuel specialists through a series of fuel calibration workshops held across the United States. The purpose of these workshops was to elicit regional expertise about fire behavior characteristics (i.e., how fire burns) in various vegetation types and structures. The calibration workshops were conducted at the extent of a LANDFIRE map zone or multiple adjacent zones. There are 80 LANDFIRE map zones across the continental U.S., Alaska, and Hawaii, ranging in size from 4 to 60 million acres (Figure 2).

The LANDFIRE total fuel change tool (formally known as ToFu Δ) (LFTFCT 2011) is a custom ArcGIS toolbar that links to the national fuel mapping rules through a Microsoft Access database (Figure 14). This tool, originally developed for use in the national calibration workshops, can now be downloaded from the Wildland Fire Management Research, Development and Application website and is highly useful in local LANDFIRE fuel data critiques.

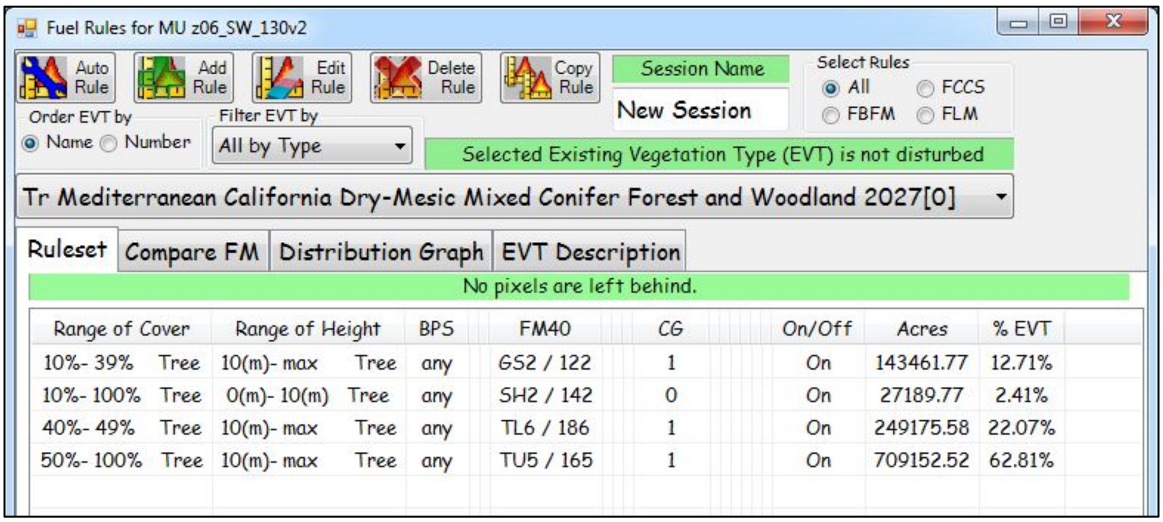


Figure 14. Example LANDFIRE Total Fuel Change Tool (LFTFCT) rule set. The LFTFCT is a custom ArcGIS toolbar that links to the LANDFIRE fuel mapping rules through a Microsoft Access database.

Canopy Fuels

The LANDFIRE canopy fuel data products include canopy base height, canopy bulk density, forest canopy cover, and forest canopy height. Forest canopy cover and canopy height values represent the midpoint of the existing vegetation cover and height data layer classes, respectively. These values are used in some fire behavior modeling systems as variables in predicting dead woody fuel moisture, wind reduction, and crown fire spotting potential. All four variables are required to model crown fire behavior using U.S. fire behavior modeling systems.

Canopy base height is defined as the lowest height above the ground at which there is sufficient available fuel (i.e., ≤ 0.25 inch diameter) to propagate fire vertically through the canopy. It is important to differentiate canopy base height—which is a property of the group of trees represented by the pixel—from *crown* base height, which is a property of an individual tree. Canopy base height is an important variable for fire behavior modeling, as it is used to predict whether crown fire initiation is possible under a given set of environmental conditions (Scott and Reinhardt 2001; Scott 2012). Prior to LANDFIRE 2012, canopy base height was mapped based on plot level averages. Various combinations of existing vegetation type, cover, and height values were crosswalked to an average canopy base height value of associated plots. For the LANDFIRE 2012 canopy base height data layer, a predictive modeling approach was implemented where field referenced plot data were used to develop regression equations based on statistical relationships between canopy base height and existing vegetation type, cover, and height (USGS 2010).

Canopy bulk density is the mass of available canopy fuel per unit canopy volume (Scott and Reinhardt 2001). Like canopy base height, canopy bulk density is a property of a group of trees—*crown* bulk density is a property of an individual tree. In fire behavior modeling, canopy bulk density is used to predict whether an active crown fire is possible under a given set of environmental conditions assuming that a crown fire has initiated (Scott and Reinhardt 2001; Scott 2012). LANDFIRE maps canopy bulk density using a predictive modeling approach based on forest canopy cover, forest canopy height, and membership to a pinyon-juniper existing vegetation type as input to a generalized linear model (Reeves et al. 2009).

In deciduous forest vegetation types—typically not considered prone to crown fire—LANDFIRE assigns canopy base height and canopy bulk density values that prevent fire behavior modeling systems from predicting crown fire. Forest canopy cover and forest canopy height values are still mapped to account for the canopy's effect on fuel moisture and wind reduction.

Fuel Updates

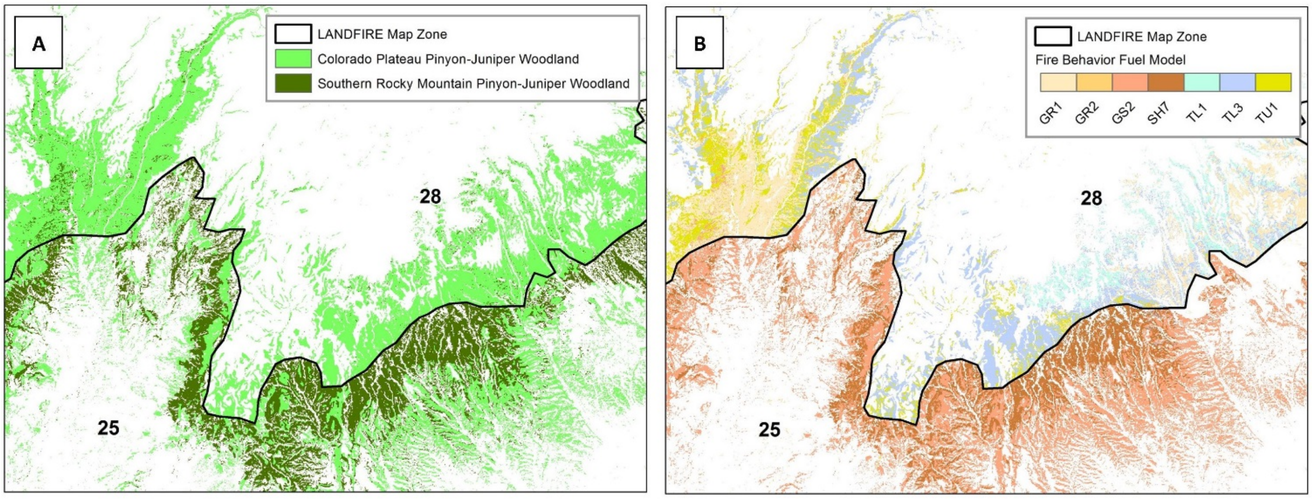
Because surface and canopy fuel mapping rules are tied to existing vegetation type, cover, and height, updates to existing vegetation data due to growth and vegetation succession automatically account for updates to fuels in non-disturbed areas. Whether an update to the fuel data layers occurs or not depends on the magnitude of the change and the threshold values in the mapping rules.

Rules for disturbed areas are independent of rules for non-disturbed areas so that the disturbance type, severity, and time-since-occurrence can be taken into account in combination with the post-disturbance vegetation characteristics, including unique lifeform and species specific disturbance response as discussed in the previous sections. The one-year time-since-disturbance category is used by LANDFIRE to update the immediate post-fire effects to canopy fuels but not used in the assignment of post-disturbance fire behavior fuel model. Fire behavior fuel model is the same for the one-year and two- to five- year time-since-disturbance categories, which are considered to represent the second growing season after the event (C. Martin, personal communication, July 10, 2015).

Considerations

Map Zone Boundaries

As mentioned earlier, fire behavior fuel model mapping rules are developed for individual map zones or groups of adjacent zones based on unique combinations of existing vegetation type, cover, and height; biophysical setting; and, disturbance. It is common to find differences in mapping rules between adjacent map zones that may lead to an "artificial edge" at the zone boundary (Figure 15). In situations where your analysis area overlaps more than one LANDFIRE map zone, a primary consideration is whether there are differences in mapping rules between the zones. If so, determine whether those differences are legitimate or if the rules from one zone more appropriately fit the analysis area as a whole. If working in an area with pinyon-juniper vegetation types, a specific mapping rule issue to watch for is whether or not there are differences between zones in the assignment of canopy fuels. In some cases, the rules for one map zone will consider the canopy fuels in pinyon-juniper vegetation types as part of the surface fuel model, while the rules for an adjacent map zone will not. This may lead to prediction of crown fire on one side of the zone boundary and surface fire on the other. The discrepancies are due to differences in mapping methodology rather than actual fire behavior potential. There may be valid reasons for each case but consistency should be strived for when an analysis area intersects multiple map zones, to ensure consistent interpretation of the results across the entire analysis area.



Existing Vegetation Type	Zone 28				Zone 25			
	Tree Cover (%)	Tree Height	FM	CG	Tree Cover (%)	Tree Height	FM	CG
Colorado Plateau Pinyon-Juniper Woodland	10-29	Any	GR1	1	10-19	Any	GR2	0
	30-39	Any	TU1	1	20-49	Any	GS2	0
	40-59	Any	TL3	1	50-100	Any	TL3	1
	60-100	Any	TL1	1	-	-	-	-
Southern Rocky Mountain Pinyon-Juniper Woodland	10-19	Any	GR2	1	10-29	Any	GS2	0
	20-59	Any	GS2	1	30-49	Any	SH7	0
	60-100	Any	TL3	1	50-100	Any	TL3	1

Figure 15. Example of variation in fire behavior fuel mapping rules by existing vegetation type and map zone. Panel A shows the existing vegetation type at the map zone boundary; panel B shows the fire behavior fuel model. FM refers to the standard Fire Behavior Fuel Model (Scott and Burgan 2005). CG refers to the canopy guide feature in the LANDFIRE Total Fuel Change Tool that controls how canopy fuels are mapped.

Multiple inconsistencies between map zones can be seen in Figure 15. The predominant pinyon-juniper existing vegetation type in map zone 28 is Colorado Plateau pinyon-juniper woodland; in map zone 25 it is southern Rocky Mountain pinyon-juniper woodland (Figure 15A). The fire behavior fuel model mapping rules for these two vegetation types vary both by type and by map zone, resulting in the obvious difference in fuel model seen at the boundary (Figure 15B). Furthermore, in map zone 28, the rules for both vegetation types include the assignment of canopy fuels (i.e., canopy guide of 1); in map zone 25 the rules do not assign canopy fuels to pixels with less than 50% canopy cover, indicating that the trees are part of the surface fuel stratum. This inconsistency forces a different interpretation of fire behavior modeling results for each map zone.

Application Scale and Location

As stated earlier, fire behavior fuel model mapping rules were developed at regional workshops for application to individual, or groups of adjacent, map zones. While these rules may be appropriate at this scale, they may need to be adjusted for application at finer scales. In other words, the “best fit” for an entire map zone may be a compromise between different parts of the zone. For finer-scale applications, fire behavior fuel model mapping rules should be locally critiqued whenever possible. We recommend doing this in a workshop setting, where local specialists with expertise in local fire behavior critique the national mapping rules and make adjustments as needed. Remember, the objective is to choose the fire behavior fuel model that most appropriately simulates the observed or expected fire behavior under a range of fire-environment conditions. It is therefore invaluable to have workshop participants who have seen fire burn under a range of conditions in the local vegetation types.

Another consideration common in more arid locations is whether the fuel models that are appropriate under a typical, or average, yearly weather scenario are appropriate in an atypical scenario. For example, in a typical year, fire behavior in many desert ecosystems may be best represented using a shrub fire behavior fuel model. However, in a year when an unusually wet winter is followed by an influx of annual grasses, the primary carrier of fire will be the herbaceous component and thus fire behavior would be better represented using a grass or grass-shrub fuel model. In this case, two separate versions of fuel data layers could be created to represent the different fuel scenarios.

Similarly, areas with a heavy deciduous tree component may experience very different fire behavior depending on the time of year. In fall, winter, and spring the leaves have fallen from deciduous trees, therefore adding to the load and structure of the surface fuels and associated surface fire behavior. As mentioned above, in deciduous forest vegetation types, LANDFIRE assigns pseudo canopy-fuel values that prohibit the simulation of crown fire in fire behavior modeling systems but retain the actual forest canopy cover and height values for modeling the influence of canopy cover on wind-reduction and fuel moisture. However, in mixed deciduous-conifer existing vegetation types LANDFIRE does not account for the proportion of deciduous-to-conifer cover; canopy bulk density is estimated from the total forest canopy cover. Depending on the proportion of conifer and deciduous trees, canopy bulk density may therefore be overestimated in these stands throughout the year, and wind-reduction and shading may be overestimated during the leaf-off times of the year.

Disturbance

Disturbances may affect both surface and canopy fuels depending on their type and severity. As with undisturbed fuels, the fuel mapping rules for disturbed areas should be critiqued by local fire specialists before application to finer-scale analyses.

In grass and shrub vegetation types the post-disturbance fire behavior fuel model is influenced by the affected species' response to disturbance. For example, wildfire in grass vegetation types is typically high-severity by nature—consuming all of the above-ground biomass. Most grasses, however, return to their pre-fire condition relatively quickly (i.e., one or two growing seasons) and in some cases will respond with increased biomass compared to the pre-fire condition due to an influx of nutrients and more favorable growing conditions. In shrub vegetation types, low-severity fire (less than 25% overstory mortality) may have little effect on the fuel load, fuelbed depth, and other components of a shrub-based fire behavior fuel model, whereas high-severity fire (greater than 75% overstory mortality) may result in immediate resprouting of shrubs or conversion to grass for some period of time, all dependent on the particular species' response to fire.

In tree-dominated vegetation types, low-severity fire will, generally speaking, consume litter (small dead branches and needles on the forest floor) and grass with minimal effect on understory shrubs and small trees. Moderate-severity fire may have a wide range of effects on litter and understory vegetation, but at the pixel level can generally be assumed to have consumed most of the litter and understory vegetation. By LANDFIRE severity definitions, moderate-severity fire in forested vegetation types results in 25% to 75% overstory tree mortality. High-severity fire results in greater than 75% mortality of the overstory trees. The mortality of overstory trees will influence the availability of light, water, and nutrients to understory vegetation, as well as contribute litter (through needle and branch fall) and large woody debris (as dead trees fall) as surface fuels over time.

These same principles apply to non-fire disturbance types. Ask yourself what is the response of the vegetation to the particular disturbance, what influences will this response have on post-disturbance fuel, how fire burns in the disturbed area, and what is the effect of time-since-disturbance.

As discussed in Chapter 3, the generalization of mechanical disturbance types to two categories—mechanical add and mechanical remove—may lead to a misrepresentation of effects. Critique of the LANDFIRE events polygon and individual year disturbance data by local experts can often confirm or provide additional information about the disturbance's effect on fuels.

The effect of time-since-disturbance varies by location and fuel type. Time-since-disturbance is split into three categories, the first of which is "one year". The need for the one year time-since-disturbance category can be evaluated based on your location, how you plan to apply the data, and how frequently you plan, or need, to update it.

Modeling

In-depth discussion of wildfire behavior modeling concepts is beyond the scope of this guidebook. Scott (2012) provides a comprehensive review of the topic in his *Introduction to Wildfire Behavior Modeling* guide. Nevertheless, a few considerations warrant discussion here. Wildfire behavior modeling requires an understanding of how the interaction among vegetation, fuels, and topography—as characterized in LANDFIRE data—influences modeling results. Wildfire analyst support may therefore be desired when critiquing and updating fuel data, depending on local wildfire behavior modeling expertise.

First, there is no direct, repeatable method for measuring canopy base height in the field, and multiple observers will often estimate significantly different values in the same stand. Methods exist to indirectly estimate canopy base height from plot data (Sando and Wick 1972; Cruz et al. 2003; Reinhardt and Crookston 2003; Scott and Reinhardt 2005), but canopy base height is challenging to map at a landscape scale because it is not well-related to characteristics that can be measured by remote sensing techniques. Canopy base height may include ladder fuels such as lichen, dead branches, needle drape, small trees, and shrubs. However, if shrubs and small trees are being considered as part of the fire behavior fuel model, they should not also be included in the canopy base height.

Next, understanding the interaction of fire behavior fuel model and canopy base height on modeling results is crucial in critiquing fuel data. The fire behavior fuel model predicts the surface fire intensity under a given set of environmental conditions (e.g., wind speed, slope steepness, fuel moisture). The lower the canopy base height, the milder these conditions can be in order to initiate crown fire. Given the difficulty of measuring and mapping canopy base height, working backwards— that is, adjusting canopy base height based on the conditions expected to initiate crown fire—is an effective way to critique canopy base height in relation to other variables. Tools such as NEXUS (Scott 1999) and BehavePlus (Andrews 2013) can provide information on the torching index—20' wind speed required for crown fire initiation—under various fuel and fire environments. The LFTFC tool also includes an option for calculating the critical canopy base height needed for crown fire initiation for different combinations of fire behavior fuel model, fuel moisture, and wind speed (Figure 16).

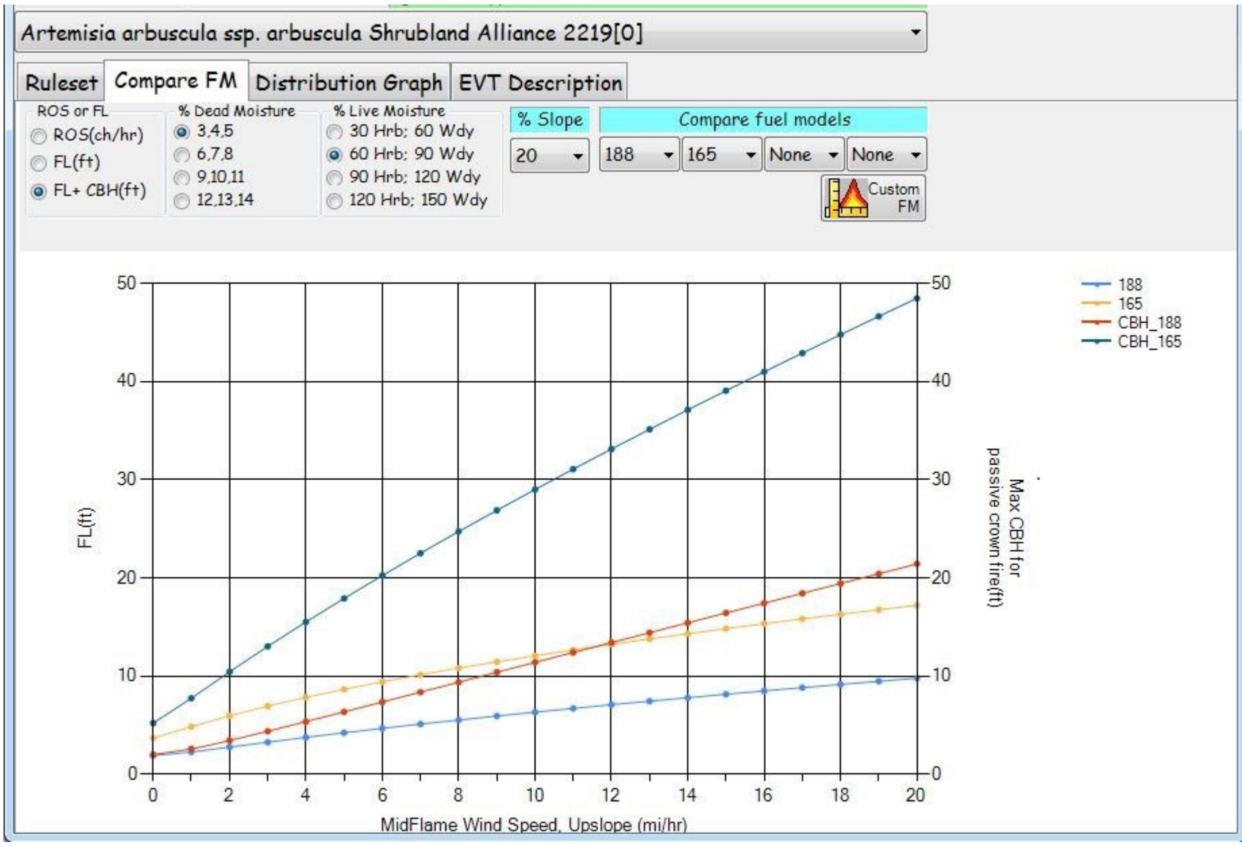


Figure 16. LANDFIRE Total Fuel Change Tool compare fuel model tab. This allows the user to calculate the critical canopy base height needed for crown fire initiation for different combinations of fire behavior fuel model, fuel moisture, and wind speed.

Lastly, in fire behavior modeling, canopy bulk density is a factor in determining whether an active crown fire can be sustained once initiated. Since the existing vegetation height classes used to predict canopy bulk density are rather coarse, they influence the resulting precision of the canopy bulk density values as well. Again, tools such as NEXUS and BehavePlus can be useful in determining if the data will predict the expected fire behavior under various conditions. Analysts are also encouraged to run geospatial fire behavior modeling systems to see if patterns in the results reveal any potential calibration issues that warrant a closer look. This is the *analyze* component of the data critique and modification framework discussed in Chapter 2.

Chapter 6: Fire Regime and Vegetation Departure

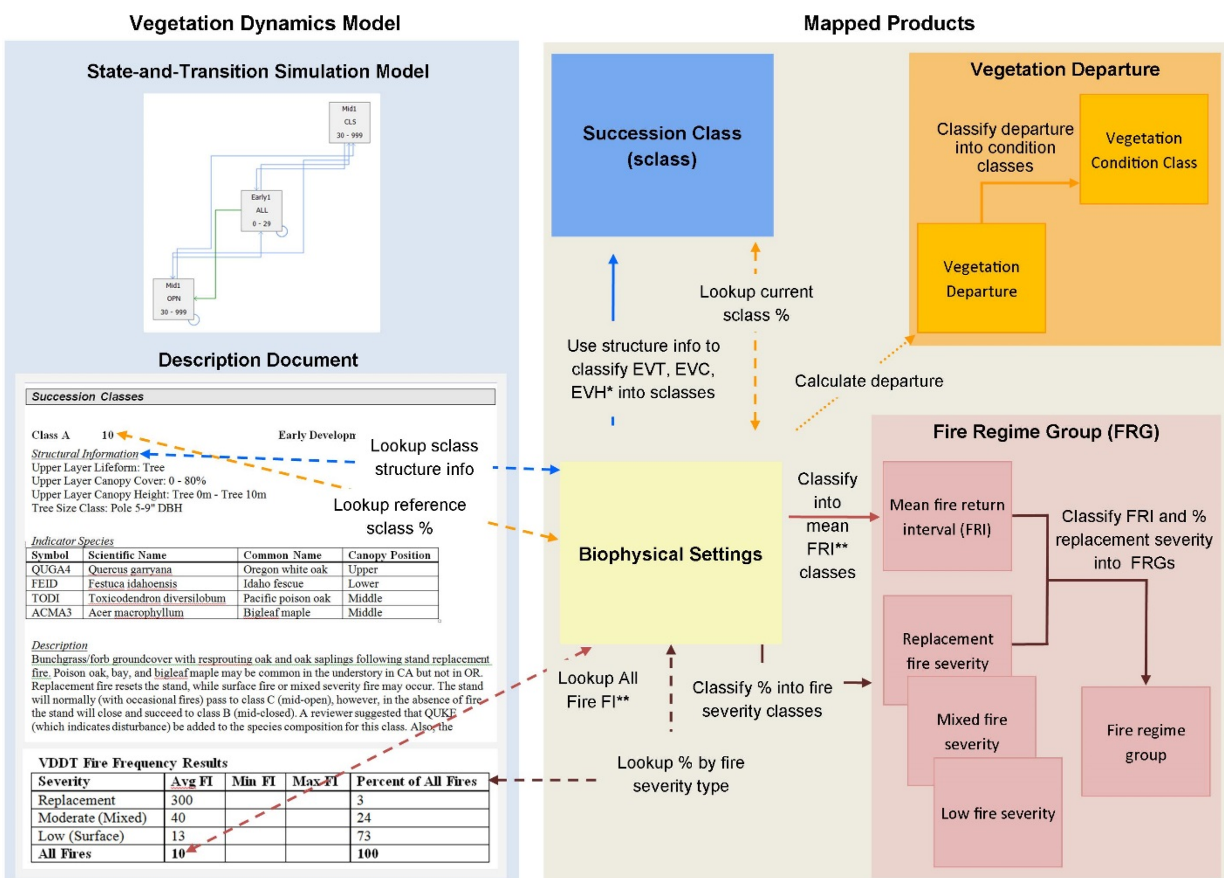
The fire regime and vegetation departure products are useful for understanding historical fire regimes and the current condition of vegetation on the landscape within the context of the historical disturbance regime. The fire regime products include fire regime group, mean fire return interval, percent low-severity fire, percent mixed-severity fire, and percent replacement-severity fire. The vegetation departure products include vegetation departure and vegetation condition class. The departure products were created for the LANDFIRE National, 2001, 2008 and 2012 versions but not for LANDFIRE 2010.

This chapter begins with an overview of the vegetation dynamics models, which form the basis of the fire regime and vegetation departure products, and then describes how those products are mapped by LANDFIRE. The chapter concludes by presenting common considerations for critiquing these data layers and provides examples of common pitfalls.

Fire Regime Mapping Process

Vegetation Dynamics Models

The foundation of the fire regime and vegetation departure products is a set of models that describe the vegetation dynamics and reference conditions of each biophysical setting mapped by LANDFIRE (Figure



*Existing vegetation type (EVT), cover (EVC) and height (EVH)
**Fire Interval (FI) and Fire Return Interval (FRI) refer to the average fire frequency modeled.

17). This section therefore begins with a brief overview of the models and how they relate to the fire regime and vegetation departure products. More information on the vegetation dynamics models can be found on the LANDFIRE program website.

Figure 17. The fire regime and vegetation departure products are created through crosswalks that link each BpS on the BpS data layer to the reference condition values modeled in the corresponding vegetation dynamics model.)

LANDFIRE collaborated with vegetation and fire ecology experts to create a vegetation dynamics model to estimate the reference (i.e., pre-Euro-American settlement) condition for each biophysical setting. The models were created in the Vegetation Dynamics Development Tool (VDDT, ESSA Technologies Ltd. 2007). A model represents a single biophysical setting and consists of five or fewer successional states, or classes, that compose the biophysical setting (Figure 18). Each state is equivalent to a succession class and each succession class is mapped in the succession class data layer (Chapter 4). A state has an age range that indicates how long it typically persists before it transitions to the next state. Disturbance pathways between states are used to represent the impact of important disturbances, and each pathway is defined by a probability that describes how often it occurs. The models were attributed based on scientific literature, available data, and the experience and judgment of the modelers (Rollins 2009).

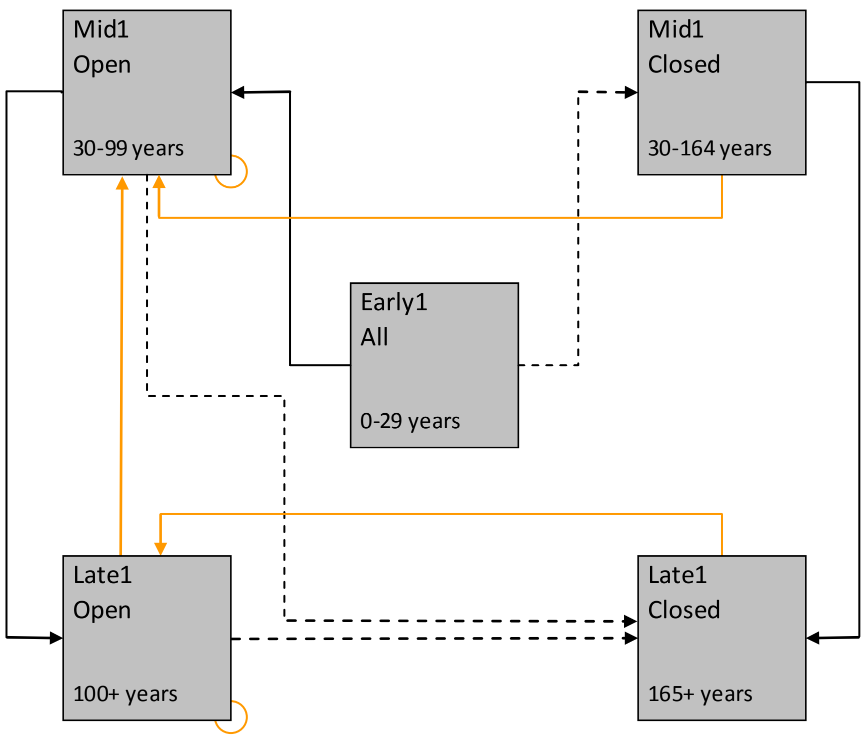


Figure 18. State-and-transition model of the Pacific Northwest Mixed Conifer BpS. This model is comprised of five successional states (boxes). Each state has an age range and is linked to other states through main successional pathways (solid lines), alternative succession pathways (dashed lines) and disturbance pathways (yellow line represent mixed fire transitions).

Once attributed, each model was run for ten 1,000-year simulations, in VDDT, and the results were averaged to estimate the biophysical setting reference conditions. The reference conditions include:

- the fire frequency expressed as a mean fire return interval,

- the fire severity expressed as the relative percent of low-, mixed-, and replacement-severity fire, and
- the relative amount represented by each succession class expressed as a percent.

The reference conditions are published in the LANDFIRE model descriptions along with the VDDT models available from the LANDFIRE website (Figure 19).

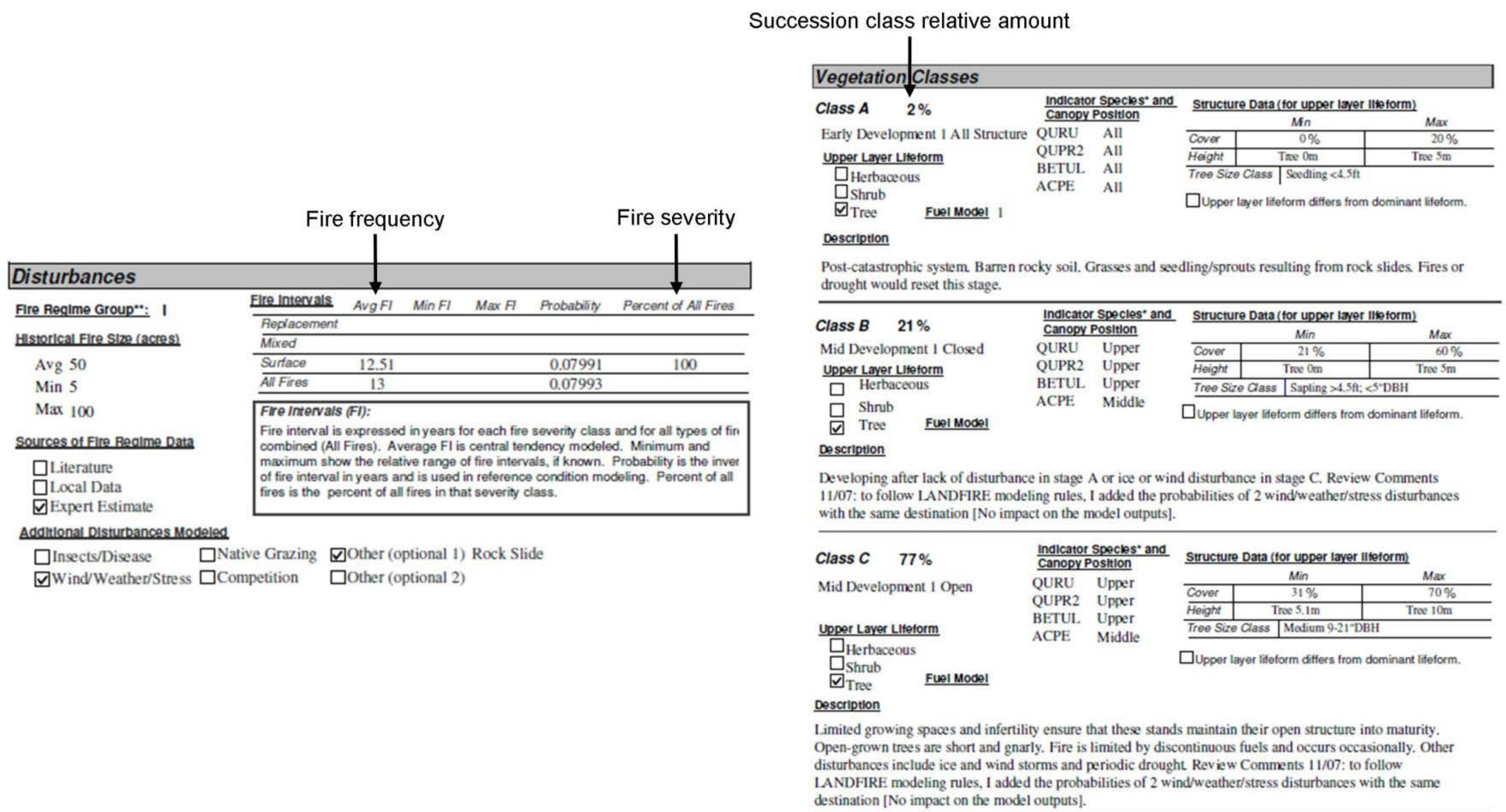


Figure 19. The biophysical setting model descriptions contain the reference conditions. The fire frequency and severity are found in the Disturbances section. The relative amount represented by each succession class is expressed as a percent and is found after the class letter name in the upper left of each vegetation class description.

Vegetation Dynamics Models and Biophysical Setting Map Units

The LANDFIRE biophysical setting data layer contains attributes for two nested map units: biophysical setting and biophysical setting groups. The biophysical setting attribute is the original biophysical setting classification based on NatureServe's Ecological Systems and described in the vegetation dynamics model description documents. LANDFIRE created biophysical setting groups to simplify the mapping of the fire regime products and to reduce the complexity of the vegetation dynamics model set for users. The original units were placed into groups based on similar vegetation and fire regime characteristics. Each biophysical setting group is represented by a single "exemplar" model chosen from the original model set. The fire regime products and the succession class data layer in LANDFIRE 2001 and 2008 are based on the biophysical setting groups and their associated exemplar models. All other LANDFIRE versions, including the most recent versions, use the original biophysical setting attribute. Although the biophysical setting and the biophysical setting groups are related, they can have different succession class definitions and different reference conditions, including different succession class proportions and fire frequency and severity values. Users need to pair the correct biophysical setting attribute in the biophysical setting data layer with the correct model based on the version of LANDFIRE data they are using. The relationship between biophysical setting and biophysical setting groups is described in the "BpS Groups Table" located on the LANDFIRE website (Figure 20).

Original biophysical settings | Biophysical settings group

Table

bps130gb

VALUE	COUNT	BPS_CODE	ZONE	BPS_MODEL	BPS_NAME	GROUPMODEL	GROUPNAME
499	108981	10800	13	1310800	Inter-Mountain Basins Big Sagebrush Shrubland	182	Wyoming Big Sage-Rubber Rabbitbrush-4
637	2754028	10800	9	910800	Inter-Mountain Basins Big Sagebrush Shrubland	178	Wyoming Big Sage-Spiny Hopsage-1
669	1443011	10800	8	810800	Inter-Mountain Basins Big Sagebrush Shrubland	178	Wyoming Big Sage-Spiny Hopsage-1
824	713470	10800	7	710800	Inter-Mountain Basins Big Sagebrush Shrubland	178	Wyoming Big Sage-Spiny Hopsage-1
1061	4428603	10800	12	1210800	Inter-Mountain Basins Big Sagebrush Shrubland	182	Wyoming Big Sage-Rubber Rabbitbrush-4
1091	1984361	10800	17	1710800	Inter-Mountain Basins Big Sagebrush Shrubland	182	Wyoming Big Sage-Rubber Rabbitbrush-4
1232	414495	10800	18	1810800	Inter-Mountain Basins Big Sagebrush Shrubland	182	Wyoming Big Sage-Rubber Rabbitbrush-4
642	4533089	11250	9	911250	Inter-Mountain Basins Big Sagebrush Steppe	220	Wyoming Big Sage-Wheatgrass-3
674	1098430	11250	8	811250	Inter-Mountain Basins Big Sagebrush Steppe	220	Wyoming Big Sage-Wheatgrass-3
834	4467517	11250	7	711250	Inter-Mountain Basins Big Sagebrush Steppe	220	Wyoming Big Sage-Wheatgrass-3
1068	670979	11250	12	1211250	Inter-Mountain Basins Big Sagebrush Steppe	221	Wyoming Big Sage-Wheatgrass-4
1102	784843	11250	17	1711250	Inter-Mountain Basins Big Sagebrush Steppe	221	Wyoming Big Sage-Wheatgrass-4
1238	5522530	11250	18	1811250	Inter-Mountain Basins Big Sagebrush Steppe	221	Wyoming Big Sage-Wheatgrass-4

Figure 20. LANDFIRE biophysical settings were placed into groups based on similar vegetation and fire regime characteristics. Each biophysical setting group is represented by a single "exemplar" model chosen from the original model set. For example, in the table above notice that the seven original Inter-Mountain Basins Big Sagebrush Shrubland biophysical settings models were lumped into two groups: Wyoming Big Sage-Rubber Rabbitbrush-4 and Wyoming Big Sage-Spiny Hopsage-1.

Fire Regime: Frequency, Severity, and Fire Regime Group

The mean fire return interval data layer depicts the presumed historical fire frequency for each biophysical setting. The layer is created by linking the biophysical setting to the VDDT-modeled fire frequency results described in the vegetation dynamics model description document. The mean fire return interval is classified into 22 categories that vary in length to provide greater temporal resolution for frequently burned biophysical settings and less temporal resolution for biophysical settings that burn infrequently.

The fire severity data layers depict the relative percent of low-, mixed-, and replacement-severity fire under the presumed historical fire regime for each biophysical setting. Fire severity is defined as the percent mortality of the overstory vegetation: less than 25% mortality is classified as low-severity, 25-75% mortality is classified as mixed-severity, and greater than 75% mortality is classified as high-severity. The layer is created by linking the biophysical setting to the VDDT-modeled relative amount of each fire severity type as reported in the vegetation dynamics model description document. The results range from 0-100% and they are classified and mapped in 5% increments.

The fire regime group data layer characterizes the presumed historical fire frequency and percent replacement severity fire for each biophysical setting in five classes (Table 8). The fire regime group layer is created by linking the biophysical setting to the fire frequency and severity results described in the vegetation dynamics model description document.

Table 8: Fire regime group mapping rules. The fire regime group layer is created using a rule set that classifies combinations of fire frequency and relative percent replacement severity fire into one of five fire regime groups for each biophysical setting.

Fire Regime Group	All Fire Frequency (years)	Relative Percent Replacement Severity Fire
I	0-35	<66%
II	0-35	>=66%
III	36-100	<80%
	101-200	<66%
IV	36-100	>=80%
	101-200	>=66%
V	>=201	Any fire severity

All of the fire regime products include additional map units for water, snow/ice, barren, and sparsely vegetated systems which are mapped from the existing vegetation type data layer. The value "indeterminate fire regime characteristics" identifies a biophysical setting without fire disturbance in its associated vegetation dynamics model. These are typically biophysical settings that are either too wet or too dry to carry fire (e.g., Alaskan Pacific Maritime Sitka Spruce Forest biophysical setting).

Vegetation Departure

LANDFIRE provides geospatial data that characterize two metrics of vegetation departure: stratum vegetation departure and stratum vegetation condition class. Vegetation departure and vegetation condition class are calculated following the methodology described in the FRCC Guidebook (Barrett et al. 2010) and the FRCC Mapping Tool User's Guide (Jones and Ryan 2012). Both metrics describe the overall departure of the current vegetation conditions from the historical, or reference, vegetation conditions across all succession classes within a particular biophysical setting (i.e., stratum). The historical proportion, or relative amount, of each succession class in a biophysical setting is based on the average proportion modeled in the vegetation dynamics model and reported in the model description document (Figure 17). Current succession class proportions are calculated directly from the succession class data layer.

Stratum vegetation departure is calculated by determining the succession class "similarity" (the smaller of the reference, or the current proportion, for each succession class), summing the similarities, and then subtracting from 100. This provides the percent departure for a biophysical setting and that value is mapped in the vegetation departure data layer. To create the vegetation condition class data layer, the percent departure is classified into three classes: 0-33% departure in condition class 1, 34-66% departure in condition class 2, and 67-100% departure in condition class 3 (see sidebar).

Calculating Vegetation Departure

Stratum vegetation departure is calculated by comparing the reference distribution of succession classes (i.e., the proportion that each contributes to the whole expressed as a percent) to the current distribution of succession class for individual biophysical settings. In the table below, departure is calculated for a biophysical setting with three reference succession classes (A, B, and C), which are defined in its vegetation dynamics model. The Uncharacteristic succession class only includes a current value because by definition it does not occur under the reference condition. The uncharacteristic class proportion is the sum of the uncharacteristic native and uncharacteristic exotic proportions.

The first step in calculating stratum vegetation departure is to determine the succession class similarity (i.e., the lower of the reference or current percent) of each succession class. Next, stratum similarity is calculated by summing the succession class similarity values. Then, the current stratum vegetation departure is calculated by subtracting the stratum similarity value from 100. This is the value mapped in the LANDFIRE vegetation departure grid. Finally, the vegetation condition class is calculated by classifying the current stratum vegetation departure value into the three condition classes (1 = ≤ 33%, 2 = > 33% to ≤ 66%, 3 = > 66%). This is the value mapped in the LANDFIRE Vegetation Condition Class grid:

Succession Class (S-Class)	Reference Percent	Current Percent	S-Class Similarity
A-Early	15	3	3
B-Mid	40	25	25
C-Late	45	31	31
Uncharacteristic	0		
Stratum Similarity	**59**		
Current Departure	**41**		
Vegetation Condition Class	**2**		

Departure is calculated for a specific geographic area referred to as the landscape summary unit. For LANDFIRE National, departure was calculated for ecological subsections (Cleland et al. 2005) within a LANDFIRE map zone. In LANDFIRE 2001 and 2008 departure was calculated within nested hydrologic unit codes (HUCs; Seaber et al. 1987). Departure for biophysical settings in fire regime groups I and II was calculated at the sub-watershed level (HUC 12); biophysical settings in fire regime group III were calculated at the watershed level (HUC 10); and biophysical settings in fire regime groups IV and V were calculated at the sub-basin level (HUC 8). In LANDFIRE 2012 the landscape summary unit was defined as a biophysical setting with identical reference condition values regardless of map zone. To understand this, imagine that a biophysical setting is mapped in map zones 1, 2, and 3 and that zones 1 and 2 have identical reference conditions in their associated vegetation dynamics models but that zone 3 has a unique set of reference conditions. In this case, the departure would be calculated using the biophysical setting's extent in zones 1 and 2 as one summary unit and zone 3 as another summary unit.

Considerations

Understanding the Source Data

All of the fire regime and vegetation departure products are derived from other LANDFIRE products. Any assumptions, limitations, and issues associated with the source data are inherited by the fire regime and vegetation departure products. To understand and critique these products, the user must therefore understand the source data. The fire frequency, fire severity, and fire regime group values come from the vegetation dynamics models. Vegetation departure and vegetation condition class results are derived from the modeled reference conditions, the biophysical setting data layer, the succession class data layer, (which is itself derived from the biophysical setting, existing vegetation type, existing vegetation cover, and existing vegetation height data layers; see Chapter 4), and the landscape summary unit. The information from other chapters in this guide will help the user critique these geospatial data inputs. For more information on critiquing the vegetation dynamics models, refer to the *Reviewing and Modifying LANDFIRE Vegetation Dynamics Models* (The Nature Conservancy 2011a) user's guide.

Knowledge Uncertainty

The quality of the fire regime and vegetation departure products depends to a great extent on the quality and quantity of the information used to create the vegetation dynamics models. In general, there are more data to attribute models for economically valuable and heavily studied biophysical settings, such as forested ecosystems, than there are for biophysical settings with little economic value and those that are rare (Blankenship et al.2012). The quantity and quality of fire regime information also varies considerably based on the characteristics of the vegetation comprising the biophysical setting. Fire history from recent centuries tends to be most reliably documented in systems where the evidence of low- and moderate-severity fires is recorded and persists within the annual rings of long-lived tree species (Swetnam et al. 1999) such as longleaf pine and ponderosa pine, and/or where the time since the last stand-replacing fire can be determined from the stand age. In non-forested systems, little direct evidence persists for inferring the characteristics of historical fire regimes (Swetnam et al. 1999) although historical records, charcoal and pollen records, and dependence or sensitivity of long-persisting species provide clues to the fire frequency and severity. The vegetation dynamics model description documents often provide information about the sources and the quality of the information on which they are based and can provide users with valuable information for evaluating the fire regime products derived from them.

Map Zone Boundaries

The vegetation dynamics models were developed to apply at the level of a LANDFIRE map zone (Figure 2). Sometimes the same biophysical setting may have different succession class mapping rules, succession class reference proportions, and fire frequency and severity information in different map zones. This can lead to abrupt changes in the fire regime and vegetation departure products at map zone boundaries, even for the same biophysical setting. Users performing an independent departure analysis can address this issue (see Vegetation Departure Analysis below).

Changes in Departure Methods

The methods LANDFIRE used to create the departure products have changed between versions (Table 9). Users should be cautious when comparing the departure products (vegetation departure and vegetation condition class) between different LANDFIRE versions because changes in the biophysical setting map units and the landscape summary unit discussed above, as well as the source of the reference conditions, can change the departure score. Theoretically the LANDFIRE 2001 and 2008 departure data layers are comparable because they were calculated using the same method, but it may be too short a time period to see substantial change across broad areas. LANDFIRE 2001, 2008 and 2012 departure data layers are not comparable to LANDFIRE National because of the changes in the methods (USFS [n.d.] Fire Regime Data…).

Table 9: Comparison of the methods and input data used to create the LANDFIRE departure data products by data version.

Version	Departure Products Mapped	Departure BpS Unit[a]	Summary Unit	Reference Condition Source[b]
National	Yes	BpS	Ecological Subsections within Mapzones	VDDT & LANDSUM
2001	Yes	BpS Group	Nested Hydrologic Unit Codes	VDDT
2008	Yes	BpS Group	Nested Hydrologic Unit Codes	VDDT
2012	No	BpS	Unique Combination of BpS Code and BpS Model	VDDT

[a]Vegetation departure products were calculated for the biophysical setting (BpS) or the BpS groups depending on the version. In versions where departure products were not mapped, the Departure BpS Unit refers to the units used to map the fire regime and succession class layers.
[b]The reference conditions were derived from the Vegetation Dynamics Development Tool (VDDT) and the Landscape Succession Model (LANDSUM). For LANDFIRE versions 2001 and greater the reference condition source is as described in this guide. The reference conditions source for the National version is described in the document "Developing the LANDFIRE Fire Regime Data Products" on the LANDFIRE Program website.

Vegetation Departure Analysis

Rather than use the LANDFIRE vegetation departure products as-is, many users prefer to complete their own, local, departure analysis. Performing an independent departure analysis allows users to address the issues discussed above, critique and refine the succession class mapping rules, and integrate ancillary data (e.g., locally mapped invasive species distribution). The Fire Regime Condition Class Mapping Tool also allows for the calculation of additional vegetation departure metrics beyond stratum vegetation departure and stratum vegetation condition class, as well as fire *regime* departure analysis. In addition to the considerations listed above, there are some considerations specific to an independent departure analysis using LANDFIRE data.

Biophysical Setting Thematic Resolution

Users performing an independent departure analysis may want to consider the thematic resolution (Chapter 1) of the biophysical setting data layer in relation to their analysis objectives (Chapter 2), especially if there are concerns about the source data or knowledge uncertainty as discussed above. Using the biophysical setting group attribute is one way to "coarsen" the biophysical setting data layer to a more appropriate thematic resolution, but careful critique of the "exemplar" vegetation dynamics model associated with the biophysical setting group is critical. In some cases, the user may want to choose a different "exemplar" model that better represents the biophysical setting group for their analysis location.

Biophysical setting classes may also be grouped using local, ancillary information. For example, in a vegetation condition analysis of Southern Sierra National Forests, analysts grouped models based on similarity of vegetation characteristics and fire regimes following a crosswalk between LANDFIRE biophysical setting and presettlement fire regime groups presented in Van De Water and Safford (2011), thus reducing the number of biophysical settings from 25 to 15.

If biophysical settings are grouped to coarsen the thematic resolution of the biophysical setting data layer, the user will usually be required to manually map succession class due to differences in succession class definitions between the original and chosen vegetation dynamics models. The guide *How to Map Successional Stages Using LANDFIRE Products* (The Nature Conservancy 2013) provides step-by-step instructions on how to do this.

Biophysical Settings that Cross Map Zone Boundaries

In situations where the analysis area overlaps more than one LANDFIRE map zone, a primary consideration is whether there are differences in the vegetation dynamics models between zones, and if such differences reflect reality. If the map zone boundary reflects an ecological transition, then differences between models for the same biophysical setting may be acceptable and necessary. However, if the map zone boundary creates an artificial demarcation in the analysis area, users will want to choose the model that best fits the analysis area and make the appropriate modifications to the related geospatial data. If a new biophysical setting model is chosen, the succession class data layer will need to be adjusted so that it reflects the succession class mapping definitions of the new model (the guide *How to Map Successional Stages Using LANDFIRE Products* provides instructions for re-mapping succession classes) (The Nature Conservancy 2013).

Succession Class Mapping Rules

It is particularly important to critique the succession class mapping rules because the vegetation departure calculation is very sensitive to the amount of area mapped to each succession class. The LANDFIRE succession class data layer is created by applying rule sets to combinations of biophysical setting, existing vegetation cover, existing vegetation height, and to a lesser extent existing vegetation type (Chapter 4). Any problems in the input data layers will carry through to the succession class data layer. Three general concerns with the succession class mapping rules that can impact departure assessments are: 1) the mappability of the classes; 2) the completeness of the succession class rule set; and, 3) the classification of uncharacteristic types.

> *Mappability of Succession Classes*. Succession class is a concept that can be difficult to translate into mappable criteria. Height and cover, the primary variables LANDFIRE uses to map succession class, may not always be the best surrogate for vegetative development and can be difficult to map (Chapter 4). In particular, the height classes for shrub and herbaceous lifeforms are difficult to discern using LANDFIRE's two dimensional satellite imagery. For example, it may be difficult to distinguish 0.5m tall grass from 1.0m tall grass using Landsat data, but some succession classes are mapped based on this distinction. In forests, the height classes tend to be mapped more accurately (see Chapter 4 - Existing Vegetation), but they may be too coarse to adequately differentiate succession classes (e.g., 10 to 25m and 25 to 50m).
>
> *Completeness of the Rule Set.* Ideally, the succession class rule sets would cover all possible mapped combinations of existing vegetation type, existing vegetation cover, and existing vegetation height for every biophysical setting without gaps or overlaps. In other words, the rules should be mutually exclusive and exhaustive, but this is not the case for all LANDFIRE succession class rules.

Take, for example, a hypothetical shrub biophysical setting with two succession classes defined as follows:

A - shrubs 10-100% cover and height < 1m

B – shrubs 50-100% cover and height > 0.5m

In this case shrubs .5-1m tall with >50% cover could be classified in either succession class A or B; the rule set is not mutually exclusive.

Take another hypothetical example of a tree-dominated biophysical setting:

A – trees 0-100% cover and < 5m height; or herbs or shrubs 0-100% cover and “any” height

B – trees 0-100% cover and 5-10m height

C – trees 0-100% cover and 10-25m height

In this example, if trees are not established or trees are less than 5m in height, the pixel is mapped as succession class A. Trees that are 5-10m in height are mapped as succession class B and trees 10-25m in height are mapped to succession class C. What about trees greater than 25m in height? Did the model developers intend for this condition to be mapped as uncharacteristic? In many cases this is not the intent; rather, the rule was developed before the geospatial data were mapped and the modelers chose the most reasonable height class without knowledge of the possible mapped height range. When the rules are not exhaustive and/or mutually exclusive, pixels can be mapped into an inappropriate class.

Users also should watch for rules that overlap in structure (cover and height) but differ by species composition. Some vegetation dynamics model descriptions use existing vegetation type as criteria for distinguishing between succession classes, but it was not a primary variable used in mapping—although this varies by biophysical setting and data version. In these cases the succession class assigned by LANDFIRE may not be in agreement with the vegetation dynamics model description. For example, in LANDFIRE map zone 21, the Rocky Mountain Aspen Forest and Woodland vegetation dynamics model differentiates between succession classes C and E by species composition (Figure 12). Both classes have the same structural criteria but succession class C represents a “relatively pure aspen stand,” whereas succession class E represents “aspen replaced by other vegetation types or a mixed aspen-conifer overstory that is changing to a conifer dominated forest.” These classes should be differentiated by existing vegetation type, but as recent as LANDFIRE 2010 no pixels were mapped to succession class E because existing vegetation type was not used in the succession class mapping process.

If manually mapping succession class, the existing vegetation type data layer can be used to mitigate this issue. For instance, where the structural criteria are met, succession class C would be assigned to pixels classified as the Rocky Mountain Aspen Forest and Woodland existing vegetation type; succession class E would be assigned to pixels classified as an aspen-mixed conifer or a pure conifer existing vegetation type.

Classification of Uncharacteristic Types. LANDFIRE classifies uncharacteristic vegetation as either uncharacteristic native or uncharacteristic exotic (Chapter 4). The uncharacteristic native class indicates that the existing characteristics (i.e., cover, height, and composition) of native vegetation are outside the reference condition range. When conducting a local vegetation departure analysis users may want to critique the mapping rule thresholds for local relevance. For example, if the maximum canopy cover in the vegetation dynamics model is 40%, any cover greater than 40% will be mapped as uncharacteristic native. Does local research of reference conditions corroborate the 40% threshold? Another instance in which the succession class might be mapped as uncharacteristic native is when a *native* riparian existing vegetation type is mapped to a non-riparian biophysical setting. As discussed in Chapter 4 this situation may be due to differences in the mapping methodologies for biophysical setting and existing vegetation type (see Chapter 4 - Potential vs. Existing Vegetation Type Rectification). Users may wish to further critique the data in such situations.

The uncharacteristic exotic class indicates that an exotic species has become established in an area. Succession class is mapped as uncharacteristic exotic wherever an "introduced" existing vegetation type is mapped (e.g., introduced upland vegetation-perennial grassland and forbland). A consideration related to the presence of exotic species is that LANDFIRE classifies less than 10% vegetation cover as "sparsely vegetated." For some analysis objectives, it may be important to identify sparse cover of exotics, such as cheatgrass (*Bromus tectorum)*, and this may require ancillary data sources (Provencher et al. 2009).

Landscape Summary Unit

Independent vegetation departure analyses are not tied to the landscape summary units used by LANDFIRE. The key criterion for landscape delineation is that the summary unit needs to be large enough to encompass the full range of succession classes expected under the historical disturbance regime (Barrett et al. 2010). Careful consideration should be given to the choice of the landscape summary unit using the guidance in the Fire Regime Condition Class Guidebook and Fire Regime Condition Class Mapping Tool User's Guide, keeping in mind that departure scores may vary with changes in the summary unit. If the landscape summary unit is so small that it would not contain the full range of succession classes under the historical disturbance regime, misleading departure scores can result, and lead to errors in the subsequent planning process (Barrett et al. 2010). In contrast, summary units that are too large may make it difficult to discern changes in departure due to planned (e.g., restoration treatments) and unplanned disturbances (Barrett et al. 2010). This may be the case for some biophysical settings under the LANDFIRE 2012 methodology for mapping departure, in which the full extent of the biophysical setting in one or multiple map zones is used as the summary unit to calculate departure. However, it is the intent of the off-the-shelf LANDFIRE products to assess departure at a much broader scale than that of a typical local analysis.

Chapter 7: Interpreted Examples

In this chapter, we (the authors) illustrate the data critique and modification process in two example applications. The first example critiques LANDFIRE data for use in fire behavior analysis of the Rogue Basin located in southwest Oregon (Figure 21). The second example focuses on the critique of LANDFIRE data for use in vegetation departure analysis in the southern Sierra Nevada Mountains of California (Figure 21).

Figure 21. Project area boundaries for interpreted examples.

There are multiple approaches and tools available for critiquing and modifying geospatial data. In these examples we demonstrate the use of common approaches and tools that are available to most natural resource professionals. The following examples summarize the concepts and considerations for modifying LANDFIRE data discussed in previous chapters and therefore should be beneficial to all readers regardless of expertise in working with geospatial data. Details on geospatial analysis and data manipulation tasks, however, are beyond the scope of this document and are only outlined here.

Example 1: Critiquing LANDFIRE data for local fire behavior analysis

Define objectives

For this example we turned to the 3.3 million-acre Rogue Basin in southwest Oregon, where the Southern Oregon Forest Restoration Collaborative and its partners are undertaking the development and implementation of a cohesive forest restoration strategy. A key component in the development of this strategy was an understanding of the current wildfire hazard and associated risk to the Basin's natural resources and assets. Our objective was to conduct a wildfire hazard analysis using LANDFIRE data and geospatial wildfire behavior modeling software.

Identify data requirements

Eight geospatial data layers are required inputs for simulating the full range of wildfire behavior—surface through active crown—in the geospatial fire modeling systems used in this analysis. These layers characterize surface fuels (fire behavior fuel model), canopy fuels (canopy base height and canopy bulk density), forest canopy structure (canopy cover and canopy height), and topography (elevation, aspect, and slope). Each geospatial data layer is available from LANDFIRE.

Given our objective to geospatially analyze wildfire hazard, it was important that the geospatial data represent the fuels and wildfire potential as appropriately as possible for the scale of the analysis. To evaluate the LANDFIRE fuels data we would use the LANDFIRE Total Fuel Change Tool (LFTFCT 2011), which allows for the critique, modification, and analysis of fuel mapping rules and their effect on simulated fire behavior within the tool itself. Because LANDFIRE fuel data (Chapter 5) are derived from existing vegetation type, cover, and height (Chapter 4), biophysical setting (Chapter 4), and disturbance (Chapter 3), the tool requires these geospatial data layers as input, thus increasing our data requirements. We downloaded the additional data layers using the LANDFIRE Data Access Tool (Figure 22, LFDAT 2012).

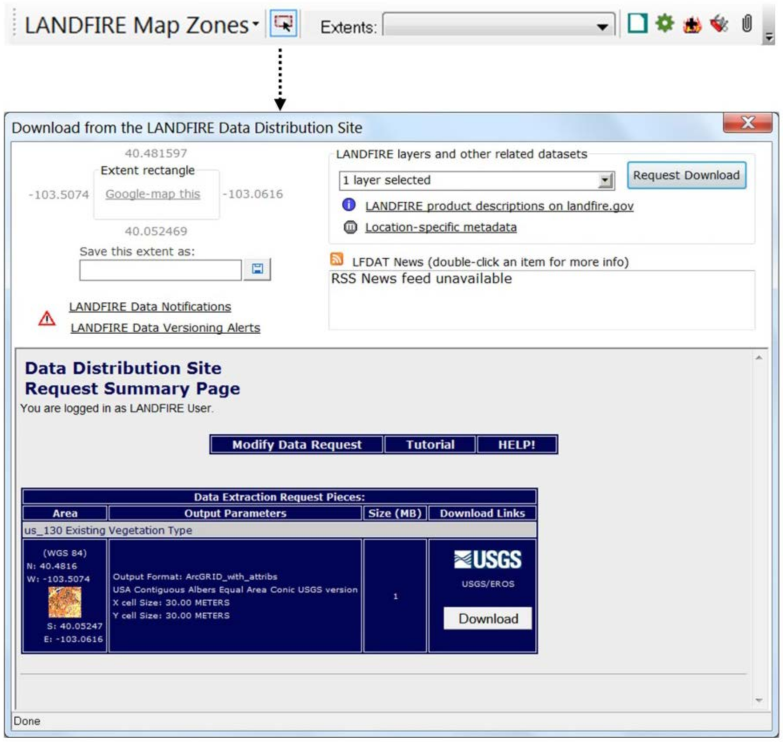


Figure 22. The LANDFIRE Data Access Tool (LFDAT). The LFDAT is a custom ArcGIS toolbar that links to the LANDFIRE Data Distribution Site.

Critique

The fundamental question of our critique was whether LANDFIRE data would be appropriate for simulating wildfire behavior at the analysis location and scale. The LANDFIRE Total Fuel Change Tool would be used to assess the fuel mapping rules in addressing this question; however, *data currency* and *map unit accuracy* (Chapter 1) are also important to accurately simulate the current wildfire hazard so we began our critique there.

The wildfire analysis component of this project began in January 2015, just after LANDFIRE version 1.3.0 (LANDFIRE 2012) data were released for the region. This meant the data were two years out-of-date at the time of acquisition. A critical first task was therefore to determine how much the landscape had changed in the preceding two years.

Approximately 200,000 acres of wildfire and 11,500 acres of mechanical disturbance had occurred over 2013 and 2014 within the wildfire simulation landscape. Given this information, it was clear that currency updates to the LANDFIRE vegetation and disturbance data inputs would be required prior to critiquing the fuel mapping rules with the LANDFIRE Total Fuel Change Tool.

The input data were also critiqued for map unit accuracy. Upon field review, local resource managers felt that oak woodland ecological systems were underrepresented in the LANDFIRE existing vegetation type data layer and that ancillary data would be required to address this issue. In critiquing the LANDFIRE disturbance data, local resource specialists also determined that certain disturbance type assignments were not correct for the local area. For example, the assignment of mechanical remove to all silvicultural treatments (i.e., clearcut, harvest, thinning) was not appropriate for the Rogue Basin because not all local harvesting methods are accompanied by activity-fuel treatments such as hand-pile burning or biomass extraction. Similarly, there were activities assigned to the "other mechanical" event type (a mechanical-add disturbance) that participants felt should be assigned to mechanical-remove. In addition, participants felt that although mastication event types add fuel to the surface fuelbed, they should be differentiated from the other mechanical add disturbances due to the effect of the structure and compactness of masticated fuel on fire behavior.

Finally, as discussed in Chapter 3, LANDFIRE does not currently use a cumulative effect approach to assign disturbance attributes in the composite fuel disturbance data layer. Rather, if multiple treatments occurred in the same location within the update period, the attributes of the most recent treatment are assigned (except where fire has occurred; see Chapter 3). This was also a potential cause of inaccurate map unit assignment.

To summarize, the following information was gathered from the data critique and used to modify the geospatial data inputs to the LANDFIRE Total Fuel Change Tool.

- Data is not current through 2014.
- Oak woodland ecological systems are underrepresented.
- Some disturbance type map unit assignments are inaccurate due to generalization of treatment types at the national scale and/or incorrect accounting of cumulative treatment effects.
- Grouping of mastication treatments with other mechanical add disturbances does not represent the unique fire behavior of masticated fuel.

Modify LANDFIRE Total Fuel Change Tool inputs

As discussed above, the LANDFIRE Total Fuel Change Tool requires geospatial data layers of: existing vegetation type, cover, and height; biophysical setting; and disturbance as inputs. The amount of updating required for these layers varies depending on analysis objectives. The following sections describe the modifications that were made, or why modification was determined to be unnecessary, for each of the required geospatial data layers based on our data critique.

Disturbance

Data Currency

Because the LANDFIRE 2012 composite fuel disturbance data layer only represents conditions through 2012, two data currency updates were required to create an up-to-date 2014 disturbance layer: 1) the time-since-disturbance attribute needed to be updated to reflect the two additional years that had passed, and 2) new disturbances—those that occurred in 2013 and 2014—would need to be added. The following methods were used to create the updated disturbance layer.

First, we determined the years for which the time-since-disturbance attribute would need to be updated (Table 6). Disturbances that occurred from 2005-2007 would remain in the 6-10 year time-since-disturbance class. Likewise, disturbances that occurred in 2010 and 2011 would remain in the 2-5 year time-since-disturbance class. However, disturbances that occurred in 2008 and 2009 would need to be updated to the 6-10 year class and disturbances that occurred in 2012 would need to be updated to the 2-5 year class.

The 2003 and 2004 disturbances would now be greater than ten years old. LANDFIRE removes disturbances greater than ten years old from the composite vegetation and fuel disturbance data layers (Chapter 3) and may also update existing vegetation layer map units to reflect a vegetation transition based on the ecology of the region. For example, a forested, existing vegetation type that experienced a high-severity wildfire, and was subsequently reassigned as an herbaceous or shrub existing vegetation type, may be reassigned to a forest vegetation type after ten years if reestablishment of trees is expected. More information on LANDFIRE vegetation transition rules is available on the program's website. Based on our analysis objectives we determined that we could leave the 2003 and 2004 disturbances in the 6-10 year time-since-disturbance class since we were only concerned with the fuel data layers required for wildfire hazard analysis and therefore not required to update existing vegetation layers.

Next, we downloaded the individual-year disturbance data layers for the years 2008-2012 using the LANDFIRE Data Access Tool. These layers were used to create two "geospatial masks" using the ArcGIS Spatial Analyst extension—one representing the 2008-2009 disturbances and one representing the 2010-2012 disturbances (Figure 23). Masks are used in geospatial analysis to constrain operations to certain pixels within a raster dataset. In our case, we used the masks to identify and update the time-since-disturbance of pixels where a disturbance had occurred in 2008 or 2009 without subsequent disturbances in 2010-2012. As in the LANDFIRE mapping process, if a fire disturbance occurred prior to 2008 we retained the time-since-disturbance of the fire (Chapter 3).

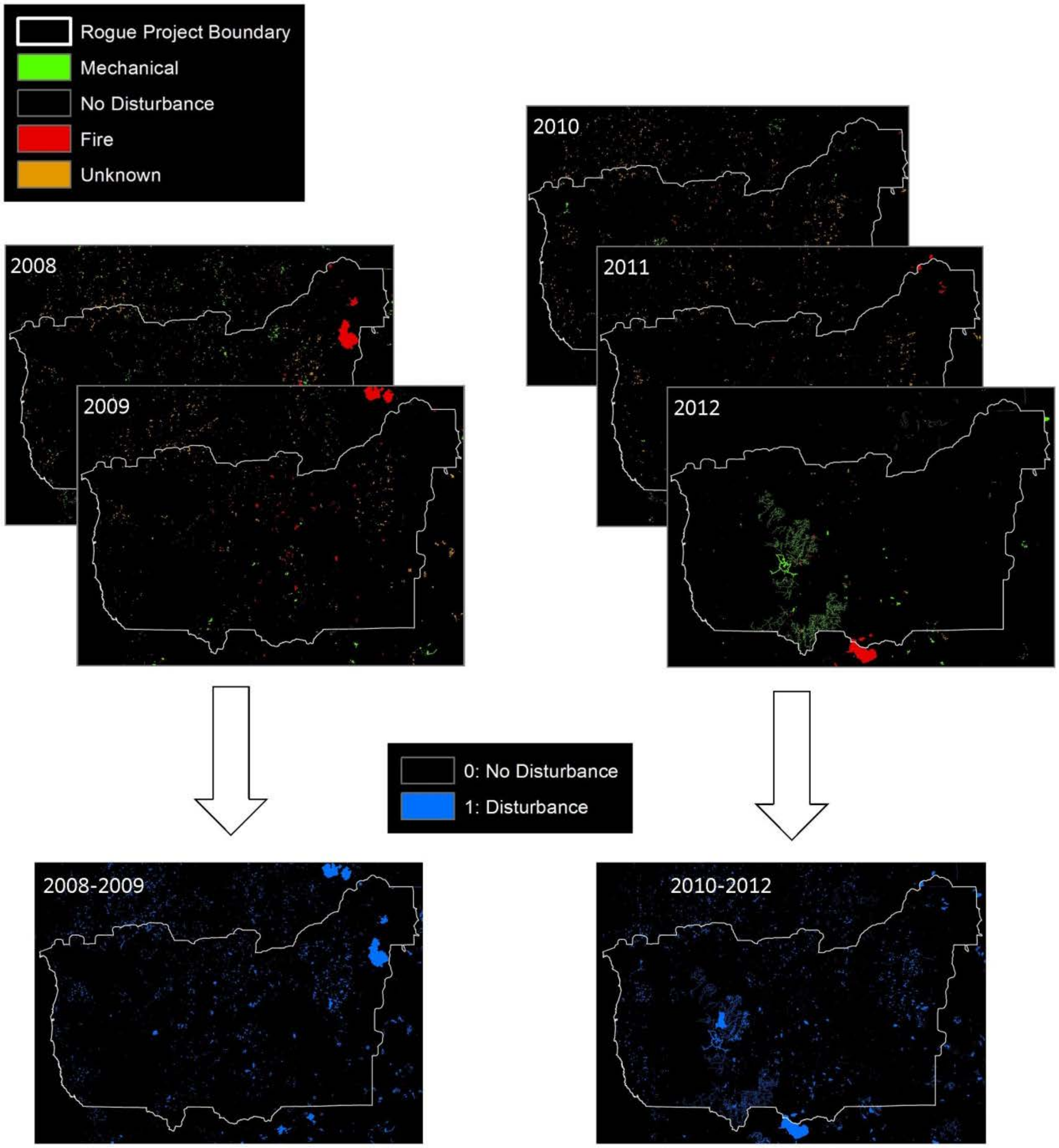


Figure 23. Updating time-since-disturbance. Two geospatial masks were created from the LANDFIRE individual year disturbance layers: one representing disturbances from 2008-2009 and the other representing disturbances from 2010-2012. Time-since-disturbance was updated from the 2-5 year class to the 5-10 year class only where disturbances occurred in 2008-2009 without subsequent disturbance in 2010-2012

With the time-since-disturbance updates complete, we next needed to incorporate 2013 and 2014 disturbances into our updated composite fuel disturbance layer. To reflect large wildfires (> 1,000 acres), we acquired wildfire severity data from the Forest Service Rapid Assessment of Vegetation Condition after Wildfire (RAVG) program website. Recall from Chapter 3 that the LANDFIRE disturbance severity classes represent the effect of disturbances on the vegetation cover of the dominant lifeform. The RAVG program produces a raster data layer representing canopy cover reduction, as a result of fire, through a process that correlates percent change in canopy cover to a remote sensing change detection protocol (Miller and Thode 2007, Miller et al. 2009). We used this data layer to further update the composite fuel disturbance layer based on the percent canopy cover reduction using the ArcGIS Spatial Analyst Extension *Reclassify* tool (Figure 24).

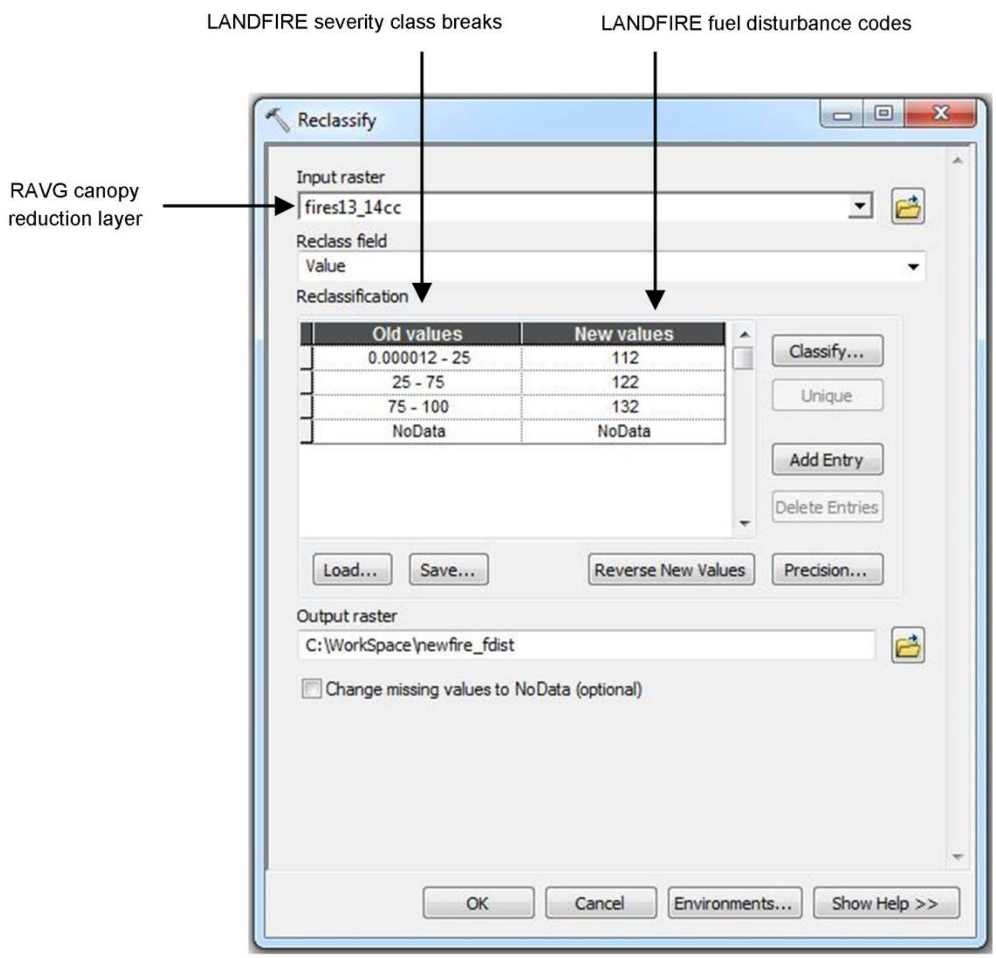


Figure 24. Reclassification of canopy cover reduction estimates from the Rapid Assessment of Vegetation of Condition after Fire (RAVG) program data to LANDFIRE fuel disturbance codes.

We followed a similar process for non-wildfire disturbances. First we acquired 2013 and 2014 Forest Service activities data from the agency's Forest Activities Tracking System (FACTS) and Bureau of Land Management activities from the National Fire Plan Operations and Reporting System (NFPORS). Forest Service and Bureau of Land Management personnel assigned the LANDFIRE disturbance type (mechanical add, mechanical remove, or prescribed fire), severity, and time-since-disturbance codes to each of the activity polygons. If subsequent activities occurred in the two-year time frame, the cumulative effect of those activities was used to determine the most appropriate disturbance attributes. We converted the polygon data to raster format and used ArcGIS Spatial Analyst Extension tools to further update the composite fuel disturbance layer.

Map Unit Accuracy

As mentioned above, our data critique identified two map unit accuracy issues in the disturbance data layer: 1) disturbance type map unit assignments were inaccurate due to generalization of treatment types at the national scale and/or incorrect accounting of cumulative treatment effects, and 2) the grouping of mastication treatments with other mechanical add disturbances does not represent the unique fire behavior of masticated fuel.

We used the ArcGIS Spatial Analyst *combine* function to combine the composite fuel disturbance layer with the individual disturbance layers from 2003-2012. The combine function creates a new raster where each unique combination of values from the input layers represents a single row in the attribute table. Using this table we were able to identify four unique situations and make adjustments based on local resource specialist input (Table 10).

Table 10: Adjustments made to mechanical disturbance type based on local input.

Criteria	Acres	Adjustment
Silvicultural treatments only	200,039	Disturbance type was changed from mechanical remove to mechanical add.
Mastication treatments only	9,188	Created a mask of mastication only pixels and changed the final fuel model values to a "post-mastication" fuel model within the mask during post-processing.
'Other mechanical' treatments only	75,936	Modified disturbance type only if local resource specialists felt the cumulative effect of the treatments was incorrectly assigned.
Combination of mechanical treatment types	289,248	Typically a combination of "other mechanical" and silvicultural treatment. Modified disturbance type only if local resource specialists felt the cumulative effect of the treatments was incorrectly assigned.

Biophysical Setting

Since the biophysical setting data layer represents potential vegetation based on the biophysical characteristics and historical disturbance regime of the site (Chapter 4), disturbances by definition do not have an effect on this layer[2]. Furthermore, because biophysical setting criteria are infrequently used in the LANDFIRE fuel mapping rules for the Northwest Geographic Area, we did not critique this layer for content accuracy.

Existing Vegetation Type

As mentioned above, our data critique identified that oak woodland ecological systems were underrepresented in the existing vegetation type layer. We therefore acquired ancillary geospatial vegetation data developed by the Landscape Ecology, Modeling, Mapping, and Analysis team (LEMMA). We extracted the oak woodland vegetation cover types from this data and augmented the LANDFIRE existing vegetation type data layer using ArcGIS Spatial Analyst tools.

Disturbances may result in a change to the existing vegetation type. For example, tree- or shrub-dominated vegetation may transition to herbaceous-dominated vegetation as a result of high-severity fire. If the existing vegetation type layer was to be used for purposes beyond the critique and development of fuel data, a separate data layer would need to be created to account for any post-disturbance effects to the existing vegetation type. However, since we were only concerned with post-disturbance effects on fuels, we were able to omit this step and rely on our updates to canopy structure and the *canopy guide* feature of the LANDFIRE Total Fuels Change Tool (see below) to correctly assign post-disturbance fuel attributes.

Existing Vegetation Cover

Two updates to the existing vegetation cover layer were required based on our data critique. First, because we used the LEMMA cover type data to augment our existing vegetation type data layer for oak woodland, we also updated the existing vegetation cover layer with LEMMA canopy cover values to ensure consistency across layers. That is, wherever existing vegetation type was updated with LEMMA data, we also updated existing vegetation cover with LEMMA data. Second, we needed to update existing vegetation cover to reflect the 2013 and 2014 disturbances added to the composite fuel disturbance layer.

The structural characteristics of existing vegetation are what the fire behavior fuel model mapping rules are keyed to (Figure 14). We were therefore required to adjust the existing vegetation cover for the new (i.e., 2013 and 2014) disturbances we added to the composite fuel disturbance layer. The post-disturbance canopy cover of forested vegetation types is also required for calculating post-disturbance canopy base height and canopy bulk density.

For the 2013 and 2014 large wildfire disturbances we used the RAVG canopy cover reduction data layer directly to adjust existing vegetation cover. For the non-wildfire disturbances we first assigned a canopy cover reduction value to each severity class midpoint (low severity: 12.5%, moderate severity: 50%, high severity: 87.5%). We did not allow values to be reduced below the lowest canopy cover class (10%-20%) because with few exceptions (e.g., clearcuts), even high-severity disturbances leave some cover. In the case of forested vegetation, leaving 15% forest canopy cover allows for simulating a slight effect of shading and wind reduction to surface fuel from the standing dead trees.

Existing Vegetation Height

[2] Although there are exceptions that could lead to a biophysical setting type conversion, such as those influenced by climate change, uncharacteristic disturbances, and/or exotic species, these occurrences are rare and even if present would have little effect on the assignment of fuel model in this analysis area—that is, biophysical setting criteria are infrequently used in the LANDFIRE fuel model mapping rules in the western states.

As with existing vegetation cover we first updated the existing vegetation height with the LEMMA data in the oak woodland vegetation type.

LANDFIRE existing vegetation height represents the basal-area weighted average of the dominant and co-dominant trees (Chapter 4). In forested vegetation types it is therefore typically not necessary to reduce forest canopy height due to disturbance, as most disturbances would not change the average height significantly enough to reduce existing vegetation height to a lower height class (Table 7). Certain silvicultural methods that target dominant trees, such as clearcuts or thinning from above, are exceptions. For high-severity wildfire, we retained the pre-disturbance canopy height. In combination with the low canopy cover value we assigned, retaining a canopy height value would allow us to simulate a slight effect of the standing dead trees on shading and wind reduction to surface fuel. We were able to prohibit crown fire from being predicted in the post high-severity fire pixels through use of the LANDFIRE Total Fuel Change Tool "canopy guide" function (see below).

Integration of steps with the LANDFIRE Total Fuel Change Tool

With the preliminary critique and updates to the required vegetation and disturbance data layers complete, we then critiqued the LANDFIRE fuel mapping rules using the LANDFIRE Total Fuel Change Tool. A user's guide, tutorial, and information on training for this tool are available on the Wildland Fire Management Research Development and Application – Fuels and Fire Ecology Program website. In this section we will highlight key features of the tool that were used to critique and update fuels for the Rogue Basin analysis.

The LANDFIRE Total Fuel Change Tool provides users the ability to critique and modify the LANDFIRE fire behavior fuel model mapping rules. Additionally, the tool will create canopy fuel data layers (canopy base height and canopy bulk density) using LANDFIRE's methodology, or allow users to "hardcode" base height and bulk density values to unique combinations of vegetation and disturbance attributes. This allows the user to "fine-tune" the interaction of fuel model, canopy base height, and canopy bulk density that is so critical to accurately simulating wildfire behavior.

Critique and Modification of Fire Behavior Fuel Model

The fuel critique was done in a workshop setting where local fire and vegetation specialists from the Forest Service, Bureau of Land Management, and The Nature Conservancy participated. This collaborative approach not only provides a wide range of local knowledge and expertise but also facilitates a sense of ownership and confidence in the end product.

We critiqued the fire behavior fuel model mapping rules for each of the major existing vegetation types in the analysis area. For each existing vegetation type, we first reviewed its description and where it was mapped. If photos were available they would be displayed to provide further context. Next, we discussed which factors—canopy cover; canopy height (a surrogate for stand age); biophysical setting; and disturbance type, severity, and time-since-occurrence—influenced the surface fuels and reviewed how the mapping rules used different combinations of these variables.

Adjustments to the fuel model mapping rules can be made in one of two ways, either to the fuel model assignment itself or to the combination of variables that define a rule (Figure 25). Adjustments to the fuel model assignment were made if workshop participants felt the specified fuel model didn't represent the expected surface fire behavior for the vegetation type and structure identified (that is, if the flame length was too high/low or the rate of spread was too fast/slow). The LFTFC tool provides an interface for comparing the flame length and rate of spread of different fuel models under varying combinations of fuel moisture, slope, and wind speed (Figure 26) as an aid to making modification decisions. Adjustments to the canopy cover and height thresholds, or addition of biophysical setting criteria will influence the spatial distribution and proportion of area assigned to each fuel model. We modified these criteria if participants felt the location or distribution of fuel models did not reflect on-the-ground conditions.

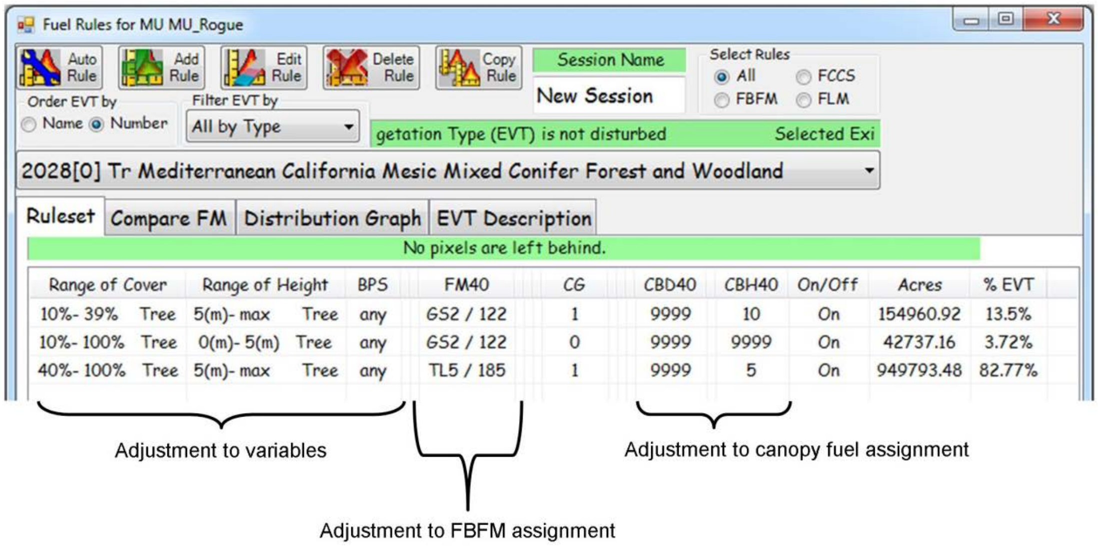


Figure 25. LANDFIRE Total Fuel Change Tool rulesets. Adjustments can be made to the range of variables, fire behavior fuel model (FBFM), and canopy fuel.

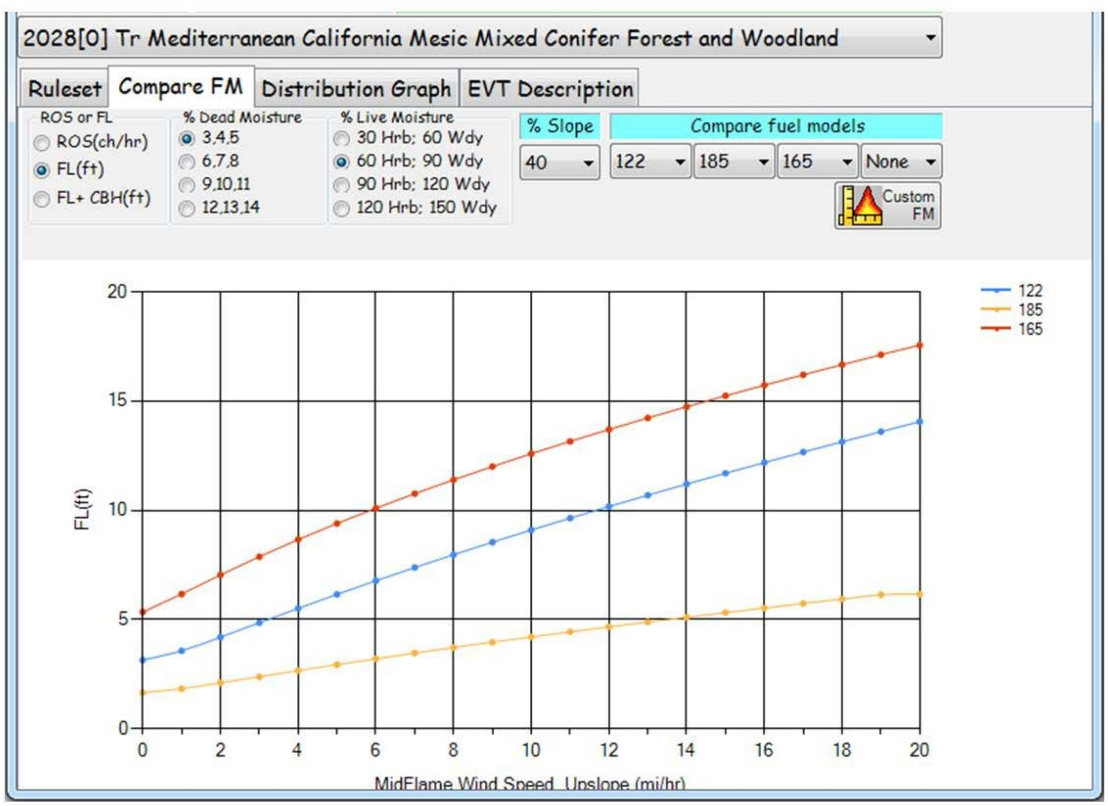


Figure 26. Comparing fuel models. The LANDFIRE Total Fuel Change Tool has built-in functionality to compare fire behavior between fuel models under a variety of fuel moisture and slope conditions.

Finally, for areas where a mastication treatment occurred we assigned the fire behavior fuel model outside of the LANDFIRE Total Fuel Change Tool using ArcGIS Spatial Analyst tools.

Critique and Modification of Canopy Fuels

There are two ways a user has control over how canopy fuels are mapped with the LANDFIRE Total Fuel Change Tool. The first is to use the tool's canopy guide feature; the second is to "hardcode" canopy fuel values. The canopy guide options are as follows:

- 0: No forest canopy structure characteristics (i.e., cover and height) or fuels are assigned. In forested existing vegetation types this may be used to represent a disturbance that removes the forested canopy (e.g., clearcut) or when the "forested" canopy is already considered in the fire behavior fuel model assignment (e.g., short trees).
- 1: The standard LANDFIRE methodologies (Chapter 5) are used to calculate canopy structure and canopy fuel values.
- 2: The canopy base height and canopy bulk density are artificially set to a point where crown fire—passive, active, or conditional (Scott and Reinhardt 2001)—will not be simulated (canopy base height of 10m and canopy bulk density of 0.012 kg/m-3). This value may be used in cases where canopy height and canopy cover values are still desired due to their influence on reducing wind speed and dead fuel moisture content through shading (Chapter 5) but where crown fire is unlikely (e.g., broadleaf forests).

We set the canopy guide value to 2 for all high-severity fire disturbances. As discussed previously, this technique allows for the standing dead trees to still have some, albeit minimal, influence on dead fuel moisture content and wind reduction, but eliminates crown fire and spotting from being modeled in fire behavior modeling systems. The use of a canopy guide value of 2 also served as an alternative to modifying the existing vegetation type due to high-severity fire. That is, by "turning off" crown fire and assigning the appropriate fire behavior fuel model for the expected change in the dominant vegetative lifeform, we accomplished the same goal.

For non-disturbed, and low- and moderate-severity fire disturbances, we assessed the effect of fire behavior fuel model and the LANDFIRE default canopy base height values on crown-fire initiation using the NEXUS (Scott 1999) fire modeling system (Figure 27). Canopy base height values were "hardcoded" (Figure 25) in the fuel rules if workshop participants felt that simulated crown-fire initiation didn't accurately represent expected crown-fire initiation. There are many factors to consider when assigning a canopy base height value. Knowledge of local wind patterns and/or analysis of the wind data that will be used in your analysis are paramount. We accepted the LANDFIRE default canopy base height assignments for all mechanical disturbances.

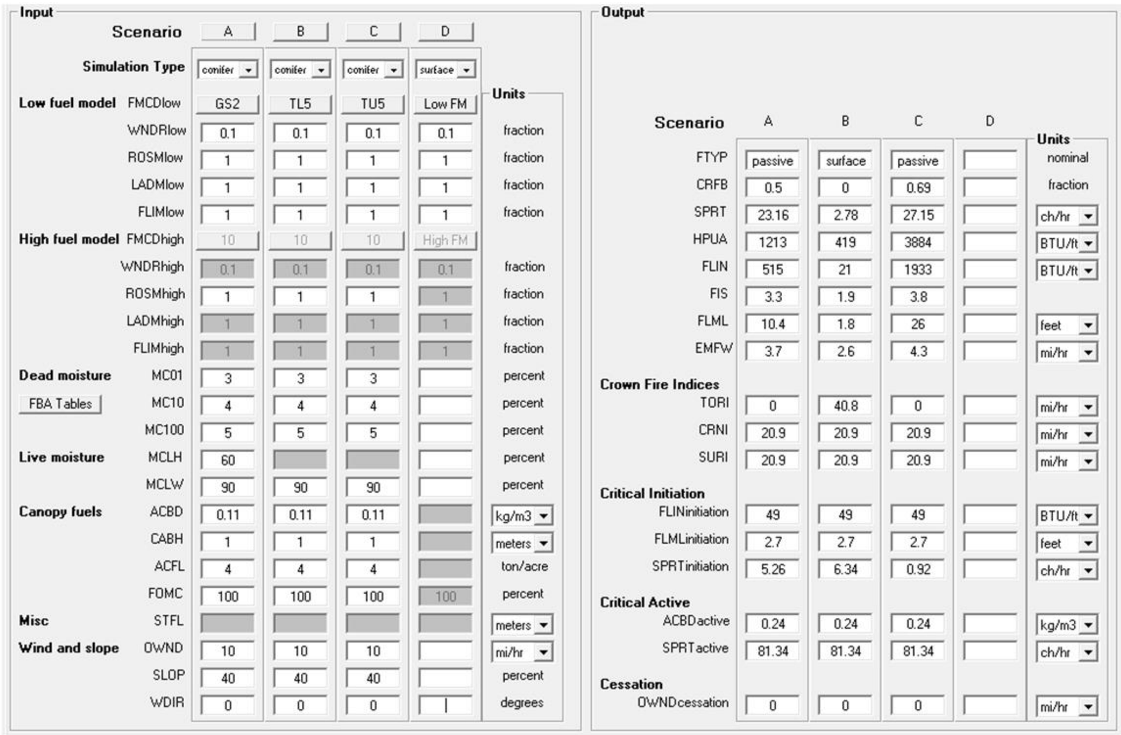


Figure 27. NEXUS fire modeling system. NEXUS facilitates in depth fire behavior critique and is particularly useful in assessing the environment conditions required to transition surface fire to crown fire based on fire behavior fuel model and canopy base height values.

Analysis

We created a new fire behavior modeling landscape (i.e., LCP file) based on our updated disturbance data layers and fuel model mapping rules. We then used this LCP to run basic fire behavior simulations as an additional critique. This final analysis step was used to highlight issues that were possibly overlooked or might have been hard to detect during the fuel calibration, thus necessitating further data modifications. After completion of this final step, the modified fuel data layers were used to analyze wildfire hazard in the Rogue Basin.

Example 2: Using LANDFIRE for local vegetation departure analysis

In this example we illustrate the data critique and update tasks conducted as part of an analysis of vegetation departure in the southern Sierra Nevada Mountains. The 12.5 million-acre planning area includes the Inyo, Sequoia, and Sierra National Forests; Sequoia and Kings Canyon National Parks; and portions of Yosemite and Death Valley National Parks (Figure 21). Because LANDFIRE provides wall-to-wall geospatial vegetation data, it was an obvious choice for vegetation departure analysis at such a broad spatial extent.

Define objectives

The objective of this project was to conduct a vegetation departure analysis using the FRCC Mapping Tool (Hutter et al. 2012) and LANDFIRE data. The results of this analysis would be further integrated into a wildfire hazard and risk assessment. The analysis was conducted in the fall of 2013.

Identify data requirements

Vegetation departure analysis requires data that characterize both the historical and current vegetation condition. LANDFIRE vegetation dynamics models (Chapter 6) would be used to describe the baseline historical conditions for each biophysical setting mapped to the analysis extent. LANDFIRE vegetation data would be used to characterize the current vegetation composition and structure. LANDFIRE 2008 vegetation data layers were acquired and updated for disturbance through 2012 by USDA Forest Service regional office geospatial analysts.

Critique and modification

We began our critique by listing biophysical settings by analysis area acreage from largest to smallest. A team of regional ecologists, vegetation specialists, and GIS and remote sensing specialists reviewed the data list to determine which biophysical settings to assess for departure. Biophysical setting classes comprising insignificant acreage, those that were difficult to accurately map (Chapter 6), and those determined not important to the analysis objectives were dropped. The review team further determined that the thematic resolution (Chapter 1) of the biophysical setting data layer was too fine, given local knowledge of historical vegetation dynamics and disturbance regimes (Chapter 6). Biophysical setting classes were therefore grouped (Table 11) based on recently developed presettlement fire regime groups that summarize presettlement fire frequency estimates for California ecosystems dominated by woody plants (Van de Water and Safford 2011).

Because the analysis area intersects multiple LANDFIRE map zones, we next reviewed the vegetation dynamics models for each of the biophysical settings for consistency across zones. It is common for the vegetation dynamics model to differ across zones for the same biophysical setting. If the map zone boundary reflects an ecological transition, then the differences between models may be appropriate (Chapter 6). However, if the map zone boundary creates an artificial demarcation in the analysis area, users will want to choose a single model that best fits the analysis area. The review team chose the most representative vegetation dynamics model for each biophysical setting or group of biophysical settings to be assessed. The LANDFIRE 2008 biophysical setting data layer was reclassified using the *reclassify* tool in the ArcGIS Spatial Analyst extension to the final 15 classes represented in Table 11.

Table 11: LANDFIRE biophysical setting (BpS) model groupings for the Southern Sierra vegetation departure analysis.

<table>
<tr><th>LANDFIRE Biophysical Setting Name</th><th>LANDFIRE BpS Code</th><th>Presettlement Fire Regime[a]</th><th>LANDFIRE Model Used in VCA[b]</th></tr>
<tr><td>Inter-Mountain Basins Big Sagebrush Shrubland</td><td>10800</td><td rowspan="3">Big Sagebrush</td><td>610800</td></tr>
<tr><td>Inter-Mountain Basins Big Sagebrush Steppe</td><td>11250</td><td rowspan="2">611260</td></tr>
<tr><td>Inter-Mountain Basins Montane Sagebrush Steppe</td><td>11260</td></tr>
<tr><td>Great Basin Xeric Mixed Sagebrush Shrubland</td><td>10790</td><td>Black and Low Sagebrush</td><td>610790</td></tr>
<tr><td>California Mesic Chaparral</td><td>10970</td><td rowspan="5">Chaparral-Serotinous Conifers</td><td rowspan="5">611050</td></tr>
<tr><td>California Montane Woodland and Chaparral</td><td>10980</td></tr>
<tr><td>Great Basin Semi-Desert Chaparral</td><td>11030</td></tr>
<tr><td>Northern and Central California Dry-Mesic Chaparral</td><td>11050</td></tr>
<tr><td>Sonora-Mojave Semi-Desert Chaparral</td><td>11080</td></tr>
<tr><td>Mediterranean California Dry-Mesic Mixed Conifer Forest and Woodland</td><td>10270</td><td>Dry Mixed Conifer</td><td>610270</td></tr>
<tr><td>Sierra Nevada Subalpine Lodgepole Pine Forest and Woodland</td><td>10580</td><td rowspan="2">Lodgepole Pine</td><td rowspan="2">610581</td></tr>
<tr><td>Sierra Nevada Subalpine Lodgepole Pine Forest and Woodland - Wet</td><td>10581</td></tr>
<tr><td>Mediterranean California Mesic Mixed Conifer Forest and Woodland</td><td>10280</td><td>Moist Mixed Conifer</td><td>610280</td></tr>
<tr><td>California Lower Montane Blue Oak-Foothill Pine Woodland and Savanna</td><td>11140</td><td>Oak Woodland</td><td>611140</td></tr>
<tr><td>Mediterranean California Mixed Oak Woodland[c]</td><td>10290</td><td>Mixed Evergreen</td><td>410140</td></tr>
<tr><td>Great Basin Pinyon-Juniper Woodland</td><td>10190</td><td>Pinyon-Juniper</td><td>610190</td></tr>
<tr><td>Mediterranean California Red Fir Forest - Cascades</td><td>10321</td><td rowspan="2">Red Fir</td><td>610321</td></tr>
<tr><td>Mediterranean California Red Fir Forest - Southern Sierra</td><td>10322</td><td>610322</td></tr>
<tr><td>Mediterranean California Subalpine Woodland</td><td>10330</td><td rowspan="2">Subalpine Forest</td><td rowspan="3">610330</td></tr>
<tr><td>Northern California Mesic Subalpine Woodland</td><td>10440</td></tr>
<tr><td>Sierra Nevada Subalpine Lodgepole Pine Forest and Woodland - Dry</td><td>10582</td><td>Lodgepole Pine</td></tr>
<tr><td>Inter-Mountain Basins Subalpine Limber-Bristlecone Pine Woodland</td><td>10200</td><td rowspan="2">Subalpine Forest</td><td rowspan="2">610200</td></tr>
<tr><td>Rocky Mountain Subalpine-Montane Limber-Bristlecone Pine Woodland</td><td>10570</td></tr>
<tr><td>California Montane Jeffrey Pine(-Ponderosa Pine) Woodland</td><td>10310</td><td rowspan="2">Yellow Pine</td><td rowspan="2">610310</td></tr>
<tr><td>Mediterranean California Lower Montane Black Oak-Conifer Forest and Woodland</td><td>10300</td></tr>
</table>

[a] Van de Water and Safford (2011) pre-settlement fire regime vegetation types shown for reference.
[b] Vegetation Condition Assessment.
[c] Based on local knowledge and ancillary vegetation data, workshop participants felt that areas mapped as a Mediterranean California Mixed Oak Woodland biophysical setting were incorrectly classified and should be classified as Central and Southern California Mixed Evergreen Woodland (BpS model 410140).

Next, the succession class mapping rules for each of the final vegetation dynamics models were assessed. Adjustments were made to ensure rules were *exhaustive* and *mutually exclusive*, and that uncharacteristic native conditions were appropriately represented for the local area (Chapter 6). Succession class was then remapped, accounting for the adjustments, using ArcGIS software (Figure 28). First, the existing vegetation type layer was reclassified to create an exotic vegetation mask, where exotic vegetation types were assigned a value of 1 and native vegetation types were assigned a value of 0. Next, the biophysical setting, existing vegetation cover, existing vegetation height, and exotic vegetation mask data layers were combined using the ArcGIS Spatial Analyst extension *combine* tool. A new field was then added to the output combine layer and populated with the new succession class values by first selecting combinations of the data layer attributes as defined in the mapping rules and using the *field calculator* function. Finally, after all combinations had been assigned a new succession class value, the ArcGIS Spatial Analyst extension *lookup* tool was used to create a new succession class data layer.

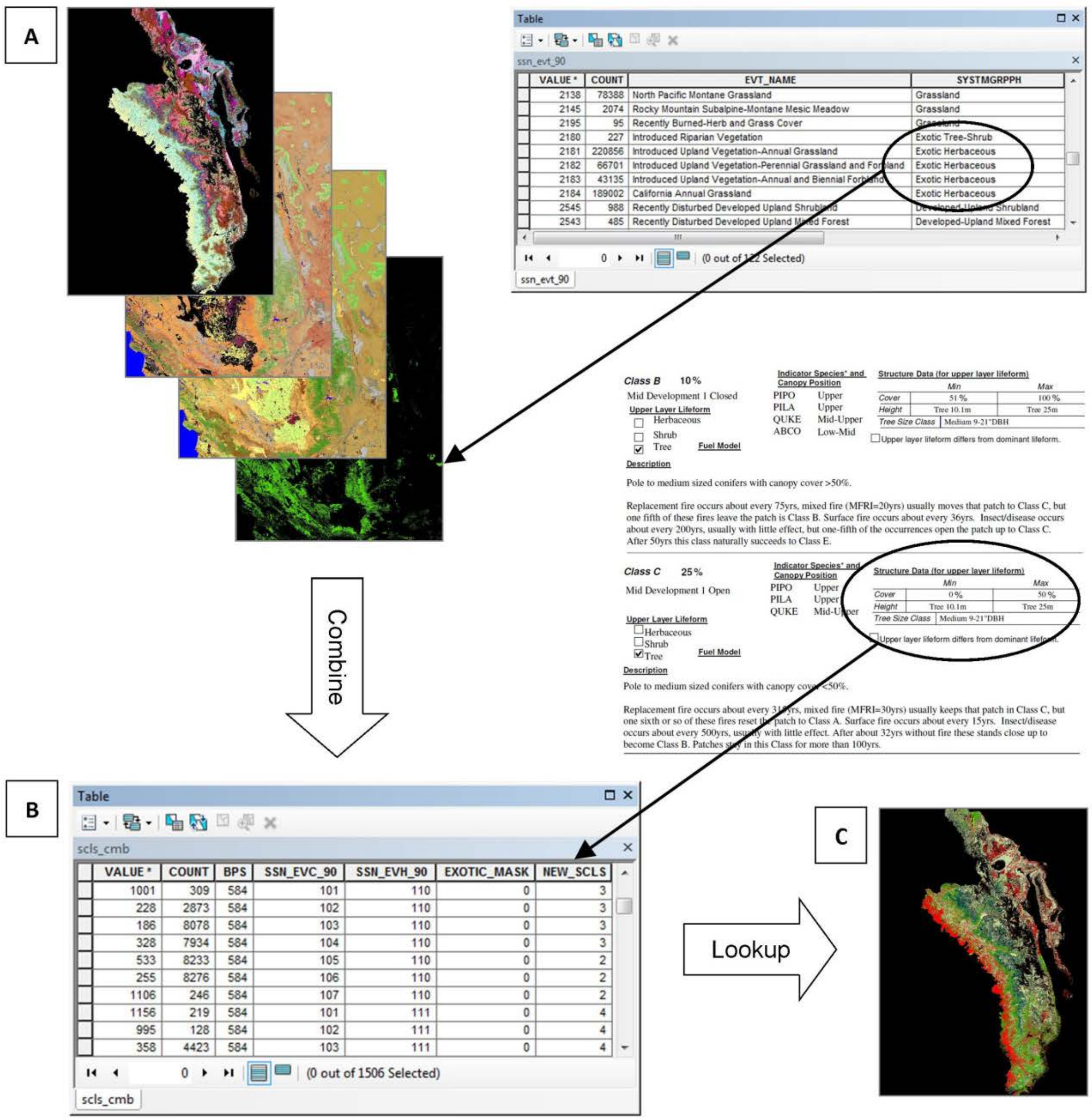


Figure 28. Succession class remapping process. (A) Biophysical setting, existing vegetation cover and height, and exotic vegetation data layers were combined using ArcGIS Spatial Analyst. (B) New succession class values were then assigned based on vegetation dynamics models and adjustments defined by local specialists. (C) Finally, the ArcGIS Spatial Analyst *lookup* tool was used to create a new succession class spatial data layer.

Analysis

We created a spatial landscape assessment unit data layer for conducting the vegetation departure analysis. Each biophysical setting was assigned to an assessment unit based on fire regime characteristics, including historical fire-size distribution (Barrett et al. 2010). Finally, we ran the Fire Regime Condition Class Mapping Tool and reviewed the results.

No issues were identified and the results informed managers where on the landscape specific vegetation development classes (i.e., succession class) were in either surplus or deficit in relation to their presettlement condition. As noted in Example 1 of this chapter, sometimes an analysis may highlight issues that were overlooked or hard to detect earlier in the data critique process that necessitate further data modifications. Analysis should be viewed as an iterative process.

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REVIEWS

"Biases" in Adaptive Natural Resource Management

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Abstract

Uncertainties about the consequences of natural resource management mean that managers are required to make difficult judgments. However, research in behavioral economics, psychology, and behavioral decision theory has shown that people, including managers, are subject to a range of biases in their perceptions and judgments. Based on an interpretative survey of these literatures, we identify particular biases that are likely to impinge on the operation and success of natural resource management. We discuss these in the particular context of adaptive management, an approach that emphasizes learning from practical experience to reduce uncertainties. The biases discussed include action bias, the planning fallacy, reliance on limited information, limited reliance on systematic learning, framing effects, and reference-point bias. Agencies should be aware of the influence of biases when adaptive management decisions are undertaken. We propose several ways to reduce these biases.

Introduction

Natural resource management is often a complex and uncertain process. The underlying environmental and physical processes are sometimes not well understood. Even when they are understood, there are likely to be uncertainties about the quantitative outcomes of management. The current actual status of the resource may be difficult to determine. Managers cannot always fully control which on-ground actions are undertaken due to lack of resources, legal powers, or capacities (Williams & Brown 2014).

These complexities and uncertainties mean that managers are required to make judgments. However, it has been shown that, in making judgments of these types, decision makers do not always undertake decisions "rationally." Simple rational decision-making models assume that agents always take decisions to maximize the achievement of their objectives, based on accurate knowledge of the outcomes, costs, and constraints. In reality, however, people have limited information, limited time, and limited cognitive capacity. As a consequence, they are restricted in formulating and solving complex problems, and they are susceptible to different types of biases (Arnott 2006; Tasic 2011)—beliefs that are inconsistent with reality (Chira *et al.* 2011) or behaviors that compromise the achievement of objectives. For example, Guthrie *et al.* (2000) found that some of the biases listed in Box 1 affect judges when they are making judicial decisions. Similarly, Hirshleifer (2008) found that financial regulators are subject to a different set of biases that influence their decisions, plans, and polices. The impacts of such biases can be substantial. For example, Kahneman (2012) reports on a 2005 study of rail projects worldwide undertaken between 1969 and 1998. Passenger usage of the rail system was overpredicted in 90% of cases. On average, planners overestimated passenger usage of new train lines by over 100%, reflecting a common bias known as the "planning fallacy."

Box 1: Selected behavioral biases with potential impact on adaptive management

- Action bias: Tendency to take actions even when it is better to delay action
- Framing effect: Tendency to respond differently to alternatively worded but objectively equivalent descriptions of the same item
- Reference-point bias: Tendency to overemphasize a predetermined benchmark for a variable when estimating the level of that variable
- Availability heuristic: Tendency to give more weights to events that can be recalled more easily
- Planning fallacy: Making judgments about a planned activity that are systematically over-optimistic, including underestimating project completion time, underestimating costs, or over-estimating benefits
- "Satisficing rule": Tendency to stop searching for a better decision once a decision that seems sufficiently good is identified
- Loss aversion: Tendency to value losses more highly than similar gains
- Reliance on limited information: Tendency to use a subset of information even when full set of information is available
- Limited reliance on systematic learning: Tendency to use information from past successful efforts rather than using information from both successful and failed efforts

For a general list of behavioral biases, see Arnott (2006) and Gino & Pisano (2008).

Managers of natural resources and the environment are likely to be just as susceptible to these biases as are other professionals who must make complex judgments, such as judges and financial regulators (Carlsson & Johansson-Stenman 2012). However, these issues have received little attention in the conservation literature. Our aim in this article is to draw from psychology, behavioral economics, and behavioral decision theory research literatures to identify key insights about biases that are relevant to conservation, and to understand their implications for managers responsible for management of environmental projects or programs.

In doing so, we focus to some extent on Adaptive Management (AM), since this is a process that has been promoted or used to manage complex and uncertain natural resource issues. AM is a process of "learning by doing" (Walters & Holling 1990) where learning from experience is combined with the need for immediate action (Westgate *et al.* 2013). Under AM, management policies are formulated as experiments that investigate ecosystems' responses to changes in people's behavior or management actions (Lee 1999). Conceptually, a set of potential models representing relationships between human actions and ecological outcomes are developed and tested. Viewing the learning process through a Bayesian lens, each model is assigned a probability of being the true model. In each time step, a management decision is made based on the current model probabilities, the current system state, and predicted future states. Model probabilities are updated after each time step based on each model's success in predicting outcomes (Conroy & Peterson 2012), and management may subsequently be modified.

Traditionally, AM has focused on learning from experimental trials or pilots of management approaches for biological and ecological systems (Wilhere 2002; McCarthy & Possingham 2007). It has been assumed that the decision makers will interpret the information collected and make their choices or decisions rationally and without bias. We will explore the extent to which research on human behavior and decision making casts doubt on this assumption. Broader implications for management of natural resources and the environment will also be discussed.

AM: Definition and Stages of Learning

AM has been defined by Williams *et al.* (2009) as "a systematic approach for improving resource management by learning from management outcomes" (p. 1). In active AM, the learning process is supported by purposefully collected information (Walters & Holling 1990), rather from observation of management actions chosen without regard to their ability to provide useful information for future decisions. In active AM, learning is often represented through single- and double-loop processes (Figure 1). Under a single-loop learning cycle, the key steps involved are: (1) define management goals with stakeholders involvement (step 1); (2) develop alternative management options, including an option to maintain the "status quo" (step 2); (3) develop models or statistical processes to trace system responses to management actions (step 2); and (4) implement management options (step 3; Westgate *et al.* 2013). Steps 4 and 5 involve monitoring and assessment of the outcomes, respectively. In a single-loop learning cycle, it is often assumed that project objectives, societal needs, and policy structures are fixed (Allen & Gunderson 2011).

In double-loop learning, on the other hand, it is assumed that policy objectives and structure could change. For example, in long-term projects, societal values and

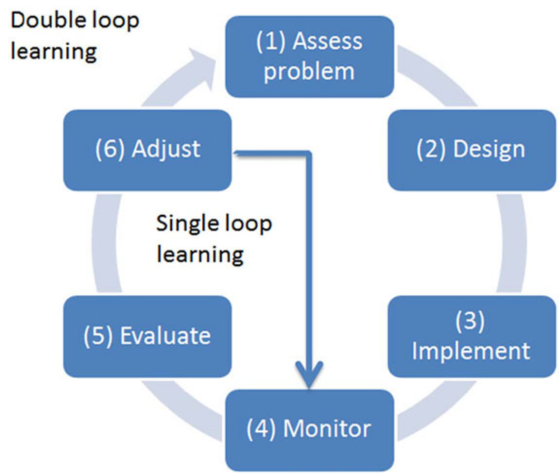

Figure 1 Different steps in active AM cycle with single- and double-loop learning (based on Williams & Brown 2014).

needs could change as time progresses and new management actions are introduced. The resource or the system under experimentation could also change to make the original project objectives unsuitable or unattainable. Therefore, the objectives, management options, or institutional arrangements might need to be changed. Under double-loop learning, original project objectives and management options are revisited after certain steps (step 6). New information from experimentation and model predictions are taken into account as well as changed policy and societal landscapes (Williams & Brown 2014).

In an AM regime, decision makers are responsible for defining management goals, identifying alternative management options, developing models, and implementing programs (Westgate *et al.* 2013). It is common to assume that in each step the resource managers would make "rational" decisions based on the information obtained from biological, physical, and social experiments. However, numerous studies inform us that people have cognitive limitation and bounded rationality, and are influenced by different types of biases. We expand on these issues in the following section.

Key Behavioral Biases

Both psychology and economics have rich literatures on the influences of different types of bias on behavior. Experimental economics serves three main purposes: testing theories, building new theories from observing experimental outcomes, and testing policy and management options. Behavioral economics also integrates insights from psychology to explain economic decision making. It studies the effect of psychological factors such as emotional, social, and cognitive factors on many decisions and economic processes (Camerer 1999). A related field is behavioral decision theory, which studies how people make decisions as well as how they should make decisions (Moore & Flynn 2008). The key biases identified in these research efforts that are relevant to AM are outlined below.

Action bias

"Action bias" occurs when the decision makers choose to take actions even when a "rational" decision maker would prefer to delay actions to allow further information collection, or to take no action. Possible reasons for action bias include that decision makers give higher weight to things that are readily observable and attributable (i.e., the management actions themselves), rather than to things that are delayed, indirect, or unobservable (i.e., potentially the outcomes from those actions; Patt & Zeckhauser 2000). For example, a study of elite soccer goalkeepers showed that they tend to jump to try to save goals even when the optimal strategy is to stay in place (Bar-Eli *et al.* 2007). In this case, taking action is valued in its own right, in addition to the value attributed to the outcome achieved. Similarly, environmental managers may feel that they will earn credit from their superiors, the general public, and the media if they take action even when it is not justified or should be of relatively low priority (Tasic 2011).

Action bias could be increased by uncertainty (Tan *et al.* 2012). In most environmental projects, knowledge of the effectiveness of interventions that will be taken on the ground is rather weak (Ferraro & Pattanayak 2006). As a result, taking action may be evaluated more positively than collecting additional information, partly because of a lack of evidence that actions would be ineffective.

The implication of "action bias" for AM is that it may be difficult to convince managers that an investment in information collection (i.e., AM) is worthwhile. They will tend to prefer to allocate the resources to additional on-ground management actions. Proponents of AM may enhance their persuasiveness by arguing that AM does not require actions to be delayed, and allows more effective or less costly actions to be taken in future. If AM is implemented, it should help to reduce action bias over time by providing additional information about whether the actions being undertaken are effective.

The planning fallacy

The "planning fallacy" is the tendency of project planners to be excessively optimistic about the performance of a project that they are developing (Kahneman & Tversky 1977; Kahneman & Lovallo 1993). For example, many investments in abatement of dryland salinity under Australia's National Action Plan for Salinity and Water Quality program were too small to make a notable difference to salinity outcomes (Auditor General 2008; Pannell & Roberts 2010). Apparently, managers choosing these investments greatly overestimated the effectiveness of the actions being funded, despite ample scientific evidence being available (Prosser *et al.* 2001; Dawes *et al.* 2002). The extent of bias due to the planning fallacy can be substantial. According to Griffin & Buehler (1999), only 1% of the U.S. military high-technology equipment purchases were delivered on time and on budget.

There are various factors that contribute to the planning fallacy. Buehler *et al.* (1994) observed that people estimate a project's expected completion time by constructing mental scenarios of how the project may develop. However, due to cognitive limitations, they generate a smaller range of scenarios than is realistically possible, overlooking many barriers and risks. The scenarios generated tend to reflect their hopes and preferences (Newby-Clark *et al.* 2000) and to neglect their own previous negative experiences with similar projects (Koole & van't Spijker 2000). To some extent, overoptimism is likely to reflect strategic biases adopted to increase the competitiveness of projects when funding is being allocated (Flyvbjerg 2007), but overoptimism is often present even when planners are attempting to be realistic (Kahneman 2012).

A strategy to reduce the planning fallacy is to ask managers to forecast the completion time, cost, or benefits for a range of comparable projects rather than a single project. This strategy, known as Reference Class Forecasting (Kahneman & Tversky 1977), has been effective in reducing time and cost overruns of large infrastructure projects (Buehler *et al.* 2010).

Where the planning fallacy is in evidence, AM may help to reduce its adverse consequences. AM, involving information collection and refinement of project design, helps in correcting decisions that were initially made on an excessively confident or optimistic basis. If necessary, targets can be modified or the project can be terminated following the collection of improved information (Dvir & Lechler 2004).

Reliance on limited information

Decision makers sometimes use only a subset of information even when the full-set information is available. In a series of experiments with common-pool resources, Apesteguia (2006) studied the impact of additional information on individual behavior and payoffs. The individual payoff depended on player's own investment as well as investments made by others. In one treatment, participants had complete information about the expected payoffs from their choices, while in another they had no relevant information. The experimenter observed that the aggregate outcomes (in terms of investment decisions and actual payoffs from the decisions made) were not significantly different between these two treatments (Apesteguia 2006). More-or-less similar observations have been made in other studies (Mookherjee & Sopher 1994; Oechssler & Schipper 2003; Van Huyck *et al.* 2007). One hypothesis to explain this phenomenon is that decision makers follow a "satisficing rule" to limit the cognitive costs of decision making (Hertwig & Pleskac 2010). Under such a rule, the decision maker stops searching for a better decision once he or she identifies a decision that seems sufficiently good.

Another version of this bias is "availability bias" in which people give more weights to certain types of events that can be recalled more easily (Tversky & Kahneman 1974). For example, a manager may assess the risk of bushfire higher than the risk of plant disease spread if bushfires have been more common or more salient in recent times. Underutilization of information is often observed in environmental planning. For example, it has been observed that many existing environmental planning systems fail to account for project costs (Mazor *et al.* 2013), for the effectiveness of management actions (Maron *et al.* 2013), or for behavior change (Pannell & Roberts 2010).

AM potentially provides a mechanism to counter this tendency of decision makers to ignore relevant information. It has been shown in many studies that use of systematic learning through use of data and models could outperform heuristic decision making and predictions by experts (Camerer 1981). It has also been shown that decision makers may employ information more comprehensively if they are asked to make a decision several times sequentially (with time delays) and to explain their decisions to third parties (Vul & Pashler 2008; Herzog & Hertwig 2014). By emphasizing the importance of using accurate information and encouraging use of a structured approach for doing so, AM may prompt a general strengthening of the evidence base for environmental decision making. There can also be a social aspect to AM, with different people contributing to decisions about how management should be adapted in response to new information. This socialization of the process may reduce the tendency of any individual to ignore information.

Limited reliance on systematic learning

Active AM involves systematic experimentation and learning from the outcomes. However, experimental studies on learning reveal that humans are not good at systematic learning. Instead, learning is often messy, noisy, and based on trial-and-error (Hertwig & Pleskac 2010). In practice, people hardly use systematic learning models where they compute and compare expected outcomes from every option before making a decision. Rather, they use heuristics and repeat their past successful choices without fully considering other potentially superior alternatives (Erev & Haruvy 2009).

One implication of limited reliance on systematic learning is that managers will try to learn only from their past “successful” project rather than learning from both “successful” and “failed” projects. In doing so, risk-averse managers are more likely to repeat their past successful choices instead of trying new management interventions (Denrell & March 2001). They are less likely (relative to risk-neutral managers) to invest resources to collect more information about the past unsuccessful strategy (Erev & Haruvy 2009). By contrast, a systematic AM approach would seek to learn from previous mistakes to avoid repeating them, and to enhance the resilience of the management system. AM encourages a systematic approach to learning, and to the use of new information for decision making. It makes explicit the importance of obtaining and using new information, at least partially countering tendencies not to do so.

An institutional barrier to systematic learning is staff turnover, which can be high in the environmental sector, sometimes due to the short duration of funding programs (Grafton 2005). Unless new staff commence before the departures of experienced staff, they must rely on written or verbal communication to learn about the existing or past project (Shogren & Taylor 2008). If the logic behind past decisions is not well-documented, new staff cannot integrate the successes or failures of past decision-making processes into their decision making. There are also differences in the way a new and an experienced manager would approach a problem. A new manager would use facts in a context-free manner whereas, for an experienced manager, problem recognition and action selection would be more intuitive (Hayes 2013).

One potential way to promote systematic learning is through the use of decision support systems (DSSs) that enable the storing of such information. There can be synergies between the use of DSSs and AM. Depending on the type of DSS, it may increase the transparency and evidence base of the initial decision to support a project. This transparent information can be updated as the AM process proceeds, allowing the DSS to inform decisions about modifications to the project (Dicks *et al.* 2014).

Framing effect and reference-point bias

The “framing effect” refers to a situation when people respond differently to statements that are worded differently but are objectively equivalent. Among the many ways of framing an environmental management issue, we mention three that are commonly discussed in the literature: (1) risky choice framing, where the expected outcomes of a risky option are described in different ways; (2) attribute framing, where some characteristics of an object or event are highlighted or focused on; and (3) goal framing, where different potential objectives of the program or activity are emphasized (Levin *et al.* 1998). In a risky choice, framing the outcomes from a lottery could be presented as a loss (say 50% chance of losing) or as a gain (50% chance of winning). In attribute framing, we might focus on only one or a few features of a project (say number of days required to complete a project) rather than all relevant features. For example, we could say that the project is successful if it is completed within a certain number of days (and ignore other features such as the achievement or nonachievement of environmental outcomes). In goal framing, we could focus on gain from undertaking a project (such as “Native animal population will increase if fox control bait is used”) or loss from not undertaking the project (such as “Native animal population will continue to decline if fox control bait is not used”; Krishnamurthy *et al.* 2001).

Reference-point bias may cause managers to respond differently to a program or activity depending on the level of a predetermined reference point or benchmark. For example, the same level of environmental improvement could be seen as a success if it is well above a benchmark level of improvement or a failure if it is less than a benchmark, even if the benchmark is arbitrary (Kühberger 1998). It has been shown that people are more sensitive to losses relative to a benchmark than to gains (Camerer 1998). This may mean that managers are strongly motivated to prevent their program from being perceived to be a failure relative to the reference point, but less strongly motivated to seek to make a program perform above the reference point, even if a stronger performance would be feasible and worthwhile.

By regular monitoring and evaluation of project outcomes, AM may help to enhance flexibility in the setting of project goals and to reduce dependence on a fixed reference point. AM, in conjunction with a DSS could help in reducing the impacts of framing effect and reference-point bias by helping managers to assess potential strategies more comprehensively and objectively. Reasons why DSSs are not more commonly used by environmental managers include: lack of adequate training, no clear policy guideline to use the best possible information or DSS, and pressure to spend money within a deadline that is too short to allow time for using the DSS (Shtienberg 2013). To address the last of these issues, in particular, agencies should ideally plan and prepare for potential programs or the next phase of an existing program well before the existing program has concluded.

Discussion

Although many natural resource managers claim to use AM, rigorous and systematic applications are rare (McFadden *et al.* 2011; Westgate *et al.* 2013; Williams & Brown 2014). This is surprising given the theoretical attractiveness of AM in the face of risk and uncertainty (Stankey *et al.* 2005). There has been little research about the impact of psychological biases on decision making by managers of environmental or natural-resource programs (Westgate *et al.* 2013). Based on a survey of the economics and psychology literature, we have identified a set of biases that have implications for AM in particular and NRM in general. As a result of this review, there are grounds to expect that: (1) the managers are likely to take on-ground actions even when these are not worthwhile (Patt & Zeckhauser 2000); (2) they could suffer from the cognitive illusion of being more in control of the system than they actually are (Koole & van't Spijker 2000); (3) they could be overconfident about the expected outcome of their decisions (Flyvbjerg 2007); (4) they may be overly optimistic in terms of expected completion time of the project (Kahneman 2012); (5) they might rely on a partial set of information for decision making even when fuller information is available (Hertwig & Pleskac 2010); (6) they might rely on trial-and-error learning and repeat their past successful choices instead of collecting and comparing information about the full set of decision options (Erev & Haruvy 2009); and (7) managers could try to achieve predefined goals rather than the best possible outcomes from a project (Kühberger 1998; Table 1).

Different biases could influence various steps of the AM cycle differently. For example, action bias could influence the design phase of the AM cycle and lead the planners and managers to design projects with more emphasis on on-ground actions and less on the expected outcomes. Similarly, overconfidence and reliance on limited information would mean the managers would fail to consider all relevant information during the design and monitoring phases. Limited use of systematic learning process would mean failure to learn from previous mistakes during the evaluation phase. Lack of systematic learning would also make mangers susceptible to framing effect and reference-point bias (Klayman & Brown 1993). Agencies should be cautious about the impact of these biases and take remedial measures (Fischhoff 1982).

First, the agencies need to promote a culture of learning (e.g., García-Morales *et al.* 2012). It needs to be recognized that both successful and failed projects generate valuable information about the future state and expected impacts of the management interventions. This could be done by providing appropriate incentives (tangible and intangible) for the managers and decision makers to consider the full range of options before making any decision (Arnott 2006), requiring them to repeat the same decision several times before finalizing it (Vul & Pashler 2008; Herzog & Hertwig 2014), or asking mangers to justify their decisions to external parties (Gollwitzer & Sheeran 2006).

Second, adoption of a DSS could facilitate retention and storing of relevant information (e.g., Behrens & Ernst 2014). It may also make learning from past projects easier and help in systematic evidence-based decision making. Relevant staff should be adequately trained and properly incentivized to use DSSs (Dicks *et al.* 2014).

Third, conducting benefit-cost analyses of planned options would help to refine and prioritize the options during the design phase of the AM cycle (e.g., Pannell *et al.* 2012, 2013). Benefit-cost analysis provides a systematic and objective framework to include all relevant costs and benefits (both market and nonmarket goods and services) related to a project. In the process of identifying benefits and costs, it also helps in identifying if there is complementarity among them (to avoid double counting) and the time lag and uncertainty attached to realization of each benefits and costs. Thus, benefit-cost analysis could be used as a tool to comprehensively assess the expected benefit of a project (Sunstein 2000; Atkinson & Mourato 2008).

Fourth, involvement of external third-party reviewers may also help in designing more realistic and feasible projects (Chen & Volden 2013; Behrens & Ernst 2014). Finally, scenario analysis should be conducted as part of the assessment and design phase of AM cycle to anticipate the expected outcomes of different options (Lautenbach *et al.* 2009). The likely impact of different types of biases, their impact and the effectiveness of potential remedial measures should be systematically analyzed and studied before making any final recommendation for use in decision making for natural resources.

Table 1 Potential psychological biases, their impacts on different steps of the AM cycle, and potential remedial measures to overcome the impact of the biases

Biases	Potential impact on behavior	Potential impact on different steps of AM cycle	Potential remedial measures
Action bias	• Tendency to rely more on actions rather than on results	• During design phase (step 2) projects with visible actions will be prioritized which may lead to wastage of valuable resources (money and time)	• Emphasize the value of information and learning from the AM cycles during the evaluation (step 5), adjustment (step 6), and assessment (step 1) phases rather than on the actions undertaken on ground • Conduct a benefit-cost analysis during the design phase (step 2) of the cycle
Planning fallacy	• Overoptimistic or wrong judgments on the expected benefits, completion time, and costs of the project	• Failure to implement the project (step 3) in due time • During the monitoring phase (step 4), all relevant indicators may not be included, which lead to inadequate assessment during the evaluation phase (step 5)	• Conduct feasibility study as part of the assessment of the problem (step 1) and design of the options (step 2) • Involve external third parties during design phase (step 2) to review proposed actions and their underlying assumptions.
Reliance on limited information	• Make quick judgment • Lack of clearly specified project goals	• During assessment of the problem (step 1), full set of information will not be considered, which will lead to faulty prioritization of projects	• Develop DSSs which will automate incorporation of available information and facilitate consideration of full range of available information during assessment (step 1) and design (step 2) phases
Limited reliance on systematic learning	• Failure to consider the full range of the options • Repetition of the "safe" options • Failure to learn from previous mistakes	• Failure to consider learning from "failed" projects during the evaluation phase (step 5) may lead to missed opportunities to learn and realize the full potential of the situation	• During the evaluation (step 5) and adjustment (step 6) phases, consider learning from all projects (complete/incomplete, successful/failed, etc.) • Always conduct a scenario analysis with a range of options and expected future states during assessment (step 1) and design (step 2) phases
Framing effect and reference-point bias	• Failure to understand the real implications of an option • Success as well as failure is measured relative to a reference point • Follow a satisficing rule rather than a maximization rule while making decisions	• Use wrong measures to evaluate a project (step 5) • Managers may not give their full efforts if they think that they have performed better than others (or with respect to a predefined goal) already (step 3)	• Use DSSs and train managers on how best to use it • A scenario analysis could demonstrate the best possible outcomes from a given situation

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Global fire emissions buffered by the production of pyrogenic carbon

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Landscape fires burn 3–5 million km^2 of the Earth's surface annually. They emit 2.2 Pg of carbon per year to the atmosphere, but also convert a significant fraction of the burned vegetation biomass into pyrogenic carbon. Pyrogenic carbon can be stored in terrestrial and marine pools for centuries to millennia and therefore its production can be considered a mechanism for long-term carbon sequestration. Pyrogenic carbon stocks and dynamics are not considered in global carbon cycle models, which leads to systematic errors in carbon accounting. Here we present a comprehensive dataset of pyrogenic carbon production factors from field and experimental fires and merge this with the Global Fire Emissions Database to quantify the global pyrogenic carbon production flux. We found that 256 (uncertainty range: 196–340) Tg of biomass carbon was converted annually into pyrogenic carbon between 1997 and 2016. Our central estimate equates to 12% of the annual carbon emitted globally by landscape fires, which indicates that their emissions are buffered by pyrogenic carbon production. We further estimate that cumulative pyrogenic carbon production is 60 Pg since 1750, or 33–40% of the global biomass carbon lost through land use change in this period. Our results demonstrate that pyrogenic carbon production by landscape fires could be a significant, but overlooked, sink for atmospheric CO_2.

Globally, landscape fires, which include wildfires, deforestation fires and agricultural burns, emit approximately 2.2 Pg C yr^{-1} to the atmosphere (1997–2016)[1]. The majority of this total emission flux is contributed by non-deforestation and non-peatland fire emissions, which are approximately balanced by vegetation regrowth and thus have no net influence on atmospheric stocks of carbon on decadal timescales[2,3]; however, around ~0.4 Pg C yr^{-1} are emitted during tropical deforestation and peatland fires, which contribute to the net global emissions of carbon due to land use change (~1.1–1.5 Pg C yr^{-1} (Fig. 1))[4–6]. These global carbon budget (GCB) estimates are generated by models that represent the temporally distinct processes of immediate carbon emission from burned areas and decadal-scale sequestration through vegetation (re)growth in a spatially explicit manner[1,7,8]. However, such models routinely overlook the coincident flux of biomass carbon to recalcitrant by-products of fire, which can be stored in terrestrial and marine pools for centuries to millennia, and thus provide a long-term buffer against fire emissions (Fig. 1)[9,10–13]. Consequently, the legacy effects of fire that operate on the longest timescales are systematically excluded from models of the carbon cycle and from GCBs[12,14].

These legacy effects are due to the incomplete combustion of vegetation during landscape fires, which transforms part of the remaining organic carbon in the biomass to a continuum of thermally altered products that are collectively termed pyrogenic carbon (PyC)[10,12,15]. The majority of the PyC produced during landscape fires remains initially on the ground in charcoal particles of varying size and is subsequently transferred to its major global stores in soils[16–18], sediments[19,20] and water bodies[21,22]. A smaller fraction of fire-affected vegetation carbon is emitted as PyC in smoke[23,24]. PyC includes labile products of depolymerization reactions as well as aromatic molecules that result from condensation reactions, the latter of which are depleted in functional groups and thus chemically and biologically recalcitrant[25–27]. The enhanced resistance of PyC to biotic and abiotic decomposition leads to its preferential storage in environmental pools[15,20] and a residence time that is typically 1–3 orders of magnitude greater than that of its unburnt precursors[12]. This makes PyC one of the largest groups of chemically discernible compounds in the soil with a contribution to the soil organic carbon stocks of 14% globally[16]. A fraction of the PyC is also conserved across the land-to-ocean aquatic continuum and thus accounts for approximately 10% of riverine dissolved organic carbon[28], 16% of riverine particulate organic carbon[29] and 10–30% of the organic carbon in ocean sediments[13,19,30,31].

A series of reviews and data syntheses have recognized the potential of PyC production to invoke a drawdown (sink) of photosynthetically sequestered CO_2 to pools that are stable on timescales relevant to anthropogenic climate change and its mitigation[9,10,12,13,32–37]. Owing to the relative recalcitrance of PyC, the conversion of biomass carbon to PyC represents an extraction of carbon from a pool cycling on decadal timescales to a pool cycling on centennial or millennial timescales[13,19,20,25,38]. This storage potential contrasts with that of dead vegetation, which degrades on timescales of months to decades or enters soil pools with a shorter residence time than that of PyC[7,11,25,39,40]. Consequently, postfire PyC pools emit carbon to the atmosphere over a significantly longer time period than would be the case in the absence of PyC production and also provide a buffer that moderates atmospheric CO_2 stocks (Fig. 1)[9,12,13]. At present, the fire-enabled vegetation models that are used to make GCB calculations account for short-term fire emissions but routinely exclude fluxes of carbon from biomass to PyC or the delayed emission of carbon from legacy PyC stocks to the atmosphere (Fig. 1)[7,8,14,41,42]. This introduces systematic errors to GCBs through misrepresentation of the effects of modern and historical fires on the exchange of carbon between the atmosphere and terrestrial–marine pools[12–14].

Although PyC has been recognized as a major component of the postfire ecosystem carbon stocks for a number of decades[10,35],

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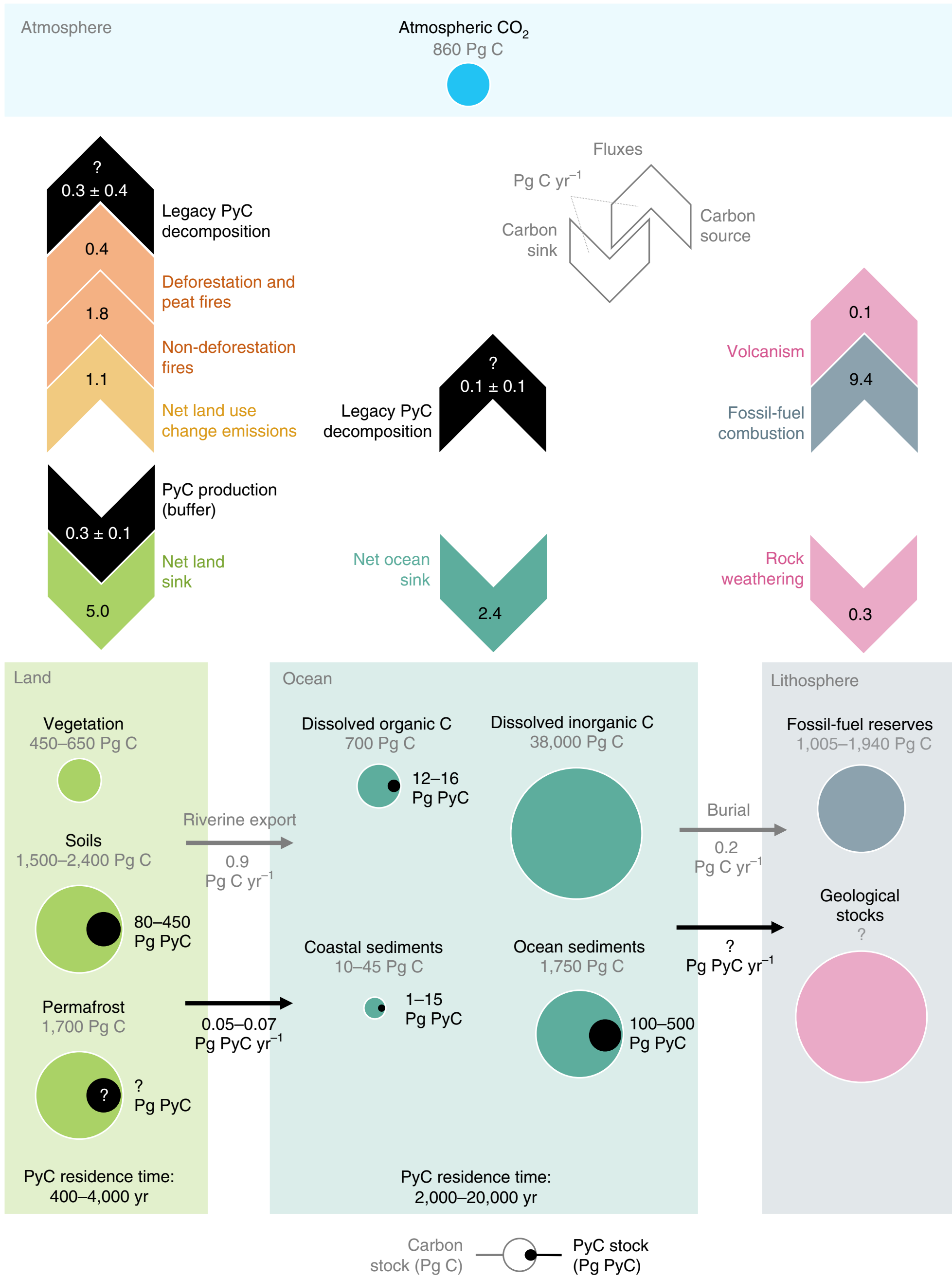

Fig. 1 | Schematic of the global carbon cycle including the buffer and legacy roles of PyC. Stocks (Pg C (1 Pg C = 1×10^{15} g of carbon)) and fluxes (Pg C yr^{-1}) of the global carbon cycle are represented by values from the GCB assessment of the decade 2008–2017[4] and the Intergovernmental Panel on Climate Change fifth assessment report of the decade 2000–2009[6]. Fluxes of carbon due to the net land sink are modified from the GCB to exclude non-deforestation fire emissions, whereas net land use change emissions are modified to exclude deforestation fire emissions. Carbon emissions from deforestation and peat fires and from non-deforestation fires were derived from GFED4s (ref. [1]) and relate to the period 1997–2016. PyC production fluxes due to deforestation and non-deforestation fires are based on estimates from GFED4s+PyC (this study). PyC stocks in soils, ocean dissolved organic carbon and ocean sediments are based on representative PyC/organic carbon ratios in the literature[13,16,68] applied to the estimates of organic carbon stocks and fluxes. PyC fluxes through rivers are the sum of global dissolved and particulate PyC export fluxes[28,29]. Residence times shown for soils derive from a meta-analysis of PyC decomposition in space-for-time substitution studies[69] and incubation experiment estimates extrapolated to field conditions[25]. Residence times for oceanic PyC pools are derived from the literature[19,70]. First-order estimates for legacy PyC decomposition fluxes and their uncertainties are calculated in quadrature for land and ocean pools as the product of the PyC stocks and the reciprocal of the residence times for PyC in these pools, assuming that the low- and high-end estimates for each term represent a consistent portion of normally distributed uncertainty.

quantification of its production rate at the global scale has been problematic and estimates vary by roughly an order of magnitude (50–379 Tg C yr^{-1}) (refs. [12,13,34,36]). A cause of the large range of production estimates is that calculations previously relied on incomplete information regarding the spatial distribution and type of fires, the allocation of carbon among the biomass fuel

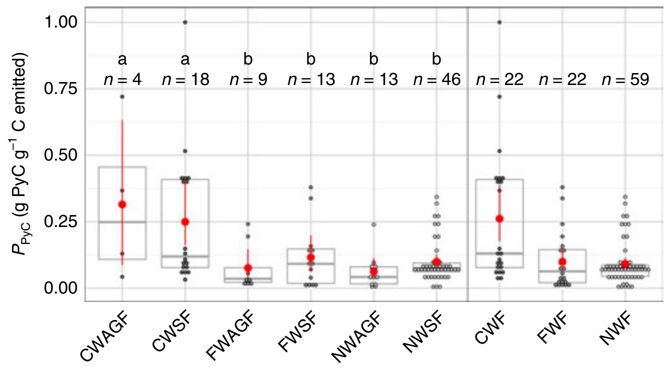

Fig. 2 | The box plots show the distributions of P_{PyC} values for each of the biomass component classes in the production factor dataset. Dots mark the distribution of P_{PyC} values across 1% intervals on the *y* axis. Red dots show the mean P_{PyC} values and red lines show the bootstrapped 95% confidence interval (Methods). Boxes illustrate the median and interquartile range of values. Letters a and b indicate biomass components with statistically similar P_{PyC} distributions at the 95% confidence level according to Tukey honest significant difference tests. The number of data entries (*n*) is also shown. CWF includes both CWSF and CWAGF. CWAGF, coarse woody aboveground fuels; CWSF, coarse woody surface fuels; FWAGF, fine woody aboveground fuels; FWSF, fine woody surface fuels; NWAGF, non-woody aboveground fuels; NWSF, non-woody surface fuels; FWF, fine woody fuels (includes both FWAGF and FWSF); NWF, non-woody fuels (includes both NWAGF and NWSF).

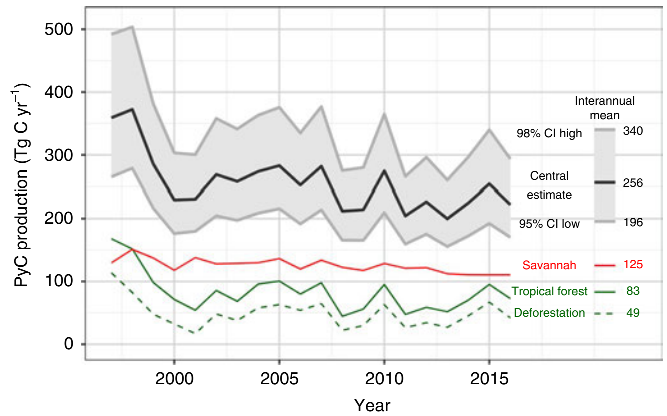

Fig. 3 | Annual global PyC production estimates from GFED4s+PyC for the period 1997–2016. The black line plots the modelled rate of production based on central P_{PyC} ratios (g PyC g^{-1} C emitted) from the global dataset. The shaded area indicates the uncertainty range of the modelled values based on the 95% confidence intervals (CIs) of P_{PyC} values (Fig. 2). The contributions of savannah burning (red line) and tropical forest burning (green solid line) to global PyC production totals are shown, the latter of which includes tropical deforestation fires (green dashed line).

components in burned areas and the specific PyC production factors for these distinct biomass fuel components. To alleviate these issues, we enhanced the Global Fire Emissions Database version 4 with small fires (GFED4s)[1], which is one of the principal process-based models used to make estimates of carbon emission from landscape fires[41,43,44]. Specifically, PyC production was incorporated by following a three-step approach that consisted of: (1) the assembly of the most comprehensive global database of PyC production factors (P_{PyC} (g PyC g^{-1} C emitted)) compiled to date, (2) the assignment of production factors for individual fuel classes stratified as coarse or fine and as woody or non-woody (Fig. 2) and (3) the application of P_{PyC} values to fuel-stratified carbon emissions (grams of C emitted) modelled by the native fuel consumption model in GFED4s. The output is the first global gridded dataset for monthly PyC production at a resolution of 0.25 × 0.25°, covering the years 1997–2016.

Global PyC production

Our central estimate for global PyC production in the period 1997–2016 was 256 Tg C yr^{-1} (Fig. 3), with an uncertainty range of 196–340 Tg C yr^{-1}, which includes variability in the measured P_{PyC} and interannual variability in global production, but excludes uncertainty in GFED4s emissions estimates (Methods). Interannual variability in global PyC production, expressed as the s.d. around the mean, was 47 Tg C yr^{-1} and was most strongly associated with variability in woody fuel combustion, which includes standing wood and coarse woody debris (CWD) (Supplementary Section 1 and Supplementary Fig. 1). Coarse woody fuels (CWF) produce PyC at a greater rate than finer fuels (Fig. 2) and consequently forest fires have disproportionate potential to influence global rates of PyC production (Supplementary Fig. 2).

The El Niño–Southern Oscillation (ENSO) is the primary driver of interannual variability in the burned area in the tropics[45] and previous analyses conducted with GFED showed that carbon emissions from tropical forest ecosystems more than doubled on average during the positive (El Niño) phases relative to the negative (La Niña) ENSO phases[46]. Correspondingly, we calculated that global rates of PyC production in tropical forests were 111% greater during the main fire season of the El Niño phases than during the La Niña phases (Supplementary Table 1). As rates of PyC production by non-forest fires were not sensitive to ENSO (Supplementary Table 1), the major driver of interannual variability in the total PyC production was variability in the tropical forest burned area (Fig. 3). The production of PyC was anomalously high in 1997–1998 (366 Tg C yr^{-1}), which aligns with a particularly strong positive El Niño phase that promoted extensive burning of (tropical) forests in South and Central America and in Southeast and Equatorial Asia[1,46].

Major production regions

The PyC production rates modelled by GFED4s+PyC conformed to a latitudinal pattern (Fig. 4) in which the tropical latitudes clearly dominated production at the global scale. Of the global production, 91% occurred in the tropics and subtropics (0–30° N and 0–30° S), whereas temperate (30–60° N and 30–60° S) and high-latitude (60–90° N) regions provided small contributions to the global total (8% and 1%, respectively).

The global distribution of PyC production also shows intricate regional patterns driven by variation in both the frequency at which fuel stocks were exposed to fire and the magnitude of the fuel stocks that were combusted during the fires that occurred (Supplementary Figs. 3 and 4). Fire frequency was ultimately the key determinant of PyC production rate, which explains why the tropics and subtropics were the dominant source regions. Although savannah fires affect low fuel stocks (Supplementary Section 2), these fires occur frequently and were spatially extensive (Supplementary Fig. 5 and Supplementary Table 2). They thus made the largest contribution to the global PyC production flux (125 Tg C yr^{-1}). Although tropical deforestation fires affected approximately 1% of the area of savannah fires, they affected large stocks of fuel (Supplementary Table 2) and were thus the second largest driver of global PyC production, at 49 Tg C yr^{-1}. The area affected by non-deforestation tropical forest fires was more than a factor of four larger than that of deforestation fires, but fuel consumption was relatively low (Supplementary Table 2). These fires provided the third major component of the global PyC production flux (34 Tg C yr^{-1}). Overall, 81% of the total global PyC production in the period 1997–2016 occurred in savannahs (49%) and tropical forests (32%).

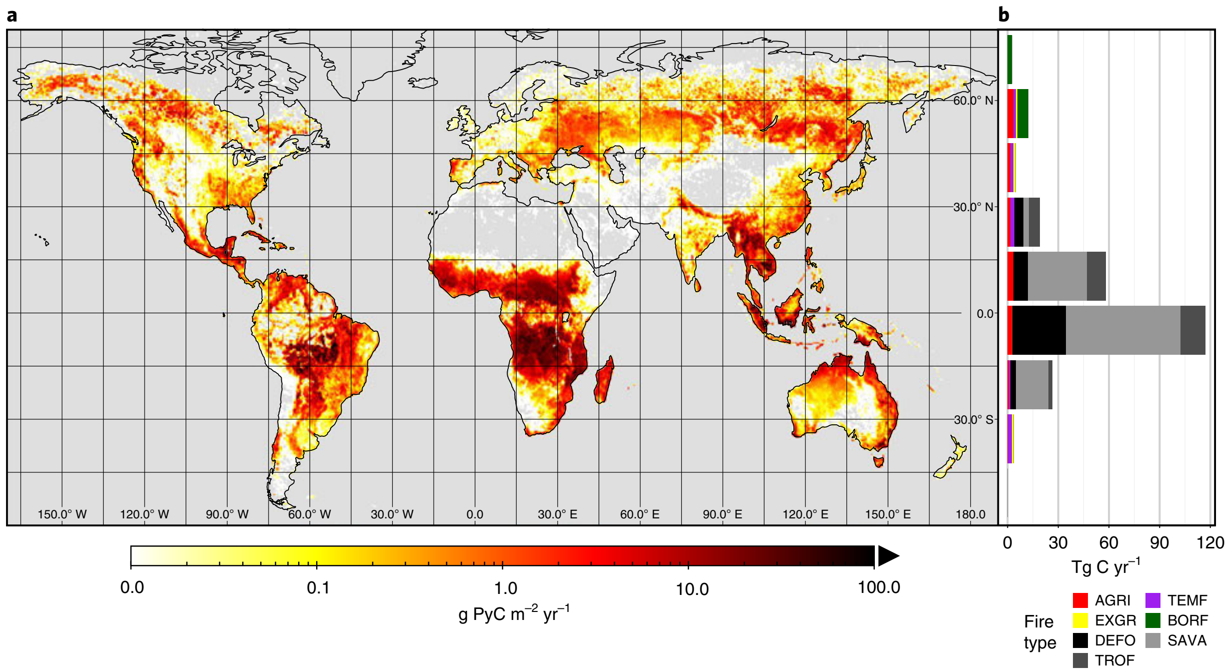

Fig. 4 | Annual average PyC production rates for the period 1997–2016 from GFED4s+PyC, based on central production factors (Fig. 2). a, The average global distribution of PyC production (g C m^{-2} yr^{-1}; note the log scale). **b**, The total production of PyC (Tg C yr^{-1}) in 15° latitudinal bands segregated according to the fire type, which includes savannah fires (SAVA), non-deforestation tropical forest fires (TROF), tropical deforestation fires (DEFO), agricultural fires (AGRI), temperate forest fires (TEMF), extratropical grassland fires (EXGR) and boreal forest fires (BORF).

Global carbon budget implications

Here we have quantified the global gross sink of atmospheric carbon caused by the transfer of photosynthetically sequestered biomass carbon to stocks of PyC during vegetation fires. Our central global PyC production flux estimate (256 Tg C yr^{-1}) is non-trivial within the context of the global carbon cycle (Fig. 1), as it equates to 12% of the global carbon emissions flux due to biomass burning and ~8% of the land sink for atmospheric CO_2 (~3.0–3.2 Pg C yr^{-1}) (refs. [4,6]). The global PyC production flux also equates to 75% of the carbon emitted from tropical deforestation and peat fires, which are the main categories of fire that cause a net loss of carbon to the atmosphere[1,9,47]. The PyC flux modelled here occurs in addition to the smaller global flux of 2 Tg C yr^{-1} caused by the emission of PyC in smoke from vegetation fires (according to equivalent estimates made using GFED4s in the years 1997–2016)[1].

The magnitude of our global estimate for PyC production indicates that the production of PyC during vegetation fires has the potential to significantly influence the atmospheric stock of carbon. A net sink of atmospheric carbon to stocks of PyC can be expected to develop if the flux associated with its production is unmatched by remineralization fluxes from legacy PyC stocks in terrestrial–marine pools (Fig. 1). Earth system models (ESMs) are the most sophisticated tools available to quantify the exchange of carbon between the atmosphere and these pools in time periods for which robust empirical data are sparse or unavailable. Despite previous attempts to highlight the importance of PyC production for carbon storage over timescales relevant to anthropogenic climate change and its mitigation[34,35,48], the absence of the PyC cycle from ESMs has restricted the scope to quantify its role in the carbon cycle[14]. The method introduced here allows for the routine integration of PyC production into fire-enabled vegetation models in a manner that systematically considers the spatial distribution of fire, the composition of the fuel stocks affected and the specific PyC production factors that apply to individual fuel components. This procedure is simple to implement in other fire-enabled vegetation models, which means that the major outstanding challenge to quantifying the net exchange of carbon between the atmosphere and PyC stocks with ESMs is to improve constraints over its storage and residence time in terrestrial and marine pools (Fig. 1)[13,14].

We also show that the PyC cycle must be integrated into ESMs if they are to represent accurately the role of fire in Earth's carbon cycle. The production flux of PyC represents the quantity of carbon that models would otherwise treat either as emitted or as unburned biomass with a residence time in terrestrial pools on the order of months to decades[7,11,25,39,40,49]. At present, the fate of 11% of the global biomass carbon stocks affected annually by fire is misrepresented in global models. As PyC dynamics are not represented in the ESMs used to make GCB calculations[4], this pool may represent a quantitatively significant missing sink or source of carbon to the atmosphere[14,50]. Recent estimates suggest that total carbon emissions from biomass burning in the period 1750–2015 amounted to ~500 Pg C (averaging 1.9 Pg C yr^{-1}) (ref. [41]). Under the assumption that the modern global PyC production flux maintained a constant ratio with the carbon emissions flux throughout this period, we estimate that since the beginning of the industrial revolution ~60 Pg C was transferred to the PyC stocks. This value is equivalent to 33–40% of the carbon lost from biomass pools due to land use change in the same time period (145–180 Pg C) (refs. [6,51]).

Our estimates for the modern and historical PyC production incorporate the best current understanding of PyC production through the combustion of vegetation biomass; however, the limitations of these estimates are worthy of mention. Notably, we do not include the production of PyC through the combustion of organic matter in soils, which may be an important process that drives the accumulation of PyC stocks in environments with deep organic layers, particularly peatlands[52]. We also do not account for the recombustion of PyC in locations that experience secondary burns, which can drive losses of the PyC that remains exposed at the surface[53]. PyC mass losses through recombustion have been reported as <8% in savannahs[54] and 17–84% in boreal forests[53,55]; however, the long fire return intervals in the latter biome typically allow sufficient time for PyC to be protected from recombustion through its burial in soils[17]. Our exclusion of recombustion is deliberate as we consider the process to be a component of the legacy PyC decomposition flux, which we do not quantify here (Fig. 1). Finally, our dataset of PyC production factors provides values for P_{PyC} that are modulated by fuel class (Fig. 2), but does not take into account fire characteristics (for example, temperature and duration) that are relevant to the formation of PyC[36,56,57]. The continued study of PyC production, with a particular focus on regions with high or rising fire incidence[58–60] and a range of fire intensities[61], will facilitate the application of more specific production factors in spatially explicit global models and thus result in reduced uncertainties in the global PyC production.

The production of PyC may become an increasingly important process for global carbon cycling in future centuries. Although the global burned area has declined in at least the past two decades, due predominantly to the conversion of savannah and grassland to agriculture[62,63], recent fire modelling studies generally agree that this decline is unlikely to continue past the year 2050[58–60]. It is also likely that a higher fraction of global burned area will be distributed in forests in which significant stocks of vegetation carbon are held[58,64,65]. As woody fuels generate more PyC per unit of biomass carbon than other fuels (Fig. 2), the spread of fire into forests can be expected to disproportionately enhance the global PyC production (Supplementary Fig. 2). Although it is less clear how fire prevalence will change in tropical and temperate forests owing to a stronger human control over burning in these regions[58,62], recent increases in fire extent caused by an increasing drought frequency in Amazonia already counteract reductions in the extent of deforestation fires[66]. Notwithstanding the significant uncertainty that exists in model predictions of future fire regimes, there are strong indications that PyC production rates will increase in some of the Earth's most carbon-dense regions in response to a changing climate[7,9,67]. This implies that the buffer for atmospheric CO_2 emissions that results from PyC production will grow in future centuries.

Online content

Any methods, additional references, Nature Research reporting summaries, source data, statements of code and data availability and associated accession codes are available at https://doi.org/10.1038/s41561-019-0403-x.

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Author contributions

M.W.J., C.S. and S.H.D. designed the study. S.H.D. led the Leverhulme Trust Research Project grant that funded the main body of the work. M.W.J. collated the PyC production factor dataset with support from C.S. C.S. and S.H.D. provided unpublished PyC production data. G.R.vdW. provided access to the GFED4s code. M.W.J. adapted the GFED4s code to include PyC production with the support of G.R.vdW. M.W.J. conducted the formal analysis of the production factor dataset and model outputs. All the authors contributed to the interpretation of the results. M.W.J. wrote the manuscript and produced all the figures. All the authors contributed to the refinement of the manuscript.

Competing interests

The authors declare no competing interests.

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Methods

Global fuel consumption modelling in GFED4s. In GFED4s, carbon emissions to the atmosphere are quantified based on burned area and fuel consumption per unit of burned area. Burned area is derived from satellite data[71] and fires that are too small to be detected by regular burned area algorithms are derived statistically based on active fire detections and relations with, among others, vegetation indices[72]. Fuel consumption is modelled using a satellite-driven biogeochemical model[1] and tuned to match observations[73]. Most of the underlying satellite input datasets have a 500 × 500 m resolution but are aggregated to the model resolution of 0.25° × 0.25°. Total fuel consumption is based on the fuel consumption of several fuel components, which include leaves, grasses, litter, fine woody debris, CWD and standing wood. van der Werf et al.[1] give more information on the GFED4s modelling approach.

To calculate the PyC production within GFED4s we added the production factor P_{PyC}, which quantifies the production of PyC per unit carbon emitted. Until now, the principle obstacle to performing a global modelling exercise of this type was the lack of a sufficiently rich and standardized dataset with which to constrain representative values for P_{PyC}.

Our estimates of uncertainty in the annual PyC production relate only to variability in the PyC production factors and interannual variability in emissions and do not include uncertainties in carbon emission estimates propagate from GFED4s. Uncertainties in GFED4s emissions estimates are discussed at length in van der Werf et al.[1,74] and are predominantly the result of uncertainties in the satellite detection of small fires using thermal anomalies and burn scars. As carbon emissions and PyC production are codependent on the burned area, estimation errors that relate to fire detection introduce scalar uncertainties. Uncertainty in the fuel consumption is an additional component of the overall uncertainty in GFED4s emission estimates[1] and has been reduced from previous versions (for example, GFED3) through its incorporation of a global dataset of fuel consumption estimates[73]. As discussed in the primary literature that relates to the development of the GFED4s[1], a formal global-scale assessment of the uncertainties in fuel consumption cannot be completed due to a paucity of ground truth data for some input datasets. For the previous version of GFED (GFED3), Monte Carlo simulations that accounted for uncertainty in both burned area detection and fuel consumption were used to obtain first-order constraints on the uncertainty in carbon emissions, which were ±20–25% at global, annual scales as a 1 s.d. (1σ) value[74]. Developments of GFED4s included the incorporation of small-fire burned area detection, which led to important reductions in the negative bias in the emissions estimates[72]; however, small fires are also challenging to detect and a lack of validation data prevents the formal investigation of uncertainty in burned area for GFED4s[1,72]. Hence, the true uncertainty of GFED4s is not known precisely, but it is likely to be on the same order as that of GFED3 ($1\sigma = \pm 20$–25%). Nonetheless, uncertainty ranges are likely to be greater in regions where small fires are prevalent or where organic soils are affected (for example, Central America, Europe and Equatorial Asia)[1,72].

Regional-scale field studies of fire emissions have served to validate that the GFED modelling framework produces reliable estimates at large scales, for example, in Alaska[75] and the tropics[76]. Studies that involve atmospheric tracers have also provided vital diagnostics for the performance of GFED[1], and generally highlight its proficiency at large scales but reveal some weaknesses in specific regions or during isolated events[77–82]. Overall, GFED4s is highly suited to the investigation of the effects of fire in global-scale biogeochemical cycles and is thus regularly used in GCB assessments[4] and as a reference point for the fire modules of ESMs[7].

Collating a global dataset of PyC production factors. We compiled a new database of P_{PyC} factors (Supplementary Dataset) from a global collection of 22 published studies that reported on PyC production in 91 burn units, as well as two new datasets produced by the authors with 23 burn units reported for the first time here, and we standardized their reporting. All the studies used one of the following two broad approaches to quantify the impacts of fire on the biomass carbon stocks, either prefire and postfire stocks of biomass carbon and PyC are measured or space-for-time substitution is used to constrain burned and unburned stocks of biomass carbon and PyC, which are assumed to be equivalent to prefire and postfire stocks, respectively. Hereafter, the terms 'prefire' and 'postfire' are used to refer to both types of assessment. Here we focus only on PyC present in charcoal and ash[83] on the ground following fire and on charred vegetation. PyC emitted with smoke, transported in the atmosphere and deposited on a regional-scale area is not included as this process has been studied in separate dedicated studies conducted by atmospheric scientists[23] and represents a relatively small flux in comparison (see main text)[12,13].

The P_{PyC} values were calculated for each of the six classes of widely used biomass components: CWSF, which includes CWD or downed wood defined by typical diameter thresholds of >7.6 cm or >10 cm (refs. [84,85]); FWSF, which includes fine woody debris or any other woody debris with diameters below the thresholds for CWSF; CWAGF, which includes trees or branches with diameters greater than the thresholds for CWSF; FWAGF, which includes material described as shrubs, trees or branches with diameters below the thresholds for CWSF; NWSF, which includes litter, understory vegetation, grass, root mat and any other form of non-woody material directly in contact with the ground surface[85,86] and, finally, NWAGF, which includes foliage, leaves, needles, crown fuels and any other forms of non-woody material that attach to standing wood structures above the ground surface.

For each biomass component, P_{PyC} (PyC produced per C emitted) was calculated using the following equation:

$$P_{\mathrm{PyC}} = \frac{C_{\mathrm{Py}}}{C_{\mathrm{PRE}} - C_{\mathrm{POST}} - C_{\mathrm{Py}}}$$

where C_{Py} is the mass of PyC created during the fire that was attributed to the component, C_{PRE} is the prefire stock of biomass carbon in the component and C_{POST} is the postfire stock of biomass carbon in the unburnt component. C_{Py}, C_{PRE} and C_{POST} are all expressed in the units g C km^{-2}.

Criteria were applied as filters to the dataset to ensure that P_{PyC} could be calculated in a consistent and representative manner. Specifically, P_{PyC} was calculated if the following conditions were met: first, both prefire and postfire biomass stocks were reported and the carbon content (%) was either measured or assumed based on representative values from the literature; second, postfire stocks of pyrogenic organic matter (charcoal, ash and the charred components of partially affected vegetation) were reported and their PyC content (%) was either measured or assumed based on representative values from the literature; third, the type of fire that occurred was representative of a widespread regional fire type (for example, wildfires, slash-and-burn deforestation and prescribed fire) and fourth, in experimental fires, the biomass carbon stock was designed to replicate the density and structure of biomass carbon stocks observed in the field and the burning efficiency was not optimized or adapted as a factor of the study design.

The set of criteria outlined above does not exclude studies that assess the PyC content of charcoal using one of the various chemical or thermochemical techniques available for the separation of PyC from bulk organic carbon[87,88]. Such techniques are frequently used for the detection of PyC in well-mixed soil, sediment and aquatic matrices. However, we note that none of the studies included in our dataset utilized a chemical or thermochemical approach to separate PyC from non-PyC; instead, these studies considered all the organic carbon in residual products of interest (charcoal, ash and the charred components of partially affected vegetation) to be PyC. Thus, we highlight that our estimates of P_{PyC} are free of the intermethod variability in PyC quantification that often confounds the comparison of PyC concentration in environmental matrices across studies and contributes to the notable uncertainty in the magnitude of Earth's major PyC stocks[12,13] (Fig. 1).

Like biomass carbon, total PyC stocks are distributed across several components, which include charcoal and ash on the ground, charcoal attached to CWD and charcoal attached to aboveground vegetation[12]. The majority of the studies included in the production factor dataset matched the studied PyC components to individual biomass carbon components from which they were known to derive. However, as some individual components of the PyC stocks can have a mixture of sources that are indistinguishable from their location or appearance alone, it was occasionally necessary to make assumptions about the biomass components that were sources of these components. This was done on a study-by-study basis. In cases where the source of each PyC component was not explicitly stated, the following procedural steps were adhered to. On a first basis, the PyC component was assigned to a biomass component according to the most probable source inferred, but not explicitly stated, in the primary literature. Second, where more than one biomass component was inferred to be a source of the PyC stock in the primary literature, the PyC stock was weighted proportionally to the prefire stock of carbon present in each of the implicated biomass components. Otherwise, if no sources of PyC were inferred in the primary literature it was necessary to make independent assumptions about the source of PyC in a manner that was consistent with the other studies included in the dataset and our collective experience of quantifying PyC production in the field.

Summary of the production factor values for use in GFED4s+PyC. Our global database suggested that CWSF and CWAGF produce significantly more PyC, relative to carbon emitted, than other fuel classes (their P_{PyC} averaged at 0.25 and 0.31 g PyC g^{-1} C emitted, respectively (Fig. 2)). In contrast, the mean P_{PyC} values for FWSF and FWAGF (0.12 and 0.076 g PyC g^{-1} C emitted, respectively) did not differ significantly from those of NWSF and NWAGF (0.099 and 0.062 g PyC g^{-1} C emitted, respectively). These results are consistent with previous studies, which suggest that large-diameter woody fuels burn less completely and produce PyC in greater proportions than finer fuels[34,35].

For each class, the mean PyC production factor was used as the central estimate for P_{PyC} and the confidence interval around the mean P_{PyC} was calculated through a bootstrapping procedure. Specifically, the available PyC production factors from the dataset were resampled 50,000 times, the mean P_{PyC} was calculated for each resample and the 95% confidence interval was calculated as the middle 95% of the observed 50,000 means (that is, those ranked 1,250th to 48,750th).

According to an analysis of variance with a Tukey honest significant difference post hoc test, no significant differences in mean P_{PyC} were observed between the distributions of P_{PyC} for coarse, fine and non-woody fuels positioned at the ground surface and those same fuels located above the ground surface. Therefore, the

P_{PyC} values applied in GFED4s+PyC are based on the distribution of values in three simplified fuel classes (Fig. 2): CWF (mean 0.26 g PyC g^{-1} C; 95% confidence interval, 0.18–0.39 g PyC g^{-1} C), FWF (mean 0.096 g PyC g^{-1} C; 95% confidence interval, 0.064–0.15 g PyC g^{-1} C) and NWF (mean 0.091 g PyC g^{-1} C; 95% confidence interval, 0.074–0.11 g PyC g^{-1} C).

Assigning PyC production factors in GFED4s+PyC. P_{PyC} values were assigned to each of the native fuel classes of GFED4s[1], which are leaves, grasses, surface fuels (which include litter and fine woody debris), CWD and standing wood (which includes trunks, stems and branches). Mean P_{PyC} values and bootstrapped confidence interval values for CWF, FWF and NWF from the global dataset were used to define representative P_{PyC} values for each of the GFED4s fuel classes (Fig. 2). Full details as to the assignment of P_{PyC} values to each GFED4s fuel class are provided in Supplementary Section 3 and Supplementary Table 3). Briefly, leaf, litter and grass were assigned the relevant P_{PyC} values of NWF, fine woody debris and CWD were assigned the values of FWF and CWF, respectively, and P_{PyC} values for standing wood were applied in a spatially explicit manner as weighted combinations of the P_{PyC} values for CWF (carbon in trunks) and FWF (carbon in branches). The weighted CWF:FWF ratio was assigned according to empirical relationships that defined biomass carbon apportionment to branches and trunks in the various forest types of the GFED4s land cover scheme (Supplementary Section 3 and Supplementary Table 4)[89].

Quantifying ENSO impacts on PyC production. To investigate the influence of pantropical climatic variability driven by the ENSO on the production of PyC, we replicated the analysis presented by Chen et al.[46] with a focus on PyC production rather than on carbon emissions. The pantropics were defined as consisting of Central America, Northern Hemisphere South America, Southern Hemisphere South America, Northern Hemisphere Africa, Southern Hemisphere Africa, Southeast Asia, Equatorial Asia and Australia (Supplementary Fig. 6). The PyC production in El Niño and La Niña phases was compared for the major fire season periods defined in each tropical region by Chen et al.[46]; their study gives a thorough explanation of the rationale for selecting these comparison periods. We summed PyC production in the major fire season period of each region and disaggregated this total to forest and non-forest fires according to the dominant land cover type in the GFED4s land cover scheme (based on the MODIS Land Cover Type Climate Modelling Grid product MCD12C1)[90].

Apportioning sources of PyC. After the GFED4s+PyC model runs, PyC production was assigned to specific sources following a method developed previously for use in GFED4s model runs[1,74]. Specifically, PyC production that occurs as a result of non-deforestation fires was disaggregated in each cell to tropical forest, savannah/grassland, boreal forest, temperate forest and agricultural fires using an existing algorithm that utilizes fractional tree cover, climate and fire-persistence variables. van der Werf et al.[74] give a full discussion of this algorithm. We added an additional latitudinal constraint (30° N to 30° S) to further disaggregate the savannah compartment, which thus separates tropical savannahs and grasslands from extratropical grasslands.

Data availability

The global dataset of the PyC production factors is available as a supplementary data file (GlobalPyC_supplementarydataset.xlsx). This dataset will also be uploaded to the GFED website (http://www.globalfiredata.org) and updated with new data as it becomes available. Supplementary Section 4 contains full references to the studies included in the production factor dataset. Burned area and fire emissions data are publicly available at the GFED website. Additional ancillary data are available from the corresponding author on request.

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Livestock Impacts on Riparian Ecosystems and Streamside Management Implications... A Review

J. BOONE KAUFFMAN AND W.C. KRUEGER

Historically, riparian vegetation has been defined as vegetation rooted at the water's edge (Campbell and Franklin 1979). Quite often, however, the stream influences vegetation in many ways and well beyond the water line. In lotic systems, the stream is not only responsible for increased water availability, but also for the soil deposition, unique microclimate, increased productivity, and the many consequential, self-perpetuating biotic factors associated with riparian zones. These factors all contribute in the formation of a unique assemblage of plant communities quite distinct from upland communities surrounding the riparian zone. Therefore, along streambanks, other lotic systems, and even ephemeral drainages, riparian ecosystems could best be defined as those assemblages of plant, animal, and aquatic communities whose presence can be either directly or indirectly attributed to factors that are stream-induced or related (Kauffman 1982).

Riparian zones can vary considerably in size and vegetation complexity because of the many combinations that can be created between water sources and physical characteristics of a site (Odum 1971, Platts 1979, Swanson et al. 1982). Such characteristics, include gradient, aspect, topography, soil type of streambottom, water quality, elevation, and plant community (Odum 1971). However, riparian zones, particularly those bordering streams or rivers, have several characteristics in common. They are ecotonal, with high edge to area ratios (Odum 1978). As functional ecosystems they are very open with large energy, nutrient, and biotic interchanges with aquatic systems on the inner margin (Cummins 1974, Odum 1978, Sedel et al. 1974) and upland terrestrial ecosystems on the other margin (Odum 1978).

Thomas et al. (1979) stated that all riparian zones within managed rangelands of the western United States have the following in common: (1) they create well-defined habitat zones within the much drier surrounding areas; (2) they make up a minor proportion of the overall area; (3) they are generally more productive in terms of biomass—plant and animal—than the remainder of the area; and (4) they are a critical source of diversity within rangelands. Both density and diversity of species tends to be higher at the land/water ecotones than in adjacent upland, especially where regional climates are characterized by dry periods (Odum 1978). Ganskopp (1978) described 44 vegetation communities in a 49-hectare riparian zone in the Blue Mountains of northeastern Oregon. Kauffman et al. (1984) stated that the several biotic, environmental and other abiotic factors interacting in a riparian zone in Oregon created a disproportionately greater number of niches compared to surrounding upland ecosystems. Two-hundred and fifty-eight stands of vegetation representing 60 discrete plant communities were identified within this study area. The higher diversity, productivity, and other unique factors associated with the riparian zone when compared to the surrounding uplands are the primary factors that create the importance of these areas as focal points for the management of the livestock, fishery, and wildlife resources.

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Importance of Riparian/Stream Ecosystems

Importance to Instream Ecosystems

Vegetation along small streams is an important component of the riparian/stream ecosystem (Campbell and Franklin 1979, Jahn 1978). Riparian vegetation produces the bulk of the detritus that provides up to 90% of the organic matter necessary to support headwater stream communities (Cummins and Spengler 1978). In these tributaries of forest ecosystems 99% of the stream energy input may be imported from bordering riparian vegetation (i.e., it is heterotrophic) and only 1% derived from stream photosynthesis by attached algae (periphyten) and mosses (Cummins 1974). Berner (in Kennedy 1977) found that even in large streams such as the Missouri River, 54% of the organic matter ingested by fish is of terrestrial origin. The riparian zone vegetation functions both in light attenuation and as the source of allochthonous inputs, including long-term structural and annual energy supplies (Cummins 1974).

Vegetation along streams exercises important controls over physical conditions in the stream environment. It acts as a roughness element that reduces the velocity and erosive energy of overbank flow during floods (Li and Shen 1973). The result is a higher flood peak than a channel without riparian vegetation but lower erosional factors acting on the floodplain and bank (Schumm and Meyer 1979). Healthy riparian vegetation tends to stabilize streambanks, determines bank morphology and may help reduce streambank damage from ice, log debris, and animal trampling (Platts 1979, Swanson et al. 1982).

Channel and floodplain obstructions such as branches, logs, and rocks enhance detention and concentration of organic matter, thereby facilitating its use locally rather than washing downstream (Everest and Meehan 1981, Jahn 1978, Swanson et al. 1982). In addition, wood debris in channel bottoms appears to play an important role in the dynamics of stream morphology. Large pieces of woody debris in streams dissipate stream energy, control routing of sediment and water through channel systems, and serve as substrates for biological activity by microbial and invertebrate organisms (DeBano 1977, Swanson et al. 1982).

Streamside vegetation strongly influences the quality of habitat for anadromous and resident coldwater fishes (Duff 1979, Everest and Meehan 1981, Marcuson 1977, Meehan et al. 1977). Riparian vegetation provides shade, preventing adverse water temperature fluctuations (Meehan et al. 1977). The roots of trees, shrubs, and herbaceous vegetation stabilize streambanks, providing cover in the form of overhanging banks (Marcuson 1977, Meehan et al. 1977). Streamside vegetation acts as a "filter" to prevent sediment and debris from man's activities from entering the stream (Meehan et al. 1977). Riparian vegetation also directly controls the food chain of the ecosystem by shading the stream and providing organic detritus and insects for the stream organisms (Cummins 1974, Meehan et al. 1977).

Importance to Wildlife

It is believed that, on land, the riparian/stream ecosystem is the single most productive type of wildlife habitat, benefiting the greatest number of species (Ames 1977, Hubbard 1977, Miller 1951, Patton 1977). The riparian zone provides an almost classic example of the ecological principles of edge effect (Odum 1978). Riparian habitat provides living conditions for a greater variety of wildlife than any other types of habitat found in California (Sands and Howe 1977), the Great Basin of southeast Oregon (Thomas et al. 1979), the Southwest (Hubbard 1977), the Great Plains (Tubbs 1980), and perhaps the entire North American continent (Johnson et al. 1977).

Examples of the wildlife values of riparian habitat are numerous (Carothers et al. 1974, Carothers and Johnson 1975, Henke and Stone 1978, Hubbard 1977, Thomas et al. 1979). Hubbard (1977) reported that 16-17% of the entire breeding avifauna of temperate North America occurs in 2 New Mexico river valleys over the course of a "few score" miles. Johnson et al. (1977) reported that 77% of the 166 nesting species of birds in the Southwest are in some manner dependent on water related (riparian) habitat and 50% are completely dependent on riparian habitats. In western Montana, 59% of the land bird species use riparian habitats for breeding purposes and 36% of those breed only in riparian areas (Mosconi and Hutto 1982). Thomas et al. (1979) stated that of the 363 terrestrial species known to occur in the Great Basin of southeastern Oregon, 299 are either directly dependent on riparian zones or utilize them more than any other habitats.

When riparian vegetation is eliminated, several wildlife species dependent on riparian ecosystems may be either severely reduced or may disappear altogether. Henke and Stone (1978) found 93% fewer bird numbers and 72% fewer avian species on 2 riprapped plots from which riparian vegetation had been removed, and 95% fewer birds and 32% fewer species on cultivated lands previously occupied by riparian forests.

The influence of riparian ecosystems on wildlife is not limited to those animal species that are restricted in distribution to the streamside vegetation. Population densities of birds in habitats adjacent to the riparian type are influenced by the presence of a riparian area (Carothers 1977). When a riparian habitat is removed or extensively manipulated, not only are the riparian species of the area adversely influenced, but wildlife productivity in the adjacent habitat is also depressed (Carothers 1977).

Riparian ecosystems are valuable to wildlife as a source of water, food, and cover (Stevens et al. 1977, Thomas et al. 1979). They also provide nesting and brooding habitat for avian species (Carothers et al. 1974, Johnson et al. 1977, Tubbs 1980). By furnishing abundant thermal cover and favorable micro-climates, especially when surrounded by nonforested ecosystems, they facilitate the maintenance of of homeostatis, particularly for big game (Thomas et al. 1979). Riparian ecosystems also serve as big game migration routes between summer and winter range (Thomas et al. 1979), and provide routes and nesting cover for migrating avian species (Stevens et al. 1977, Wauer 1977).

Importance to Livestock

Livestock grazing on rangelands is the most extensive form of land use in the interior Pacific Northwest (Skovlin et al. 1977). Cattle tend to congregate on meadows and utilize the vegetation much more intensively than the vegetation of adjacent ranges (Reid and Pickford 1946).

In northeast Oregon, Reid and Pickford (1946) stated that moist meadow soils in riparian ecosystems are generally so highly productive than an acre of mountain meadow has a potential grazing capacity equal to 10-15 acres of forested range. Although riparian meadows cover only about 1-2% of the summer range area of the Pacific Northwest, potentially they can produce 20% of the summer range forage (Reid and Pickford 1946, Roath and Krueger 1982). However, Roath and Krueger (1982) found that because of livestock concentrations, limits on livestock movements imposed by steep slopes, and erratic distribution of watering areas away from the creek, the riparian zone (covering about 2% of a Blue Mountain grazing allotment) accounted for 81% of the total herbaeous vegetation removed by cattle.

Cattle exhibit a strong preference for riparian zones for a number of the same reasons other animals prefer and use these areas. The main attributes believed to attract and hold cattle to riparian areas are the availability of water, shade, and thermal cover, and the quality and variety of forage (Ames 1977, Severson and Boldt 1978). In addition, sedges (Carex spp.) tend to retain relatively constant crude protein levels until the first killing frost. Several sedges common to riparian zones of the Pacific Northwest outrank key upland forage species in sustained protein and energy content (McLean et al. 1963, Paulsen 1969, Skovlin 1967).

Livestock Riparian Relationships

The impact of livestock on riparian zones in public grazing lands of the western states has received much attention recently. Several studies are presently underway examining the impact of livestock grazing on stream ecology, water quality, channel stabilization , salmonid fish habitat and physiology, terrestrial riparian wildlife populations, and riparian vegetation.

It is often difficult for one to interpret science from opinion in the literature. Many of the studies reported in this paper have not necessarily followed the generally accepted "scientific method" for research today. However, it is not the purpose of this paper to determine, even if possible, which published reports represent quality scientific results and which are little more than a forum to express one's opinion. Rather the purpose of this paper is to familiarize the reader with the accepted facts and management theories available today concerning livestock interactions in riparian zones with the other valid resources also dependent or utilizing this resource. Where possible, in this paper, results of properly conducted research are reported using terms such as "significant", referring to a statistically significant result and those of reports relying on observational data or "hearsay" will be reported as suggestions or observations.

General Considerations for Livestock-Riparian Management

The quality of the riparian habitat and its associated aquatic environment, both formed over geologic time, are fragile ecosystems which currently serve as focal points for management of livestock, recreation, and fisheries and timber resources. It has been reported that inappropriate livestock management results in overuse and subsequent degradation of the riparian/stream ecosystem (Behnke and Raleigh 1978, Oregon-Washington Interagency Wildlife Council 1978, Platts 1979). Davis (1982) suggested that one of the most destructive forces in riparian ecosystems is the long-term impact of overgrazing by cattle. Livestock grazing can affect 4 general components of an aquatic system-streamside vegetation, stream channel morphology, shape and quality of the water column and the structure of the soil portion of the streambank (Behnke and Raleigh 1978, Marcuson 1977, Platts 1979, Platts 1981). Improper livestock use of riparian ecosystems can affect the streamside environment by changing, reducing, or eliminating vegetation bordering the stream (Ames 1977, Behnke and Raleigh 1978, Platts 1979). The channel morphology can be changed by widening and shallowing of the streambed, gradual stream channel trenching, or braiding, depending on soils and substrate composition (Behnke and Raleigh 1978, Gunderson 1968, Marcuson 1977, Platts 1979). The water column can be altered by increasing water temperatures, nutrients, suspended sediments, bacterial counts and by altering the timing and volume of water flow (Behnke and Raleigh 1978, Johnsen et al. 1978, Rauzi and Hanson 1966, Platts 1979). Overgrazing can cause bank sloughoff creating false setback banks, accelerated sedimentation, and subsequent silt degradation of spawning and food producing areas (Behnke and Raleigh 1978, Everest and Meehan 1981, Platts 1979, Platts 1981). These impacts on the water column due to abusive livestock practices result in decreased fish biomass and in percent of salmonid fishes in the total fish composition (Behnke and Raleigh 1978, Bowers et al. 1979, Duff 1979, Gunderson 1968, Marcuson 1977).

Livestock abuse of riparian areas can severely impact terrestrial wildlife habitat causing a subsequent decrease in wildlife species and numbers (Ames 1977, Townsend and Smith 1977, Tubbs 1980, Wiens and Dyer 1975).

Improper grazing can have a considerable effect on vegetation, resulting in decreased vigor, biomass and an alteration of species composition and diversity (Ames 1977, Bryant et al. 1972, Evans and Krebs 1977, Knoph and Cannon 1982, Pond 1961).

While various other land management activities have caused serious losses or reductions in wildlife habitat productivity, livestock grazing has been suggested as the major factor identified in numerous studies throughout the 11 western states (Oregon-Washington Interagency Wildlife Council 1978). Conversely, Busby (1979) suggested that it was not reasonable to conclude that livestock grazing is the only, nor necessarily the major cause of impacts to riparian ecosystems.

Impacts of Livestock on the Instream Ecology

A healthy instream environment is vital for the aquatic life forms inhabiting the stream, as well as for various human needs that directly depend on water quality. High concentrations of suspended solids or other sediment loads, and fecal coliforms or fecal streptococci are usually associated with the degree of impact of man's activities, and can have a major impact in altering an existing stream ecosystem or even creating an entirely new ecosystem (Johnson et al. 1977, Johnson et al. 1978, McKee and Wolf 1963).

During the grazing season, Johnson et al. (1978) could not find any significant differences in physical and chemical properties of streamwater (suspended solids, total dissolved solids, and orthophosphates) between an area grazed at 1.2 ha/AUM and an ungrazed area. After the grazing season, however, there was a significant increase in total dissolved solids which indicated that some livestock waste products may have eventually reached and enriched the stream, probably from the action of rain showers. The presence of cattle significantly elevated the fecal coliform and fecal streptococci for about 9 days after cattle were removed.

Winegar (1977) found sediment loads were reduced 48–79% while flowing through 3.5 miles of a stream protected from grazing.

Rauzi and Hanson (1966) found a nearly linear relation between runoff and infiltration to the degree of grazing intensity. They found that runoff from a heavily grazed watershed (1.35 acre/AUM) was 1.4 times greater than from a moderately grazed watershed (2.42 acre/AUM) and 9 times greater than from a lightly grazed watershed (3.25 acre/AUM).

Changes in water temperature have been shown to have drastic effects on fisheries and aquatic insect populations (Johnson et al. 1977). Changes in average temperature or daily fluctuations can in effect create an entirely new aquatic ecosystem (Johnson et al. 1977).

Van Velson (1979) found average water temperatures dropped from 24°C to 22°C after 1 year of livestock exclusion on a creek in Nebraska. Claire and Storch (unpublished) compared stream temperatures between an area that had been grazed season long (June 1–October 15) and an area that had been rested for 4 years and, thereafter, grazed only after August 1. The maximum water temperatures outside and downstream from the exclosure averaged 7°C higher than those sampled within the exclosure. Daily fluctuations of water temperatures averaged 15°C outside the exclosure as compared to 7°C inside the exclosures. Winegar (pers. comm. 1982) observed similar results in an exclosure along Beaver Creek in central Oregon.

The effects of livestock grazing have been shown to vary greatly depending upon several factors, in particular, the nature of the stream studied. Duff (1979) stated that introduction of livestock for 6 weeks into a riparian area rested for 4 years resulted in elimination of overhanging banks and a fracturing of the streambank, causing it to erode into the stream. In contrast, after 6 weeks of mid-summer grazing by cattle, Roath (1980) gave a visual estimate of 90% bank stability with little indication that trampling was contributing to or causing erosion. He attributed nearly all erosion present to geologic erosion caused by the actions of streamflow.

Buckhouse et al. (1981) could find no particular relationship between streambank erosion and various grazing treatments (including nonuse) in northeastern Oregon. There appeared to be no significant patterns of accelerated streambank deterioration due to moderate livestock grazing (3.2 ha/AUM and 60–65% utilization of the riparian vegetation). Most bankcutting losses in this system were associated with over-winter periods where ice floes, high water, and channel physiognomy were critical factors involved in the erosional process.

Hayes (1978) found that stream channel movement did not occur more frequently in grazed riaprian meadows under a rest-rotation grazing scheme compared to ungrazed meadows after 1 year of study. Rather, streambank degradation appeared to occur more often and to a greater magnitude along ungrazed streams. However, Hayes stated that sloughoff increased as forage removal was above 60%. High forage removal, high amount of foraging time along banks, and high percentages of palatable sedges along the bank were shown to significantly increase the probability of sloughoff occuring during the grazing season.

Kauffman et al. (1983b) measured significantly greater streambank losses in grazed areas (1.3–1.7 ha/AUM) compared to ungrazed areas in northeastern Oregon. The grazed pastures had utilization levels greater than 35% and less than 85% on the different vegetation stands while utilization by native animals was less than 20% on every stand. During 2 late season grazing periods (late August–mid-September), a mean of 13.5 cm of streambank was lost in grazed areas and 3.0 cm was lost in ungrazed areas. Total annual streambank losses were 30 cm in grazed areas and 9 cm in ungrazed areas.

Marcuson (1977) found the average channel width to be 53 meters in an area grazed season long at 0.11 ha/AUM and an average channel width of only 18.6 meters in areas that were ungrazed. Marcuson (1977) also recorded 224 meters of undercut bank/km in the grazed area and 686 meters of undercut bank/km in the ungrazed area. Heavy grazing and trampling by cattle were suggested to cause the excessive erosion.

Duff (1979) found the stream channel width in a grazed area was 173% greater than the stream channel not grazed for 8 years inside an exclosure. Similar results have been reported (Behnke and Zarn 1976, Dahlem 1979, Gunderson 1968, Heede 1977) where overgrazing and excessive trampling caused a decrease in bank undercuts, increases in channel widths, and a general degradation of fish habitat.

Claire and Storch (unpublished) stated that the production of game fish in headwater streams can be used as a biological indicator of the quality of land management that is occurring within the watershed and/or streamside. Overgrazing, causing a reduction in vegetative cover and the caving in of overhanging banks, has been suggested as one of the principal factors contributing to the decline of native trout in the West (Behnke and Zarn 1976).

Bowers et al. (1979) reported an average increase in fish production of 184% for 5 independent studies where livestock use was light or eliminated by fencing. They concluded with a prediction that trout production in streams currently being heavily grazed could be increased about 200% if management decisions were made to optimize habitat conditions for trout.

Van Velson (1979) found rough fish made up 88% of a fish population before relief from grazing and only 1% of the population after 8 years' rest. Rainbow trout *(Salmo gairdneri)* made up 1% of the fish population before cessation of grazing and 97% of the population after relief from grazing. Marcuson (1977) found that an overgrazed section (.11 ha/AUM) of Rock Creek, Montana, supported only 71 kg of brown trout *(Salmo trutta)* per hectare; whereas an ungrazed section produced 238.8 kg of brown trout per hectare. Claire and Storch (unpublished) found in the Blue Mountains of Oregon that game fish were 24% of the total population in area grazed season long, contrasted to a 77% game fish composition within a livestock exclosure.

Chapman and Knudsen (1980) found 8 sections of streamside vegetation in western Washington, judged to be moderately to heavily affected by livestock, had significant reductions in total biomass for Coho salmon *(Oncorhychus kisutch),* Cutthroat trout *(Salmo clarki),* and all salmonids compared to those areas that had not been grazed. Similar relationships between livestock grazing and salmonid fish populations have been reported by Dahlem (1979), Duff (unpublished), Gunderson (1968), Keller et al. (1979), and Lorz (1974).

Impacts of Livestock on Terrestrial Wildlife

Riparian zones are the most critical wildlife habitats for many species in managed rangelands (Thomas et al. 1979). It is readily apparent that riparian ecosystems are of paramount importance in producing and maintaining a large degree of biotic diversity in North America (Hubbard 1977, Johnson et al. 1977).

Changes in plant vigor, growth form and species composition due to grazing have frequently been related to the increase or decline of various species of birds (Townsend and Smith 1977). Several studies have shown a negative impact on certain avian populations due to grazing (Dambach and Good 1940, Overmire 1963, Owens and Meyers 1973, Reynolds and Trost 1980, Smith 1940). The tendency for livestock to congregate and linger around ponds and streambanks may result in the elimination of food and cover plants and reduces nest sites and habitat diversity (Buttery and Shields 1975, Behnke and Raleigh 1978, Crouch 1978, Evans and Krebs 1977). However, grazing may improve habitat for some avian species (Burgess et al. 1965, Crouch 1982, Kirch and Higgins 1976). In areas of higher precipitation (or productivity), grazing may be highly desirable to open up "roughs" and provide more diversity and patchiness (Ryder 1980). Grazing effects on breeding avifaunas are not uniform nor easily defined, primarily because grazing varies so much in its local intensity and because of the general difficulties in unraveling cause-effect relationships in rangeland faunas (Wiens and Dyer 1975).

Several studies have shown wildlife numbers increased when a riparian area that was abused by improper grazing practices was fenced and allowed to recover (Crouch 1978, 1982, Duff 1979, Van Felson 1979, Winegar 1977). Duff (1979) reported a 350% increase in small mammal songbird and raptor use after 8 years' rest from grazing. Van Velson (1979) reported increased pheasant *(Phasianius colchicus)* production, increased deer populations, and that watefowl production occurred for the first time in the rested area. Crouch (1982) found more ducks (primarily mallards) *(Anas platyrhynchos),* more upland game animals, and twice as many terrestrial birds in an ungrazed bottomland rested for 7 years compared to adjacent grazed bottomlands on the South Platte River in northeastern Colorado. The grazed areas, utilized at "varying intensities, provided habitat for significantly more aquatic species of birds.

Mosconi and Hutto (1982) found no significant differences in total bird densities between heavily grazed riparian communities (2.5 cow-calf units/ha) and lightly grazed riparian communities (0.3 cow-calf units/ha). However, significant differences were recorded in bird species composition and foraging guilds. The majority of the bird species significantly affected were of the flycatcher, ground-foraging thrust, or foliage-gleaning insectivore guilds.

Similar results were reported by Kauffman (1982) and Kauffman et al. (1982). No significant differences in total avian densities were noted between riparian communities grazed under a late-season grazing scheme (2.0–2.5 ha/AUM) and those totally excluded from grazing. However, forage removal causing a change in habitat physiognomy did appear to cause some differential use in species and foraging guilds. These differences were particularly evident immediately after forage removal and negligible during seasons when cover and plant growth were similar between treatments. The grazed riparian communities were preferred by birds of insect foraging guilds; ungrazed riparian communities were preferred by birds of herbivorous/granivorous foraging guilds.

Livestock grazing and the subsequent removal of forage in the riparian zone has been shown to cause significant short-term decreases in small mammal composition and densities (Kauffman et al. 1982). When mammal densities before and after the grazing season in 1979 (stocking rate of 2.0–2.5 ha/AUM) were compared, small mammal communities decreased from 800 to 83 mammals/ha in Douglas hawthorn *(Crataegus douglasii)*-dominated communities; from 450 to 60 mammals/ha in riparian meadow communities; and from 129 to 42 mammals/ha in black cottonwood *(Populus trichocarpa)*-mixed conifer communities. By late summer the following year (10 months after grazing) and just prior to the grazing season, small mammal densities were not significantly different between grazed and ungrazed areas.

When properly managed, the grazing of domestic livestock is generally compatible with wildlife, and may even increase the numbers of some species (Tubbs 1980). Nongame wildlife which depend on riparian ecosystems have intangible values which are very hard to evaluate (Peterson 1980). It has been demonstrated that livestock can graze streamsides without causing serious damage, and the capability to achieve positive on-site livestock control appears to be the limiting factor (Claire and Storch unpublished).

Impacts of Livestock on Riparian Vegetation

Recently there has been much published research and opinion on the effects of livestock in riparian ecosystems. Specifically, these reports have dealt with soil compaction and its relationship to root growth; plant succession and productivity; and species diversity and vegetation structural diversity. Opinions on the subject have varied from there being no evidence of heavy, season-long cattle grazing affecting the productivity of a riparian zone, or causing bank deteriorations by trampling (Roath 1980) to grazing only a few days seriously impairing a riparian zone's reproductive capability.

Impacts to riparian vegetation induced by livestock can basically be separated into: (a) compaction of soil, which increases runoff and decreased water availability to plants; (b) herbage removal, which allows soil temperatures to rise and increases evaporation to the soil surface; and (c) physical damage to vegetation by rubbing, trampling, and browsing (Severson and Boldt 1978).

Impacts of Trampling

The impact of livestock trampling on soil compaction bulk density and subsequent effects on forage growth have been documented. Alderfer and Robinson (1949), Bryant et al. (1972), Orr (1960), and Rauzi and Hanson (1966) all found soil compaction increased linearly with increases in grazing intensity.

Alderfer and Robinson (1949) found grazing and trampling Kentucky bluegrass *(Poa pratensis)* upland pastures to a 1-inch (2.5 cm) stubble height reduced vegetation cover, lowered yields, decreased noncapillary porosity, and increased the volume weight of the 0-1 inch (0-2.5 cm) layer of soil.

Rauzi and Hanson (1966) found water intake rates on silty clay and silty clay loam soils to be 2.5 times greater in an area grazed at 1.35 acres/AUM compared to an area grazed at 3.25 acres/AUM. After 22 years of grazing at this intensity, not only had species composition been altered but soil properties had been changed as well.

In a riparian zone continuously grazed season long, Orr (1960) found bulk density and macropore space to be significantly greater in grazed areas over exclosures. Differences in total pore space (both macro- and micro-pores) between grazed and exclosed areas were small because of a transformation of macropore spaces to micropore spaces by trampling. Macropore space is a more sensitive indicator of compaction or recovery from compaction than either micro or total pore space (Orr 1960).

Bryant et al. (1972) found increasing trampling pressure had an adverse effect on Kentucky bluegrass swards, particularly during the months of June and September. After one overwinter period, there was a significant difference in soil compaction between an area trampled by 120 cow trips over bluegrass plots and an area that was untrampled.

Impacts of Herbage Removal

Impacts of herbage removal can be divided into 2 categories according to vegetation structure: (1) utilization of herbaceous vegetation and subsequent impacts on species composition, species diversity, and biomass produced and (2) utilization of woody vegetation and subsequent impacts on foliage cover, structural height diversity and stand reproduction.

A major vegetation change that has taken place in mountain riparian systems of the Pacific Northwest is replacement of native bunchgrass with Kentucky bluegrass. It has successfully established itself as a dominant species in native bunchgrass meadows as a result of overgrazing by herbivores and subsequent site deterioration (Volland 1978).

Pond (1961), in Wyoming, found clipping native bunchgrass meadows every 2 weeks for 4 years caused a marked reduction in native sedges (*Carex* spp.), tufted hairgrass *(Deschampsia caespitosa)* and fostered the appearance of Kentucky bluegrass where it was not present before. Kauffman et al. (1983a) found that when grazing was halted in moist meadows, succession towards a more mesic/hydric plant community occurred. Exotic grasses such as meadow timothy *(Phleum pratense)* and forbs more attuned to drier environments were decreasing and were being replaced by native sedges and mesic forbs.

In central Oregon, Evenden and Kauffman (unpublished) compared plant communities on each side of a fence that was heavily grazed on one side and protected from grazing on the other. The grazed site was dominated by Kentucky bluegrass and Baltic rush *(Juncus balticus),* while the ungrazed site was dominated by panicled bullrush *(Scirpus microcarpus).* Twenty herbaceous species were recorded in the grazed area with 12 herbaceous species recorded in the ungrazed area. Dobson (1973) also found an increase in species numbers due to grazing in a riparian zone in New Zealand. He concluded the effect of grazing had been to open up the vegetation, creating more niches in which weeds could establish themselves. Hayes (1978) in central Idaho also observed that the abundance of forb species appeared to be higher in grazed areas than in pristine areas.

The impact of cattle on herbaceous productivity in riparian zones has been examined along several streamsides in the western United States. Duff (1979), Gunderson (1968), Kauffman et al. (1983a), Marcuson (1977), McLean et al. (1963), and Pond (1961) found either decreases in biomass due to herbage removal or increases in biomass due to cessation of grazing in riparian ecosystems.

Kauffman et al. (1983a) compared grazed and ungrazed responses on 10 riparian plant communities in northeastern Oregon from 1978 to 1980. Three of 10 communities displayed significant standing biomass differences. Production in ungrazed moist meadows dominated by Kentucky biomass, meadow timothy, and sedges was significantly less after 2 years of rest compared to grazed meadows but was not significantly different after 3 years of rest. Standing biomass in a Douglas hawthorn-dominated community and in a Kentucky bluegrass-dominated community was significantly greater in ungrazed stands compared to grazed stands after 3 years. Conversely, Volland (1978) could find no significant differences in biomass between a Kentucky bluegrass meadow grazed annually and one that had been rested for 11 years.

Effect of herbivory on shrub and tree production is a critical impact in riparian ecosystems, because of the importance of the woody vegetation to wildlife habitat and its dominant influence in altering the riparian microclimate. While mature vegetation approaches senescence, excessive grazing pressures have prevented the establishment of seedlings, thus producing an even-aged nonreproducing vegetative community (Carothers 1977, Glinski 1977).

The effects of excessive herbivore use on woody vegetation bordering streamsides can generally be termed as negative. Knopf and Cannon (1982) found that cattle significantly altered the size, shape, volume, and quantities of live and dead stems of willows. Cattle grazing was also found to influence the spacing of plants and the width of the riparian zone. Marcuson (1977) found shrub production to be 13 times greater in an ungrazed area than in a severely overgrazed area. Cover was 82% greater in the natural area. On a stream rested from continuous grazing for 10 years, Claire and Storch (unpublished) found alders (*Alnus* sp.) and willows (*Salix* spp.) provided 75% shade cover over areas that had been devoid of shrub canopy cover before exclosure. Similar herbivore-woody vegetation relations have been reported by Crouch (1978), Davis (1982), Duff (1979), Evenden and Kauffman (1980), Gunderson (1968), and Kauffman (1982).

Management of Riparian Ecosystems

Recognizing and understanding the impacts on the streamsides which resulted from all previous land use practices is a prerequisite to streamside planning (Claire and Storch unpublished). Because of their small extent, riparian zones in the past were considered "sacrifice areas" (Oregon-Washington Interagency Wildlife Council 1978, Skovlin et al. 1977). Riparian vegetation has been intensively used by livestock over several decades causing a reduction in the productivity of fish and wildlife habitats and degrading water quality as well as promoting increases in flow fluctuations (Oregon-Washington Interagency Council 1978).

Platts (1979) indicated that riparian ecosystems are the most critical zones for multiple-use planning and offer the most challenge for proper management; therefore, stream habitats should be identified as separate management units from the surrounding upland ecosystems. Even among riparian zones the need to identify and classify them adequately is important for proper stewardship of these systems (Claire and Storch unpublished, Platts 1978, 1979).

However, there have been few attempts to come up with a viable classification scheme of riparian vegetation that is feasible for land management activities (Cowarden 1978, Norton et al. 1981, Padgett 1982, Pase and Layser 1977, Tuhy and Jenson 1982). The major problem has been the lack of successional knowledge to formulate classification schemes based upon potential climax communities. Other problems have been the lack of continuity of terminology. For example, terms such as riparian dominance type (Padgett 1982), community type (Tuhy and Jenson 1982), and riparian type (USFS-R-4 file data) have all been used to define the basic unit of land which supports a riparian community.

Land management agencies responsible for managing livestock grazing have not adequately considered the influence of grazing on the other uses and users of riparian ecosystems (Platts 1979). Often what is good range or timber management (in short-term economic terms) is not good riparian or stream management (Platts 1979). On the other hand, it has been suggested that proper stream management practices that protect stream banks from damage also improve the potential for riparian zones to enhance fisheries, wildlife, and livestock uses (Gunderson 1968, Marcuson 1977).

Methods discussed for riparian zone rehabilitation include exclusion of livestock grazing, alternative grazing schemes, changes in the kind or class of animals, managing riparian zones as "special use pastures," in-stream structures and several basic range management practices (eg. salting, alternative water sources, fencing, range riders, etc.).

The use of instream structures as a method of riparian rehabilitation has met with some success where instream structures are combined with rest from livestock grazing (Duff unpublished, Heede 1977). Bowers et al. (1979) indicated that some instream structures (e.g., trash catchers, gabions, small rock dams, individual boulder placement, rock jetties, and silt log drops) could serve the dual purpose of increasing the water table in areas of former wet meadows as well as improving salmonid habitat.

Heede (1977), combining rest from grazing with construction of check dams, obtained vegetation cover improvements, a change from an ephemeral stream flow to a perennial flow and a stabilization of gully erosion.

After losing 23 out of 26 instream structures in a grazed area in Utah, Duff (unpublished) suggested that stream improvement structures cannot work effectively to restore pool quality and streambank stability as long as livestock grazing continued. Keller et al. (1979) in Idaho found that rest from grazing negated the need for artificial instream structures intended to enhance trout production for stream ecosystems. Kimball and Savage (in Swan 1979) found aquatic ecosystems can be restored through intensive livestock management at a lower cost than through installation of instream improvement structures.

Grazing systems have achieved some success in riparian rehabilitation and much success in riparian ecosystem maintenance. The damage caused by heavy season or yearlong grazing is well documented (Evans and Krebs 1977, Gunderson 1968, Marcuson 1977, Severson and Boldt 1978). It appears that rest-rotation grazing schemes and/or specialized grazing schemes in which riparian zones are treated as special use pastures have been the most successful.

Hayes (1978), in Idaho, stated that species composition appeared to be improved under a rest-rotation grazing system and bank sloughoff occurrences were not increased if utilization was under 60%. In other Idaho mountain grazing studies, Platts (1982) stated that when rest-rotation strategies call for livestock to utilize riparian vegetation at a rate of 65% or more, some riparian habitat alteration occurs. He also indicated that riparian alteration may be insigificant when utilization is equal to 25% or less.

Claire and Storch (unpublished) found a rest-rotation system to be favorable for achieving desired streamside management objectives if 1 year's rest out of 3 is included in the scheme.

Davis (1982), in Arizona, found that a four-pasture rest-rotation system was a cost-effective and successful method for rehabilitation of the riparian resource when each pasture received spring-summer rest for 2 years out of 3. On 2 grazing allotments, cottonwood and willows had a mean increase from 78 plants/ha to 2,616 plants/ha, 2 years after implementation of the system. A rest-rotation system also obtained a very favorable response for vegetation surrounding a livestock pond in South Dakota (Evans and Krebs 1977).

Criticism of rest-rotation systems includes reports that objectives for herbaceous vegetation were not being achieved within desired time limits (Storch 1979), and that rest-rotation systems may increase trailing and trampling damage, causing streambank erosion and instability (Meehan and Platts 1978).

Fencing and managing riparian zones separately from terrestrial upland sites as special use pastures has been shown to be an adequate multiple use system of riparian zone management (Kauffman 1982, Winegar 1977). Simulated grazing of a fenced riparian zone annually after August 1 had no measurable effect on production or species composition in riparian meadows, contrasted to decreased production and composition in a simulated season-long scheme in northcentral Wyoming (Pond 1961).

Kauffman (1982) suggested that positive characteristics of a late season grazing scheme on a riparian zone in Oregon included increased livestock production, good plant vigor and productivity, minimal soil disturbance, and minimal short-term disturbance to wildlife populations dependent on riparian ecosystems.

Another grazing system for fenced riparian zones includes winter grazing, where possible, to minimize damage (Severson and Boldt 1978). For riparian meadows dominated by Kentucky bluegrass, Volland (1978) recommended an initial year's rest, then late spring grazing alternated with late fall grazing to discourage flowering, increase tiller development, maintain plant vigor, and maximize productivity.

Changes in the kind or class of animal as well as selective culling and breeding may be another positive tool for riparian rehabilitation or maintenance. Roath (1980) found that cattle exhibited distinctive home range patterns in which certain groups of cattle preferred upland sites and groups preferred riparian sites. As forage became limiting on stream bottoms, some cattle actually decreased intake rather than move away from the riparian zone. Selective culling of these cattle and replacing them with those that prefer uplands may be beneficial for the livestock operator as well as for the riparian zone.

Platts (1982) stated that because sheep grazing on public lands is usually controlled by the use of herders, it may be possible to graze a watershed without exerting direct significant influence on riparian habitats. May and Davis (1982) suggested that sheep have been shown to exert a lesser influence on certain riparian and aquatic ecosystems and converions back to a sheep operation may be necessary to improve some riparian areas.

The most successful riparian management alternative on public lands to date has been intensive livestock management by permit holders (Storch 1979). Herding livestock on a somewhat daily basis has been successful in limiting the number of livestock that visit streambottoms and improving utilization of upland areas. Proper stewardship of riparian ecosystems is, in effect, money in the bank for the floodplain rancher (Marcuson 1977). Proper management of riparian zones means decreased streambank erosion and floodplain losses (Duff 1979, Gunderson 1968, Marcuson 1977), increased forage production (Evans and Krebs 1977, Pond 1961, Volland 1978), and an increased wildlife and fisheries resource (Buttery and Shields 1975, Duff 1979, Tubbs 1980, Van Velson 1979).

In conclusion, public grazing lands must be managed on a true multiple use basis that recognizes and evaluates the biological potential of each ecological zone in relation to the present and future needs of our society as a whole (Behnke et al. unpublished). Management strategies that recognize all resource values must be designed to maintain or restore the integrity of riparian communities (Behnke et al. unpublished).

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The Status of Our Scientific Understanding of Lodgepole Pine and Mountain Pine Beetles – A Focus on Forest Ecology and Fire Behavior

A synthesis of our current knowledge about the effects of the mountain pine beetle epidemic on lodgepole pine forests and fire behavior, with a geographic focus on Colorado and southern Wyoming.

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Introduction

Major lodgepole pine forest changes and how they affect us. Mountain pine beetle populations have reached outbreak levels in lodgepole pine forests throughout North America. The geographic focus of this report centers on the southern Rocky Mountains of Colorado and southern Wyoming. The epidemic extends much more widely, however, from the southern Rocky Mountains in Colorado in the United States to the northern Rocky Mountains in British Columbia and Alberta, Canada.

Worries about large-scale tree mortality in lodgepole pine forests have created public concerns across the West. The appearance of red trees during the last decade, a clear sign of recent beetle attack, has been followed by bare dead tree skeletons throughout this large area. Unquestionably, millions of dead trees foretell large forest changes in the near future, and more might be anticipated in areas where the mountain pine beetle has not yet reached epidemic levels.

People are concerned for many reasons. At a minimum, the loss of mature lodgepole pine trees will significantly change the present and future appearance of affected forests for half a century or more. Extensive areas of dead trees and snags are not as aesthetically appealing as live forests. Perhaps more seriously, dying and dead trees raise fears of increased fire danger. Some people worry that the dead needles and wood generated by the mountain pine beetle epidemic will lead, perhaps quickly, to severe wildfires that threaten lives, property, wildlife, and watersheds. Many are concerned that trees not yet attacked will succumb to the epidemic. Some people worry that the forest in and around our communities and recreation areas will become sparse or disappear forever, and that these forest changes will affect timber commodities, game habitat, and recreation resources.

Some contend that the current epidemic with synchronous outbreaks at many locations is unprecedented and a clear warning of global climate change impacts on ecosystems around the world. Scientists and others point to other changes occurring in our region – *Ips* beetle-caused mortality of piñon pine in the Southern Rocky Mountains, aspen decline, and large fires in Front Range ponderosa pine forests and elsewhere. It is difficult to prove cause and effect, but all of these changes began during the last 10-15 years, coinciding with recent warm climatic conditions, increasing numbers of large trees, and advancing age of many forests. Whether or not the current epidemic is unprecedented is a question to which there is currently no clear answer because of the lack of precise information on extent and severity of beetle outbreaks prior to the early 1900s. Nevertheless, many in the scientific community believe the probability of a similar event historically over at least the past few 100 years is low.

There are many insights and opinions about lodgepole pine being discussed by stakeholders of all kinds -- forest managers, agency administrators, researchers, policy-makers, politicians, the news media, industries, and the general public. Some concerns and fears are supported by scientific evidence. Others are probably justified given the current status of our scientific knowledge, but lack clear scientific support. Still others are myths with little or no basis in science. A further complication is that some of the information emerging from the science community has appeared on the surface to be somewhat contradictory.

The reason for this report. This document is written to report our current scientific understanding of the ecology and fire behavior of lodgepole pine, with a focus on the direct and indirect effects of the current mountain pine beetle epidemic that is so dominant in our minds. We recognize that important socio-economic implications stemming from the mountain pine beetle epidemic exist, and we hope that examining the status of science will aid in addressing these issues. While this document focuses on lodgepole pine and mountain pine beetles, there are also many other forest types and non-forested systems subject to extreme or at least unexpected impacts of climate, other insect and pathogen species, and other disturbances including fire and wind.

This report results from a meeting in January 2008 convened in Colorado by The Nature Conservancy, bringing together expertise of scientists who study lodgepole pine throughout its geographic range. We hope to provide as much scientific help to stakeholders as possible by sorting out what is known with a high degree of certainty, what we are confident about but with less certainty, and what is truly not understood and in need of more research. While our primary geographic focus during the workshop was Colorado and southern Wyoming, some of the findings may be appropriate for lodgepole pine throughout much of its natural range of distribution. *We urge caution, however, in applying our findings beyond our initial area of focus or to other forest types in the region.*

During the workshop and through subsequent email dialogue, the lodgepole pine team reached consensus on nine key points. As always, science is a work in progress, and uncertainties surfaced during discussion of some key points. For some points we provide what is known with adequate confidence rather than waiting for more definitive information, when this information is useful to interested stakeholders. This report provides the nine key points along with explanatory material intended to help the reader understand the degree of confidence we have from scientific study for these key points. To help the reader, we provide a list of suggested reading at the end of this report for more detailed information on many of the topics discussed. We begin with the obvious.

A. Lodgepole pine forests are being heavily impacted by the ongoing mountain pine beetle epidemic.

From British Columbia to Colorado, forests are experiencing high mortality of lodgepole pine trees from attack by mountain pine beetles. An insect epidemic with multiple outbreaks at this scale has not been observed during the last century of scientific study, though small outbreaks have occurred. This mortality is changing forest structure and composition, and modifying fuels in ways that will affect fire behavior for decades.

Many believe the mountain pine beetle epidemic, now nearly a decade in duration, might be unprecedented at least in recent centuries, stemming from a unique alignment of factors. These factors include extensive forests of trees at the right age, size, and density to support large numbers of mountain pine beetles, and a climate warm enough over the last decade to favor beetle reproduction and survival. But records are short. Modern records cover little more than a century, and for this period there is no account of a similar severe mountain pine beetle epidemic in lodgepole pine over such a large area.

For earlier periods, however, little scientific evidence exists to suggest that severe mountain pine beetle outbreaks either did or did not occur. Forest fires, another important natural disturbance, often scar living trees, which provides physical evidence indicating dates, locations, and severity of fires back through much of the last millennium. Fire-scarred wood is often resistant to rot and may persist for centuries, preserving a record of fire. But mountain pine beetle attacks that might have occurred more than a century ago leave little or no physical evidence helpful for determining dates or severity of such attacks. Wood from trees killed by beetles rots quickly, especially where wood moisture is high (e.g. fallen trees). Both stand-replacing fires and beetle epidemics that kill large numbers of trees allow stands of trees of the same age to establish in the wake of the disturbance. The ages of these trees can be used to estimate the time of the last stand-replacing disturbance, but it is often not possible to tell what kind of disturbance initiated the stand, and disturbances such as beetles, fire, and wind may act synergistically. Thus we cannot exclude the possibility that factors aligning so perfectly to result in the current epidemic could not have aligned equally in past centuries or millennia.

Regardless of whether or not the current mountain pine beetle epidemic and lodgepole pine mortality are within the historical range of variability at some time scale, the epidemic and associated tree mortality are large and are having immediate effects on forest structure and function over a vast area.

B. Not all lodgepole pine forests are the same.

Some forests are composed of nearly pure lodgepole pine established following large fires decades or centuries ago. Others are mixtures of lodgepole pine with subalpine species such as Engelmann spruce, subalpine fir, and aspen at higher elevations, or with mixed conifer species such as ponderosa pine, Douglas-fir, and aspen at lower elevations. Each type of forest has unique features of ecology and fire behavior. And lodgepole pine trees in all three types are vulnerable to attack by mountain pine beetles.

Lodgepole Pine Ecology 101. Lodgepole pine is found over a large area in western North America, from northwestern Canada in the northern Rocky Mountains; Washington, Oregon, and California in the Cascades and Sierra Nevada; Idaho, Montana, and Wyoming in the central Rockies; down to Colorado and even northern New Mexico in the southern Rockies. It comes as no surprise that across this large area and also locally, lodgepole pine trees are found in diverse forest conditions. In Colorado and southern Wyoming, pure stands of lodgepole pine occur. Even where pure stands occur, lodgepole pine forests may range from extremely dense to open and savanna-like. Elsewhere, lodgepole pine is mixed with other species. These differences in species composition of forests influence the way forests are affected by mountain pine beetles and fire, and how forests may change in the future.

Two key features of lodgepole pine are especially important in the way the species interacts with the environment and with other trees. Lodgepole trees are relatively intolerant of shade, and they are adapted to reproduce prolifically after fire. Unshaded lodgepole trees survive and grow more readily than trees overtopped either by larger lodgepole pines or by other species. Fire adaptation in trees occurs in two primary forms: the capacity to survive fire, or the ability to reproduce after fire even if killed. While species like ponderosa pine are adapted to survive fire, lodgepole pine is adapted to reproduce readily after fire.

Many lodgepole pine trees have serotinous cones that remain closed and store viable seeds in the crowns of trees for years, actually requiring the heat of a fire for seed release and dispersal. When crown fires kill trees, the resin sealing the cones melts, allowing the cones to open shortly after the fire. Huge numbers of seeds are released at once to the forest floor, falling on exposed soil that is an excellent seedbed for germination and seedling establishment. It is not uncommon to find 50,000 or more seedlings per acre several years after a stand-replacing fire. Competition then thins out trees naturally as these young forests grow to maturity. After a mountain pine beetle epidemic, lodgepole pine stands also generally regenerate, because serotinous cones on branches that have fallen near the ground heat adequately to release seeds, and seeds previously released from non-serotinous cones may exist in the forest litter. However, the role of serotinous and non-serotinous cones as seed sources, and the effect of cone serotiny on subsequent stand density, are not well understood.

The three most common natural agents influencing lodgepole pine in Colorado and southern Wyoming other than fire are mountain pine beetles, dwarf mistletoe, and wind. Of these, mountain pine beetles have the capacity like fire to change forests at large scales. Beetle populations can occasionally reach epidemic densities over large areas, though not usually as large as the current epidemic. The spatial extent of the current epidemic is probably related to large numbers of suitable host trees existing over much of the range of lodgepole pine in the West. Mountain pine beetles are a native insect that has evolved with lodgepole pine. They normally exist in endemic populations that kill a few trees but are regulated by weather. Endemic populations of beetles typically infest diseased or stressed trees. Because temperature regulates beetle development, prolonged warm periods may help trigger outbreaks. Natural enemies also help regulate endemic bark beetle populations but their role under epidemic populations is not as effective.

Dwarf mistletoe typically occurs in localized patches. While mistletoe slowly spreads, it often remains only locally significant, and trees may live for decades with mistletoe. This native parasite, which also evolved closely with lodgepole pine, is periodically reduced by fires that kill the infected trees. Major wind events may topple trees and create small to large openings. In many places lodgepole pines are shallowly rooted in rocky soils or on steep slopes. Typically even the largest blowdowns affect forests only locally, and while they contribute to the landscape mosaic of forest age and composition, they are unlikely to affect forests regionally unless they become centers of another disturbance agent (e.g. spruce beetle).

Three kinds of lodgepole pine forest. Lodgepole pine forests occur along gradients of elevation and latitude that control the length of growing season, available moisture, and frequency of natural disturbances. Fire and mountain pine beetles affect forest structure and composition differently in each ecosystem, just as environmental conditions regulate the occurrence and intensity of the disturbances. To understand this, it is useful to identify three specific types of forest in which lodgepole pine occurs. In Colorado and southern Wyoming, these are pure lodgepole pine, subalpine forest, and mixed conifer forest.

Pure lodgepole pine forests may occur where environmental conditions are poorly suited for other tree species, or where human impacts such as logging followed by burning eliminate other species. Lodgepole pines are tolerant of cold, dry conditions and poor, rocky soils. Individuals rarely live more than 400 years. Typically, pure lodgepole pine stands result after stand-replacing fires have killed all or most trees, leaving behind lodgepole seeds stored in serotinous cones as the only significant seed source. Alternatively, fire-killed stands without serotinous cones may still reproduce if lodgepole pine seeds are blown in from unburned trees nearby. Stand-replacing fires may occur in healthy, green forests under extreme weather conditions. Similar fires might occur under more moderate conditions when mountain pine beetle mortality or mistletoe infestation in stands creates additional dry fuels, though there is no firm evidence thus far confirming this. Pure lodgepole pine stands are often established within a few years after the fire and have one dominant age class or cohort for the life of the new stand, although some stands may develop continuously over longer periods of time and have multiple age classes. However, if aspen is present even in small amounts before large fires, its sprouting capability may lead to aspen patches which often give way over time to slower-growing lodgepole pine.

The spatial extent of pure lodgepole pine forests typically reflects the size of the fires that established them. As a general rule, pure lodgepole pine forests occur more commonly at upper elevations in the mixed conifer (upper montane) zone and the lower portion of the subalpine forest zone, between 9,000 and 10,000 feet elevation in Colorado and southern Wyoming. Less commonly, pure stands exist because sites are unsuitable for other tree species. Historically, past fires may have been tens to hundreds of thousands of acres in size, resulting in large lodgepole pine stands that dominate the landscape for several hundred years. However, even large intense fires do not burn uniformly, and within a fire perimeter, some patches of trees or individuals may survive intact. The 1988 Yellowstone fires are a good example of this. Alternatively, smaller crown fires may have created patches of pure lodgepole pine as small as an acre or less.

If not renewed by fire every few centuries, pure lodgepole pine stands often but not always experience ingrowth by other tree species, especially those tolerant of moderate shade. Ingrowth of other species depends strongly on site suitability for the other species, and availability of seeds. Eventually these species may replace lodgepole pine as the dominant trees in the stand. Lodgepole pine may persist in these mixed stands even if only a limited number of seedlings become established periodically, usually as a consequence of minor local disturbances such as very small fires, wind, insects, or disease.

Subalpine forests at higher elevations (usually above 10,000 feet elevation but as low as 9,000 feet) often include lodgepole pine as a component along with Engelmann spruce, subalpine fir, and aspen. Stand-replacing fires may occur in subalpine forests, but intervals between fires are usually several to many centuries (compared to one to several centuries for pure lodgepole pine forests). After stand-replacing fire, lodgepole pine seedlings grow faster than spruce or fir seedlings and may dominate stands during early developmental stages, even when spruce and fir seeds are available nearby. When aspen is present, however, creation of openings by fire or other disturbances may shift species dominance to aspen because of its sprouting ability.

Mixed conifer forests at lower elevations (usually between 7500 and 9000 feet elevation) often include lodgepole pine along with ponderosa pine, Douglas-fir, aspen, and perhaps small amounts of subalpine fir, Engelmann spruce, and limber pine. Large stand-replacing fires can occur in mixed conifer forests and may lead to pure lodgepole pine stands. More typically, however, mixed-severity fires create smaller openings providing opportunities for patches of lodgepole pine establishment and persistence within the complex landscape mosaic of mixed conifer. Once again, aspen may become temporarily dominant if it existed prior to the fire.

C. Forests are living systems subject to constant change.

It is normal and expected that many natural agents, including mountain pine beetles, fire, and wind, change forests over time. Some changes are so gradual that we barely notice them, while others are relatively sudden and extensive. The forests that are presently losing many trees to insect attack will not look the same in our lifetimes, but healthy and vigorous forests will eventually return in most locations.

We tend to think of forests as static over time because their change is slow relative to human time scales. Yet forests are non-equilibrium systems, and we should expect them to change. Our adult human experience is measured in years or decades at most, and we often fail to notice all but the more dramatic changes that occur in forests. Thus we may believe that the structure and composition of forests typically do not (and even should not) change, and, when they do, it means something alarming has happened. However, lodgepole pine and other tree species live several centuries or more and during their life cycles a number of very natural, and ecologically predictable, forest-changing events or processes often occur. The 1988 Yellowstone fires are often cited as an example of natural change in lodgepole pine ecosystems.

Taking this more comprehensive view, it is clear that combinations of fire and other natural disturbance agents, along with differences in ecological characteristics of the various tree species suited for the landscape, result in frequent changes in forest landscapes over time. The overall forest mosaic is in fact not static, but rather experiences significant shifts and adjustments, all a part of the natural ecology of forests. Thus at any location in a given landscape, the species composition, distributions of tree sizes and ages, and stand density all are subject to change, even if in our memory they do not appear to.

Understanding and predicting the consequences of natural disturbance effects on landscapes is difficult. All of the natural disturbance factors – fire, insects, pathogens, wind, drought, etc. – are capable of affecting forest landscapes at various scales and may act individually or in combination. In the current mountain pine beetle epidemic, interactions between fire and beetle effects are certain, because the insects are changing fuel characteristics of forests significantly.

D. Lodgepole pine will not disappear from the southern Rocky Mountains.

The make-up of our forests is already changing where mountain pine beetles cause high mortality of lodgepole pine. However, this event will not cause the extinction or disappearance of lodgepole pine, and forests dominated by or including lodgepole pine will persist in the southern Rockies, though they may look different from those of the past due to changing climate. Future forests will continue to provide valuable ecological services and aesthetic and recreational benefits.

When viewed from a distance, it may appear that many pure lodgepole pine forests in Colorado and southern Wyoming are being completely killed. It even appears that in some places all the lodgepole pine trees in subalpine or mixed conifer forests are being killed. Yet there is wide variability in the amount of tree mortality, and even where all the mature trees have died, understory saplings may be released and new lodgepole pine seedlings are likely to emerge. Thus it is untrue that lodgepole pine will disappear from our forests.

Scientific knowledge is not complete, however, and there is considerable uncertainty about the composition of future forests after the epidemic. Clearly, major changes in these forests are occurring, but multiple factors will affect what kind of new forest will result. A high proportion of larger lodgepole pine trees (diameters greater than six inches) are dying, and in many places many smaller trees are being killed as well. Mortality may approach 100% in pure lodgepole pine stands having few small trees.

Recovery of lodgepole pine forests following previous beetle outbreaks suggests, however, that in many places significant numbers of lodgepole seedlings and small saplings will survive. These may produce new pure stands of lodgepole pine if no other species are present, or help sustain a lodgepole component in stands of mixed species. Height growth of Engelmann spruce or subalpine fir seedlings is slow compared with that of lodgepole seedlings. Where small seedlings of spruce or fir existed beneath a pure lodgepole pine overstory, lodgepole pine may still predominate after the first decade because of their more rapid growth. However, if saplings of spruce and fir trees are left under the dead pines, they may grow quickly into the canopy and dominate the site. If aspen is present, sprouting and rapid early growth may result in an aspen forest, perhaps with the shade tolerant conifer species in the understory. However, aspen sprouting after mountain pine beetle mortality is not as well understood as it is for disturbances that more directly affect aspen trees or roots.

In pure lodgepole pine forests with few or no surviving trees, it is reasonable to expect a new lodgepole forest to regenerate on suitable sites, but difficult to predict with certainty. The existing seed bank (seeds stored in cones of dead trees and in the litter) or seeds produced by non-serotinous trees near the time of tree death may produce enough new seedlings to regenerate a new lodgepole pine forest. It is also possible that other species will colonize the sites, including other wind-dispersed trees such as spruce and fir, ponderosa pine and Douglas-fir, or bird-dispersed trees such as limber pine. Grasses, forbs, and shrubs may flourish in the new openings for periods of time, and tree establishment may be limited or slowed. Under such conditions, the landscape is likely to become more diverse than it was in the previously pure, single-aged lodgepole forests. This in itself may be beneficial for reducing the risk of a future large-scale mountain pine beetle epidemic or other monolithic disturbance.

E. Active vegetation management is unlikely to stop the spread of the current mountain pine beetle outbreak.

Mountain pine beetles are so numerous and spreading so rapidly into new areas that they will simply overwhelm any of our efforts where trees have not yet been attacked, and no management can mitigate the mortality already occurring. However, judicious vegetation management between outbreak cycles may help mitigate future bark beetle-caused tree mortality in local areas.

In the current epidemic, it is impractical to expect that silvicultural treatment of lodgepole pine forests will prevent or even impede the advance of the epidemic in Colorado and southern Wyoming. There are simply too many suitable host trees over too large an area, and unusually high insect populations. Unless climatic conditions become less favorable for beetle reproduction and spread, the most likely scenario is that the epidemic will be sustained until host trees are depleted.

Preventive spraying of high-value trees with insecticides is effective in protecting trees from bark beetle attack. Direct control measures such as removing infested trees may provide some mitigation on a small local scale but are not be effective at a landscape scale.

The current epidemic is so extensive and severe in part because large areas of lodgepole pine forest are suitable hosts for mountain pine beetles. As noted earlier, it is unclear if epidemics occurred at such a large scale historically, though smaller-scale or less severe epidemics most likely did occur and are expected in the future. Active vegetation management *between* periods when lodgepole pine forests are vulnerable to a mountain pine beetle epidemic may reduce the magnitude of future landscape-scale outbreaks, if that is chosen as a management objective. Creating diverse patch ages and sizes (including young patches) and perhaps more mixed-species forests across the landscape may or may not reduce the spread of future mountain pine beetle outbreaks, but it likely would reduce the amount of forest susceptible through time to a monolithic disturbance, including mountain pine beetle attack or fire. Thus while unproven, this increased landscape heterogeneity may be effective for limiting the scale and severity of future mountain pine beetle impacts. The effectiveness of such measures cannot be assured, nor are all the ecological consequences known, though even in the current epidemic, stands and patches of younger lodgepole pine trees appear to have survived the epidemic with no or only limited mortality.

F. Large intense fires with extreme fire behavior are characteristic of lodgepole pine forests, though they are infrequent.

Very dry and windy conditions can lead to large intense fires in lodgepole pine forests. Such fires are a natural way for lodgepole pine to be renewed and are largely responsible for extensive pure lodgepole pine forests.

Fire history studies based on fire scars and stand structure evidence extending over at least the past 500 years show that large, severe fires (often involving multiple ignitions) occurred in subalpine lodgepole pine forests of Colorado and southern Wyoming during periods of exceptionally warm and dry weather. These studies also show that long intervals (e.g. of 80 to 100 years) during which large fires were absent from study areas extending over 10,000 or more acres were common during the past five centuries in subalpine lodgepole pine forests. Climatic variation at annual and multi-decadal time-scales has been the major driver of fire occurrence in these forests and is the key explanation for the non-equilibrium behavior of these ecosystems. Large fires shaped the amounts and locations of extensive lodgepole pine forests on the landscape and this process is relatively well understood, but additional research would be helpful to characterize stand history in local areas, especially in relation to past climate.

Fire is complex, and its behavior varies with variations in weather, ignitions, fuel amounts and arrangement, and fuel moisture. Historically, most ignitions in lodgepole pine forests were caused by lightning. The role of Native American ignitions is unknown, but given that extensive fire occurs in these forests only under dry and windy conditions, their contribution was probably small. Young and mature stands of pure lodgepole pine are relatively unlikely to burn except under the most extreme weather conditions. Unless residual fuels remain from the effects of previous fire or insect epidemic, fuels commonly are sparse in the understory, and closed canopies help keep the forest floor cool and somewhat moist. The snow-free period above 9000 feet elevation is relatively short, leaving little time for fuels to dry. The term “asbestos forest” has been applied to these forests, attesting to their low probability of an intense crown fire except under extreme weather conditions.

As lodgepole pine forests mature they become increasingly vulnerable to natural disturbances such as mountain pine beetles and wind. Even with only partial overstory mortality, openings created in the forest canopy allow more air circulation beneath the canopy, and drying of surface fuels. In addition, fuel amounts may be increased by the localized tree mortality, including fuel ladders provided by fallen trees, young understory trees, and shrubs that may help fire reach the overstory. Such changes may increase the probability of fuel ignition from lightning and may alter fire behavior in several ways. Fire behavior in maturing stands is not fully understood, however, and more research would be beneficial.

These remarks about fire behavior apply especially to pure lodgepole pine forests. In subalpine mixed forests, the likelihood of dry fuels is even less as the snow-free period is shorter. In mixed conifer forests below 9000 feet, the complexity of the landscape, greater productivity and longer and more frequent fire season encourages mixed-severity fires which have both surface and stand-replacing components. Even in the mixed conifer forests, however, fire extent is highly variable due to climatic variation, and fire-scar studies show that years of widespread fires during past centuries were dependent on exceptional drought. Typically, lodgepole pine occurrence can be suppressed with shortened fire intervals because its long-term presence depends on seed germination after fire and trees growing to reproductive maturity before the next fire. In general, fire history and potential fire behavior are less well understood in mixed conifer forests than in pure lodgepole pine forests.

G. In forests killed by mountain pine beetles, future fires could be more likely than fires before the outbreak.

Large intense fires with extreme fire behavior are again possible.

There is considerable uncertainty about fire behavior following a mountain pine beetle epidemic on this scale. In pure lodgepole pine forests, crown fires are possible both before an epidemic and after while needles are still on trees. Intense surface fires are possible after most dead trees have fallen to the ground. The probabilities of such fires are uncertain, and more research is needed to learn in what ways and how long the fuels and fire environment are altered by the beetles. Nevertheless, protection of communities and other values at risk continues to be imperative.

More research is required to fully understand fire behavior over time following a mountain pine beetle attack. Nonetheless, the extensive epidemic now occurring is precipitating enormous changes in fuel structure over large areas in Colorado and southern Wyoming, through changes in the condition and arrangement of the forest biomass (which is fuel for forest fires). The mature lodgepole pine trees that provided abundant but moist living fuels are now dead, dry, and falling, and have the potential to contribute to extreme fire behavior in post-beetle forests similar to historical fires in lodgepole pine forests. However, the realization of that potentially extreme fire behavior will depend on a number of contingencies, particularly future climatic conditions.

Empirical data are very limited. One study of fire extent and severity of wildfires that burned in subalpine forests in Colorado in the extreme drought of 2002 did not find that fire extent or severity were greater in stands recently killed by mountain pine beetle. The authors cautioned, however, that the conclusions regarding the influence of the recent beetle outbreak on fire extent and severity are limited by spatial and temporal limitations associated with aerial detection of the outbreak. More importantly, any broader applications of this case study would need to be tested by additional studies considering different initial forest (fuel) conditions and especially weather conditions that drive fire behavior. Even though only limited scientific information is available to predict likely fire behavior during and in the decades following a mountain pine beetle epidemic and under varying climate conditions, we believe that both field observations of fire behavior and modeling provide some insights into what could be expected. We offer these insights as preliminary guidance for those concerned with management of beetle-killed forests, even as new research is being conducted to clarify our scientific understanding.

Pure lodgepole pine. In the initial phases of the epidemic when trees are being killed, needles die, turn red and dry out but persist on trees for two or three years. During this phase, needles and small branches provide dry fine fuel that could burn in a crown fire. The amount of fuel is relatively unchanged compared with the pre-epidemic forest. However, fuel moisture is lower, and some think it likely that a crown fire could ignite and spread under somewhat less extreme fire weather conditions than were required for initiating a crown fire in an equivalent forest of live trees.

The fuel structure of dead lodgepole pine stands changes significantly when needles fall to the ground. During this phase, little fine fuel remains in the forest canopy to support an active crown fire that spreads from tree to tree. Furthermore, the fallen needles lie close to the ground surface and, in the absence of other fuels near the ground, provide a relatively poor fuel bed for generating significant flame heights. Increased growth of grasses, low shrubs and forbs may create a moist fuel bed during the growing season but provide dry fine fuels near the end of the growing season. However, large amounts of biomass in the boles and branches of standing trees remain well above typical flame heights, and without needles these canopy fuels are relatively unlikely to burn. Thus surface fires in years following needle fall may not be intense and crown fires may be nearly impossible (assuming the forest is relatively pure lodgepole pine and most or all large trees are dead). In some areas, rapid development of a tall shrub community (which may precede tree regeneration) may provide shade and protection from drying of fuels on or near the ground. However, this is unlikely in most lodgepole pine forests in Colorado and southern Wyoming (the focus area of this report), because few tall shrub species occur in these relatively dry forests. Instead, low shrubs such as huckleberry and buffaloberry are more common.

Trees killed by mountain pine beetle may remain standing for a number of years, but as they progressively decay and fall to the ground (often aided by wind), the fuel structure changes once again. In this phase (typically 10-20 years or more after death), a large amount of biomass becomes available as fuel within flame heights that can be generated by the fine surface fuels. Some of the biomass is elevated above the ground where it dries out more easily and becomes available to support intense fire with a large release of heat. Such a fire is relatively hard to control and nearby structures may be hard to protect. Furthermore, fire intensities under these conditions could cause high mortality of young trees that survived or regenerated after the mountain pine beetle attack. If widespread fire mortality occurs before trees have matured to cone production age, rapid re-establishment of lodgepole pine on this site is less likely.

At the scale of a stand, none of the changes in fire behavior that we have described would be outside the historical range of variability for this ecosystem. Even in stands with tremendous wood accumulation on the ground, fire behavior may differ little from historical fires within blow-downs or areas recently burned by stand-replacing fires. However, we are uncertain about fire behavior at landscape or regional scales because we have not seen systems with such heavy fuel loads over such extensive areas; and we know little about the ecological consequences of such fires at these scales.

Lodgepole pine with other species. Similar transitions in fuel structure also will occur in the lodgepole pine-dominated component of subalpine and mixed conifer stands. But the mixture of dead lodgepole pine with live trees of other species creates a more complex fuel structure. An important effect of lodgepole pine mortality in mixed stands is a change in the environmental conditions and thus the fuel moisture near the forest floor. Prior to beetle mortality of the overstory, solar radiation is largely intercepted by the forest canopy, and air movement beneath the forest canopy is moderated by the overstory. The understory beneath the canopy remains relatively cool and moist.

When lodgepole pine trees die and needles fall from dead trees, radiation reaching the forest floor and air movement beneath the residual live tree canopy are increased, and both contribute to fuel drying. More open canopies also contribute to greater understory vegetation growth. The consequences of these changes on fire behavior are not fully understood, but such conditions may favor ignition and spread of fire more readily than in forests having few canopy gaps or fuels created by mountain pine beetle mortality, particularly later in the growing season when fuels near the ground become drier. Because several associated species, firs and spruces, typically have low crown bases due to poor self-pruning, higher surface fire intensity from added lodgepole pine fine fuels coupled with drier, warmer, windier surface conditions, could lead to an increase in potential for passive crown fire (torching). Furthermore, increased human activity in today's forests has increased fire ignitions compared with the historical period.

H. Mountain pine beetle outbreaks are not likely to cause increased erosion.

Soils are not disturbed and protective ground cover is not reduced when mountain pine beetles kill lodgepole pine trees. If anything, understory plants may grow more vigorously in the increased light and with the higher available soil moisture and nutrients. Where tree mortality is high, annual streamflow may increase and the timing of water delivery may be changed, because of reduced canopy interception of precipitation and reduced water uptake by the trees.

Interactions between forest structure and hydrology have been studied extensively, and there is little question that major changes in the structure of Colorado and southern Wyoming forests alter several key hydrologic characteristics of these forests. Forests are widely viewed as important for protecting sloping terrain in watersheds from extreme runoff and erosion. Wildfire severe enough to kill forests is viewed as a major threat to watersheds, because protective vegetation, litter, and duff are often consumed. In many cases, fire exposes soils directly to precipitation, and runoff during heavy precipitation events (often exacerbated when fire makes soils temporarily hydrophobic) can result in extreme erosion for several months following a fire.

Soil type, steepness of slope, precipitation intensity and duration, and timing of understory vegetation recovery all affect the severity of erosion after fire.

Death of forest trees during a mountain pine beetle epidemic affects the forest floor and soil much differently than fire. Tree mortality caused by beetles leaves behind protective layers of litter and duff, and often quickly results in more productive understory vegetation. Thus severe erosion events are not expected as a result of the mountain pine beetle epidemic. In fact, mulching and seeding after fire often are used in attempts to mimic the stabilizing effects of litter, duff, and understory vegetation found after overstory mortality by beetles.

The potential for erosion from wildfire still exists, however, if extensive fire occurs in the decades following the epidemic, when large amounts of fuel are on the ground. Thus while the mortality of trees does not increase erosion significantly, erosion remains a possibility if a post-beetle fire occurs with heavy fuel loading on the ground. We note, however, that erosion is a natural process, and concerns about extreme erosion may be more a human issue than an ecological one.

There may be other hydrologic effects of mountain pine beetle mortality. Paired watershed studies around the world support the conclusion that substantially decreasing forest density results in increased runoff, though many factors affect the degree to which this occurs. Subalpine forest studies in Colorado and elsewhere are among the best examples supporting these findings. While no empirical studies of runoff in relation to the current mountain pine beetle epidemic have been completed in Colorado and southern Wyoming, it is reasonable to expect that the total annual runoff will increase where pure stands of lodgepole pine are killed by mountain pine beetles. More research is needed to determine how the hydrologic features of watersheds change during and after such epidemic changes in forest structure.

I. Climate changes will most likely contribute to substantial forest changes in the decades ahead.

Given the climate changes in the last several decades and projected changes for coming decades, large fires and other natural disturbances and shifts in vegetation composition and distribution are anticipated in many ecosystems of Colorado and southern Wyoming. These large disturbances and other changes in growing conditions will likely contribute to restructuring many forest landscapes.

Many uncertainties about climate and vegetation exist for the years, decades, and centuries ahead. As noted in our introduction, we have seen a number of significant ecological events in the last decade, including the mountain pine beetle epidemic in lodgepole pine. All of them coincide with warmer climatic conditions than were typical for the past century or more for which we have records. Warming temperatures (especially winter minimum temperatures), longer growing seasons, and growing season drought may be playing major roles in the widespread bark beetle outbreaks in Colorado and southern Wyoming and elsewhere. Fuel quantity and arrangement and fire behavior may be influenced directly or indirectly by the same variables. Germination and establishment success of new seedlings may be affected. However, it is difficult to prove whether climate changes, consequences of past forest management practices on forest conditions, or both are the primary causes of these ecological changes.

Implications for future forests. Models for predicting future climates have progressed dramatically in recent years, but their accuracy is questionable for planning purposes, particularly at local levels. Nonetheless, model predictions suggest significant alterations in climate from past observed patterns. These predictions are supported by recent climate events that themselves had largely been predicted several years ago. Therefore, the potential for future changes justifies thinking about future ecosystem dynamics that are very different from what we have seen in the last few centuries, including vegetation responses involving natural disturbance agents, species distribution, habitat suitability, and conservation of biodiversity. Areas at the elevational and latitudinal edges (ecotones) of lodgepole pine distribution may be the most likely to experience notable changes following the beetle epidemic.

Our understanding of natural disturbance phenomena such as fire, drought, and insect epidemics under new climatic scenarios is inadequate for us to judge the likely consequences of future climatic conditions. We all observe and acknowledge that natural disturbances can be major change agents regardless of their cause. Climate warming may be contributing to substantial forest changes now, but there may be more subtle changes in the future as well. Through time forest species (including insect associates and other animal species, shrubs, grasses, and forbs as well as trees) may shift to other elevations and latitudes where habitats have become more suitable for them. Some species with rapid generation times, such as mountain pine beetle, may adapt to the changing climate. Alternatively, without adaptation local extinction in bark beetle populations could occur with increased warming due to a disruption of their tightly coupled developmental timing with local weather. Groups of species may migrate together or separately, perhaps leading to unanticipated new forest communities. We cannot make firm predictions about the makeup of future forests or the biodiversity associated with these forests. Regeneration and plant community restructuring in the landscape may follow novel pathways. Information is lacking, however, and extensive research (including use of monitoring data and reconstructions of past changes) is needed to relate potential future climate and the requirements and environmental amplitudes of species and communities.

Is re-establishment of lodgepole pine assured after the mountain pine beetle epidemic? Undoubtedly, but subtle or even large shifts in its location and plant associations are not out of the question. Nonetheless, most of the area experiencing a mountain pine beetle epidemic will likely remain forest. Even if future forests differ from those of today, such forests are likely to provide valuable (if different) benefits and opportunities, both ecologically and socially. Monitoring of changes in forests as they occur is important for enabling research on such changes, and to allow managers to adapt practices to achieve desired effects as conditions change and consequences of past actions are better understood.

J. Summary

The current mountain pine beetle epidemic affecting lodgepole pine forests is an important ecological event with significant socio-economic implications. What will be the consequences for the affected ecosystems? How do we protect our communities and other human values at risk in ways that are socially and economically (as well as ecologically) feasible? These are difficult questions. This report has focused specifically on the ecology and fire behavior issues associated with lodgepole pine and the mountain pine beetle epidemic. We recognize that the socio-economic aspects are as important as the ecological issues, but they are beyond the scope of this report.

Ecologically, much is known about lodgepole pine and mountain pine beetles. Even though the scale of the current epidemic is unprecedented over the past approximately 100 years of reliable observations, beetle-caused tree mortality at some scale has long been part of the dynamics of the lodgepole pine ecosystems. Similarly, fire behavior and its role in ecological processes and fuel management practices are relatively well understood. While we are confident about our general understanding, we have identified at least some scientific uncertainties about lodgepole pine, mountain pine beetle effects, and fire behavior that should be acknowledged and further researched.

We are most concerned about several wildcard issues that create some uncertainty in applying what we know from science. The scale of this epidemic is larger than any mountain pine beetle epidemic studied thus far. We do not fully understand if or how the magnitude of this ecological event will affect future forests in terms of regeneration of the present species or transitions to different vegetation types. Furthermore, there is the question – both tantalizing and troubling – about possible climate change (including its rate, direction and magnitude) and the degree to which scientific findings need to be qualified as they are applied.

If humans were not a part of the equation, forests would simply mature, die, and regenerate or be replaced by other vegetation types, following ecological trajectories over time driven by climate, environment, and species capabilities.

Because humans cause changes in forests by choosing to live there and deriving economic services from them, our communities are impacted by forest changes, whether they are natural or not. Thus both the scale of the mountain pine beetle epidemic and the uncertainties about future forests leave us with questions that are important to us but may not be answerable with the knowledge we have now.

Knowledge from scientific research about lodgepole pine and mountain pine beetles is valuable in two ways. It offers answers to some of the questions we have about forest ecology and provides valuable insight for management of these forests for ecological and community protection purposes. It also clarifies what we do not know. This is valuable not just to direct new research, but also to inform stakeholders of the degree of confidence they should have as land and natural resource management practices are considered.

As noted in the introduction, science is a work in progress. Many of the scientific uncertainties discussed in this report already are receiving attention in the research community. Even as research continues, however, the scientific knowledge already available is usable by a wide variety of stakeholders and in the collaborative and adaptive management process. Adaptive management is perhaps best described as managing while learning on the fly. In this report, the scientific community provides information to managers and other stakeholders, but the scientific community also will help advance the knowledge base through lessons learned as management practices are planned, implemented, monitored, and evaluated. We humans must decide how to manage forests based upon their intrinsic value and natural processes as well as some desired future condition contingent on human wants and needs. We must be realistic about the degree to which we as observers, managers and stewards of the forest can affect what is happening now and what will happen in the future. Whatever we do from here should be done together.

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Chapter 3

c0015
Using Bird Ecology to Learn About the Benefits of Severe Fire

Richard L. Hutto, Monica Bond and Dominick DellaSala Au1

s0010
3.1 INTRODUCTION

p0010 In this chapter we do not provide an encyclopedic review of the more than 450 published papers that describe some kind of effect of fire on birds. In other words, we are not systematically proceeding through a litany of fire effects on birds of southeast pine forests, California chaparral, Australian eucalypt forests, South African fynbos, and so forth. Instead, we have chosen to highlight underappreciated principles or lessons that emerge from selected studies of birds in ecosystems born of, and maintained by, mixed- to high-severity fire. Those lessons show how important and misunderstood basic fire ecology is when it comes to managing fire-dependent forest lands and shrublands, and the lessons apply to all fire-dependent ecosystems that have historically experienced severe fire—fires that are severe enough to stimulate an ecological succession of plant communities (as described in Chapter 1). We also focus our attention primarily on conifer forest ecosystems of the western United States because they undergo an amazing transformation following severe fire and because studies of these systems clearly reveal how birds evolved with, and now require, severe fire. Insight that emerges from the study of bird populations is overlooked in management circles worldwide. This is unfortunate because the insight one can gain by studying the ecology of individual bird species argues strongly that severe fire needs to be maintained in the landscape if we hope to maintain the integrity of most fire-dependent ecological systems.

p0015 Most studies of fire effects on birds are disappointingly "empty" because they are merely lists of birds that benefit from or are hurt by fire; they are not placed in the broader context of what a self-sustaining fire-dependent system looks like. To understand whether a particular change in abundance is "good" or "bad" requires insight into what ought to be, which requires an understanding of the patterns that occur under conditions that are as natural as possible for any given vegetation system. That, in turn, requires replicated study of what we can expect to find after "natural" fire in any given system.

The Ecological Importance of Mixed-Severity Fire: Nature's Phoenix.

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Thus, a study of the effects of, say, prescribed understory fire on birds is meaningless without knowing what a "natural" fire in that system would ordinarily produce. Many studies might show that bird species A increases after a prescribed fire, but is that a good thing? If bird species B increases after postfire salvage logging, is that a good thing? If bird diversity is higher in one fire treatment versus another, is that a good thing? For studies of fire effects to be useful, we need to address questions that inform management by tapping into a solid understanding of what constitutes a "natural" response to fire, and that requires knowing something about the fire regime under which a given system evolved. Only through distribution patterns and adaptations of individual species (not through effects on bird guilds or on diversity and similar composite metrics) can we begin to understand which kind of fire regime necessarily gave rise to specific patterns of habitat use and to adaptations that have evolved over millennia. Birds are excellent messengers; they carry all the information we need to reconstruct the historical conditions under which they evolved. All we have to do is listen.

s0015 3.2 INSIGHTS FROM BIRD STUDIES

s0020 Lesson 1: The Effects of Fire Are Context Dependent; Species Respond Differently to Different Fire Severities and Other Postfire Vegetation Conditions

p0020 One extremely important lesson that has emerged from studies of the fire effects on birds is that a given effect depends entirely on the vegetation type, the kind of fire, and the time since the fire (Recher and Christensen, 1981; Woinarski and Recher, 1997). For years, individual bird species have been labeled as "positive responders" or "negative responders" or "mixed responders" when, in fact, any species can be all of the above. The actual response of a bird species (or of any species) to fire, then, is dependent on context. The earliest papers on fire effects rarely provided details about the nature of the fire being studied, so the first attempt to conduct a meta-analysis based on a compilation of published results of fire effects (Kotliar et al., 2002) necessarily generated a lot of "mixed" responses by birds because some papers said a species was positively affected and others said the same species was negatively affected by fire. The seeming disagreement among studies was, in most cases, a simple result of researchers looking at different postfire vegetation conditions and times since fire. It was not until Smucker et al. (2005) separated their data into categories of fire severity and time since the fire that responses began to look much more consistent among studies that share a particular vegetation type, fire type, and time since the fire. As soon as one accounts for these factors, it becomes clear that the responses of most bird species are quite consistent and that most bird species benefit from severe fire (as we discuss below).

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s0025 *Time Since Fire*

p0025 Species that benefit from severe fire are not only those that flourish during the first year or two following the disturbance event. The same can be said for species that are restricted to years 2-4, years 5-10, or even years 50-100 following severe fire. In fact, *most* plant and animal species are present only during a limited time period following a disturbance. Therefore, *most* plant and animal species in disturbance-based systems depend on disturbance to periodically create the conditions they need. Many bird species that thrive after fire have been mislabeled as species hurt by fire because studies of bird response to fire typically involve only a brief period of time soon after the fire. For example, although Williamson's sapsucker (*Sphyrapicus thyroideus*) was labeled a "mixed responder" and brown creeper (*Certhia americana*) a "negative responder" in the meta-analysis by Kotliar et al. (2002), and the change in house wren (*Troglodytes aedon*) abundance was labeled "insignificant" in a recently published study by Seavy and Alexander (2014), each of these species typically reaches its peak abundance several years after a fire, as revealed in an 11-year postfire study conducted after the Black Mountain fire, which burned near Missoula, Montana, in 2003 (Figure 3.1). Thus, each species clearly benefits from severe fire when viewed in the proper (and perhaps very restricted) time frame after fire.

p0030 By extending the duration of a postfire study beyond the first few years after a fire, most bird species reveal a unimodal response to time since fire, and most benefit from fire; they reveal a greater probability of detection in the burned forest at some point during that postfire period than in the same forest before fire or in the surrounding unburned forest (Taylor and Barmore, 1980; Reilly, 1991a, 2000; Taylor et al., 1997; Hannon and Drapeau, 2005; Saab et al., 2007; Chalmandrier et al., 2013; Hutto, 2015). These results force one to appreciate that if for a period of time after a fire conditions remain better than they are in very old plant communities near the end of the late seral stage of succession, then disturbance is periodically necessary to create the conditions needed by that species. Thus a species being "hurt" in the short term by fire is not evidence that fire is somehow "bad" for that species and that it would have been better off without fire. In fact, once a system is beyond the ideal postdisturbance time period for a species, the only way to periodically "restore" conditions needed by that species is to disturb the system with another severe fire and then wait for the appropriate time period following disturbance again. The lesson is this: one cannot assess the effects of fire on any plant or animal species without examining whether the species is restricted to a period of time preceding the oldest possible vegetation condition.

p0035 A necessary consequence of different species occurring at different points in time following fire (in association with changes in vegetation type and structure) is that we must embrace natural severe disturbance processes because they create starting points for the development of the full range of vegetation-age

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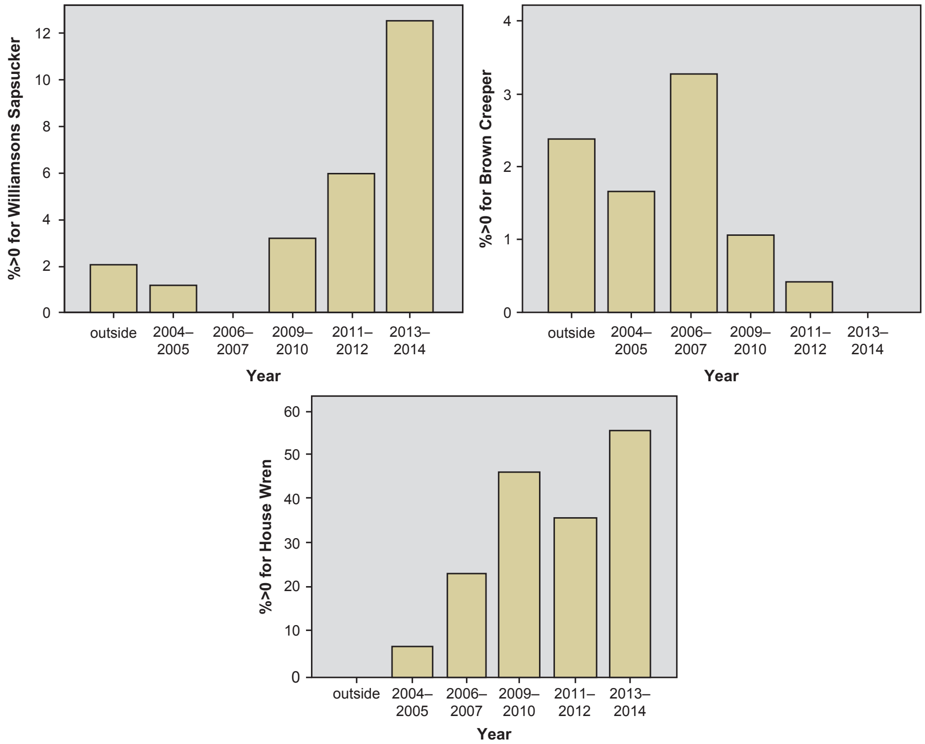

f0010 **FIGURE 3.1** The probabilities of occurrence of Williamson's sapsucker (~~*Sphyrapicus thyroideus*~~), brown creeper (~~*Certhia americana*~~), and house wren (~~*Troglodytes aedon*~~) were significantly greater several years after the 2003 Black Mountain fire than they were either before the fire (as determined from survey data "outside" the burn perimeter in unburned, mixed-conifer forest of the same type) or during the first 2 years following the fire (R.L. Hutto, unpublished data; sample sizes exceed 150 point counts for each time period; $P < 0.05$, log linear analyses). Therefore, the benefit of severe fire for some species cannot be detected without restricting data collection to within a specific time period after the fire event.

categories, which, in turn, are needed for the maintenance of biological diversity (in particular beta diversity, the turnover in species number across gradients). Moreover, mixed-severity fires (which can result only from high-severity fire events) help provide a variety of kinds of starting points, which, in turn, also help maintain biological diversity (Smucker et al., 2005; Haney et al., 2008; Rush et al., 2012; Sitters et al., 2014; see also Chapters 4-6).

s0030 Old Growth

p0040 As already emphasized, most bird species clearly depend on severe fire to reset the clock, which stimulates development of the particular postdisturbance "age" to which they are best adapted. Still, many bird species are restricted in their habitat distribution to an end-of-the-line successional stage—they are dependent on old growth. There are also ecosystems (e.g., eucalyptus forests, chaparral) where severe fire is natural but where there are few, if any, early

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f0015 **FIGURE 3.2** Resprouting eucalyptus trees following a severe fire that burned through the area only months earlier. *(Photograph by Richard Hutto, taken in November 1999 near −34.284030°S, 150.725373°E in the tablelands above Wollongong, New South Wales, Australia.)*

fire-dependent bird species because many of the dominant plant species resprout, yielding a plant community structure and composition that "recovers" rapidly after fire (Figure 3.2). In these instances most bird species are associated with "mature" forms of those plant communities and would appear to do well if there were no fire at all (e.g., Taylor et al., 2012).

p0045 In all vegetation types that undergo plant succession following mixed- to high-severity fire, there will always be some bird species that depend on long-unburned vegetation. Therefore, discovering that those species are absent in the short term or "hurt" by fire is not unexpected, nor is it a necessarily a problem that needs to be addressed. The fact that fire temporarily removes large parts of a landscape from the pool of suitable conditions for those species is not a problem because the loss of suitable conditions is temporary, and there are usually nearby "refuges" of suitable conditions in places that have not burned for a long time (Bain et al., 2008; Leonard et al., 2014; Robinson et al., 2014; Winchell and Doherty, 2014). Natural systems exist as an ever-changing mosaic of different postfire ages—all vegetation ages are present at some point in space all the time. A significant problem emerges only when humans remove or degrade so much of the older vegetation through timber harvesting or land conversion that there is now a perceived risk of fire to those species that depend on older vegetation stands that are too few and far between. Understand clearly, however, that the absence of late-succession forest refuges is a problem that stems from excessive logging or development, not from the presence of fire per se.

p0050 Now that we are down to the last remaining old-growth forest remnants in California and Oregon l~~andscapes that have been subjected to excessive logging~~, some feel that we should thin the forests around those remnants to protect them from fire. The effect of altering mature forest surrounding the last remaining old-growth remnants on the remnants themselves is, however, unknown.

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Moreover, as has been discussed in reference to eucalyptus forest systems, many old-growth forest patches are old precisely because they are situated in places that are relatively immune to severe fire (Bowman, 2000); the same is undoubtedly true of many old-growth mixed-conifer forest patches. Unburned forest patches surrounding unburned, old-growth forest patches also have been suggested to be important as dispersal corridors across which old-growth species may recolonize recently burned areas as succession proceeds toward later stages (Pyke et al., 1995; Robinson et al., 2014; Seidl et al., 2014). Therefore, proposals to thin the forest around remaining old-growth stands may be well intentioned but reflect a lack of appreciation for the resilience associated with plant communities born of, and maintained by, natural disturbance processes (a case in point is the spotted owl [*Strix occidentalis*]; see Box 3.1).

s0035 *Postfire Vegetation Conditions*

p0070 One must account not only for time since fire but also for fire severity and other forest conditions (e.g., vegetation composition and tree density) to adequately assess fire effects on animal species. Smucker et al. (2005) accounted for both time since fire and fire severity in an analysis of bird occurrence patterns following the Bitterroot fires of 2000 in Montana, and the results were profound.

b0010 BOX 3.1 Old-Growth Species and Severe Disturbance Events

p0055 There are a number of old-growth-dependent species in North American conifer forests, but severe fire may not pose anywhere near the threat to those species that one might suppose. Consider the spotted owl ~~(*Strix occidentalis*)~~, one of the most iconic old-growth-dependent bird species in the Pacific Northwest, California, and Southwest (extending into northern Mexico). This federally listed threatened raptor typically nests, roosts, and forages in dense conifer and mixed-conifer-oak forests dominated by large (>50-cm diameter at breast height), older trees and peppered with big decadent snags and fallen logs. High levels of canopy cover (generally >60%) from overhead foliage is an important component of nesting and roosting stands; thus, spotted owls were long presumed to be seriously harmed where severe fire burned the forest canopy. Indeed, over the past several decades, most forest management efforts in the range of the spotted owl (a Forest Service management indicator species) has been driven by logging to prevent or reduce fire to "save" the owl, including the latest U.S. Fish & Wildlife Service recovery plans for the northern and Mexican spotted owls. Yet, the forests where the owl dwells have experienced mixed- and high-severity fire for millennia. So how do these birds actually respond when severe fire affects habitat within their home ranges?

p0060 Several studies have demonstrated that all three subspecies of spotted owl can survive and thrive (i.e., successfully reproduce) within territories that have experienced moderate- and high-severity fire (Bond et al., 2002; Jenness et al., 2004;

Continued

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b0010 **BOX 3.1 Old-Growth Species and Severe Disturbance Events—Cont'd**

Roberts et al., 2011; Lee et al., 2012, 2013). Exceptionally high levels of severe fire in a nest stand can cause spotted owls to abandon that territory (Lee et al., 2013), but only a small fraction of sites ever exceed that threshold in any given fire. Moreover, a higher probability of abandonment after fire was documented only in a geographical region where prefire forest patches were limited or isolated; this did not occur in areas where prefire forest cover was more abundant (Lee et al., 2012, 2013) or in areas that were ~~salvage~~ logged after fire (Lee et al., 2013; Clark et al., 2013). For example, the year after the 2013 Rim Fire—one of the largest fires to occur in California within the past century—at least six pairs of California spotted owls were detected in sites where >70% of the "suitable habitat" around their nest stands burned at high severity. (At one occupied site severe fire burned 96% of the habitat!) Why do they stick around in burned territory? One study found California spotted owls selectively hunted (mostly for woodrats and gophers) in stands recently burned by severe fire when those burned forests were available to them and relatively near the nest or roost stand (Bond et al., 2009, 2013). Another study showed that during winter, Mexican spotted owls moved up to 14 km into burned forests where prey biomass was 2-6 times greater than in their breeding-season nesting areas (Ganey et al., 2014). Spotted owls are perch-and-pounce predators, so it is not surprising that they avoided foraging in areas that were logged after fire, as there were no longer any perch trees (Bond et al., 2009), nor is it surprising that postfire logging reduced site occupancy and survival rates (Clark et al., 2013; Lee et al., 2013). In these studies, spotted owls still preferred to nest and roost in green forests, underscoring the importance of unburned/low-severity refuges within the larger landscape mosaic of mixed-severity fire. Still, the point is that where severe fire is natural, even old-growth species can partake of its bounty. The spotted owl, too, is sending a message here: A natural fire regime provides a bedroom, nursery, and kitchen for even old-growth-dependent species, as long as the burned forest is left standing.

p0065 Despite this evidence, the U.S. Fish & Wildlife Service is now calling for aggressive, large-scale thinning in northern spotted owl habitat in dry forests as a means of reducing fire intensity (U.S. Fish and Wildlife Service, 2011). This "recovery" objective for the owl was developed over objections raised by scientists (Hanson et al., 2009, 2010) and professional societies such as The Wildlife Society and Society for Conservation Biology. Notably, Odion et al. (2014b) simulated changes in owl habitat over a four-decade period following fire and the kind of thinning proposed by federal land managers. The simulation study showed that thinning over large landscapes would remove 3.4-6.0 times more of their dense, late-successional habitat in the Klamath and dry Cascades, respectively, than forest fires would, even given a future increase in the amount of high-severity fire. Further, Baker (2015) documented that before extensive Euro-American settlement, mixed- and high-severity fires shaped dry forests in the Eastern Cascades of Oregon and provided important habitat for northern spotted owls there. These studies challenge the paradigm that severe fire is a serious threat to spotted owls, which evolved in landscapes shaped by such fire, and that extensive logging is needed to ameliorate this widely believed but overstated threat.

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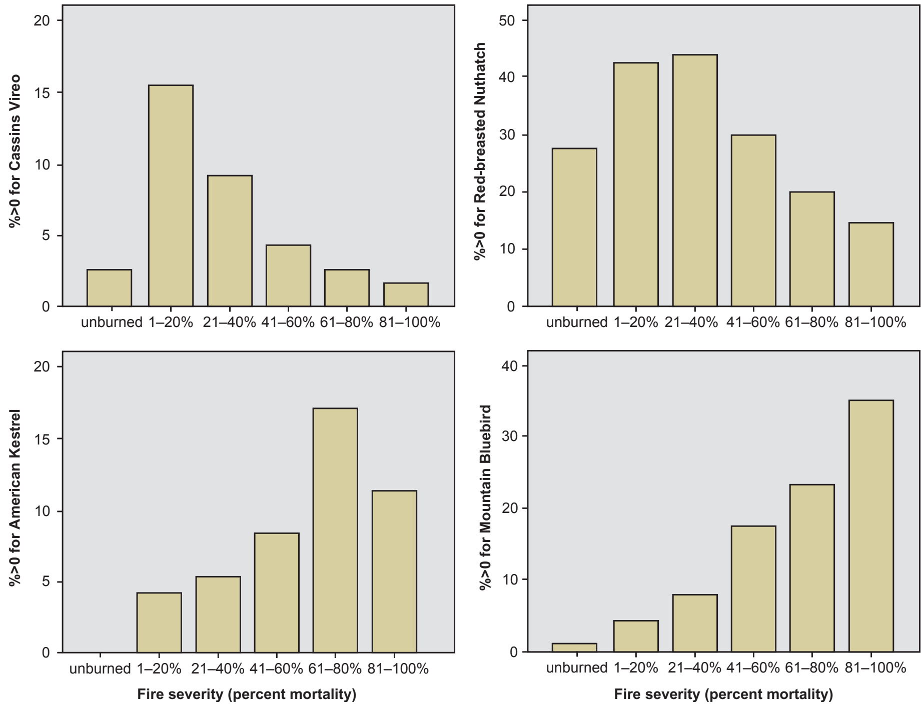

f0020 **FIGURE 3.3** Example plots of the percentage occurrence of four mixed-conifer bird species in relation to fire severity in the first few years after fire. Data were drawn from 7043 survey points distributed across 110 different fires that burned since 1988 in western Montana. Sample sizes exceed 700 point counts per severity category. All patterns are significant ($P < 0.05$, log linear analyses). Note that each species [illegible] abundant in burned than in unburned forest, and each is relatively abundant at a level o[illegible]everity (percentage of tree mortality) that differs from that occupied by the other species.

Once they accounted for fire severity alone, it became abundantly clear that many of the same bird species that had been labeled as “mixed responders” to fire by others (e.g., Kotliar et al., 2002) were not at all mixed in their response to fire. The importance of fire severity is strikingly apparent in even the simplest graphs of percentage occurrence across severity categories (Figure 3.3).

s0040 Lesson 2: Given the Appropriate Temporal and Vegetation Conditions, Most Bird Species Apparently Benefit from Severe Fire

p0075 After we combine information on the time since fire, fire severity, and perhaps one or two additional vegetation variables, most bird species apparently benefit from severe fire. For each species there is a particular combination of burned forest variables that creates ideal conditions for that species, as evidenced by an abundance that exceeds that in a long-unburned patch of the same vegetation type. Indeed, when Hutto and Patterson (2015) considered just two fire-context variables (time since fire and fire severity), they found 46 of 50 species to be

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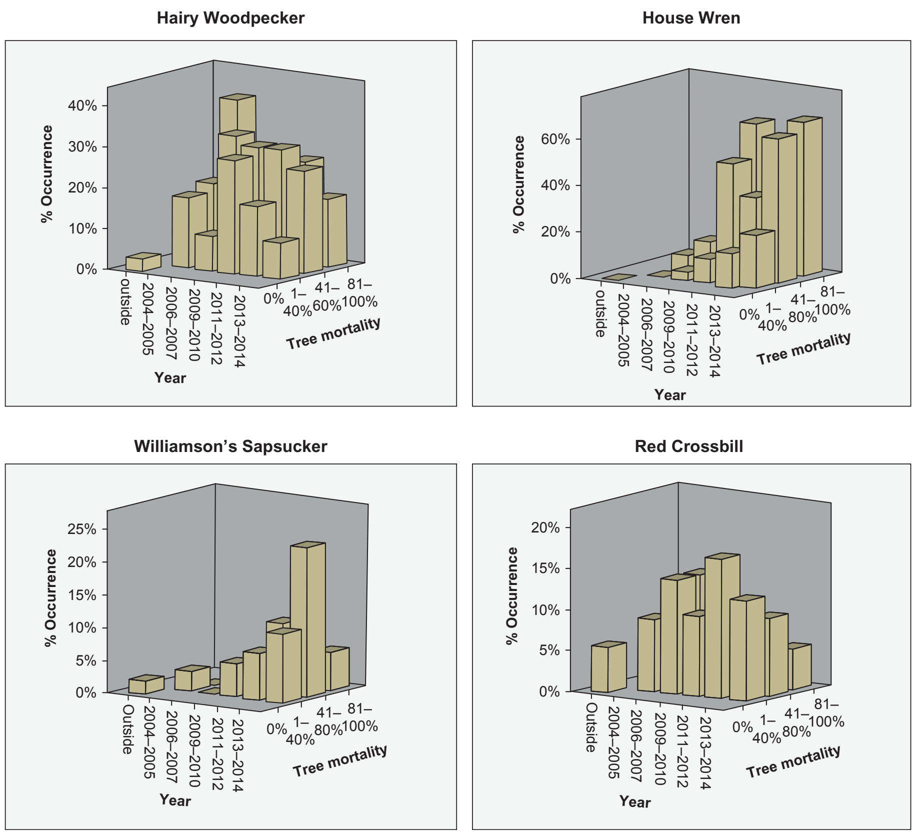
more abundant in some combination of those two variables than in long-unburned stands (Figure 3.4). Thus, not only are most species relatively abundant in one burned forest condition or another, but the average point in space and time occupied by each species is also species specific (Figure 3.5).

p0080 As an introduction to some of the fascinating biology surrounding severely burned forests, consider the following bird species. The black-backed woodpecker (*Picoides arcticus*), American three-toed woodpecker (*Picoides dorsalis*), hairy woodpecker (*Picoides villosus*), northern flicker (*Colaptes auratus*), and Lewis's woodpecker are all more abundant in severely burned than unburned mixed-conifer forest (see patterns of habitat occurrence for four of the five species in Figures 3.11 and 3.12) because of an abundance of food (beetle larvae and ants) and potential nest sites associated with standing dead trees.



f0025 **FIGURE 3.4** Example plots of percentage occurrence for various mixed-conifer bird species in relation to both time since fire and fire severity after the 2003 Black Mountain fire near Missoula, Montana (R.L. Hutto, unpublished; sample sizes exceed 35 point counts for each time-by-severity category; all patterns are significantly nonrandom as determined by log linear analyses [$P < 0.05$]). The examples were selected to illustrate that each species is more abundant in burned than in unburned forest (the occurrence rate in unburned forest shown in the first time period), and each is most abundant in a different combination of time since fire and burn severity (percentage of tree mortality).

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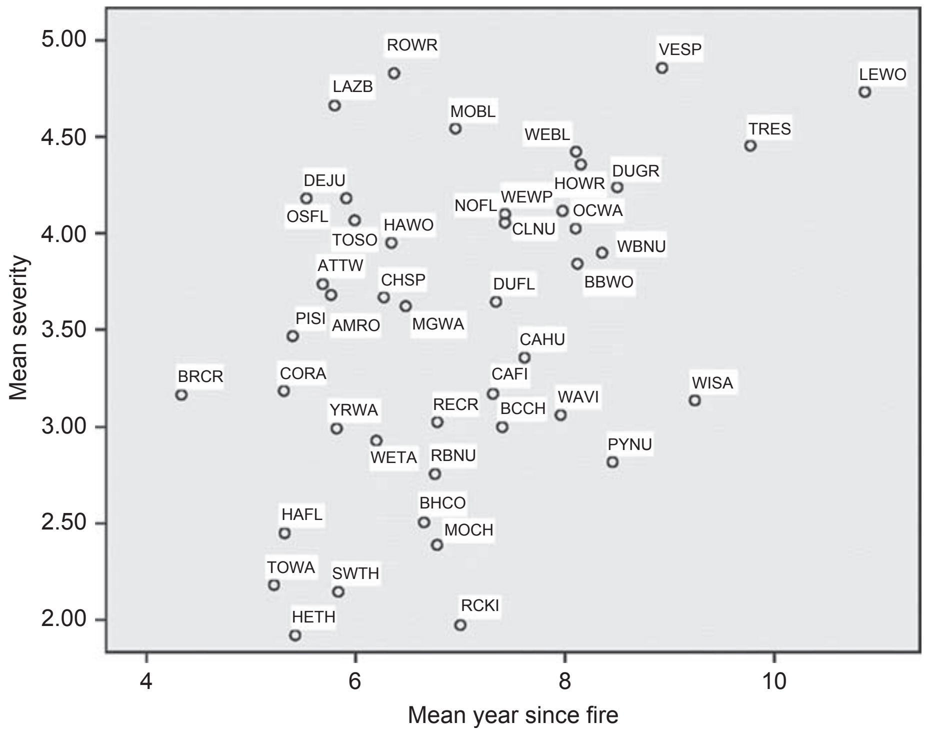

f0030 **FIGURE 3.5** In combination, the mean time since fire and mean fire severity at points of occurrence for each of 46 (mnemonically coded) species differs from that of every other species. Mean values were calculated from the kind of data presented in Figure 3.4.

The Williamson's sapsucker and olive-sided flycatcher (*Contopus cooperi*) find the abrupt edges between severely burned and unburned forest to be ideal nest locations (Figure 3.6). A host of secondary cavity-nesting and snag-nesting species (e.g., northern hawk owl [*Surnia ulula*], great gray owl [*Strix nebulosa*], mountain bluebird [*Sialia currucoides*], western bluebird [*Sialia mexicana*], house wren, and tree swallow [*Tachycineta bicolor*]) benefit from new forest openings, where they find a mature-forest legacy of already existing broken-top snags (Figure 3.7), where a disproportionately large number of nest sites are located (Hutto, 1995). These species depend on the kinds of snags that become common only after a forest reaches the mature- to old-growth stage and then burns in a severe fire. A variety of species (e.g., flammulated owl [*Psiloscops flammeolus*], mountain bluebird, Townsend's solitaire [*Myadestes townsendi*], and dark-eyed junco [*Junco hyemalis*]) make use of the cavities created by burned-out root wads or uprooted trees that happen to blow down in the first few years after severe fire (Figure 3.8). Many species (e.g., Clark's nutcracker [*Nucifraga columbiana*], Cassin's finch [*Haemorhous cassinii*], red crossbill (*Loxia curvirostra*], and pine siskin [*Spinus pinus*]) take advantage of seeds that are released or made available in cones that open after severe fire

f0035 **FIGURE 3.6** Williamson's sapsucker ~~(*Sphyrapicus thyroideus*~~; left) and olive-sided flycatcher (~~*Cont*~~ ~~*pperi;*~~ right) are known to nest disproportionately often near the abrupt edges between severely burned and unburned forest. *(Photographs by Richard Hutto (left) and Bruce Robertson (right).)*

f0040 **FIGURE 3.7** Compared with burned trees with intact tops, broken-top snags that were already snags before the fire burned are used disproportionately more often as nest sites by cavity-nesting bird species. The black-backed woodpecker also roosts almost entirely in burned-out hollows, forked trunks, or other relatively unusual structures that create crevices in "deformed" snags that existed before the forest burned (Siegel et al., 2014). Pictured (left to right) are a young hairy woodpecker ~~(*Picoides villosus*)~~ in its nest cavity, an American robin (*Turdus migratorius*) nest, and a northern flicker ~~(*Colaptes auratus*)~~ nest. The implications are profound—old-growth elements (snags) are really important to birds that depend on burned forest conditions, so burned, old-growth forests are as valuable to wildlife as unburned old-growth forests. *(Photographs by Richard Hutto.)*

f0045 **FIGURE 3.8** The architecture of a burned forest becomes modified after trees begin to blow down in the first few years after a fire, and a number of bird species make use of the root wads as nest sites. A Townsend's solitaire ~~(*Myadestes townsendi*~~) nest is highlighted here. *(Photograph by Richard Hutto.)*

f0050 **FIGURE 3.9** Few people seem to realize how i[illegible]t Clark's nutcrackers ~~(*Nucifraga columbiana*~~) are as seed dispersers after severe fire in pond[illegible]pine forests. Pictured here are examples of a nutcracker extracting seeds from a ponderosa pine cone that opened after fire (left) and a nutcracker with a throat pouch full of seeds in the scorched ground beneath a ponderosa pine canopy. *(Photographs by Richard Hutto.)*

(Figure 3.9). Still more bird species (e.g., calliope hummingbird [*Selasphorus calliope*], lazuli bunting [*Passerina amoena*], and MacGillivray's warbler [*Geothlypis tolmiei*]) use the shrub-dominated early seral stage for feeding and nesting and as display sites (Hutto, 2014).

s0045 Lesson 3: Not only Do Most Bird Species Benefit from Severe Fire, but Some also Appear to *Require* Severe Fire to Persist

p0085 The black-backed woodpecker has become an iconic indicator of severely burned forests because its distribution is nearly restricted to such conditions. Bent (1939) provided the first description of the unusual association between this woodpecker species and burned forests when he noted that Manly Hardy wrote to Major Bendire in 1895 about finding the woodpecker to be "… so abundant in fire-killed timber areas that I once shot the heads off six in a few minutes when short of material for a stew." This anecdote, reflecting the importance of severe fire, went largely unnoticed until the 1970s, when Dale Taylor undertook a study of birds in relation to time since fire in the Yellowstone and Grand Teton National Parks. His more systematic study uncovered the same remarkable pattern. Taylor was the first person to evaluate data drawn from a series of burned conifer forest stands of differing ages, and he found the appearance of the black-backed woodpecker to be restricted to the first few years after fire (Taylor and Barmore, 1980). A subsequent before-and-after fire study by Apfelbaum and Haney (1981) and studies of burned versus adjacent unburned forest by Niemi (1978), Pfister (1980), and Harris (1982) provided additional evidence that this bird species is strongly associated with burned forest conditions. Following the Rocky Mountain fires of 1988, Hutto (1995) conducted a more comprehensive study of the distribution of black-backed woodpeckers across a broad range of vegetation types. That study served to reinforce the notion that this species is an ideal indicator of severely burned mixed-conifer forest. More specifically, Hutto provided a meta-analysis of his own and already published bird survey data collected from burned forests and from more than a dozen unburned vegetation types; those data showed the black-backed woodpecker to be relatively restricted to burned forests. To address the potential problem of putting too much faith in distribution patterns derived from bird occurrence rates that were based on a variety of study durations and methods, Hutto subsequently coordinated the collection of standardized bird survey data from more than 18,000 points distributed across every major vegetation type in the U.S. Forest Service Northern Region. The results (Hutto, 2008) were strikingly similar to what earlier studies showed: one is hard pressed to find a black-backed woodpecker anywhere but in a recently burned forest (Figure 3.10).

p0090 Numerous studies (most published just in the past decade) provide additional detail that can help us better understand this remarkable association between the black-backed woodpecker and severely burned forests. Here we list some of the insights we have gained:

o0010 1. The magical appearance of woodpeckers within weeks of a fire (Blackford, 1955; Uxley, 2014) suggests that either smoke, or perhaps the fire or burned landscape itself, provides a stimulus for birds to colonize newly burned forests.

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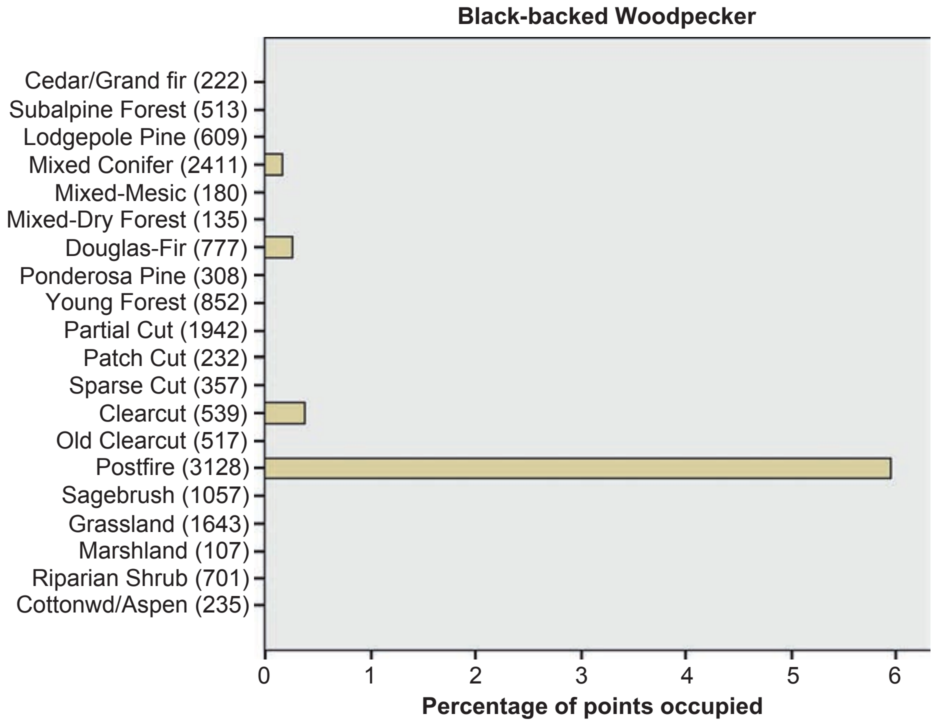

f0055 **FIGURE 3.10** Histogram bars indicate the percentage of points (sample sizes in parentheses) at which the black-backed woodpecker was detected in each of 21 distinct vegetation types within northern Idaho and western Montan[illegible] distribution is nonrandom ($X^2 = 559.43$; $df = 19$; $P < 0.0001$) and reveals that the black-[illegible] woodpecker is highly specialized in its use of burned conifer forest. *(Data from Hutto (2008).)*

o0015 **2.** Breeding and nest densities increase more rapidly than expected on the basis of recruitment alone (Yunick, 1985; Youngman and Gayk, 2011), which suggests that the process of immigration after fire is significant.

o0020 **3.** Woodpecker diet, which is based mainly on wood-boring beetle larvae that feed almost exclusively on recently burned and killed trees (Murphy and Lehnhausen, 1998; Powell et al., 2002; Fayt et al., 2005), reflects the broad postfire change in animal community composition that accompanies severe fire.

o0025 **4.** The woodpecker's nonrandom use of forest patches containing dense, larger-diameter trees (Saab and Dudley, 1998; Saab et al., 2002, 2009; Nappi and Drapeau, 2011; Dudley et al., 2012; Seavy et al., 2012) that have burned at high rather than low severity (Schmiegelow et al., 2006; Koivula and Schmiegelow, 2007; Hanson and North, 2008; Hutto, 2008; Nappi and Drapeau, 2011; Youngman and Gayk, 2011; Siegel et al., 2013) is striking and consistent among studies.

o0030 **5.** The window of opportunity for occupancy by this species is not only soon after fire, but generally lasts only about a half-dozen years before the birds

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(and the abundant native beetle populations) disappear (Taylor and Barmore, 1980; Apfelbaum and Haney, 1981; Murphy and Lehnhausen, 1998; Hoyt and Hannon, 2002; Saab et al., 2007; Nappi and Drapeau, 2009; Saracco et al., 2011).

o0035 **6.** The size of the home ranges of black-backed woodpeckers within burned forests are significantly smaller (indicating better quality habitat) than those outside burned forests (Rota et al., 2014b; Tingley et al., 2014). Even more telling is that nest success is significantly higher inside than outside burned forests (Nappi and Drapeau, 2009; Rota et al., 2014a).

o0040 **7.** Estimated population growth rates are insufficient to maintain a growing population outside burned forests (Rota et al., 2014a). Thus, although one could argue that low woodpecker densities in green-tree forests multiplied by a much larger unburned forest area might yield even more woodpeckers in green forests (Fogg et al., 2014), a sink area alone (no matter how large) can never yield a viable population of woodpeckers (Odion and Hanson, 2013).

o0045 **8.** The importance of severely burned forests as foraging locations for wintering black-backed woodpeckers is virtually unknown; the only detailed work so far (Kreisel and Stein, 1999) revealed densities that were an order of magnitude greater in burned than in unburned forests.

p0135 The biology surrounding this single bird species clearly reflects not only the ecological importance but also the necessity of severely burned forests, but major environmental organizations have yet to focus conservation efforts on burned forests (Schmiegelow et al., 2006), and management guidelines developed by state agencies to designate important wildlife habitats (e.g., https://www.dfg.ca.gov/biogeodata/cwhr/) do not even have burned conifer forests on their radar.

p0140 The distributional stronghold of the black-backed woodpecker might be considered to lie within the boreal forests of Canada, which nobody doubts are among the most severe-fire-dependent ecosystems in the world, but the bird's distribution south into the California Sierras and Rocky Mountains of the Intermountain West confirms that severe fires in those areas have been historically important as well. A North American forest bird species that is more narrowly restricted to a single forest condition does not exist; the black-backed woodpecker is the definition of a specialist. Everything about this bird species, including its distribution, territory size, breeding success, and even coloration pattern (which matches blackened trees), all indicate that this species needs expansive patches of severely burned forest to persist (Figure 3.11).

p0145 We have taken the liberty to provide extensive detail on this particular species because its ecological story carries significant management implications. Because public land managers have a responsibility to manage for the maintenance of all vertebrate species, finding even a single species that depends on severe fire should be enough to raise their awareness that severely burned

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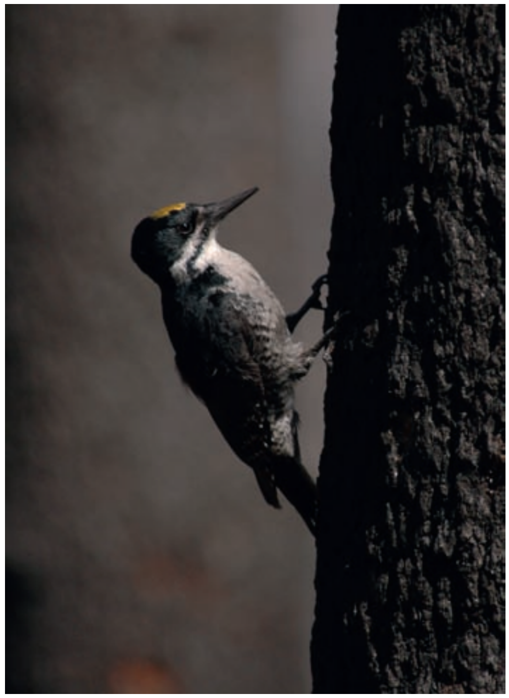

f0060 **FIGURE 3.11** Black-backed woodpecker—a species that is relatively restricted in its distribution to severely burned forests. *(Photograph by Richard Hutto.)*

mixed-conifer forests provide necessary habitat as well. Thus the black-backed woodpecker is an ideal focal species for bringing attention to the fact that burned forest conditions are important to maintain in the landscape (DellaSala et al., 2014). The evolutionary history that has led to a strong association between burned forests and the woodpecker also raises questions about whether (as many assume) severe fires in mixed-conifer forests are really beyond the historical natural range of variation, whether we need to be thinning forests outside the wildland-urban interface to reduce fire severity, whether we need to be suppressing fire outside the wildland-urban interface, and whether we should "salvage" logging trees (including important legacy trees; see Chapter 11) after fire. Yes, the story surrounding this focal species is important.

s0050 *Bird Species in Other Regions That Seem to Require Severe Fire*

p0150 Do any other bird species seem not only to benefit from but also to require severe fire to persist? The presence of a species in a specific environment and its absence elsewhere would be a clear indication that it depends on that particular environment. For species that occur across a range of environmental conditions, the places where they are relatively abundant are also likely to represent places that are required for population persistence because they persist in source areas and they are generally less abundant in, and their abundance is

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more variable through time in, more marginal areas (Pulliam, 1988; Sergio and Newton, 2003). Although the same level of biological detail that has been amassed for the black-backed woodpecker has not been collected for most other fire-associated bird species, the habitat distribution patterns of numerous bird species reveal that they are nowhere more abundant than in recently burned forests. For example, Hutto (1995) listed 15 species that were more abundant in recently burned forests than in any of 14 other vegetation types. Graphs generated from surveys conducted across an even broader range of vegetation types show just how striking these habitat distribution patterns can be: numerous species are nowhere more abundant than they are in severely burned forests (Hutto and Young, 1999) (Figure 3.12).

p0155 Many mixed-conifer bird species (e.g., black-backed woodpecker, American three-toed woodpecker, hairy woodpecker, northern flicker, olive-sided flycatcher, western wood-pewee [*Contopus sordidulus*], dusky flycatcher [*Empidonax oberholseri*], mountain bluebird, Townsend's solitaire, house wren, tree swallow, lazuli bunting, Clark's nutcracker, red crossbill) fall consistently into a short-term "benefit" category, as revealed either by some measure of abundance or nest success in studies of burned versus unburned or before versus after fire (Bock and Lynch, 1970; Bock et al., 1978; Taylor and Barmore, 1980; Apfelbaum and Haney, 1981; Raphael et al., 1987; Hutto, 1995; Kotliar et al., 2002; Hannah and Hoyt, 2004; Smucker et al., 2005; Mendelsohn et al., 2008; Seavy and Alexander, 2014). Even severely burned patches within conifer forests that we have come to associate with low-severity fire can provide critically important habitat for species like the buff-breasted flycatcher (Kirkpatrick et al., 2006; Conway and Kirkpatrick, 2007; Hutto et al., 2008).

p0160 One of the most celebrated examples of a fire specialist involves the federally endangered Kirtland's warbler (*Setophaga kirtlandii*). It occurs almost exclusively in young (5- to 23-year-old) jack pine (*Pinus banksiana*) forest historically created by severe fire (Walkinshaw, 1983). In addition, pairing success is significantly higher in burned than in unburned forests (98% vs. 58% success; Probst and Hayes, 1987). The need for severe fire is obvious not only because, historically, it must have taken severe fires to stimulate forest succession but also because of how its critically endangered population increased dramatically after a fire accidentally escaped within its breeding range (James and McCulloch, 1995). Managers have had difficulty trying to recreate conditions that mimic natural postfire conditions through the use of logging techniques (Probst and Donnerwright, 2003; Spaulding and Rothstein, 2009), and efforts to use these artificial means to maintain warbler populations miss the point. Conservation efforts should be directed toward maintaining severely burned forests, not toward finding a way around the natural fire disturbance process.

p0165 In Australia, where few species are thought to be restricted to recently burned shrubland or forest conditions, early colonists are viewed as generalists, and management concerns are focused on postfire decreases in late-succession specialists (Serong and Lill, 2012). Nevertheless, recent data from Lindenmayer

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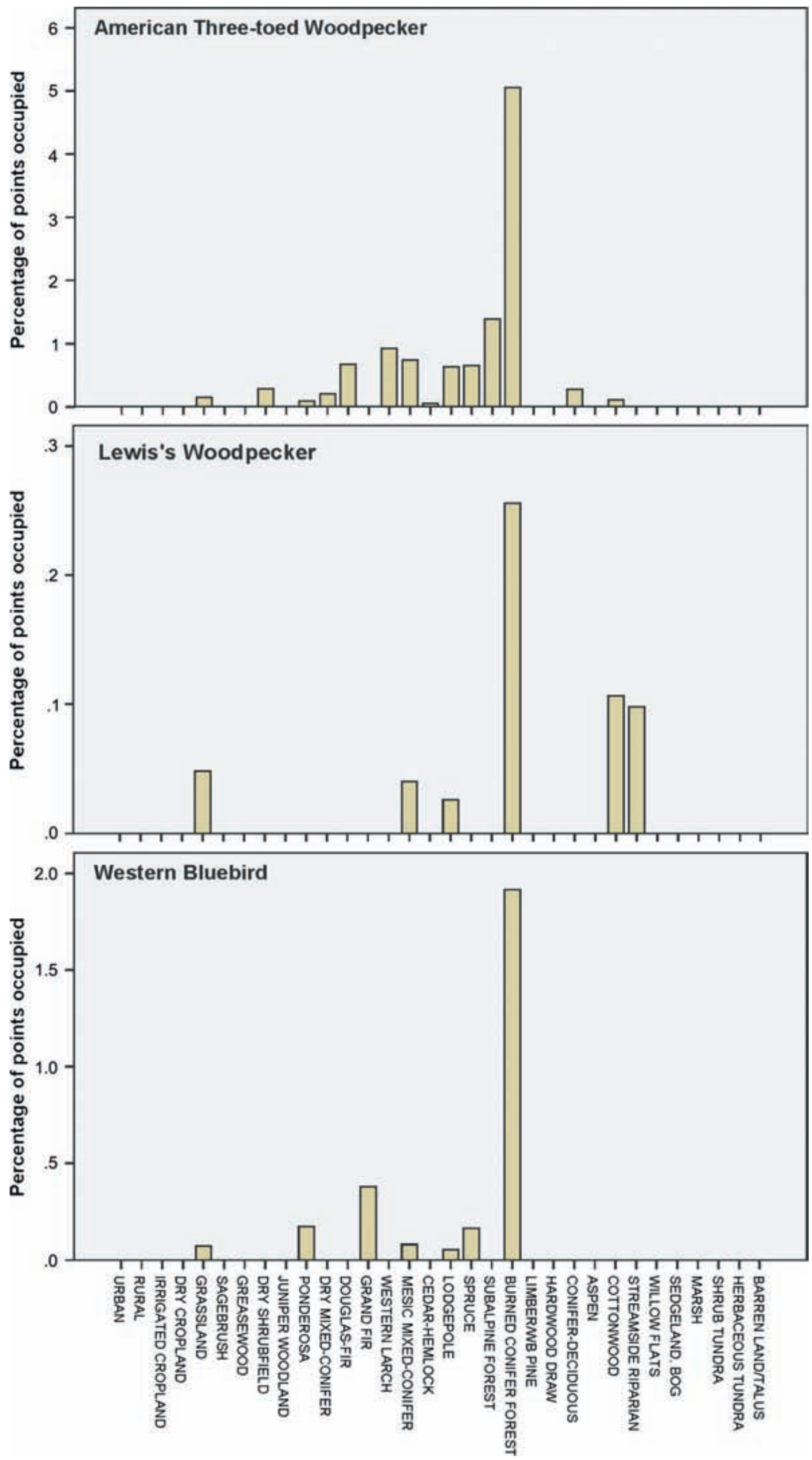

f0065 **FIGURE 3.12** Several graphs depicting species that seem to be more abundant in burned forests than in any other vegetation type in the northern Rocky Mountains. Data were drawn from a subset of the Northern Region Landbird Monitoring Program database consisting of 20,000 survey points distributed across northern Idaho and western Montana.

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et al. (2014) show that a number of bird species decline in abundance 1-2 years after moderate to severe fire but then return to levels comparable to, or *higher* than, those in unburned forests within 3 years following fire. Indeed, upon further inspection, we found that the superb fairywren (*Malurus cyaneus*), gray fantail (*Rhipidura albiscapa*), yellow-faced honeyeater (*Lichenostomus chrysops*), white-fronted honeyeater (*Purnella albifrons*), dusky robin (*Melanodryas vittata*), flame robin (*Petroica phoenicea*), willie wagtail (*Rhipidura leucophrys*), gray shrike-thrush (*Colluricincla harmonica*), varied sittella (*Daphoenositta chrysoptera*), apostlebird (*Struthidea cinerea*), white-browed scrubwren (*Sericornis frontalis*), brown thornbill (*Acanthiza pusilla*), spotted pardalote (*Pardalotus punctatus*), welcome swallow (*Hirundo neoxena*), dusky woodswallow (*Artamus cyanopterus*), black-faced woodswallow (*Artamus cinereus*), and silver-eye (*Zosterops lateralis*) each have been shown by one or more authors to be more abundant in severely burned than in long unburned, dry sclerophyll forests (Christensen and Kimber, 1975; McFarland, 1988; Reilly, 1991a,b, 2000; Turner, 1992; Taylor et al., 1997; Fisher, 2001; Leavesley et al., 2010; Recher and Davis, 2013; Lindenmayer et al., 2014). Thus many eucalyptus forest species also seem to require severe fire to create the early successional forest conditions within which they are most abundant, but most of those species are not restricted to conditions that occur during the first year or two after fire. In comparison with the dramatic change in bird species composition following severe fire in mixed-conifer forests, there is, in fact, a notable lack of turnover in bird species composition following severe fire in eucalyptus forests (compare before-and-after fire data from Australia and the western United States in Figure 3.13). This difference in response to fire is presumably because eucalyptus trees resprout rapidly from epicormic shoots (Figure 3.2). Lindenmayer et al. (2014) also note that in montane ash forests, "... very rapid vegetation regeneration and canopy closure on severely burned sites ... may limit the influx of open-country birds and preclude the evolutionary development of early successional species" (p. 474). Nevertheless, the bird species listed above suggest that many may depend on slightly later stages of succession before the development of a fully mature forest and that a slightly different perspective might be needed to expose the ecological importance of severe fire to birds of Australian eucalypt forests.

p0170 Taken together, we hope we have provided enough ecological information derived from birds to solidify the notion that severe fire in most severe-fire-dependent shrublands and forests is both natural and necessary for maintenance of the ecological integrity of such systems.

s0055 Postfire Management Implications

p0175 Severe fire is natural and necessary in most—not relatively few—conifer forest types and in many other vegetation types worldwide as well (see Chapters 1 and 2). Current management practices designed to prevent fire,

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Australian eucalyptus forest		
Species	unburned (n=39)	burned (n=35)
New Holland Honeyeater	0.161	0
Little Wattlebird	0.095	0
Scarlet Robin	0.019	0
Yellow-faced Honeyeater	0.040	0.231
Painted Button-Quail	0	0.035
Grey Shrike-thrush	0	0.058
Olive-backed Oriole	0	0.131

Western North American mixed-conifer forest		
Species	unburned (n=1143)	burned (n=638)
Townsend's Warbler	0.4	0.03
Solitary Vireo	0.238	0.021
Golden-crowned Kinglet	0.235	0.021
Gray Jay	0.084	0.009
Pileated Woodpecker	0.052	0.006
Swainson's Thrush	0.43	0.062
Varied Thrush	0.103	0.015
White-breasted Nuthatch	0.017	0.003
Black-capped Chickadee	0.053	0.012
Red-breasted Nuthatch	0.591	0.145
Ruby-crowned Kinglet	0.316	0.086
Hammond's Flycatcher	0.091	0.027
Hermit Thrush	0.048	0.015
Orange-crowned Warbler	0.098	0.036
Western Tanager	0.398	0.163
Mountain Chickadee	0.219	0.092
MacGillivray's Warbler	0.201	0.095
Yellow-rumped Warbler	0.521	0.249
Warbling Vireo	0.145	0.098
Clark's Nutcracker	0.022	0.047
Pine Siskin	0.111	0.257
Rufous Hummingbird	0.014	0.038
Northern Flicker	0.076	0.21
Calliope Hummingbird	0.01	0.03
Song Sparrow	0.004	0.015
Olive-sided flycatcher	0.025	0.107
Rufous-sided Towhee	0.01	0.044
Cassin's Finch	0.029	0.13
American Kestrel	0.003	0.015
Mourning Dove	0.004	0.021
Hairy Woodpecker	0.021	0.124
Three-toed Woodpecker	0.007	0.056
Northern Waterthrush	0.003	0.033
Green-tailed Towhee	0.001	0.012
White-crowned Sparrow	0.002	0.027
Lazuli Bunting	0.01	0.148
House Wren	0.004	0.086
Western Wood-pewee	0.003	0.104
Mountain Bluebird	0.004	0.281
American Robin	0.185	0.441
Lincoln's Sparrow	0	0.015
Tree Swallow	0	0.089
Rock Wren	0	0.044
Black-backed Woodpecker	0	0.05

f0070 **FIGURE 3.13** Probabilities of the occurrence of bird species in burned and unburned Australian eucalypt forests in the tablelands above Wollongong, New South Wales, and in burned and unburned mixed-conifer forests in western Montana (R.L. Hutto, unpublished data). Numbers of survey points are given in parentheses. Birds are ordered by the unburned-to-burned ratio of abundance, and species that are completely absent from or are significantly (Mann-Whitney *U* tests) less abundant in the opposite condition are highlighted in yellow. In both locations are bird species restricted to either early or later successional stages, but the amount of species turnover (degree of replacement of late with early succession specialists) is less pronounced after severe fire in Australia than after severe fire in the western United States.

suppress fire, mitigate fire severity, "restore" or "rehabilitate" burned forests after fire, and mimic the effects of severe fire are incompatible with the maintenance of ecosystem integrity (Chapter 13). Below we use results from bird research as evidence to support this statement, and we offer positive suggestions about what land managers could be doing differently.

s0060 *Fire Prevention Should Be Focused on Human Population Centers*

p0180 The dependence of so many bird (and many other plant and animal) species on conditions created by severe fire is clear. It necessarily follows that we cannot prevent fire and still retain anything close to a natural world. The obvious

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alternative is to focus prevention efforts toward population centers that are most at risk from severe fire so that fire can be left to periodically restore forest conditions elsewhere. Smokey Bear needs to refine his message so that it reflects a desire to save human lives and property, *not* a desire to save trees from fire in our wildlands (see Chapter 13).

s0065 *Fire Suppression Should Be Focused on the Wildland-Urban Interface (or Fireshed)*

p0185 Because many species depend on severe fire, it also necessarily follows that we should focus suppression efforts on areas immediately adjacent to human settlements (see Chapter 13). Wildland firefighters should serve primarily as support for firefighters who defend homes and human lives. Efforts to suppress fire beyond settled areas should be viewed as little more than efforts to save the forest from itself—forests need fire in the same way that they need sunlight and rain.

s0070 *High-Severity Fires Beget Mixed-Severity Results*

p0190 In contrast with high-severity fire, low-severity understory fires cannot create as broad a range of postfire conditions as severe fires can, nor can they stimulate the postfire process of ecological succession like a severe fire can. Therefore, managing for the maintenance of biodiversity requires more conscientious management for the maintenance of severe fires and the mixed-severity landscape effects that result from such fires (Nappi et al., 2010; Taylor et al., 2012).

s0075 *Mitigate Fire Severity Through Thinning only Where such Fuel Reduction Is Appropriate*

p0195 Because many species depend on severe fire, it necessarily follows that we should focus forest-thinning efforts in the wildland-urban interface and perhaps beyond that in what are basically artificial tree plantations that have resulted from past timber harvesting (see Odion et al., 2014a for review of this topic). The distributions of black-backed woodpeckers and many other fire-dependent plant and animal species make it abundantly clear that a reduction in fire severity is ecologically justified in only a very small proportion of vegetation types (Odion et al., 2014a; Sherriff et al., 2014). The presence of numerous fire-dependent species in most conifer forests throughout the American West (as illustrated by the abundance of bird research results considered in this chapter) is the strongest possible indication that the same forests have burned severely for millennia and are well within the historical range of natural variation.

p0200 The distribution of birds like the black-backed woodpecker and other fire-dependent plant and animal species, which blanket most of the forested land in the American West, are clearly at odds with claims (e.g., Haugo et al., 2015) that as much as 40% of public forested lands in parts of the United States are in need of restoration to prevent or mitigate the effects of severe fire. Lower-severity fires do not produce the mixed- and high-severity conditions needed by the most

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fire-dependent bird species, so efforts to mitigate fire severity in most places is incompatible with maintenance of the ecological integrity of most conifer forest systems (Odion et al., 2014a). So, what should we be doing differently? We could realize that modeled estimates indicating that our forests are in conditions that lie beyond the historical natural range of variation are just that—modeled estimates that rest strongly on many untested assumptions. We should always compare modeled results with insight gained by ecologists who can also draw strong inferences about historical conditions and, more specifically, about the kind of environments that necessarily led to adaptations of plants and animals—adaptations that reflect the distant past much more accurately than other methods commonly used to reconstruct natural fire regimes.

s0080 *Postfire "Salvage" Logging in the Name of Restoration or Rehabilitation Is Always Inappropriate*

p0205 Postfire "salvage" logging, seeding, planting, and shrub removal have overwhelmingly negative effects on natural systems (Lindenmayer et al., 2004; Lindenmayer and Noss, 2006; McIver and Starr, 2006; Swanson et al., 2011; DellaSala et al., 2014; Hanson, 2014), and birds have been instrumental in uncovering that fact. There is nothing as obvious to a birdwatcher as the negative effect of postfire salvage logging on the most fire-dependent birds (Uxley, 2014), and these anecdotal impressions are backed up by the strongest and most consistent scientific results ever published on any wildlife management issue (Hutto, 1995, 2006; Morissette et al., 2002; Nappi et al., 2004; Hutto and Gallo, 2006; Koivula and Schmiegelow, 2007; Hanson and North, 2008; Cahall and Hayes, 2009; Saab et al., 2009; Rost et al., 2013). One look at (Figure 3.14), or one walk through, a salvage-logged forest after knowing

f0075 **FIGURE 3.14** A vivid view of what can only be described as an ecological disaster following this postfire salvage logging operation, which took place after the 1988 Combination fire in Montana. *(Photograph by Richard Hutto.)*

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something about the biological wonder associated with a severely burned forest should be enough to convince any thinking person that there is no justification for this kind of land management activity.

p0210 It is bad enough that forests logged after fire are made unsuitable for black-backed woodpeckers and other early postfire specialists, but much worse is that postfire logging and shrub removal through mechanical or chemical means may also act as an "ecological trap" (Robertson and Hutto, 2006). This can occur when birds are attracted to burned areas that seem to be suitable and then those areas are suddenly transformed by logging or shrub removal into unsuitable habitat in an unnaturally rapid period of time. This is the most reasonable explanation for why black-backed woodpeckers are more abundant in dense, burned forests that are logged after fire than they are in burned forests that are logged before fire—birds are not attracted to the latter, where tree densities are too low and sizes are too small to provide suitable habitat, but they are attracted to the former before the trees are unexpectedly removed (Hutto, 2008). Similarly, the disproportionate use of recently logged, unburned, old-growth forests in Canada (Tremblay et al., 2009) suggests that black-backed woodpeckers sometimes make the best of a marginal situation, not that they "prefer" recently logged forests.

p0215 Although the ecological responses of birds to postfire salvage logging may differ among globally different ecosystems (Rost et al., 2012), there is absolutely no ecological justification for this kind of logging in the mixed-conifer forests of the western United States, nor is there an economic justification to salvage log after fire, because there are always better places to harvest timber without anywhere near the negative ecological consequences associated with postfire salvage logging. This is a matter of setting priorities for timber harvest, and burned forests should be at the bottom of the list. Burned forests not only provide unique ecological value, they also set the stage for the development of a variety of future forest conditions—conditions that are much more varied than those associated with development after artificial disturbance from logging. Forests have their own rules and timetables associated with the natural process of ecological succession, and we should embrace that variety and complexity. What could be done differently? Postfire rehabilitation should focus on roads, culverts, and other infrastructure issues, and nothing else. We need to recognize that new forest conditions get created after fire, and a disturbance-dependent forest does not need to be "fixed" after disturbance takes place.

s0085 *We Can Do more Harm Than Good Trying to "Mimic" Nature*

p0220 Prescribed burning, forest thinning, and the use of other forms of artificial disturbance in an effort to mimic nature are often poor substitutes for natural disturbance processes. Prescribed burning is usually done out of season, too frequently, and in a manner that is far too mild to have the necessary effects in most systems that evolved with fire (England, 1995; Tucker and

Robinson, 2003; Penman and Towerton, 2008; Peters and Sala, 2008; Arkle and Pilliod, 2010; Rota et al., 2014a). Thinning forests in a manner thought to mimic disturbance effects is also likely to be problematic because natural disturbance (the process of fire itself) produces effects that cannot be emulated through artificial means (Schieck and Song, 2006; Reidy et al., 2014). Moreover, a thinned forest that subsequently burns in a natural fire event will not be suitable as postfire habitat for early postfire specialists because of the reduction in tree densities and sizes (Hutto, 2008). Finally, the use of forest thinning in the name of forest restoration is inappropriately applied to relatively mesic mixed-conifer forests that are unlikely to be in need of restoration, as indicated by a lack of posttreatment change in bird communities toward what one would expect if the forests were actually outside the historical range of natural variation (Hutto et al., 2014).

p0225 Except in the case of an endangered species, the worst management approach is one that focuses narrowly on creating *artificial* conditions needed by a single species. This is "single-species management," which is not the same thing as using a "management indicator approach." Management indicators are not meant to be tools that enable land managers to artificially modify land conditions to benefit a single species. Instead, a management indicator species should be used as an indication of a particular kind of "natural" condition that needs to be maintained on the landscape and as a check that the land condition is indeed acceptable to a species that requires such conditions [illegible] for an endangered species, we should always be thinking about maintaining the "natural" conditions that historically maintained its population. Thus, although artificial tree plantations may provide conditions used by Kirtland's warbler (Spaulding and Rothstein, 2009), the bird historically nested beneath the canopy of young trees born of fire. Therefore we should create conditions safe enough to allow natural severe fire events to unfold throughout most of its historical range. As clearly stated in the Endangered Species Act (ESA, Section 2), "the purposes of this act are to provide a means whereby the *ecosystems upon which endangered species and threatened species depend* may be conserved . . ." (our italics). Conservation should be about the larger system (e.g., maintaining a fire disturbance-based jack pine forest system), not about ~~a~~ finding a way to maintain a species through artificial means. Thus the black-backed woodpecker is an "indicator" or "focal species" that should be used to inform us about a critically important "natural" disturbance process and vegetation condition we need to maintain—severely burned forests and all the associated organisms that thrive within them.

p0230 What could we be doing differently? We need to trust that disturbance-dependent systems need severe disturbance (yes, that means a lot of tree death) to stimulate ecological succession in a manner that is indeed natural. We also need to appreciate that modeled *means and standard deviations* associated with measures of forest structure are not the same things as historical *ranges of variation* associated with the same measures. While some places have tree densities that exceed some estimated historical average value, it does not mean they fall outside the historical range of natural variation. Land managers need to relax in response to severe fire. As long as we can reduce the frequency of human-caused fires and remain safe during naturally ignited fire events, a management option that lets nature take its course will work just fine (Gill, 2001; Bradstock, 2008). In this context, noting that safety is best achieved through mechanical treatments in small areas immediately adjacent to structures (Cohen, 2000; Cohen and Stratton, 2008; Winter et al., 2009; Stockmann et al., 2010; Gibbons et al., 2012; Syphard et al., 2014), and not through mechanical treatments in more remote wildlands, is important. Given this fact, why treatments in relatively remote, publicly owned wildlands have become the tactic most commonly used to reduce wildfire risk is puzzling (Schoennagel et al., 2009).

s0090 *Concluding Remarks*

p0235 The most important ecological lessons we can take away from the bird research described in this chapter are that (1) many species have evolved to the point where they now require severe fire to create the conditions they need, and (2) even though some ecological systems may have departed significantly from what are believed to be historical conditions (e.g., tree plantations in the Pacific Northwest), birds are telling us (through their behavior and distribution patterns) that the vast majority of fire-dependent ecosystems are still well within the historical range of natural variation, are plenty "resilient," and are fully capable of proceeding quite naturally through the process of succession following a severe-fire event. Therefore, thinning forests in the name of restoration is largely unnecessary. If this were not true, the world would be full of places that experienced a severe fire disturbance and then underwent an unnatural transformation or "type conversion" following the disturbance event, never to return to what was there before disturbance. It is most telling that those kinds of places are rare indeed.

p0240 For those who would like to read, view, or hear more about the relationship between birds and severe fire, there are excellent children's books (e.g., Peluso, 2007; Collard, 2015); several informative videos, including a field trip that illustrated many of the patterns discussed here (listed in the Preface); and a Fire Ecology Lab Facebook page (https://www.facebook.com/FireEcologyLab) devoted to building an appreciation for the role of severe fire in our forests.

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Non-Print Items

Abstract

Important lessons emerge from studies of birds in ecosystems born of, and maintained by, mixed- to high-severity fire. Specifically, (1) the effect of fire on any one species is context dependent. It depends on the time since the fire, the fire severity, and vegetation type and condition. (2) Bird species respond differently to any given postfire condition and, given an appropriate time since the fire and postfire vegetation conditions, most benefit from severe fire. (3) Some bird species (the black-backed woodpecker being iconic) seem to *depend* on conditions created by severe fire, as evidenced by their distribution patterns, territory sizes, nest success, and other adaptations. (4) Given these facts, current management practices designed to prevent fire, suppress fire, mitigate fire severity, "restore" or "rehabilitate" burned forests after fire, and mimic the effects of severe fire are incompatible with the maintenance of bird populations and, therefore, ecosystem integrity.

Keywords: Adaptation; Bird; Disturbance; Fire severity; Forest restoration; Salvage logging; Severe fire.

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SUPPLEMENTARY MATERIALS

www.sciencemag.org/content/354/6318/1419/suppl/DC1
Materials and Methods
Figs. S1 to S5
Tables S1 and S2
References (*22–29*)

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CONSERVATION

A global map of roadless areas and their conservation status

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Roads fragment landscapes and trigger human colonization and degradation of ecosystems, to the detriment of biodiversity and ecosystem functions. The planet's remaining large and ecologically important tracts of roadless areas sustain key refugia for biodiversity and provide globally relevant ecosystem services. Applying a 1-kilometer buffer to all roads, we present a global map of roadless areas and an assessment of their status, quality, and extent of coverage by protected areas. About 80% of Earth's terrestrial surface remains roadless, but this area is fragmented into ~600,000 patches, more than half of which are <1 square kilometer and only 7% of which are larger than 100 square kilometers. Global protection of ecologically valuable roadless areas is inadequate. International recognition and protection of roadless areas is urgently needed to halt their continued loss.

The impact of roads on the surrounding landscape extends far beyond the roads themselves. Direct and indirect environmental impacts include deforestation and fragmentation, chemical pollution, noise disturbance, increased wildlife mortality due to car collisions, changes in population gene flow, and facilitation of biological invasions (*1–4*). In addition, roads facilitate "contagious development," in that they provide access to previously remote areas, thus opening them up for more roads, land-use changes, associated resource extraction, and human-caused disturbances of biodiversity (*3*, *4*). With the length of roads projected to increase by >60% globally from 2010 to 2050 (*5*), there is an urgent need for the development of a comprehensive global strategy for road development if continued biodiversity loss is to be abated (*6*). To help mitigate the detrimental effects of roads, their construction should be concentrated as much as possible in areas of relatively low "environmental values" (*7*). Likewise, prioritizing the protection of remaining roadless areas that are regarded as important for biodiversity and ecosystem functionality requires an assessment of their extent, distribution, and ecological quality.

Such global assessments have been constrained by deficient spatial data on global road networks. Importantly, recent publicly available and rapidly improving data sets have been generated by crowd-sourcing and citizen science. We demonstrate their potential through OpenStreetMap, a project with an open-access, grassroots approach to mapping and updating free global geographic data, with a focus on roads. The available global road data sets, OpenStreetMap and gROADS, vary in length, location, and type of roads; the former is the data set with the largest length of roads (36 million km in 2013) that is not restricted to specific road types (table S1). OpenStreetMap is more complete than gROADS, which has been used for other global assessments (*7*), but in certain regions, it contains fewer roads than subglobal or local road data sets [see the example of Center for International Forestry Research data for Sabah, Malaysia (*8*); table S1]. Given the pace of road construction and data limitations, our results overestimate the actual extent of global roadless areas.

The spatial extent of road impacts is specific to the impact in question and to each particular road and its traffic volume, as well as to taxa, habitat, landscape, and terrain features. Moreover, for a given road impact, its area of ecological influence is asymmetrical along the road and can vary among seasons, between night and day, according to weather conditions, and over longer time periods. We conducted a comprehensive literature review of 282 publications dealing with "road-effects zones" or including the distance to roads as a covariate, of which 58 assessed the spatial influence of the road (table S2). All investigated road impacts were documented within a distance of

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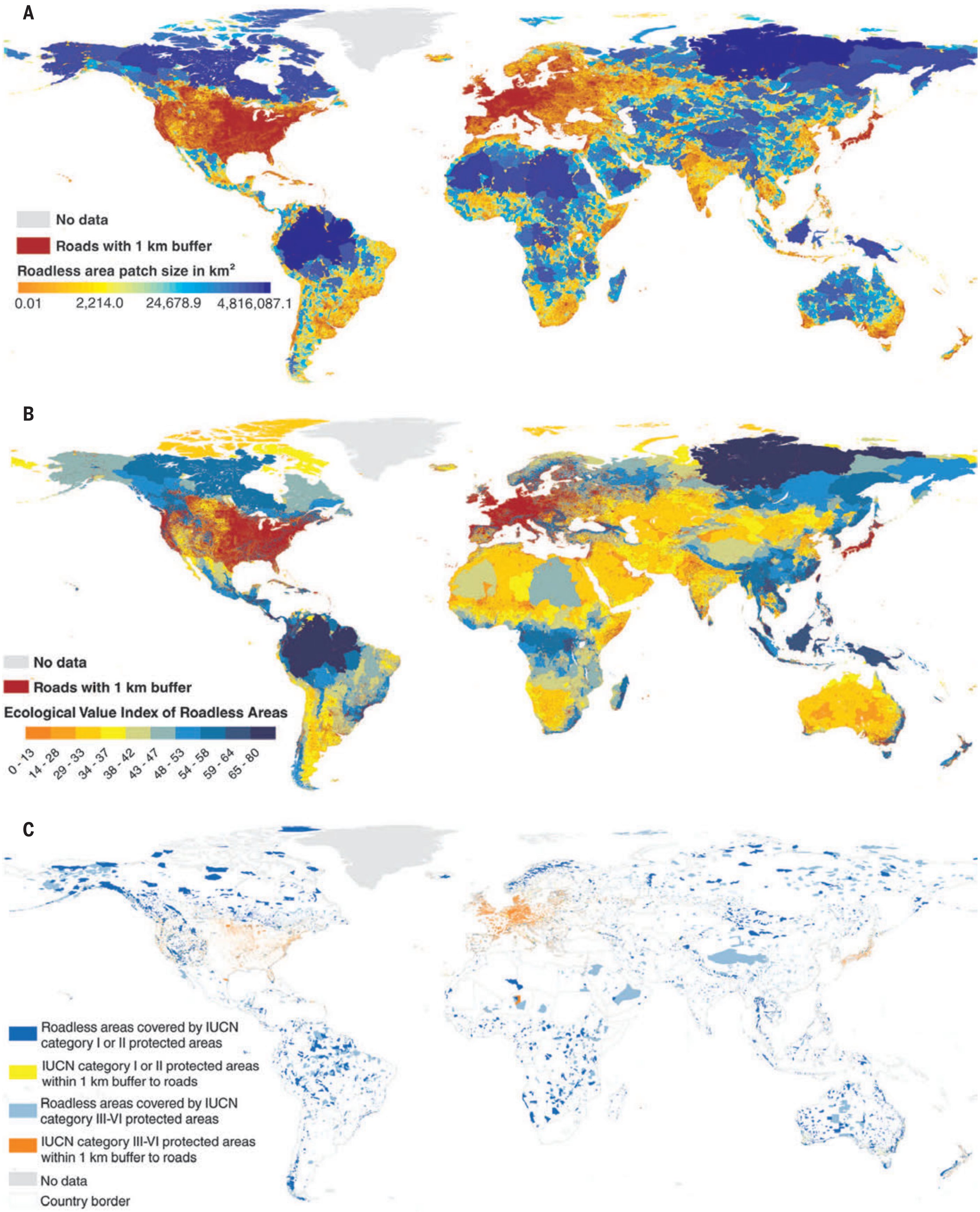

Fig. 1. The global distribution of roadless areas, based on a 1-km buffer around all roads. The distribution is depicted according to (**A**) size classes, (**B**) the ecological value index of roadless areas (EVIRA; based on patch size, connectivity, and ecosystem functionality), and (**C**) representation in protected areas (*8*).

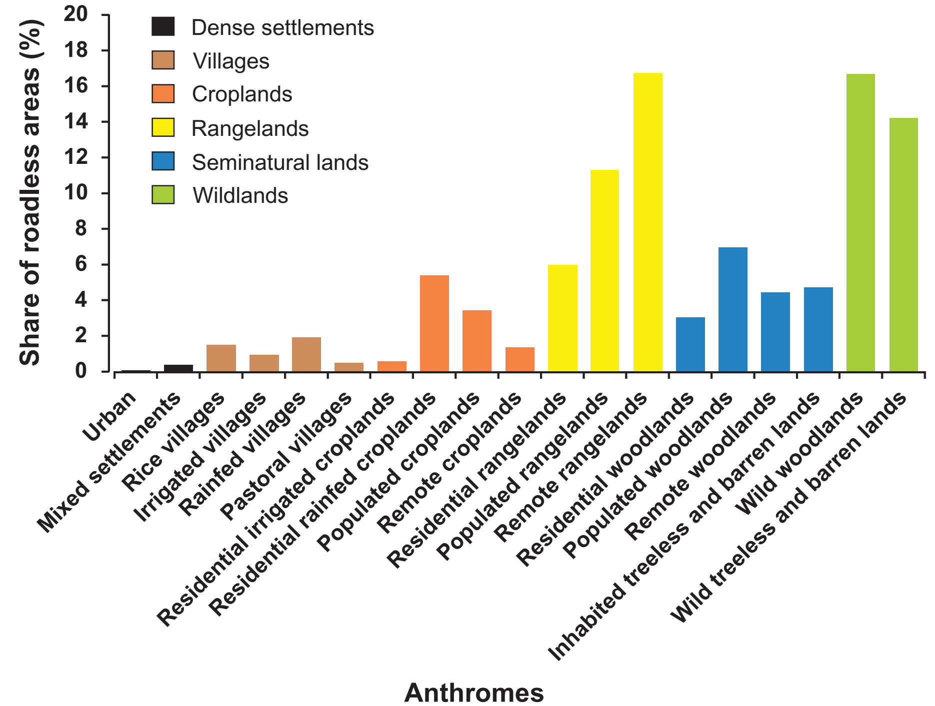

Fig. 2. Extent of roadless areas (1-km buffer) across anthromes. The majority of the world's roadless areas are in remote and unmodified landscapes, but they also occur in anthropogenically modified landscapes. The so-called anthromes were mapped according to (*10*).

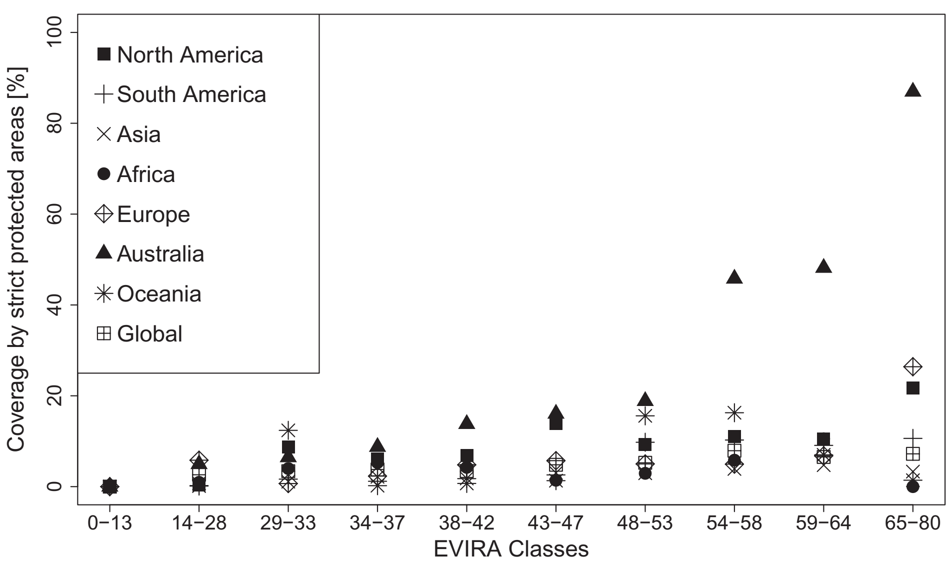

Fig. 3. Coverage of roadless areas by strictly protected areas (IUCN categories I and II) compared with global and continental EVIRA values. If priority were given to protecting roadless areas with high ecological functionality, we should see a positive correlation, with higher coverage associated with higher EVIRA values.

1 km from the road, 39% reached out to 2 km from the road, and only 14% extended out to 5 km from the road (fig. S1). Because the 1-km buffer along each side of the road represents the zone with the highest level and variety of road impacts, we defined roadless areas as those land units that are at least 1 km away from all roads and, therefore, less influenced by road effects. We compared results from using this criterion with the outcomes from using an alternative 5-km buffer (see fig. S2 and table S3). We excluded all large water bodies, as well as Greenland and Antarctica, which are mostly covered by ice, from the analyses.

Roadless areas with a 1-km buffer to the nearest road cover about 80% of Earth's terrestrial surface (~105 million km^2). However, these roadless areas are dissected into almost 600,000 patches. More than half of the patches are <1 km^2; 80% are <5 km^2; and only 7% are >100 km^2 (table S4 and fig. S3). If the buffer is extended to 5 km, there is a substantial reduction in roadless areas to about 57% of the world's terrestrial surface (~75 million km^2), dissected into 50,000 patches (fig. S2 and table S3). The occurrence, distribution, and size of roadless areas differ considerably among continents (Fig. 1A and fig. S4). For instance, the mean size of roadless patches (1-km buffer) is 48 km^2 in Europe, compared with >500 km^2 in Africa. Because of comparatively large gaps in available spatial data on roads in many segments of the tropics, the number and size of roadless areas are overestimated and should be treated with caution (e.g., Borneo; table S1).

All identified roadless areas were assessed for a set of ecological properties that were selected to reflect their relative importance to biodiversity, ecological functions, and ecosystem resilience: patch size, connectivity, and ecosystem functionality (*9*) (table S5). We normalized these three indicators to between 0 and 100 to calculate an additive and unitless index of the ecological value of each roadless area identified (termed the ecological value index of roadless areas, or EVIRA) [Fig. 1B and fig. S5; the specific rationale and technicalities of the chosen indicators are described in table S5 (*8*)]. The EVIRA values range from 0 to 80. A sensitivity analysis shows that ecosystem functionality and patch size are the best single indicators for the final index values (table S6 and figs. S6 to S8). Areas with relatively high index values tend to have a lower coefficient of variation (fig. S9).

We used the International Union for Conservation of Nature (IUCN) and UN Environment Programme–World Conservation Monitoring Centre data set of global protected areas to determine the extent of roadless areas that are protected (*8*) (Fig. 1C). The roadless areas distribution across human-dominated landscapes was determined following the classification of so-called anthromes, defined as biomes shaped by human land use and infrastructure (*10*) (Fig. 2 and table S7).

When examining the density of roads within different biomes, large discrepancies in distribution are apparent. The tundra and rock and ice-covered biomes are nearly entirely roadless, whereas temperate broadleaf and mixed forests have the lowest share of roadless areas (41%; figs. S9 and S10). Boreal forests of North America and Eurasia still retain large tracts of roadless areas (figs. S10 and S11). In the tropics, large roadless landscapes (>1000 km^2) remain in Africa, South America, and Southeast Asia, with the Amazon having the single largest roadless segment. In relation to the anthromes (*10*), about two-thirds of the world's roadless areas can be described as remote and unmodified landscapes [26% uninhabited or sparsely inhabited treeless and barren lands; 21% natural and remote seminatural woodlands, with 17% wild woodlands therein (*8*); Fig. 2 and table S7]. The remaining one-third consists of rangelands, indicating that roadless areas can also occur in anthropogenically modified landscapes.

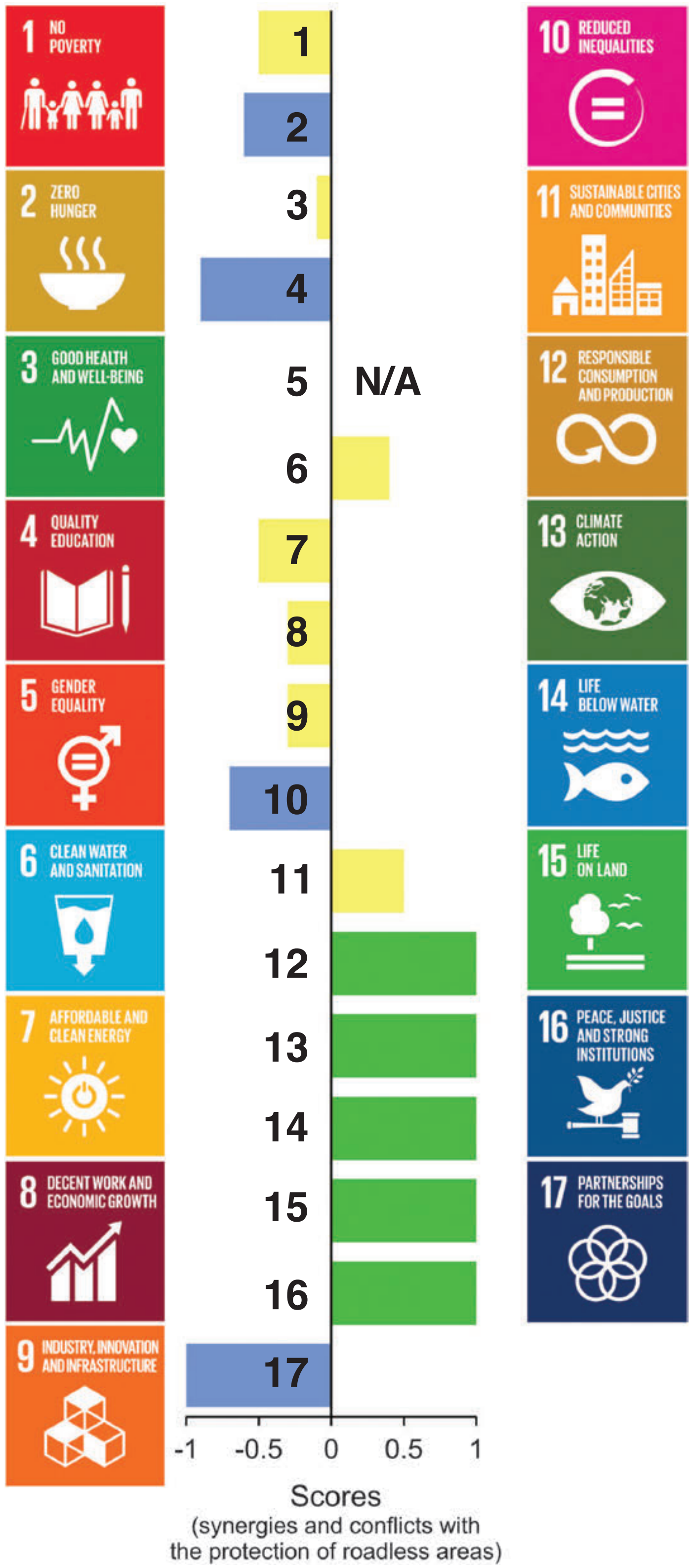

Fig. 4. Synergies and conflicts between conservation of roadless areas and the United Nations' Sustainable Development Goals. Scores <–0.5 (blue bars) indicate that conflicts with the goal prevail; scores between –0.5 and 0.5 (yellow) indicate a mixture of synergies and conflicts with the goal; and scores >0.5 (green) indicate prevailing synergies with the goal [for details, see table S11 (*8*)]. The scores reflect substantial imminent conflicts between various Sustainable Development Goals and conservation of roadless areas (table S11).

About one-third of the world's roadless areas have low EVIRA values. Patches with relatively low EVIRA values (ranging from 0 to 37; namely, <50% of the maximum value) account for 35% of the overall roadless area distribution, because most are small, fragmented, isolated, or otherwise heavily disturbed by humans. Some large tracts of roadless areas, such as arid lands in northern Africa or central Asia, occur in areas of sparse vegetation and low biodiversity and, thus, have low index values for ecosystem functionality (*9*) (Fig. 1B). High EVIRA values occur both in tropical and boreal forests. The relative conservation value of roadless areas is context-dependent. Comparatively small or moderately disturbed roadless areas have higher conservation importance in heavily roaded environments, such as most of Europe, the conterminous United States, and southern Canada.

Although the world's protected areas cover 14.2% of the terrestrial surface, only 9.3% of the overall expanse of roadless areas is within protected areas (all IUCN categories; Fig. 1C and table S8). There is no major difference in the coverage of roadless areas by strictly protected areas (IUCN categories I and II) versus the coverage of the overall landscape by strictly protected areas (3.8% roadless versus 4.2% overall). Only in North America, Australia, and Oceania are more than 6% of roadless areas under strict protection (table S8). If conservation efforts were to prioritize functional, ecologically important roadless areas, we would find a positive relation between strict protection coverage and EVIRA values of roadless areas. However, with the exception of Australia, this is not the case (Fig. 3 and table S9). Asia and Africa have particularly low protection coverage for roadless areas with high EVIRA values. For instance, we found gaps in the Asian tropical southeast, as well as in boreal biomes.

The recent Global Biodiversity Outlook (*11*) gives a bleak account of the progress made toward reaching the United Nations' biodiversity agenda as specified in the 20 Aichi Targets of the Convention on Biological Diversity (*12*). Governments have failed on several accounts to keep their use of natural resources well within safe ecological limits (target 4); to halt or at least halve the rate of habitat loss and substantially reduce the degradation and fragmentation of natural habitats (target 5); and to appropriately protect areas of particular importance for biodiversity and ecosystem services (target 11). To achieve global biodiversity targets, policies must explicitly acknowledge the factors underlying prior failures (*13*). Despite increasing scientific evidence for the negative impacts of roads on ecosystems, the current global conservation policy framework has largely ignored road impacts and road expansion. Furthermore, key policies on road infrastructure and development, such as the Cohesion Policy of the European Union, fail to take into account biodiversity.

In the much wider context of the United Nations' Sustainable Development Goals, conflicting interests can be seen between goals intended to safeguard biodiversity and those promoting economic development (*14*). We analyzed how roadless areas relate to the global conservation and sustainability agendas. As a transparent synthesis, we calculated simple scores of conflicts versus synergies of Sustainable Development Goals and Aichi Targets with the conservation of roadless areas (tables S10 and S11). Roads are explicitly mentioned in the Sustainable Development Goals only for their contribution to economic growth (goal 8), promoting further expansion into remote rural areas, and consideration is given neither to the environmental nor the social costs of road development. The resulting scores reflect substantial imminent conflicts (Fig. 4 and table S10); only in five Sustainable Development Goals do synergies with conservation of roadless

areas prevail, and four Sustainable Development Goals are predominantly in conflict with conservation of roadless areas. Maybe even more surprisingly, several of the Aichi Targets are ambivalent with respect to conserving roadless areas, rather than being in synergy entirely [six conflicting versus 11 synergistic targets (*8*); table S11].

There is an urgent need for a global strategy for the effective conservation, restoration, and monitoring of roadless areas and the ecosystems that they encompass. Governments should be encouraged to incorporate the protection of extensive roadless areas into relevant policies and other legal mechanisms, reexamine where road development conflicts with the protection of roadless areas, and avoid unnecessary and ecologically disastrous roads entirely. In addition, governments should consider road closure where doing so can promote the restoration of wildlife habitats and ecosystem functionality (*4*). Our global map of roadless areas represents a first step in this direction. During planning and evaluation of road projects, financial institutions, transport agencies, environmental nongovernmental organizations, and the engaged public should consider the identified roadless areas.

The conservation of roadless areas can be a key element in accomplishing the United Nations' Sustainable Development Goals. The extent and protection status of valuable roadless areas can serve as effective indicators to address several Sustainable Development Goals, particularly goal 15 ("Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss") and goal 9 ("Build resilient infrastructure, promote inclusive and sustainable industrialization and foster innovation"). Enshrined in the protection of roadless areas should be the objective to seek and develop alternative socioeconomic models that do not rely so heavily on road infrastructure. Similarly, governments should consider how roadless areas can support the Aichi Targets (see tables S10 and S11). For instance, the target of expanding protected areas to cover 17% of the world's terrestrial surface could include a representative proportion of roadless areas.

Although we acknowledge that access to transportation is a fundamental element of human well-being, impacts of road infrastructure require a fully integrated environmental and social cost-benefits approach (*15*). Still, under current conditions and policies, limiting road expansion into roadless areas may prove to be the most cost-effective and straightforward way of achieving strategically important global biodiversity and sustainability goals.

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SUPPLEMENTARY MATERIALS

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Materials and Methods
Figs. S1 to S11
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Data Sources
References (*16*–*180*)

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PLANT PATHOLOGY

Regulation of sugar transporter activity for antibacterial defense in *Arabidopsis*

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Microbial pathogens strategically acquire metabolites from their hosts during infection. Here we show that the host can intervene to prevent such metabolite loss to pathogens. Phosphorylation-dependent regulation of sugar transport protein 13 (STP13) is required for antibacterial defense in the plant *Arabidopsis thaliana*. STP13 physically associates with the flagellin receptor flagellin-sensitive 2 (FLS2) and its co-receptor BRASSINOSTEROID INSENSITIVE 1–associated receptor kinase 1 (BAK1). BAK1 phosphorylates STP13 at threonine 485, which enhances its monosaccharide uptake activity to compete with bacteria for extracellular sugars. Limiting the availability of extracellular sugar deprives bacteria of an energy source and restricts virulence factor delivery. Our results reveal that control of sugar uptake, managed by regulation of a host sugar transporter, is a defense strategy deployed against microbial infection. Competition for sugar thus shapes host-pathogen interactions.

Plants assimilate carbon into sugar by photosynthesis, and a broad spectrum of plant-interacting microbes exploit these host sugars (*1*, *2*). In *Arabidopsis*, pathogenic bacterial infection causes the leakage of sugars to the extracellular spaces (the apoplast) (*3*), a major site of colonization by plant-infecting bacteria. Although leakage may be a consequence of membrane disintegration during pathogen infection, some bacterial pathogens promote sugar efflux to the apoplast by manipulating host plant sugar transporters (*4*, *5*). Interference with sugar absorption by bacterial and fungal pathogens reduces their virulence, highlighting a general

Science
AAAS

A global map of roadless areas and their conservation status

Pierre L. Ibisch, Monika T. Hoffmann, Stefan Kreft, Guy Pe'er, Vassiliki Kati, Lisa Biber-Freudenberger, Dominick A. DellaSala, Mariana M. Vale, Peter R. Hobson and Nuria Selva (December 15, 2016)
Science **354** (6318), 1423-1427. [doi: 10.1126/science.aaf7166]

Editor's Summary

Too many roads

Roads have done much to help humanity spread across the planet and maintain global movement and trade. However, roads also damage wild areas and rapidly contribute to habitat degradation and species loss. Ibisch *et al*. cataloged the world's roads. Though most of the world is not covered by roads, it is fragmented by them, with only 7% of land patches created by roads being greater than 100 km^2. Furthermore, environmental protection of roadless areas is insufficient, which could lead to further degradation of the world's remaining wildernesses.

Science, this issue p. 1423

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ECOSPHERE

Toward a more ecologically informed view of severe forest fires

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Abstract. We use the historical presence of high-severity fire patches in mixed-conifer forests of the western United States to make several points that we hope will encourage development of a more ecologically informed view of severe wildland fire effects. First, many plant and animal species use, and have sometimes evolved to depend on, severely burned forest conditions for their persistence. Second, evidence from fire history studies also suggests that a complex mosaic of severely burned conifer patches was common historically in the West. Third, to maintain ecological integrity in forests born of mixed-severity fire, land managers will have to accept some severe fire and maintain the integrity of its aftermath. Lastly, public education messages surrounding fire could be modified so that people better understand and support management designed to maintain ecologically appropriate sizes and distributions of severe fire and the complex early-seral forest conditions it creates.

Key words: early succession; ecological integrity; ecological system; fire management; fire regime; forest resilience; forest restoration; severe fire; wildfire.

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Introduction

The spatiotemporal expression of fire events over time in any landscape produces a "fire regime" that influences ecosystem dynamics in that area (Heinselman 1981, Kilgore 1981). Even though the various characteristics of a fire regime (Table 1) are continuous in nature, the traditional approach in representing this variation has been to create a small number of discontinuous categories. Fire regimes in western North America, for example, are often classified into as few as three categories: (1) low-severity, (2) mixed-severity, and (3) high-severity or stand-replacement (Agee 1998, Brown 2000). Our attempt to categorize fire regimes is ". . . an oversimplification…for the convenience of humans" (Sugihara et al. 2006; p. 62), and has had the unfortunate consequence of minimizing rather than emphasizing variation in fire behavior and fire outcomes among vegetation types and across spatial scales (Morgan et al. 2014). In reality, relatively few forest types fit entirely within either of the two extremes—the low-severity (e.g., some interior ponderosa pine) or the stand-replacement (e.g., Rocky Mountain lodgepole pine) categories. Instead, as a simple analysis

Table 1. Characteristics or descriptors often used to describe disturbance regimes (from Keane 2013).

Disturbance Characteristic	Description	Example
Agent	Factor causing the disturbance	Fire is an agent that can kill trees
Source, Cause	Origin of the agent	Lightning is a source for wildland fire
Frequency	How often the disturbance occurs or its return time	Years since last fire (scale dependent)
Intensity	A description of the magnitude of the disturbance agent	Wildland fire heat output
Severity	The level of impact of the disturbance on the environment	Fuel consumption in wildland fires; change in biomass
Size	Spatial extent of the disturbance	Tree kill can occur in small patches or across entire landscapes
Pattern	Patch size distribution of disturbance effects; spatial heterogeneity of disturbance effects	Fire can burn large regions but weather and fuels can influence fire intensity and therefore the patchwork of tree mortality
Seasonality	Time of year of that disturbance occurs	Spring burn vs. fall burn
Duration	Length of time of that disturbances occur	Fires can burn for a day or for an entire summer
Interactions	Disturbance types may interact with each other, or with climate, vegetation and other landscape characteristics	Mountain pine beetles may create fuel complexes that facilitate or exclude wildland fire
Variability	The spatial and temporal variability of the above factors	Each of the above characteristics has variation associated with it

using LANDFIRE data (Rollins 2009, <http://www.landfire.gov>) reveals, roughly 85% of all forested lands within the western US fit within the mixed-severity category, which includes proportions of low-, moderate-, and high-severity (lethal to more than 70% of all trees) fire that vary widely across vegetation types and biophysical settings.

Agee (1993) captured the essence of this important idea in a graph depicting the proportion of low-, moderate-, and high-severity fire across the range of fire regimes (Fig. 1). Note that change from one fire regime to the next (movement along the x-axis) is accompanied not by the sudden appearance of a different fire severity, but by continuous changes in the proportions of each fire severity category. Thus, fire regimes blend imperceptibly into one another. More importantly, except for the two end points on the graph where the proportion of high-severity fire would be either 0% or 100%, most fire regimes consist of a mix of fire severities so, technically speaking, they fit best within a mixed-severity regime (Fig. 2). It is not the presence of a particular fire severity, but the proportion (and, presumably, the distribution and patch sizes) of each severity component that distinguishes regimes. Indeed, empirical

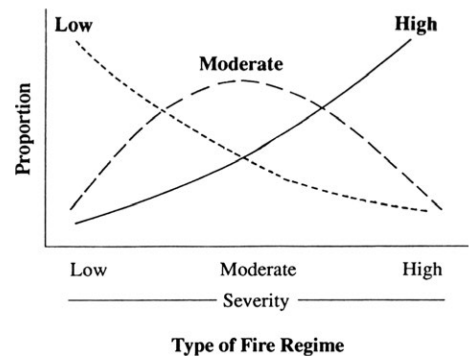

Fig. 1. This graph (from Agee 1993) illustrates that fire regimes are not characterized by the presence of only one kind of fire. Rather, it is the relative frequency of low-, moderate-, and high-severity fire in an average burn that varies among fire regimes.

data drawn from recent fires across the western United States between 1984 and 2008 (Fig. 3) reveal this continuous variation in proportions of different fire severities among fires. Thus, a more continuous view of fire regimes might be a better way to appreciate the infinite variability in fire behavior among forest types and geographic locations, and it might also promote a greater appreciation of severe fire as an integral

Fig. 2. Mixed-severity fires (fires that leave recognizable patches of low-severity, medium-severity, and high-severity effects) typify the majority of mixed-conifer forest systems in the western United States. The brown-needled and blackened areas harbor unique sets of plant and animal species found in no other forest conditions. This photograph of the North Fork of the Blackfoot River was taken 10 months after the 1988 Canyon Creek fire in Montana. Many fire-dependent plant and animal species were present in the more severely burned areas until they were helicopter logged, suggesting that unburned forests might be a better alternative for timber harvest.

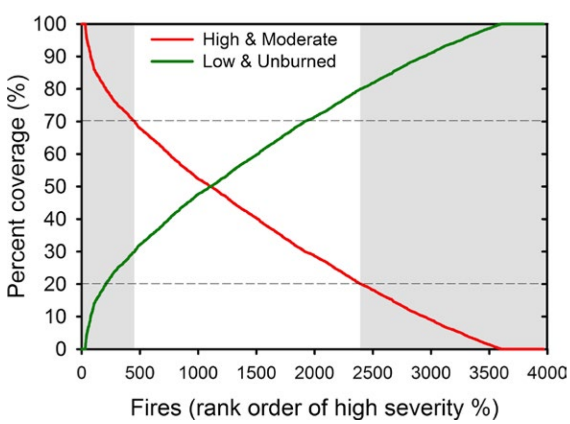

Fig. 3. The percent area within a fire perimeter that burned at low (green line) and at moderate to high (red line) severity is shown for a series of 3696 fires that burned in the western United States between 1984 and 2008 (after Belote 2015). The figure shows that the proportions of each severity category are continuously variable and that high-severity fire is a natural part of most forest fires in the West.

part of mixed- and high-severity conifer forest fire regimes.

Accordingly, we highlight the need for better information on the historical patterns and abundances of high-severity patches in different forest types. This is an important discussion because, even though our National Cohesive Wildland Fire Management Strategy (Wildland Fire Executive Council 2014) acknowledges that many fire regimes exist and that management needs to accommodate that variation and the variety of habitat such variation produces, contemporary fire management is focused heavily on the exclusion (prevention and suppression, collectively) or mitigation of severe fire. When either of those fails, management efforts seem to shift toward speeding the "recovery" of the forest after severe fire. With respect to the latter, there are repeated attempts to introduce legislation designed to expedite logging after fire (salvage logging). Although the removal of dead trees is justified near roads and structures for safety reasons, and although postfire logging can capture economic value of wood that would otherwise be lost, such logging has been shown to carry significant ecological costs (Hutto 2006, Lindenmayer and Noss 2006, Swanson et al. 2011, Lindenmayer and Cunningham 2013, DellaSala et al. 2015). The ecological benefits and necessity of severe fire (and its aftermath) has widespread implications for the flora and fauna that depend on the presence of burned forest conditions. Ecologically sound fire management includes land management designed to ensure the maintenance of ecologically appropriate mixes of fire severities within the forested landscapes of western North America while protecting homes and lives at the same time (Perry et al. 2011). An ecologically informed view of severe fire requires recognition that it is a natural component of many western conifer forests (Heinselman 1981, Arno 2000). Moreover, the severe-fire component must have been large enough and frequent enough to have favored the evolution of specialization by various plant and animal species to conditions that occur in the aftermath of severe fire. We offer the following points in an effort to better recognize and include severe fire as an integral part of fire management in mixed-conifer forest systems:

Severely burned forests create biologically unique conditions that cannot be created by other kinds of disturbances or through artificial means

Patterns in the habitat associations of plant and animal species can provide definitive evidence that severe fire plays an essential role in the ecology of mixed-conifer forests (Hutto et al. 2008). Specifically, if a plant or animal species occurs only in burned forest conditions created by severe fire events, then it cannot be using burned forest conditions merely opportunistically. Instead, the species must have evolved to depend on such conditions because it occurs rarely, if ever, in unburned habitat (Swanson et al. 2011, DellaSala et al. 2014). For example, some moss and lichen species are relatively restricted to severely burned forest conditions (Ahlgren and Ahlgren 1960), as are the fire morel mushroom (*Morchella elata*) and Bicknell's geranium (*Geranium bicknellii*) in forests throughout the West (Heinselman 1981, Pilz et al. 2004). The black-backed woodpecker (*Picoides arcticus*) is emblematic of a species that is relatively restricted to early successional conditions created by high-severity fire (Hutto 1995, Dixon and Saab 2000, Hoyt and Hannon 2002). Black-backed woodpeckers are attracted to postwildfire conditions because of the abundance of larvae of a number of wood-boring beetle species that are attracted to the fire-killed trees (Murphy and Lehnhausen 1998, Rota et al. 2015). Several of these beetle species are themselves relatively restricted to recently burned forests (Saint-Germain et al. 2004*a,b*, Boucher et al. 2012). Importantly, black-backed woodpeckers are significantly more likely to occur in the more severely burned portions of a mixed-severity fire (Hutto 2008, Latif et al. 2013). Although black-backed woodpeckers are known to occur outside severely burned forests on rare occasions, detailed study of survival and reproductive success shows that they exhibit growing populations only in forests recently burned by summer wildfires (Rota et al. 2014). The adaptations of thick bark, branch shedding, and serotiny in *Pinus* are thought to have evolved in response to a period of more intense crown fires in the mid-Cretaceous (He et al. 2012), and those adaptations also reflect the severe-fire backdrop against which pine, Douglas-fir, and larch are thought to thrive.

Many additional animal species, while not as narrowly restricted to burned forest conditions, clearly benefit from the burned forest conditions created by severe fires in mixed-conifer forests throughout the West (Hutto et al. 2015). For example, nest survival of white-headed woodpeckers is significantly higher in burned (wildfire) compared to unburned forest (Hollenbeck et al. 2011, Lorenz et al. 2015). In aquatic systems, severe fire events can rejuvenate stream habitats by causing large amounts of gravel, cobble, woody debris, and nutrients to be imported, resulting in increased production and aquatic insect emergence rates (Benda et al. 2003, Burton 2005, Malison and Baxter 2010, Ryan et al. 2011, Jackson et al. 2015). These changes can, in turn, affect food web dynamics in a way that results in higher growth rates in young trout, including young coastal cutthroat trout (*Oncorhynchus clarkii clarkii*) (Heck 2007) and rainbow trout (*Oncorhynchus mykiss*) (Rosenberger et al. 2011). Indeed, nonnative fish populations declined and native trout densities increased 3 yr after a severe fire in the Bitterroot River watershed, Montana, indicating that severe fire may help ensure ecological integrity of some western streams (Sestrich et al. 2011). In addition, native amphibians such as boreal toads (*Bufo boreas*) thrive in areas that burn severely (Dunham et al. 2007, Hossack and Corn 2007) and use severely burned areas more than expected due to chance (Hossack and Corn 2007, Guscio et al. 2008), as do some bat species (Buchalski et al. 2013).

These strong associations between organisms and severely burned forest patches suggests that many plant and animal species have evolved to rely on recurring severe wildfire events, and further indicates that severe fire events are a natural and important part of the fire regimes associated with many western mixed-conifer forest types. In other words, if one or more species occupy severely burned forests to the exclusion of other forest types (and if they do not tend to occupy forests disturbed through artificial means), then a severely burned forest would have to be considered natural, and would necessarily lie within the historical range of variation (Hutto et al. 2008). Moreover, a more intimate understanding of the biology of those plants and animals (e.g., knowledge of dispersal processes and patterns, foraging ecology, home-range sizes) can provide insight into the historical spatial scales at which severe fire operated across the broader landscape.

Fire history studies suggest that severe fire is an integral component of most fire regimes

In addition to the definitive evidence provided above, a growing body of fire history information points to the same conclusion—severe fire was historically, and is currently, an important component of many western conifer forest systems. At one end of the fire regime spectrum, conifer forests in the warmer, drier geographic areas in western North America are commonly characterized by frequent, low-severity fires that killed primarily juvenile trees historically, resulting in the maintenance of open pine forests with low densities of mature trees (Covington and Moore 1994*a,b*). Nevertheless, mixed and stand-replacement fires were possible even in these forest types after long inter-fire intervals, such as after an especially cold, wet period similar to what occurred during the Little Ice Age (Brown et al. 1999, Sherriff and Veblen 2007, Williams and Baker 2012, Odion et al. 2014, Hanson et al. 2015). At the other end of the fire regime spectrum, cooler, moister forest types, such as lodgepole pine forests, support fire regimes dominated by severe fire events (Brown and Smith 2000), although mixed- and low-severity fires are known to occur in these types as well (Barrett et al. 1991).

Between these two extremes lie the vast majority of mixed-conifer forest types in western North America. These include everything from the xeric, low-elevation, mixed ponderosa pine and Douglas-fir forest types to mesic, high-elevation, spruce-fir forest types. Unlike the forest types that are dominated by either the absence or presence of severe fire, mixed-conifer forests are best characterized by fire regimes of variable, or mixed severity (see Baker 2009: fig. 7.1), which means that the presence of sizable proportions of the three classes of fire severity characterize the fires that burn in those forest systems (Sherriff and Veblen 2006, 2007, Baker et al. 2007, Hessburg et al. 2007, Klenner et al. 2008, Perry et al. 2011, Schoennagel et al. 2011). Importantly, extreme weather (e.g., high temperature, low humidity, high wind speed) rather than quantity of woody fuels often exerts the greatest influence on fire severity and extent across that broad range of mixed-conifer forest types (Johnson et al. 2003, Schoennagel et al. 2004, Lydersen et al. 2014, Williams et al. 2015). This means that, in contrast with the situation in low-elevation or xeric-type ponderosa pine forests in some areas of the southwestern United States (Keane et al. 2008), the amount of high-severity fire in other mixed-conifer forest types is less likely to have departed significantly from historical ranges of variability, even though those forests may have experienced measurable twentieth century changes in fuels due to fire exclusion, timber harvest, and cattle grazing (e.g., Baker et al. 2007, Dillon et al. 2011, Marlon et al. 2012, Miller et al. 2012, Odion et al. 2014, Sherriff et al. 2014). We recognize the lack of relevant historical information on landscape-level distributions and spatial scales of different classes of fire severity for many forest types and regions, but severely burned forest patches have probably always occurred naturally, even in pure ponderosa pine forests of the Southwest, as Cooper (1961) and Weaver (1943) described long ago. We also know that, at least throughout the northern half of the western United States, the extent of severe-fire patches must have been both substantial enough in area and frequent enough to support those plant (e.g., lodgepole pine) and animal (e.g., wood-boring beetle and woodpecker) species that evolved to depend on severe fire itself or on the resulting severely burned forest conditions.

Maintaining ecological integrity means accommodating a broad spectrum of fire severities, including severe fire and its aftermath, in most mixed-conifer forests

We have now established two important facts: severe fire (moderate-to-high burn severity) is a natural agent of disturbance in many mixed-conifer forest types, and such fire is thought to be ecologically necessary for the presence or success of many plant and animal species. These two facts make it clear that management to maintain the ecological integrity of any ecosystem that harbors species that depend on severe fire as a disturbance agent will have to integrate severe fire and its effects into management goals. Moreover, if we better considered distribution patterns, home range sizes, movement patterns, and other animal adaptations that reflect the environment within which they evolved (e.g., Hutto et al. 2008), we could gain considerable insight into historical spatial scales under which severe fire operated as well. We are not questioning or attempting to discredit the evidence that some forest systems were historically dominated by low-severity fire; rather, we are encouraging land managers to also pay close attention to maintaining amounts and distributions of higher severity fire consistent with ecological integrity in our western mixed-conifer forests. The current science, management, and policy challenge for ecosystem managers is to estimate and incorporate amounts of low-, moderate-, and high-severity fire in a manner that maintains ecological integrity (Hessburg et al. 2007, Perry et al. 2011, Baker 2015).

While many fire ecologists understand the importance of more severe fire in forest ecosystems, politicians and the public at large have yet to reach the same understanding. Recent increases in the amount of forested area burned by wildfire over the past three decades in western North American forests (Westerling et al. 2006, Dennison et al. 2014) signaling what many believe to be the emergence of a new age of megafires (Attiwill and Binkley 2013), has created increased movement toward pre and postfire land management activities designed to reduce fire severity, mimic fire effects without the use of fire, or speed the recovery of a forest after fire. These activities may provide some societal benefits, but they can have real costs in terms of the way they negatively affect the ecological integrity of mixed-conifer forests born of mixed-severity fire. Removed from locations that pose a clear and immediate threat to human lives and property, the ecological costs associated with forest thinning may outweigh stated benefits by large margins. We highlight two types of land management (beyond fire suppression itself) that can have significant negative effects on fire-dependent species and, therefore, can interfere with our ability to maintain the ecological integrity of fire-dependent conifer forests: prefire fuel treatments and postfire salvage logging.

Prefire harvest treatments

We know a great deal about the effects of fuel treatments and restoration harvests on forest structure and vegetation recovery, but we know little about the ecological effects of such treatments on the prefire responses of most plant and animal species, and virtually nothing about postfire responses of the most fire-dependent plant and animal species after a treatment subsequently burns in a wildfire. This is because such treatments are rarely accompanied by "ecological effects monitoring," which, in contrast with implementation monitoring (evaluating whether a management activity was implemented) and effectiveness monitoring (evaluating whether the management activity achieved the stated goal), is specifically designed to address whether there are unforeseen negative ecological consequences of a management treatment (Hutto and Belote 2013).

Fuel treatments designed to restore fire-prone ecosystems should do so in the proper fire regime context; more specifically, they should produce appropriate postfire plant and animal responses when fire returns to the forest. Thus, treatments appropriate for dry forests that were historically maintained by a low-severity fire regime may be inappropriate for forests maintained by a mixed-severity fire regime. One serious negative consequence of canopy fuel reduction in forests that evolved with mixed-severity fire could be that fire-dependent species requiring high densities of large standing-dead trees created by the severe-fire component may not recruit after a subsequent fire. For example, the fire-dependent black-backed woodpecker was found to be even less abundant in mixed-conifer forests that were thinned before fire than in the same forest types logged after fire, even though the two pathways support similar standing dead tree densities. This is probably because birds rarely colonize thinned forests that burn, but they still make the best of a bad situation when trees are removed after they have already colonized a densely stocked, severely burned forest (Hutto 2008). Recent research on postfire soil conditions shows that soil C and N response following wildfire also depends on whether there have been fuel treatments, so the assessment of fuel treatment effects needs to include postfire response and not simply postharvest response (Homann et al. 2015). It has been suggested (e.g., Franklin and Johnson 2014) that variable-retention harvests could be designed to emulate early-seral conditions following natural disturbance events in forests born of mixed-severity fire, thereby avoiding the negative consequences associated with other tree harvesting methods. Unfortunately, that strategy is unlikely to satisfy the needs of those fire-dependent animal species that require high densities of fire-killed trees immediately following severe fire (Schieck and Song 2006, Hutto 2008, Reidy et al. 2014).

Postfire salvage logging

Salvage logging after fire is intended to recover economic value of timber that would otherwise be lost, to ensure human safety, and to reduce the risk of future fires. Unfortunately, salvage harvesting activities undermine the ecosystem benefits associated with fire (Lindenmayer et al. 2004, Lindenmayer and Noss 2006, Swanson et al. 2011). For example, postfire salvage logging removes dead, dying, or weakened trees, but those are precisely the resources that provide nest sites and an abundance of food in the form of beetle larvae and bark surface insects (Hutto and Gallo 2006, Koivula and Schmiegelow 2007, Saab et al. 2007, 2009, Cahall and Hayes 2009). No fire-dependent bird species has ever been shown to benefit from salvage logging (Hutto 2006, Hanson and North 2008). The ecological effects of salvage logging on aquatic ecosystems are also largely negative (Karr et al. 2004). In fact, the demonstrated negative ecological effects associated with postfire salvage logging are probably the most consistent and dramatic of any wildlife management effects ever documented for any kind of forest management activity (Hutto 2006). Therefore, because the National Forest Management Act and other legal mandates require public land managers to maintain the integrity of the larger ecological system, burned forests should perhaps be given special consideration compared with green-tree forests. Specifically, they could receive a low priority ranking when it comes to timber harvest decisions (with the obvious exception of small harvests associated with roads and other areas where safety or infrastructure are legitimate concerns). Timber can be harvested from many green-tree forests in a manner that imposes relatively little ecological cost in comparison with the costs associated with logging in burned forest (Lindenmayer and Cunningham 2013).

How do we move toward a more ecologically informed view of forest fires?

The ecological costs associated with some of the more commonly employed pre and postfire management activities in the western United States probably increase substantially as one moves from the low-elevation or xeric ponderosa pine or woodland forest types, where trees were widely spaced and severe fire historically played a spatially restricted role, to the broad array of more densely stocked mixed-conifer forest types, where severe fire historically played a major role. Therefore, a thorough understanding of the historical fire regime associated with any particular vegetation type or land area (as determined from multiple lines of evidence concerning regionally specific fire history) is critically important for land managers who concern themselves with the issues of wildfire risk, ecological restoration, or maintenance of the diversity of native species (Schoennagel and Nelson 2011). More specifically, quantification of appropriate fire rotations and proportions of low-, moderate-, and high-severity fire for any given forest landscape is critical for enlightened land management. For example, in some xeric ponderosa pine forest types, ecosystem restoration activities designed to decrease the severity of wildfire may be ecologically appropriate. The same management activities are not likely to be ecologically appropriate in many mixed-conifer forests, however, because key indicator species evolved to depend on significant amounts of severe fire in those forest types (Schoennagel et al. 2004, Hutto 2008, Klenner et al. 2008, Baker 2012, 2015, Williams and Baker 2012, Odion et al. 2014).

Land and fire managers are now facing future fires that many hypothesize will become larger and contain larger proportions of more severely burned patches under warming climate conditions (Rocca et al. 2014). Problems associated with climate change, however, must be solved through efforts directed toward the causes of climate change and not toward the symptoms of climate change. Any perceived problem with future changes in fire behavior cannot be solved by redoubling our effort to treat this particular climate change symptom by installing widespread fuel treatments that do nothing to stop the warming trend, and do little to reduce the extent or severity of weather-driven fires (Gedalof et al. 2005). Therefore, fuel management efforts to reduce undesirable effects of wildfires outside the xeric ponderosa pine forest types could be more strategically directed toward creating fire-safe communities (Calkin et al. 2014, Kennedy and Johnson 2014). A management emphasis directed toward altering conditions in and immediately adjacent to human communities is very different from an emphasis directed toward treating massive amounts of fuel on more remote public lands. Fuel treatment efforts more distant from human communities may carry the negative ecological consequences we outlined earlier and do little to stop or mitigate the effects of fires that are increasingly weather driven (Rhodes and Baker 2008, Franklin et al. 2014, Moritz et al. 2014, Odion et al. 2014).

Public land managers face significant challenges balancing the threats posed by severe fire with legal mandates to conserve wildlife habitat for plant and animal species that are positively associated with recently burned forests. Nevertheless, land managers who wish to maintain biodiversity must find a way to embrace a fire-use plan that allows for the presence of all fire severities in places where a historical mixed-severity fire regime creates conditions needed by native species while protecting homes and lives at the same time. This balancing act can be best performed by managing fire along a continuum that spans from aggressive prevention and suppression near designated human settlement areas to active "ecological fire management" (Ingalsbee 2015) in places farther removed from such areas. This could not only save considerable dollars in fire-fighting by restricting such activity to near settlements (Ingalsbee and Raja 2015), but it would serve to retain (in the absence of salvage logging, of course) the ecologically important disturbance process over most of our public land while at the same time reducing the potential for firefighter fatalities (Moritz et al. 2014). Severe fire is not ecologically appropriate everywhere, of course, but the potential ecological costs associated with prefire fuels reduction, fire suppression, and postfire harvest activity in forests born of mixed-severity fire need to considered much more seriously if we want to maintain those species and processes that occur only where dense, mature forests are periodically allowed to burn severely, as they have for millennia.

Another integral part of moving toward an ecologically informed perspective of forest fire involves getting the public, politicians, and policy-makers to better recognize and appreciate the critical role that severe fire plays in many forest systems. This has been difficult, and this difficulty has been exacerbated by public messages about severe fire that are uniformly negative. Progress toward allowing fires to burn is difficult unless the public begins to receive a message that differs markedly from the message that Smokey the Bear is sending them now. Fires in our wildlands are fundamentally natural and beneficial, so we must learn to live in a way that allows naturally occurring fires, including severe fires, to burn while minimizing risk to human property and lives (Calkin et al. 2014). That is a vastly different message from one that says severe fires are fundamentally bad and that we have to do everything in our power to prevent and suppress them, or from one that says severely burned forests are places where we should expedite efforts to capture residual economic value through "salvage" logging. We challenge ecologists and managers to pay greater attention to the degree of variation in fire regimes within mixed-conifer forests and to recognize that prefire thinning and postfire "restoration" activities may not always be compatible with maintenance of the ecological integrity of conifer forests that depend on complex mixed-severity fire disturbance.

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Research Article

A Case Study Comparison of LANDFIRE Fuel Loading and Emissions Generation on a Mixed Conifer Forest in Northern Idaho, USA

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ABSTRACT

The use of fire as a land management tool is well recognized for its ecological benefits in many natural systems. To continue to use fire while complying with air quality regulations, land managers are often tasked with modeling emissions from fire during the planning process. To populate such models, the Landscape Fire and Resource Management Planning Tools (LANDFIRE) program has developed raster layers representing vegetation and fuels throughout the United States; however, there are limited studies available comparing LANDFIRE spatially distributed fuel loading data with measured fuel loading data. This study helps address that knowledge gap by evaluating two LANDFIRE fuel loading raster options—Fuels Characteristic Classification System (LANDFIRE-FCCS) and Fuel Loading Model (LANDFIRE-FLM) layers—with measured fuel loadings for a 20 000 ha mixed

RESUMEN

El uso del fuego como herramienta de manejo de tierras es bien reconocido por sus beneficios ecológicos en varios ecosistemas naturales. Para continuar con el uso del fuego y a su vez cumplir con las regulaciones referidas a la calidad del aire, los gestores de tierras deben frecuentemente cumplir con tareas de modelado de emisiones durante el proceso de planificación de las quemas. Para alimentar tales modelos, el programa denominado *Landscape Fire and Resource Management Planning Tools* (LANDFIRE) ha desarrollado capas raster, que representan vegetación y combustibles a lo largo de todos los EEUU; desde luego, son limitados los estudios disponibles que puedan comparar los datos de carga de combustibles espacialmente distribuidos derivados del LANDFIRE, con datos similares producto de mediciones de carga de combustible en el terreno. Este estudio ayuda a dilucidar este vacío en el conocimiento mediante la evaluación de carga de combustible usando dos opciones del programa LANDFIRE—el *Fuels Characteristic Classification System* (LANDFIRE-FCCS) y el *Fuel Loading Model* (LANDFIRE-FLM) *layers*—

conifer study area in northern Idaho, USA. Fuel loadings are compared, and then placed into two emissions models—the First Order Fire Effects Model (FOFEM) and Consume—for a subsequent comparison of consumption and emissions results. The LANDFIRE-FCCS layer showed 200%* higher duff loadings relative to measured loadings. These led to 23% higher total mean total fuel consumption and emissions when modeled in FOFEM. The LANDFIRE-FLM layer showed lower loadings for total surface fuels relative to measured data, especially in the case of coarse woody debris, which in turn led to 51% lower mean total consumption and emissions when modeled in FOFEM. When the comparison was repeated using Consume model outputs, LANDFIRE-FLM consumption was 59% lower relative to that on the measured plots, with 58% lower modeled emissions. Although both LANDFIRE and measured fuel loadings fell within the ranges observed by other researchers in US mixed conifer ecosystems, variation within the fuel loadings for all sources was high, and the differences in fuel loadings led to significant differences in consumption and emissions depending upon the data and model chosen. The results of this case study are consistent with those of other researchers, and indicate that supplementing LANDFIRE-represented data with locally measured data, especially for duff and coarse woody debris, will produce more accurate emissions results relative to using unaltered LANDFIRE-FCCS or LANDFIRE-FLM fuel loadings. Accurate emissions models will aid

comparados con la medición de la carga para 20 000 ha de un área de bosques mixtos de coníferas en el norte de Idaho, EEUU. Las cargas de combustibles fueron comparadas, y luego ubicadas dentro de dos modelos—el *First Order Fire Effects Model* (FOFEM) y el *Consume*—para su subsecuente comparación de los resultados del consumo de combustibles y sus emisiones. El LANDFIRE-FCCS mostró una estimación 200%* superior en la carga del mantillo comparado con la carga medida a campo. Esto llevó a un valor 23% más alto en la media total de consumo y emisiones del combustible cuando fue modelado mediante el modelo FOFEM. El modelo LANDFIRE-FLM layer mostró menores cargas para combustibles de superficie relativo a datos medidos a campo, especialmente en el caso de restos de combustible leñoso grueso (*coarse woody debris*), que a su vez llevó a un 51% menos en el consumo y emisiones promedio cuando fueron modeladas por el modelo FOFEM. Cuando la comparación fue repetida usado el *Consume model outputs*, el consumo estimado por el LANDFIRE-FLM fue un 59% menor en relación a lo determinado en las parcelas medidas, con un 58% menos que las emisiones modeladas. Aunque ambos modelos de LANDFIRE y las cargas efectivamente medidas se ubican dentro de los rangos observados por otros investigadores en los ecosistemas mixtos de coníferas de los EEUU, la variación dentro de las cargas de combustible determinadas por las distintas fuentes fue alta, y las diferencias en carga de combustible llevan a diferencias significativas en consumo y emisiones, dependiendo éstos del modelo elegido. Los resultados de este estudio de caso son consistentes con aquellos obtenidos por otros investigadores, e indican que suplementando datos de LANDFIRE con datos locales obtenidos de mediciones a campo, especialmente para el mantillo y restos de combustible leñoso grueso, producirá resultados de consumo y emisiones más precisos que aquellos que usan solamente datos de carga provistos por LANDFIRE-FCCS o LANDFIRE-FLM. Los modelos de emisiones preci-

*Originally reported as 300%; corrected to 200% on 28 March 2018.

in representing emissions and complying with air quality regulations, thus ensuring the continued use of fire in wildland management.

sos ayudarán a representar emisiones y a cumplir con las regulaciones sobre la calidad del aire, de manera de asegurar el uso continuado del fuego en el manejo de áreas naturales.

Keywords: coarse woody debris, duff, fire effects, fuel loading models, Fuels Characterization Classification System, LANDFIRE

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INTRODUCTION

The use of fire as a land management tool is widely recognized for its ecological benefits, and as a historic disturbance that has driven succession across many ecosystems (Agee 1996, Hardy and Arno 1996, Rothman 2005). While fire science and policy has advanced in the last 50 years to better allow for the use of fire in managing wildlands (van Wagtendonk 2007), increasingly stringent air quality regulations (US EPA 1990, Hardy *et al.* 2001, US EPA 2015,) and an increased awareness of the health impacts from smoke (Liu *et al.* 2015) can make the use of fire as a management tool difficult. In a recent United States survey, prescribed fire practitioners expressed that smoke and air quality issues are the third greatest impediment to prescribed burning, following low work capacity and unfavorable weather conditions (Melvin 2012). To continue using fire as a management tool, land managers must plan to meet management objectives, while also limiting the impact of smoke on public health and keeping smoke levels within regulatory thresholds (NWCG 2014). Such planning may often require the use of models to determine the quantity of emissions generated by fire; these models require many pieces of information, including expected fire size, fuel loading characteristics, and fuel consumption. Of these, fuel loading has been identified as the most critical step in obtaining accurate smoke predictions (Drury *et al.* 2014). Unfortunately, in many areas there may be little or no measured data on fuel loading; this creates a major difficulty in estimating fuel consumed and emissions produced.

To address the lack of fuel loading information in planning, geospatial Fire Effects Fuel Model (FEFM) layers developed by the Landscape Fire and Resource Management Planning Tools (LANDFIRE) program are often used. LANDFIRE data layers were developed for the contiguous United States, Alaska, and Hawaii to provide consistent geospatial data describing the vegetation type, structure, fuel loading, and disturbances, regardless of land ownership boundaries (Rollins 2009). LANDFIRE is principally intended to inform management and planning decisions made by land management agencies in the United States. It is also the only resource available that provides the geospatial information outlined above across as wide an area as the continental US. To populate models for smoke production, LANDFIRE FEFMs describe fuel loading for duff, litter, woody fuels from timelag size classes ranging from one hour (≤0.6 cm) to 1000 hours (≥7.62 cm), and live herb and shrub loading. Currently, there are two FEFM choices available through LANDFIRE: one represents fuel loading based on the Fuel Loading Model (FLM) categories developed by Lutes *et al.* (2009), and the other based on Fuels Characteristics Classification System (FCCS) categories developed by Ottmar *et al.* (2007). Both methods are derived from extensive measured datasets; however, FCCS is stratified to represent fuel loading by vegetation type (Ottmar *et al.* 2007), while FLM is stratified to represent fuel loadings by their potential fire effects (Lutes *et al.* 2009). The two LANDFIRE FEFMs are different not only in how they stratify fuels, but also in their reported fuel loadings.

There have been few studies that detail the differences between these two LANDFIRE FEFMs. One study evaluated their mapping performance across the western United States (Keane *et al.* 2013), and another compared their loadings and resulting emissions as part of a broader comparison of factors affecting smoke predictions in Washington, USA (Drury *et al.* 2014). When Keane *et al.* (2013) compared fuel loading and mapping accuracy of FCCS and FLM LANDFIRE layers throughout the western United States to data from the Forest Inventory and Analysis (FIA) program, they found poor correlations between FIA and LANDFIRE represented loadings, mainly due to the high variability in fuel loadings. Drury *et al.* (2014) compared FLM and FCCS FEFM data with other local datasets and found the landscape fuel loadings to range from 2.7 million Mg to 8.8 million Mg for their research area in Washington, USA, depending on which fuel loading dataset they used.

Studies such as these are extremely valuable for documenting the complexity and variation within fuel loading data, and identifying the importance and challenges of applying FEFM fuels data to model emissions. Our study builds on the few evaluations of LANDFIRE FEFMs to date by comparing FEFM surface fuel loading with measured fuel loadings, and using these loadings in two popular consumption and emissions models—the First Order Fire Effects Model (FOFEM) and Consume—to compare the resulting differences in fuel consumption and emissions production, while holding the site and environmental conditions constant. This provides insight into the degree of fuel loading differences possible at smaller scales relative to the national or sub-regional scales that LANDFIRE was developed to represent. Yet this 20 000 ha area is large enough to fall within the range of fire management units that land managers are tasked to manage (USDI NPS 2005, USDA FS 2008). We compared duff, litter, herb, shrub, and woody fuel loadings measured in forest inventory plots to those shown on both LANDFIRE Fuel Loading Models (LANDFIRE-FLM) and LANDFIRE Fuels Characterization Classification System (LANDFIRE-FCCS) maps. Subsequent differences in modeled consumption and emissions using FOFEM and Consume are reported.

METHODS

Study Area

To evaluate potential differences in predicted fuel loadings and fire effects, we selected a 20 000 ha study area centered on Moscow Mountain in Latah County, Idaho, USA (Figure 1). The mountain lies in the Palouse Range of northern Idaho, with elevations ranging from 770 m to 1516 m. Moscow Mountain is dominated by mixed conifer forest tree species including ponderosa pine (*Pinus ponderosa* C. Lawson var. *scopulorum* Engelm.), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco var. *glauca* [Beissn.] Franco), grand fir (*Abies grandis* [Douglas ex D. Don] Lindl.), western red cedar (*Thuja plicata* Donn ex D. Don), western hemlock (*Tsuga heterophylla* [Raf.] Sarg.), and western larch (*Larix occidentalis* Nutt.). Ponderosa pine and Douglas-fir habitat types occur on the xeric southern and western aspects, grand fir and cedar-hemlock habitat types occur on the mesic northern and eastern aspects (Cooper *et al.* 1991). The majority of the land is owned by private timber companies, private non-commercial landowners, and public land holdings. Recent disturbances recorded between 2003 and 2009 were predominantly the result of forest man-

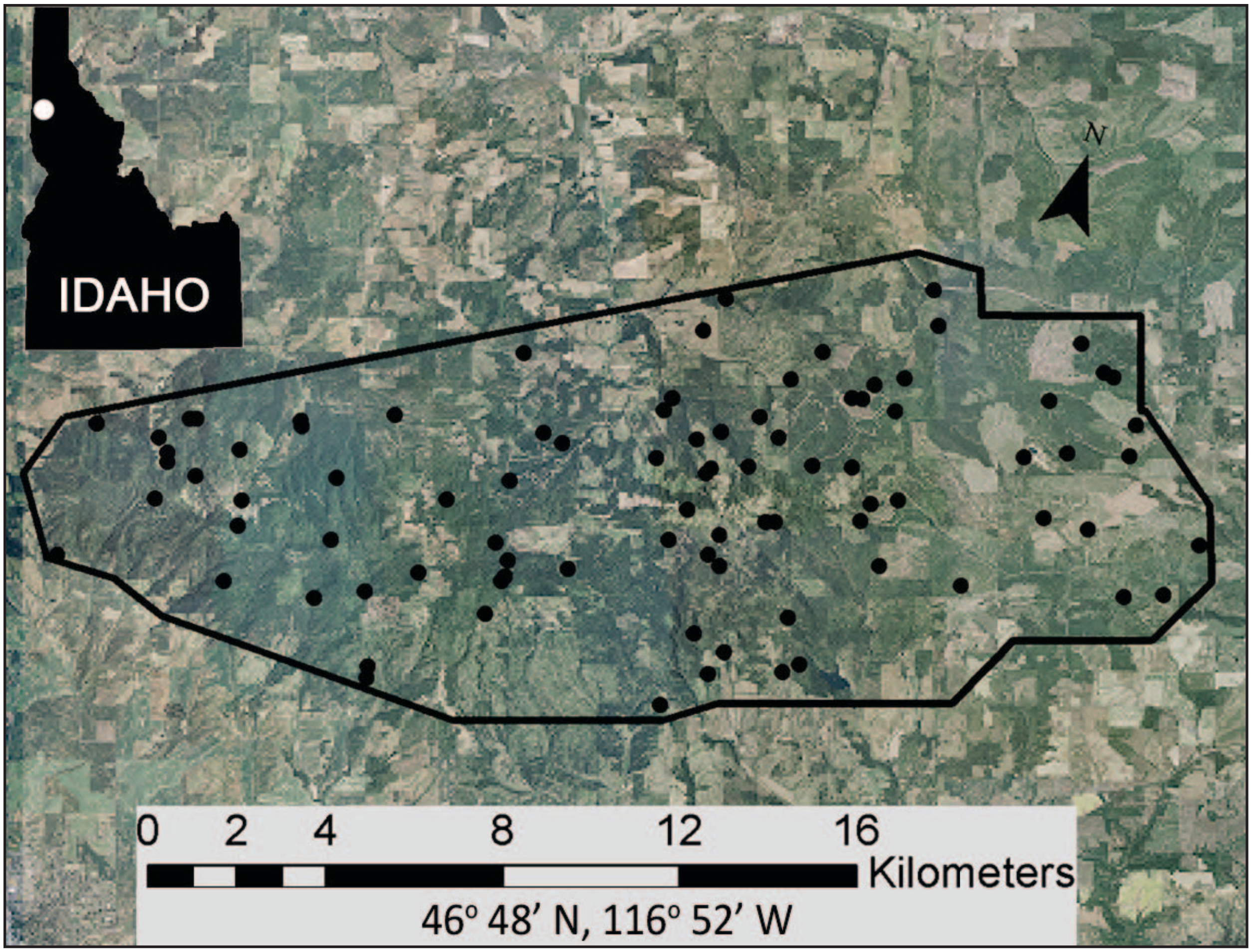

Figure 1. 2009 orthoimagery of the Moscow Mountain, Idaho, USA, study area (outlined) from the United States Geological Survey. The white dot in the inset is the study area. Plot locations are indicated with black dots.

agement practices including thinning, timber harvesting, and prescribed burning (Hudak *et al.* 2012). These activities have resulted in a forest with varying stand ages and structures that occur over a variety of biophysical settings (Falkowski *et al.* 2009, Martinuzzi *et al.* 2009, Hudak *et al.* 2012).

Plot Fuel Loadings

Plot data used in this study were collected in 2009, with information on plot placement and methodologies described in detail in Hudak *et al.* (2012). Following a stratified random sampling design of the study area, 0.04 ha fixed-radius field plots were placed randomly within strata based on elevation, slope, aspect, and percent forest cover. Plots that randomly fell within agricultural areas were subsequently excluded, leaving 87 forested plots for this analysis. Within each plot, duff; litter; coarse woody debris (CWD) in the ≥1000 hour (≥7.62 cm) size class; and fine woody debris in one hour (<0.635 cm), ten hour (0.635 cm to 2.54 cm), and 100 hour (2.54 cm to 7.62 cm) size classes were measured and loading was determined as described by Hudak *et al.* (2009), briefly summarized as follows: fuel loading was determined using two parallel 15 m Brown's transects (Brown 1974) centered 2.5 m upslope and downslope from plot center. On each transect, one hour and ten hour fuels were tallied over a 1.8 m segment, 100 h fuels over a 4.6 m segment, and 1000 h fuels over the entire length of both transects. Shrub and herbaceous cover were estimated ocularly and translated to loadings using equations from Brown (1981) and Smith and Brand (1983). Duff and litter depths were measured once at a set distance along each transect (Brown 1981), and loading was derived from relationships presented in Brown *et al.* (1982) with bulk densities from Woodall and Monleon (2008).

LANDFIRE Fire Effects Fuel Model Loadings

LANDFIRE FEFM map layers are available for both FCCS and FLM fuel classification systems. The FCCS system is composed of fuel loading data organized by vegetation type; each vegetation type is represented by loadings derived from field data collected from that vegetation type (Ottmar *et al.* 2007). FLM fuel loadings are the result of several field-collected datasets, which are grouped into statistically distinct groups based on fuel loading and modeled fire effects (i.e., emissions and soil heating; Lutes *et al.* 2009). In-depth comparisons of these approaches have been addressed by Keane (2013).

For this study, we compared LANDFIRE Refresh 2008 FEFMs to measured fuel loadings. LANDFIRE-FCCS and LANDFIRE-FLM layers were generated using different methodologies. LANDFIRE-FCCS layers were derived by matching FCCS fuelbeds to LANDFIRE vegetation communities (Comer *et al.* 2003) and vegetation type (McKenzie *et al.* 2012, LANDFIRE Team 2014*a*). LANDFIRE-FLMs were derived by a series of database queries that matched LANDFIRE data to the appropriate FLMs (Hann *et al.* 2012). More specifically, Forest Inventory and Analysis (FIA) data (Woudenberg *et al.* 2010) were keyed to FLMs (Lutes *et al.* 2009) and these FLMs were systematically matched to LANDFIRE vegetation types and cover. We should note that the scope of our study focuses on the surface fuel loadings represented in LANDFIRE map layers, not the FCCS and FLM fuel classification systems that the layers are intended to represent.

Generating Emissions within FOFEM and Consume

Consumption and emissions were generated using two common fire effects models: Consume version 4.2 (FERA Team 2014) and the Fire Order Fire Effects Model (FOFEM) version 6.0 (Lutes 2012). Consume calculates consumption and emissions based on empirical algorithms from many studies (Prichard *et al.* 2005). The FOFEM model generates consumption based on equations from the BURNUP model (Albini and Reinhardt 1997) and emission factors from Ward *et al.* (1993). Evaluating results in both models is important as FOFEM and Consume are both commonly used in fire management and are integrated into planning tools such as the Interagency Fuels Treatment Decision Support System (IFTDSS 2015). Consume is also integrated into the BlueSky Framework that is used for emissions calculations (AirFire 2015). For this study, we included the major compounds emitted by wildland fire that could be of concern for reasons of human health effects, regulatory impacts, or greenhouse gas emissions: carbon dioxide (CO_2), carbon monoxide (CO), methane (CH_4), and particulate matter 2.5 μm and 10 μm ($PM_{2.5}$ and PM_{10}). Nitrogen oxides (NO_x) and sulfur dioxide (SO_2) were also modeled using only FOFEM, and non-methane hydrocarbons (NMHC) were modeled using only Consume as these options are specific to each model. To parameterize these models we used the values in Table 1 to simulate summer fire conditions under which past fires in the region have ignited (McDonough 2003).

Table 1. Environmental parameters used to populate FOFEM and Consume under default 'Low' moisture conditions to simulate an early summer fire.

Parameter	Input
Moistures	
Duff	40%
10 hour	10%
CWD	15%
Soil	10%
Fuel type	Natural
Region	Interior West
Season	Summer

Statistical Comparison of Fuel Loadings

All analyses were conducted using R Statistical Software (R-Project 2013). We initially tested fuel loading differences using Bartlett's test for equal variance (Bartlett 1937). This indicated that the data did not meet the assumption of homoscedasticity required for parametric regression analysis. Therefore, we used non-parametric statistical methods. Analysis of variance was chosen and performed using the Anova test from the "car" package (Fox *et al.* 2014) as this version implemented the test using heteroscedasticity-corrected coefficient covariance matrices. If a significant difference was detected, further analysis was conducted with the Dunnett-Tukey-Kramer pairwise multiple comparison test adjusted for unequal variances and unequal sample sizes (Dunnet 1980) using the DTK package (Lau 2013) at the alpha = 0.05 significance level. This method was used to compare fuel loadings, consumption, and emissions. To examine the influence of different fuels on the total emissions produced, we used Hoffman and Gardner's Importance Index, a ratio of variances between total emissions generated and each individual fuel component (Hoffman and Gardner 1983 , Hamby 1994). Values close to one indicate higher significance than values closer to zero.

RESULTS

Fuel Loadings

In comparing LANDFIRE fuel loadings with measured fuel loadings, all fuel components differed at the alpha = 0.05 significance level with the exception of shrubs (Table 2, Figure 2). LANDFIRE-FCCS loadings over-represented duff and herbs; under-represented litter, 10 h, and 100 h fuels; and did not differ for 1 h fuels or CWD. LANDFIRE-FLM under-represented duff, litter, fine (1 h, 10 h, and 100 h), and CWD fuel loadings; over-represented herb loadings; but duff loading did not

Table 2. Mean fuel loads (Mg ha^{-1} and SD in parentheses) on measured plots and as modeled by LANDFIRE-FCCS and LANDFIRE-FLM. Asterisks indicate statistically significant difference relative to measured loading data at the $P < 0.05$ significance level.

	Mean plot loading		
Fuel	**Measured**	**LANDFIRE-FCCS**	**LANDFIRE-FLM**
Duff	10.55 (10.20)	31.89 (17.80)*	7.76 (12.19)
Litter	5.86 (4.13)	4.199 (1.37)*	3.66 (3.40)*
1 h	0.65 (0.47)	0.81 (0.46)	0.50 (0.32)*
10 h	2.57 (2.19)	1.85 (1.11)*	1.65 (1.13)*
100 h	4.98 (5.20)	2.47 (3.41)*	1.94 (1.66)*
CWD	20.087 (23.33)	18.45 (16.38)	2.75 (4.04)*
Herb	0.46 (0.28)	0.68 (0.76)*	0.73 (0.76)*
Shrub	1.179 (3.08)	1.36 (1.51)	3.65 (10.60)
Total fuel	46.26 (32.49)	61.63 (34.81)*	22.64 (21.16)*

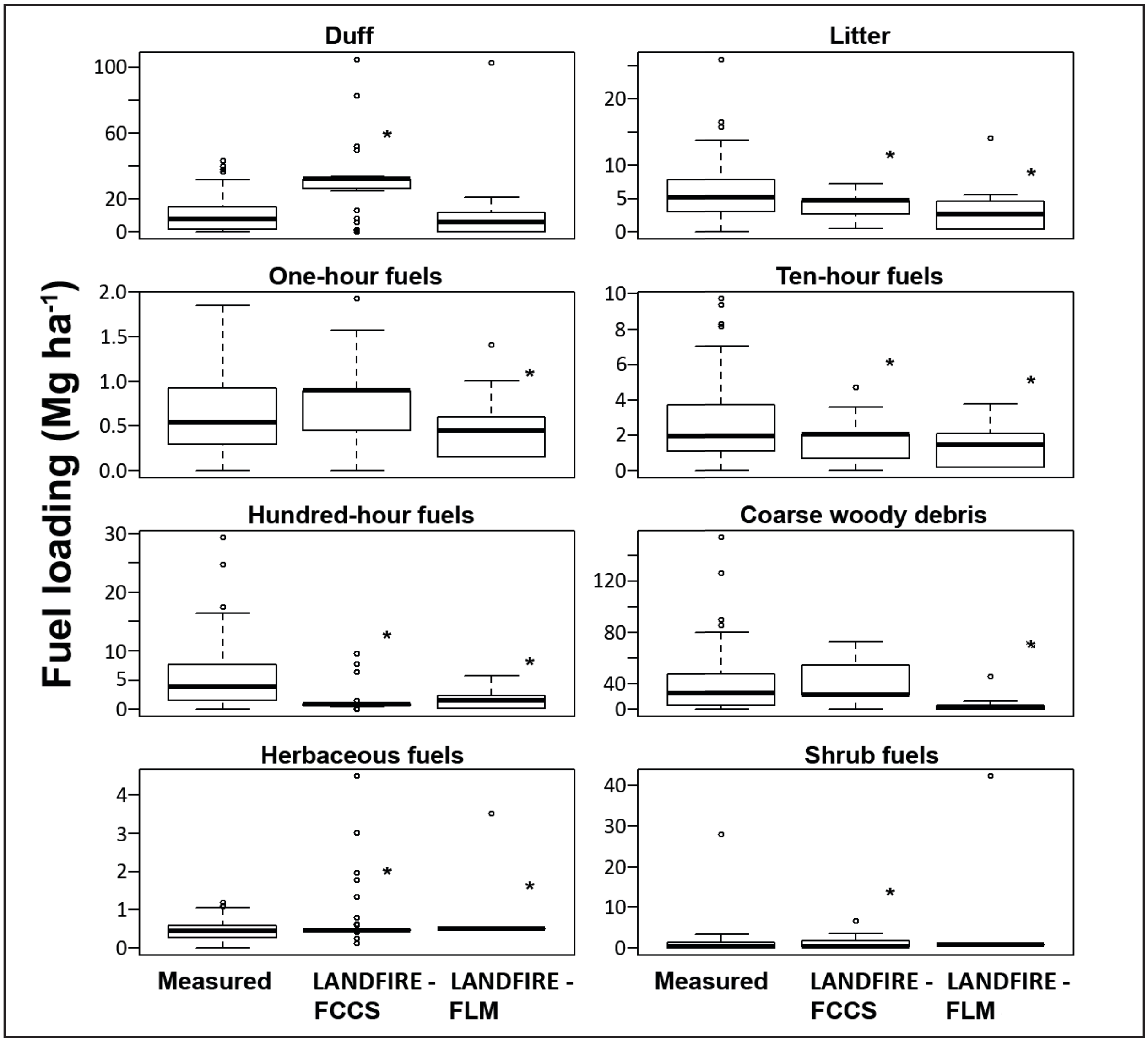

Figure 2. Differences in fuel loading for measured plots, LANDFIRE-FLM, and LANDFIRE-FCCS products. Bold horizontal lines indicate median values, asterisks represent significant differences relative to measured loadings. Circles indicate outliers, and whiskers indicate the region between the first and third quartiles.

differ. Duff and CWD fuel components showed the most pronounced difference in loadings, with LANDFIRE-FCCS duff loadings 200 % higher than measured loadings, and 300 % higher than LANDFIRE-FLM loadings. LANDFIRE-FLM CWD loading was 9 times lower than measured or LANDFIRE-FCCS loadings. When comparing LANDFIRE FEFMs to each other, only duff, CWD, and 1 h fuel loadings differed, with LANDFIRE-FCCS having the greater loadings.

Modeled Consumption and Emissions in FOFEM

The statistical relationships for fuel consumption mirrored those for fuel loading (Figures 2 and 3, Tables 2 and 3). Relative to measured consumption, the mean total surface consumption from LANDFIRE-FCCS was 23 % higher, and LANDFIRE-FLM was 51 % lower. It is apparent that the high LANDFIRE-FCCS duff loading led to the higher

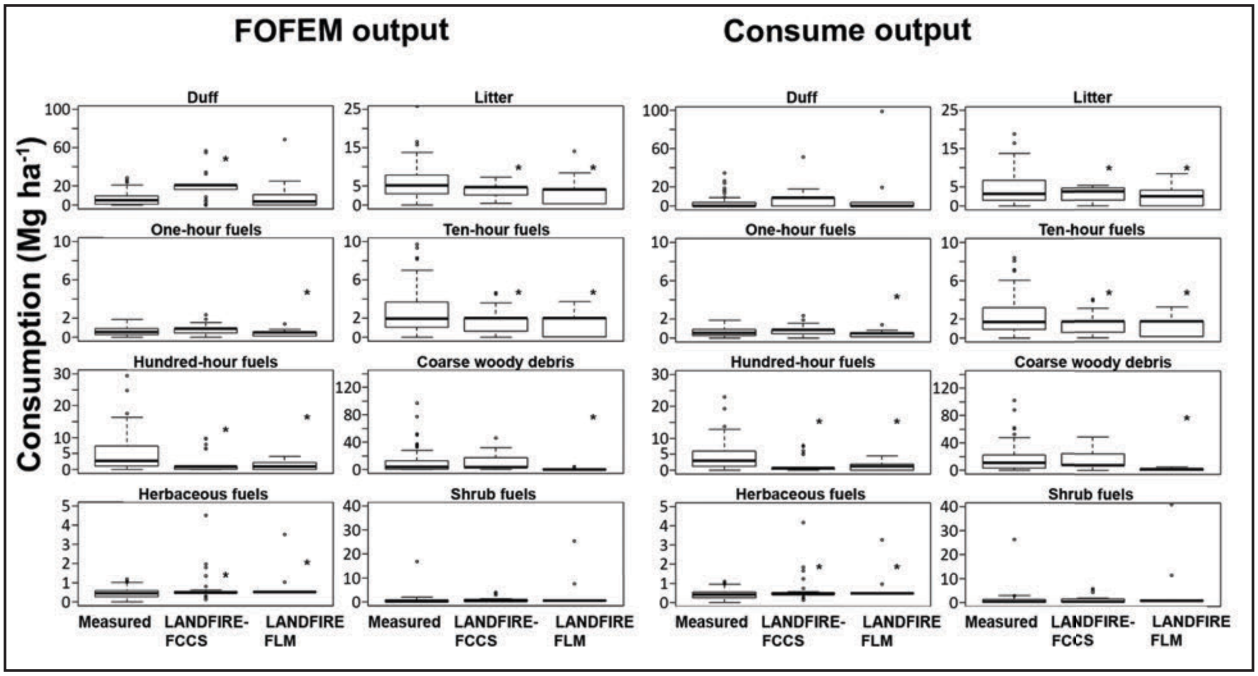

Figure 3. Differences in modeled consumption for measured, LANDFIRE-FLM, and LANDFIRE-FCCS fuel loadings. Bold horizontal lines indicate median values, asterisks represent significant differences relative to results derived from measured loadings.

Table 3. Mean fuel consumption (Mg ha^{-1} with SD in parentheses) under fixed environmental conditions or measured plots, and as modeled by LANDFIRE-FCCS and LANDFIRE-FLM, using FOFEM and Consume. Asterisks indicate statistically significant difference relative to estimates based on measured loading at the $P < 0.05$ significance level.

	Mean plot consumption in FOFEM			Mean plot consumption in Consume		
Fuel	Measured	LANDFIRE-FCCS	LANDFIRE-FLM	Measured	LANDFIRE-FCCS	LANDFIRE-FLM
Duff	6.98 (6.84)	20.64 (11.66)*	5.35 (8.38)	3.36 (6.48)	5.67 (8.48)	2.31(10.83)
Litter	5.83 (4.16)	4.14 (1.21)*	3.68 (3.23)*	4.45 (3.98)	3.11 (1.66)*	2.32 (1.71)*
1 h	0.65 (0.49)	0.76 (0.44)	0.49 (0.29)*	0.65 (0.49)	0.75 (0.44)	0.49 (0.29)*
10 h	2.58 (2.19)	1.77 (1.17)*	1.62 (1.19)*	2.24 (1.89)	1.53 (1.00)*	1.44 (0.98)*
100 h	4.56 (5.29)	2.32 (3.46)*	1.47 (1.25)*	3.93 (4.06)	1.84 (2.70)*	1.55 (1.30)*
CWD	10.59 (16.88)	8.60 (10.84)	0.53 (0.67)*	17.06 (19.41)	14.20 (14.17)	1.84 (1.56)*
Herb	0.45 (0.28)	0.66 (0.61)*	0.70 (0.70)*	0.42 (0.26)	0.61 (0.57)*	0.64 (0.65)*
Shrub	0.70 (1.85)	0.99 (1.20)	1.99 (5.87)	1.02 (2.88)	1.41 (1.74)	3.10 (9.48)
Total fuel	32.35 (25.46)	39.89 (23.40)	15.83 (14.30)	33.12 (25.90)	29.13 (18.61)	13.69 (16.08)*

overall consumption, and that the low CWD loading in the LANDFIRE-FLM contributed to less consumption. This in turn had a direct effect on the emissions modeled. All modeled emissions, with the exception of NO_x, were significantly higher when modeled using LANDFIRE-FCCS loadings, and lower when using LANDFIRE-FLM loadings, while emissions derived from measured fuel loadings fell in between (Table 4, Figure 4).

The relative importance of CWD and duff to total emissions was reaffirmed and quantified using the importance index (Table 5). Duff and CWD stood out as the primary con-

Table 4. Mean modeled emissions (Mg ha^{-1} with SD in parentheses) calculated using FOFEM and Consume for measured plots, LANDFIRE-FCCS, and LANDFIRE-FLM. Asterisks indicate statistically significant difference relative to estimates based on measured loading at the $P < 0.05$ significance level.

	Plot-level values FOFEM			Plot-level values Consume		
Effect	Measured	LANDFIRE-FCCS	LANDFIRE-FLM	Measured	LANDFIRE-FCCS	LANDFIRE-FLM
CH_4	0.32 (0.30)	0.46 (0.30)*	0.13 (0.14)*	0.19 (0.18)	0.19 (0.14)	0.06 (0.11)*
CO	6.83 (6.56)	10.00 (6.64)*	2.67 (3.00)*	3.67 (3.38)	3.65 (2.65)	1.20 (2.02)*
CO_2	45.20 (34.09)	52.81 (29.57)	23.39 (21.90)*	51.90 (39.72)	44.82 (28.11)	22.08 (25.22)*
$PM_{2.5}$	0.53 (0.50)	0.76 (0.50)*	0.22 (0.23)*	0.29 (0.25)	0.27 (0.19)	0.11 (0.15)*
PM_{10}	0.63 (0.59)	0.90 (0.59)*	0.25 (0.27)*	0.33 (0.28)	0.30 (0.21)	0.12 (0.16)*
SO_2	0.03 (0.03)	0.04 (0.02)	0.02 (0.01)*			
NO_x	0.03 (0.03)	0.02 (0.01)*	0.02 (0.03)			
NMHC				0.16 (0.14)	0.15 (0.11)	0.05 (0.08)*

tributors to total emissions in all cases, with the exception of LANDFIRE-FLM data, in which duff and shrub loadings were the primary contributors. Although shrub loadings did not statistically different in our study, shrub loadings tended to be higher in LANDFIRE-FLMs compared to other sources.

Modeled Consumption and Emissions in Consume

With the exception of duff, the relationships between fuel loading and modeled consumption when using Consume remained the same as with FOFEM; modeled duff consumption was much lower when using Consume (Table 3). Duff consumption using LANDFIRE-FCCS loadings did not significantly differ from consumption generated from measured loadings. Because of this, the overall modeled fuel consumption from LANDFIRE-FCCS did not significantly differ from the fuel consumption generated by measured loadings. However, the modeled consumption from LANDFIRE-FLM was significantly lower than consumption from measured loadings, with mean total surface fuel consumption 59% less than that derived from measured fuel loadings.

The importance index for the consumption and total emissions in Consume was similar to the FOFEM emissions importance index (Table 5). Duff consumption was still an important component with regard to emissions production, even though it did not statistically differ between measured and modeled fuel datasets when modeled with Consume. When emissions were evaluated, the LANDFIRE-FLM generated emissions were significantly lower than those generated using measured fuel loadings. Emissions generated using LANDFIRE-FCCS and measured fuel loadings did not differ from each other (Table 4).

DISCUSSION

Measured Versus Modeled Fuel Loading

Duff and CWD led to the most significant differences in modeled consumption and emissions. LANDFIRE-FLMs contained higher shrub loadings, although this number did not result in a statistically significant difference, nor was it great enough to influence the total surface fuel loading when consumption and emissions were modeled. While the cause for these LANDFIRE-FLM shrub values to be so much higher is not known, the FLM system itself was developed with very little available shrub data (Lutes *et al.* 2009). This likely influenced which FLMs were available to assign

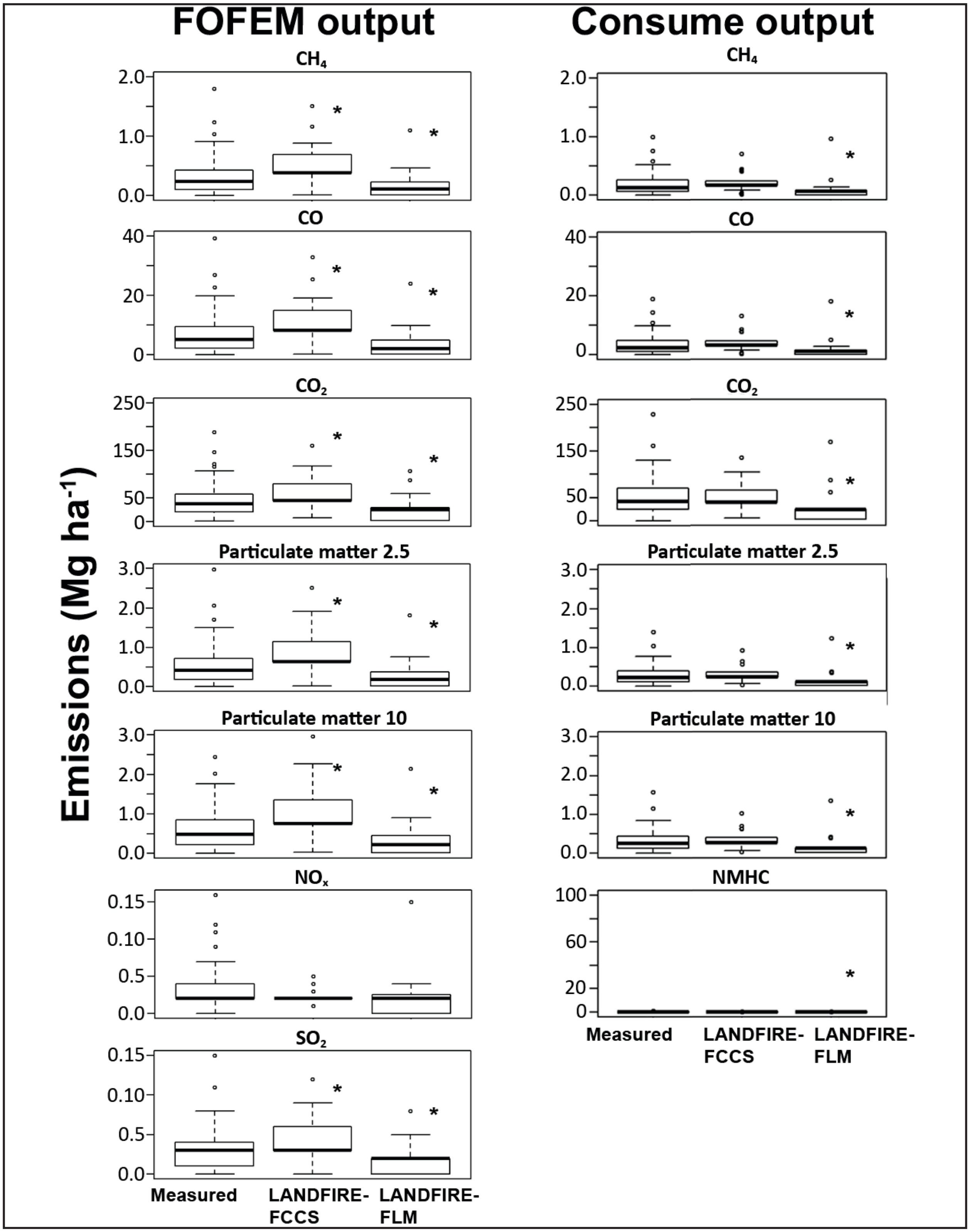

Figure 4. Differences in modeled emissions for measured, LANDFIRE-FLM, and LANDFIRE-FCCS fuel loadings. Bold horizontal lines indicate median values, asterisks represent significant differences relative to results derived from measured loadings.

Table 5. Hoffman and Gardner Importance Index for each FEFM and each fuel type shows that the fuel of relative importance to the total emissions produced varied depending by FEFM. Emissions from measured data and FCCS fuelbeds were most influenced by CWD and duff, and FLM by duff and shrubs, respectively. Highest values are indicated in bold.

	Importance Index FOFEM			Importance Index Consume		
Fuel	**Measured**	**LANDFIRE-FCCS**	**LANDFIRE-FLM**	**Measured**	**LANDFIRE-FCCS**	**LANDFIRE-FLM**
Duff	**0.012**	**0.043**	**0.053**	**0.022**	**0.073**	**0.0156**
Litter	0.002	0.000	0.004	0.008	0.003	0.004
1 h	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
10 h	0.001	<0.001	<0.001	0.002	0.001	0.001
100 h	0.003	0.002	0.001	0.009	0.007	0.002
CWD	**0.063**	**0.048**	0.001	**0.196**	**0.204**	0.003
Herb	<0.001	<0.001	<0.001	<0.001	<0.001	0.001
Shrub	0.001	0.001	**0.032**	0.004	0.003	**0.119**

to LANDFIRE maps when the LANDFIRE-FLM was created. Because the scope of this study focused on a mixed conifer ecosystem, our shrub data were somewhat limited and probably provided little insight in shrub-dominated ecosystems where shrubs are a large fuel component. Further investigation of these LANDFIRE layers in shrub-dominated systems and further fuel loading data from shrub ecosystems would be beneficial to further refining FLMs and the resulting LANDFIRE-FLM data for shrub ecosystems.

When comparing each fuel component for measured and LANDFIRE-represented loadings with those of other mixed conifer systems, all three of our fuel loading sets fell within the ranges observed by other researchers (Table 6). Focusing on duff and coarse woody debris, we found LANDFIRE-FCCS mean duff loading exceeded our measured values, but more closely resembled the ranges found in other mixed conifer forests. Thus, it is possible that our study area may have had less duff loading than other mixed conifer forests. When evaluating mean CWD loadings, we found the wide range noted in other studies, from 0.5 Mg ha^{-1} to 37 Mg ha^{-1}; LANDFIRE-FLM mean CWD loadings were at the low end of this range averaging 0.53 Mg ha^{-1}, while our measured data and LANDFIRE-FCCS were 10.6 Mg ha^{-1} and 8.6 Mg ha^{-1}, respectively.

Our results support a broader evaluation of the importance of various steps in the emissions modeling process in which Drury *et al.* (2014) compared LANDFIRE-represented loadings to a custom loading map based on measured data. Like our results, their duff loading was higher for LANDFIRE-FCCS relative to loadings represented using measured data, while in our study the LANDFIRE-FCCS total loadings were greater. Drury *et al.* found a wide range in possible fuel loadings depending upon the method chosen, as did we, and concluded that custom fuel loading layers derived from measured data produced the most reliable emissions estimates. Of the two LANDFIRE fuel layers, Drury *et al.* found the LANDFIRE-FCCS layer produced results closer to the custom loading layers. We found this to be true in our study when modeling emissions with Consume, but still found LANDFIRE-FCCS to produce higher emissions values when modeled using FOFEM.

In another study that compared classification, mapping accuracy, and fuel loadings of LANDFIRE-FCCS and LANDFIRE-FLM to Forest Inventory and Analysis (FIA) plot data across the western US, Keane *et al.* (2013) found poor performance in both LAND-

Table 6. Fuel loading for other mixed conifer forests in the western United States compared with mean fuel loading from this study (in Mg ha^{-1}). Standard deviations, when present, are indicated in parentheses. Values from this study are indicated in bold in the last three rows.

Source	Duff	Litter	1 hour	10 hour	100 hour	1000 h sound	1000 h rotten	Herb	Location	Elevation (m)
Hille and Stephens 2005	17.8 (3.6)	17.8 (3.6)	2.0 (0.2)	6.3 (0.7)	5.8 (1.6)	6.0 (3.3)	15.8 (4.3)	-	North-central Sierra Nevada, California	1200 to 1500
Sikkink and Keane 2008			0.019	1.649	0.513	0.683 (sound and rotten)		0.545	NW Rockies*	730 to 2130
Sikkink and Keane 2008			0.012	1.297	0.671	0.549 (sound and rotten)		0.659	NW Rockies	730 to 2130
Sikkink and Keane 2008			0.107	0.709	1.105	0.937 (sound and rotten)		0.581	NW Rockies	730 to 2130
Sikkink and Keane 2008			1.155	4.390	5.682	0.600 (sound and rotten)		0.615	NW Rockies	730 to 2130
Sikkink and Keane 2008			2.586	5.567	7.849	0.863 (sound and rotten)		0.636	NW Rockies	730 to 2130
Youngblood *et al*. 2008	22.27 (7.52)	5.9 (0.97)	0.94 (0.2)	1.56 (0.33)	4.16 (0.59)	9.63 (3.46)	7.31 (2.5)		Blue Mountain Region, Oregon	1040 to 1480
Youngblood *et al*. 2008	25.48 (7.03)	3.74 (0.44)	0.37 (0.12)	0.64 (0.21)	3.04 (0.67)	8.88 (4.26)	7.97 (0.61)		Blue Mountain Region, Oregon	1040 to 1480
Raymond and Peterson 2005			1.2	4.1	4.8			1.2	Oregon Coast Range	670 to 850
Raymond and Peterson 2005			4.4	6.8	8.7			1.2	Oregon Coast Range	670 to 850
Kobziar *et al.* 2006			1.25 (0.87)	4.53 (3.23)	9.93 (8.18)	7.52 (16.82)	14.18 (23.31)		North-central Sierra Nevada, California	1100 to 1410
Kobziar *et al.* 2006			1.13 (1.04)	5.53 (4.97)	6.17 (7.15)	7.91 (17.04)	29.02 (40.86)		North-central Sierra Nevada, California	1100 to 1410
Kobziar *et al.* 2006			0.9 (0.71)	2.9 (2.3)	4.25 (4.12)	2.57 (5.36)	26.62 (65.62)		North-central Sierra Nevada, California	1100 to 1410
Reinhard *et al.*1991	52 (1.3)		Other values are logging slash, not natural fuels						NW Rockies	900 to 1200
Reinhard *et al*. 1991	48.4 (1.6)		Other values are logging slash, not natural fuels						NW Rockies	900 to 1200
Measured	**10.55 (10.20)**	**5.86 (4.13)**	**0.65 (0.47)**	**2.57 (2.19)**	**4.98 (5.20)**	**20.09 (23.33)** (sound and rotten)		**0.46 (0.28)**	**NW Rockies**	**770 to 1516**
LANDFIRE-FLM	**7.76 (12.19)**	**3.66 (3.40)**	**0.50 (0.32)**	**1.65 (1.13)**	**2.47 (3.41)**	**2.75 (4.04)** (sound and rotten)		**0.73 (0.76)**	**NW Rockies**	**770 to 1516**
LANDFIRE-FCCS	**31.89 (17.8)**	**4.19 (1.37)**	**0.81 (0.46)**	**1.85 (1.11)**	**1.94 (1.66)**	**18.45 (16.38)** (sound and rotten)		**0.68 (0.76)**	**NW Rockies**	**770 to 1516**

[*] NW Rockies includes parts of Idaho and Montana, USA.

FIRE-represented FEFMs. LANDFIRE-FLM tended to under-predict loadings while LANDFIRE-FCCS tended toward over prediction. However, LANDFIRE-FLM loadings had lower root mean squared errors (Keane *et al.* 2013). Our findings here support the work of Keane *et al.* (2013) and Drury *et al.* (2014) in describing the tendency of LANDFIRE-FCCS to have higher loadings relative to LANDFIRE-FLMs.

Modeled Consumption and Emissions Using FOFEM

Relative differences in consumption values when modeled with FOFEM mirrored those of the loading values. High LANDFIRE-FCCS duff and low LANDFIRE-FLM CWD loading and consumption contributed to the total modeled emissions being highest when using LANDFIRE-FCCS inputs, and lowest when using LANDFIRE-FLM inputs. In examining the fuel loading data (Table 2), there is high variance in all fuel loading categories. This supports the work by Keane *et al.* (2013), who noted the high variance inherent in all categories of fuel loading, and the difficulties caused by spatial variation when trying to represent fuel loadings across large landscapes. Consumption followed the pattern of the total fuel loading values for the landscape, with LANDFIRE-FCCS being the highest, FLM being the lowest, and measured values in the middle. This in turn produced higher emissions from LANDFIRE-FCCS and lower emissions from LANDFIRE-FLM, highlighting the differences in emissions outcomes depending upon the choices made to represent fuel loadings.

Modeled Consumption and Emissions Using Consume

In comparing consumption and emissions from Consume, the choice of model has an effect on emissions generated. In this study, there were similar trends in modeled consumption with both fire effects models, but the lower duff consumption in Consume, relative to FOFEM, led to emissions outputs in which LANDFIRE-FCCS derived emissions did not differ from those derived from measured loadings. This difference in duff consumption is due to the fact that Consume and FOFEM calculate the consumption of duff using different equations, derived from different data sets (Reinhardt 2003, Prichard *et al*. 2005).

Research Implications

In modeling emissions, fuel loadings have been identified as the most crucial variables (Drury *et al.* 2014), yet they represent one of the greatest uncertainties in modeling emissions (French *et al.* 2011). In a detailed discussion on the topic, Keane *et al.* (2013) identified several factors creating difficulties in quantifying fuel loadings. These include lack of data to develop thorough loading maps; the use of classification systems that were developed from discrete plot locations but then applied to large, national-scale areas; and the inherent difficulty of classifying fuels into categories such as hourly size classes and duff, when each of these size classes may have different degrees of variation at different spatial scales (Keane *et al.* 2012, 2013). If existing fuel loading classification maps are to be improved, more data are necessary. The results of our study indicate that data on CWD and duff should be priorities, due to the relative importance of these fuels to overall emissions in mixed conifer forests (Table 5). While consumption didn't statistically differ for the specific case of shrubs, shrub loading accounted for a great deal of variability in emissions from LANDFIRE-FLMs (Table 5), a classification that was developed with little available shrub data (Lutes *et al.* 2009). For the case of LANDFIRE-FLMs, having additional data on shrub loadings would be beneficial.

Despite being represented at a 30 m resolution, LANDFIRE data layers are intended to be used at larger, sub-regional to national scales (LANDFIRE Team 2014*b*). Data from fuel loading maps may work for finer scales; however, there will likely be greater need to supplement that data with local knowledge. Based on our findings in a 20 000 ha area, using measured data, especially for duff and CWD loadings, is preferable relative to unaltered LANDFIRE layers. However, we understand that measured data are often unavailable, may be incomplete, or limited in availability.

Management Implications

If monitoring resources are available, emission estimates will be improved by having more information on duff loading, as differences in duff loading lead to the greatest differences in emissions, followed by CWD. For coarse woody debris, the planar intercept sampling methods have been most commonly used in forests such as in this study, although the Photoload (Keane and Dickinson 2007) method has also performed well (Sikkink and Keane 2008). Duff sampling is often performed via sampling points along a planar intercept to gather both loading and depth (Brown 1974). The fuel photoseries guides available for many ecosystems provide estimates of duff loading (Ottmar *et al.* 2003), but there are few studies comparing their performance relative to the traditional method. If measured data are not available, one could model with both the LANDFIRE-FLM and LANDFIRE-FCCS derived fuel loadings, and then average the two sets of results.

The use of systems such as the Wildland Fire Decision Support System (WFDSS) and the Interagency Fuels Treatment Decision Support System (IFTDSS) also hold potential for obtaining measured fuel loading information (IFTDSS 2015, WFDSS 2015). These systems provide online access to several models to represent fire behavior and effects (including emissions), but they also provide an easy platform from which data can be shared from user to user. In the future, it would be ideal to see a searchable database of user-provided fuel loadings within these decision support systems, similar to the searchable data available through the Fire Research and Management Exchange System (FRAMES) Resource Catalog (FRAMES 2015).

This study has characterized the potential differences in LANDFIRE-represented fuel loadings in a mixed conifer case study area, and their impact on the emissions modeling. While using measured data provides the most reliable outcome, either by itself or to help supplement the LANDFIRE data, this is not always possible. Web-based systems can aid in finding and sharing data; however, a search for the keywords "duff" and "coarse woody debris" in FRAMES returned 34 and 3 results, respectively. While online systems can be powerful sources of information, there is clearly a need for additional data with which tools such as the LANDFIRE map layers could be strengthened. In the interim, information on the relative differences in fuel loadings from LANDFIRE-represented data may be useful to managers who are tasked with quantifying emissions for fire management planning. Using all of these resources will aid in generating more accurate emissions estimates in a climate where regulatory pressure and the need to accurately represent potential emissions from fire are increasing.

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SUPPLEMENTARY MATERIALS

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CONSERVATION

A global map of roadless areas and their conservation status

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Roads fragment landscapes and trigger human colonization and degradation of ecosystems, to the detriment of biodiversity and ecosystem functions. The planet's remaining large and ecologically important tracts of roadless areas sustain key refugia for biodiversity and provide globally relevant ecosystem services. Applying a 1-kilometer buffer to all roads, we present a global map of roadless areas and an assessment of their status, quality, and extent of coverage by protected areas. About 80% of Earth's terrestrial surface remains roadless, but this area is fragmented into ~600,000 patches, more than half of which are <1 square kilometer and only 7% of which are larger than 100 square kilometers. Global protection of ecologically valuable roadless areas is inadequate. International recognition and protection of roadless areas is urgently needed to halt their continued loss.

The impact of roads on the surrounding landscape extends far beyond the roads themselves. Direct and indirect environmental impacts include deforestation and fragmentation, chemical pollution, noise disturbance, increased wildlife mortality due to car collisions, changes in population gene flow, and facilitation of biological invasions (*1*–*4*). In addition, roads facilitate "contagious development," in that they provide access to previously remote areas, thus opening them up for more roads, land-use changes, associated resource extraction, and human-caused disturbances of biodiversity (*3*, *4*). With the length of roads projected to increase by >60% globally from 2010 to 2050 (*5*), there is an urgent need for the development of a comprehensive global strategy for road development if continued biodiversity loss is to be abated (*6*). To help mitigate the detrimental effects of roads, their construction should be concentrated as much as possible in areas of relatively low "environmental values" (*7*). Likewise, prioritizing the protection of remaining roadless areas that are regarded as important for biodiversity and ecosystem functionality requires an assessment of their extent, distribution, and ecological quality.

Such global assessments have been constrained by deficient spatial data on global road networks. Importantly, recent publicly available and rapidly improving data sets have been generated by crowd-sourcing and citizen science. We demonstrate their potential through OpenStreetMap, a project with an open-access, grassroots approach to mapping and updating free global geographic data, with a focus on roads. The available global road data sets, OpenStreetMap and gROADS, vary in length, location, and type of roads; the former is the data set with the largest length of roads (36 million km in 2013) that is not restricted to specific road types (table S1). OpenStreetMap is more complete than gROADS, which has been used for other global assessments (*7*), but in certain regions, it contains fewer roads than subglobal or local road data sets [see the example of Center for International Forestry Research data for Sabah, Malaysia (*8*); table S1]. Given the pace of road construction and data limitations, our results overestimate the actual extent of global roadless areas.

The spatial extent of road impacts is specific to the impact in question and to each particular road and its traffic volume, as well as to taxa, habitat, landscape, and terrain features. Moreover, for a given road impact, its area of ecological influence is asymmetrical along the road and can vary among seasons, between night and day, according to weather conditions, and over longer time periods. We conducted a comprehensive literature review of 282 publications dealing with "road-effects zones" or including the distance to roads as a covariate, of which 58 assessed the spatial influence of the road (table S2). All investigated road impacts were documented within a distance of

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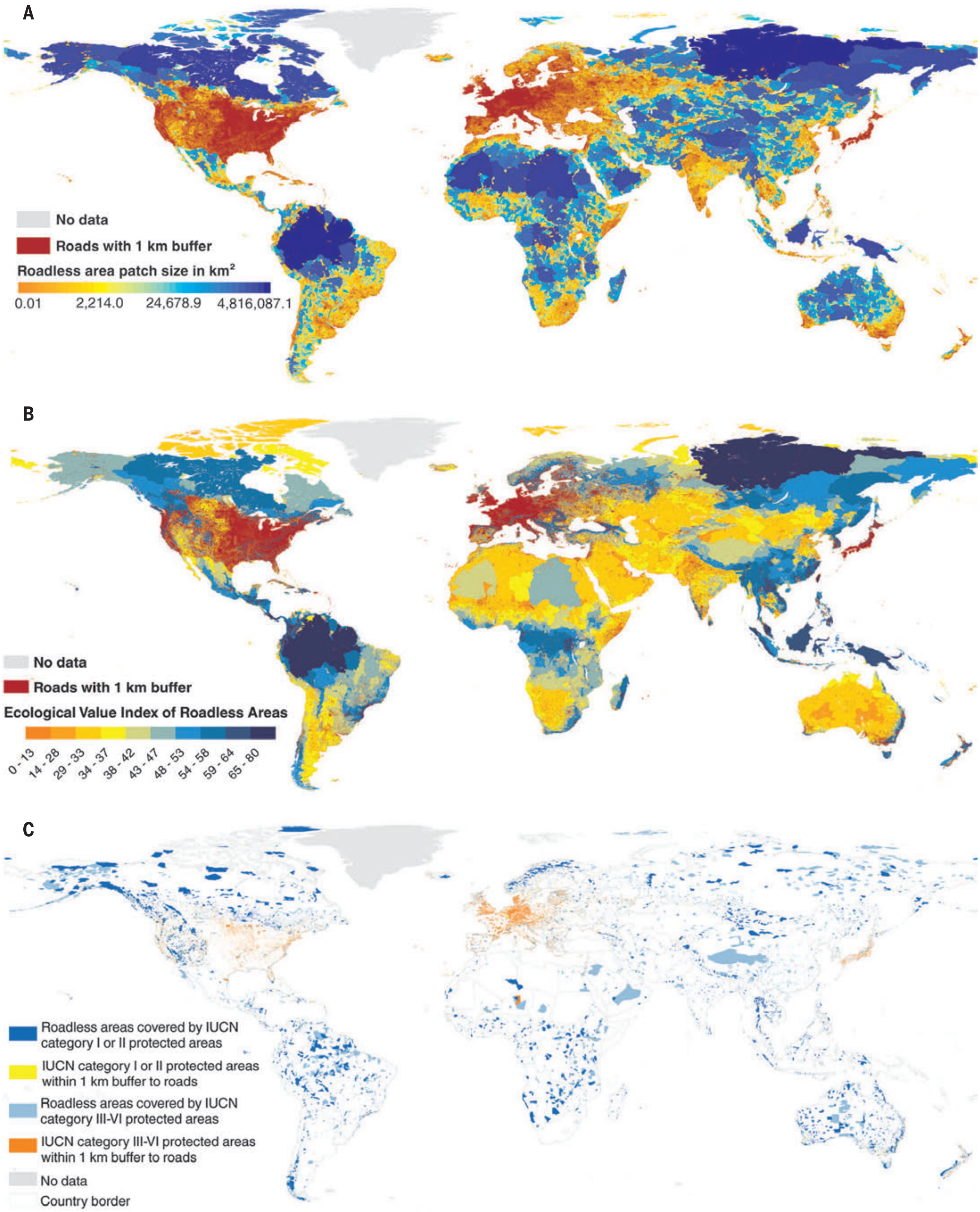

Fig. 1. The global distribution of roadless areas, based on a 1-km buffer around all roads. The distribution is depicted according to (**A**) size classes, (**B**) the ecological value index of roadless areas (EVIRA; based on patch size, connectivity, and ecosystem functionality), and (**C**) representation in protected areas (*8*).

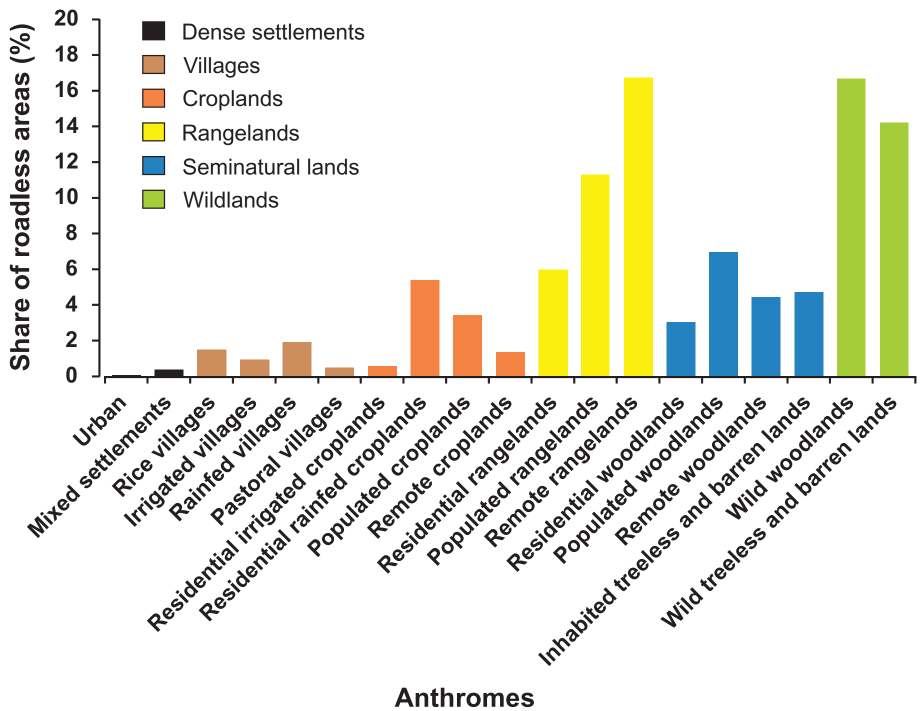

Fig. 2. Extent of roadless areas (1-km buffer) across anthromes. The majority of the world's roadless areas are in remote and unmodified landscapes, but they also occur in anthropogenically modified landscapes. The so-called anthromes were mapped according to (*10*).

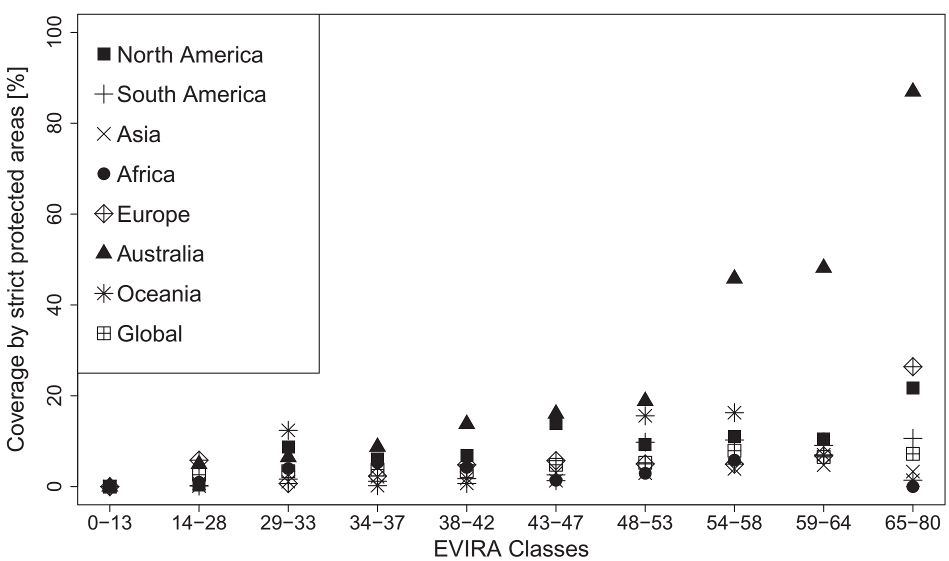

Fig. 3. Coverage of roadless areas by strictly protected areas (IUCN categories I and II) compared with global and continental EVIRA values. If priority were given to protecting roadless areas with high ecological functionality, we should see a positive correlation, with higher coverage associated with higher EVIRA values.

1 km from the road, 39% reached out to 2 km from the road, and only 14% extended out to 5 km from the road (fig. S1). Because the 1-km buffer along each side of the road represents the zone with the highest level and variety of road impacts, we defined roadless areas as those land units that are at least 1 km away from all roads and, therefore, less influenced by road effects. We compared results from using this criterion with the outcomes from using an alternative 5-km buffer (see fig. S2 and table S3). We excluded all large water bodies, as well as Greenland and Antarctica, which are mostly covered by ice, from the analyses.

Roadless areas with a 1-km buffer to the nearest road cover about 80% of Earth's terrestrial surface (~105 million km^2). However, these roadless areas are dissected into almost 600,000 patches. More than half of the patches are <1 km^2; 80% are <5 km^2; and only 7% are >100 km^2 (table S4 and fig. S3). If the buffer is extended to 5 km, there is a substantial reduction in roadless areas to about 57% of the world's terrestrial surface (~75 million km^2), dissected into 50,000 patches (fig. S2 and table S3). The occurrence, distribution, and size of roadless areas differ considerably among continents (Fig. 1A and fig. S4). For instance, the mean size of roadless patches (1-km buffer) is 48 km^2 in Europe, compared with >500 km^2 in Africa. Because of comparatively large gaps in available spatial data on roads in many segments of the tropics, the number and size of roadless areas are overestimated and should be treated with caution (e.g., Borneo; table S1).

All identified roadless areas were assessed for a set of ecological properties that were selected to reflect their relative importance to biodiversity, ecological functions, and ecosystem resilience: patch size, connectivity, and ecosystem functionality (*9*) (table S5). We normalized these three indicators to between 0 and 100 to calculate an additive and unitless index of the ecological value of each roadless area identified (termed the ecological value index of roadless areas, or EVIRA) [Fig. 1B and fig. S5; the specific rationale and technicalities of the chosen indicators are described in table S5 (*8*)]. The EVIRA values range from 0 to 80. A sensitivity analysis shows that ecosystem functionality and patch size are the best single indicators for the final index values (table S6 and figs. S6 to S8). Areas with relatively high index values tend to have a lower coefficient of variation (fig. S9).

We used the International Union for Conservation of Nature (IUCN) and UN Environment Programme–World Conservation Monitoring Centre data set of global protected areas to determine the extent of roadless areas that are protected (*8*) (Fig. 1C). The roadless areas distribution across human-dominated landscapes was determined following the classification of so-called anthromes, defined as biomes shaped by human land use and infrastructure (*10*) (Fig. 2 and table S7).

When examining the density of roads within different biomes, large discrepancies in distribution are apparent. The tundra and rock and ice-covered biomes are nearly entirely roadless, whereas temperate broadleaf and mixed forests have the lowest share of roadless areas (41%; figs. S9 and S10). Boreal forests of North America and Eurasia still retain large tracts of roadless areas (figs. S10 and S11). In the tropics, large roadless landscapes (>1000 km^2) remain in Africa, South America, and Southeast Asia, with the Amazon having the single largest roadless segment. In relation to the anthromes (*10*), about two-thirds of the world's roadless areas can be described as remote and unmodified landscapes [26% uninhabited or sparsely inhabited treeless and barren lands; 21% natural and remote seminatural woodlands, with 17% wild woodlands therein (*8*); Fig. 2 and table S7]. The remaining one-third consists of rangelands, indicating that roadless areas can also occur in anthropogenically modified landscapes.

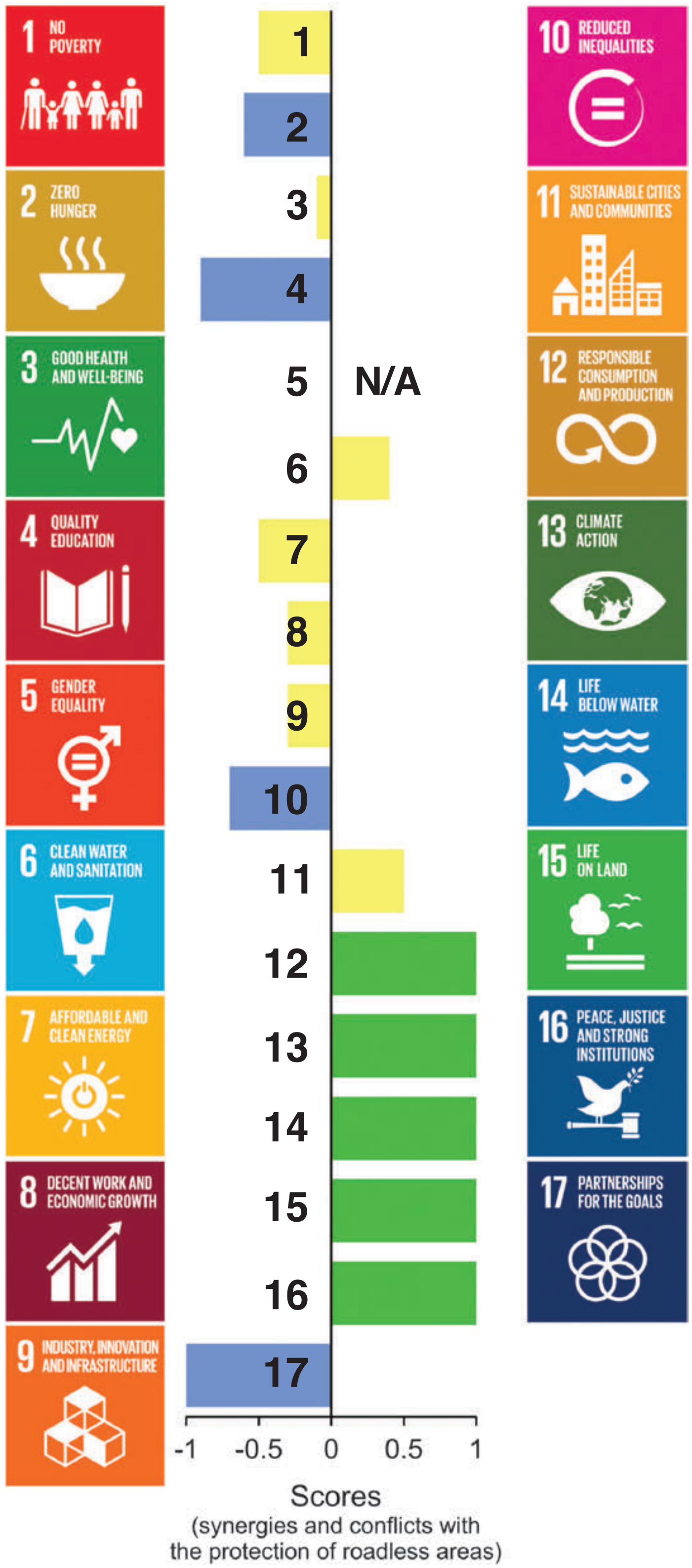

Fig. 4. Synergies and conflicts between conservation of roadless areas and the United Nations' Sustainable Development Goals. Scores <–0.5 (blue bars) indicate that conflicts with the goal prevail; scores between –0.5 and 0.5 (yellow) indicate a mixture of synergies and conflicts with the goal; and scores >0.5 (green) indicate prevailing synergies with the goal [for details, see table S11 (*8*)]. The scores reflect substantial imminent conflicts between various Sustainable Development Goals and conservation of roadless areas (table S11).

About one-third of the world's roadless areas have low EVIRA values. Patches with relatively low EVIRA values (ranging from 0 to 37; namely, <50% of the maximum value) account for 35% of the overall roadless area distribution, because most are small, fragmented, isolated, or otherwise heavily disturbed by humans. Some large tracts of roadless areas, such as arid lands in northern Africa or central Asia, occur in areas of sparse vegetation and low biodiversity and, thus, have low index values for ecosystem functionality (*9*) (Fig. 1B). High EVIRA values occur both in tropical and boreal forests. The relative conservation value of roadless areas is context-dependent. Comparatively small or moderately disturbed roadless areas have higher conservation importance in heavily roaded environments, such as most of Europe, the conterminous United States, and southern Canada.

Although the world's protected areas cover 14.2% of the terrestrial surface, only 9.3% of the overall expanse of roadless areas is within protected areas (all IUCN categories; Fig. 1C and table S8). There is no major difference in the coverage of roadless areas by strictly protected areas (IUCN categories I and II) versus the coverage of the overall landscape by strictly protected areas (3.8% roadless versus 4.2% overall). Only in North America, Australia, and Oceania are more than 6% of roadless areas under strict protection (table S8). If conservation efforts were to prioritize functional, ecologically important roadless areas, we would find a positive relation between strict protection coverage and EVIRA values of roadless areas. However, with the exception of Australia, this is not the case (Fig. 3 and table S9). Asia and Africa have particularly low protection coverage for roadless areas with high EVIRA values. For instance, we found gaps in the Asian tropical southeast, as well as in boreal biomes.

The recent Global Biodiversity Outlook (*11*) gives a bleak account of the progress made toward reaching the United Nations' biodiversity agenda as specified in the 20 Aichi Targets of the Convention on Biological Diversity (*12*). Governments have failed on several accounts to keep their use of natural resources well within safe ecological limits (target 4); to halt or at least halve the rate of habitat loss and substantially reduce the degradation and fragmentation of natural habitats (target 5); and to appropriately protect areas of particular importance for biodiversity and ecosystem services (target 11). To achieve global biodiversity targets, policies must explicitly acknowledge the factors underlying prior failures (*13*). Despite increasing scientific evidence for the negative impacts of roads on ecosystems, the current global conservation policy framework has largely ignored road impacts and road expansion. Furthermore, key policies on road infrastructure and development, such as the Cohesion Policy of the European Union, fail to take into account biodiversity.

In the much wider context of the United Nations' Sustainable Development Goals, conflicting interests can be seen between goals intended to safeguard biodiversity and those promoting economic development (*14*). We analyzed how roadless areas relate to the global conservation and sustainability agendas. As a transparent synthesis, we calculated simple scores of conflicts versus synergies of Sustainable Development Goals and Aichi Targets with the conservation of roadless areas (tables S10 and S11). Roads are explicitly mentioned in the Sustainable Development Goals only for their contribution to economic growth (goal 8), promoting further expansion into remote rural areas, and consideration is given neither to the environmental nor the social costs of road development. The resulting scores reflect substantial imminent conflicts (Fig. 4 and table S10); only in five Sustainable Development Goals do synergies with conservation of roadless

areas prevail, and four Sustainable Development Goals are predominantly in conflict with conservation of roadless areas. Maybe even more surprisingly, several of the Aichi Targets are ambivalent with respect to conserving roadless areas, rather than being in synergy entirely [six conflicting versus 11 synergistic targets (*8*); table S11].

There is an urgent need for a global strategy for the effective conservation, restoration, and monitoring of roadless areas and the ecosystems that they encompass. Governments should be encouraged to incorporate the protection of extensive roadless areas into relevant policies and other legal mechanisms, reexamine where road development conflicts with the protection of roadless areas, and avoid unnecessary and ecologically disastrous roads entirely. In addition, governments should consider road closure where doing so can promote the restoration of wildlife habitats and ecosystem functionality (*4*). Our global map of roadless areas represents a first step in this direction. During planning and evaluation of road projects, financial institutions, transport agencies, environmental nongovernmental organizations, and the engaged public should consider the identified roadless areas.

The conservation of roadless areas can be a key element in accomplishing the United Nations' Sustainable Development Goals. The extent and protection status of valuable roadless areas can serve as effective indicators to address several Sustainable Development Goals, particularly goal 15 ("Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss") and goal 9 ("Build resilient infrastructure, promote inclusive and sustainable industrialization and foster innovation"). Enshrined in the protection of roadless areas should be the objective to seek and develop alternative socioeconomic models that do not rely so heavily on road infrastructure. Similarly, governments should consider how roadless areas can support the Aichi Targets (see tables S10 and S11). For instance, the target of expanding protected areas to cover 17% of the world's terrestrial surface could include a representative proportion of roadless areas.

Although we acknowledge that access to transportation is a fundamental element of human well-being, impacts of road infrastructure require a fully integrated environmental and social cost-benefits approach (*15*). Still, under current conditions and policies, limiting road expansion into roadless areas may prove to be the most cost-effective and straightforward way of achieving strategically important global biodiversity and sustainability goals.

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SUPPLEMENTARY MATERIALS

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Data Sources
References (*16*–*180*)

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PLANT PATHOLOGY

Regulation of sugar transporter activity for antibacterial defense in *Arabidopsis*

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Microbial pathogens strategically acquire metabolites from their hosts during infection. Here we show that the host can intervene to prevent such metabolite loss to pathogens. Phosphorylation-dependent regulation of sugar transport protein 13 (STP13) is required for antibacterial defense in the plant *Arabidopsis thaliana*. STP13 physically associates with the flagellin receptor flagellin-sensitive 2 (FLS2) and its co-receptor BRASSINOSTEROID INSENSITIVE 1–associated receptor kinase 1 (BAK1). BAK1 phosphorylates STP13 at threonine 485, which enhances its monosaccharide uptake activity to compete with bacteria for extracellular sugars. Limiting the availability of extracellular sugar deprives bacteria of an energy source and restricts virulence factor delivery. Our results reveal that control of sugar uptake, managed by regulation of a host sugar transporter, is a defense strategy deployed against microbial infection. Competition for sugar thus shapes host-pathogen interactions.

Plants assimilate carbon into sugar by photosynthesis, and a broad spectrum of plant-interacting microbes exploit these host sugars (*1*, *2*). In *Arabidopsis*, pathogenic bacterial infection causes the leakage of sugars to the extracellular spaces (the apoplast) (*3*), a major site of colonization by plant-infecting bacteria. Although leakage may be a consequence of membrane disintegration during pathogen infection, some bacterial pathogens promote sugar efflux to the apoplast by manipulating host plant sugar transporters (*4*, *5*). Interference with sugar absorption by bacterial and fungal pathogens reduces their virulence, highlighting a general

A global map of roadless areas and their conservation status

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Editor's Summary

Too many roads

Roads have done much to help humanity spread across the planet and maintain global movement and trade. However, roads also damage wild areas and rapidly contribute to habitat degradation and species loss. Ibisch *et al*. cataloged the world's roads. Though most of the world is not covered by roads, it is fragmented by them, with only 7% of land patches created by roads being greater than 100 km ^{2}. Furthermore, environmental protection of roadless areas is insufficient, which could lead to further degradation of the world's remaining wildernesses.

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Supplementary Materials for

A global map of roadless areas and their conservation status

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MATERIALS AND METHODS

A. Definition of roadless areas
B. Data set and data accuracy
C. Data processing - Mapping of roadless areas and general processing
D. Data processing - Ecological Value Index of Roadless Areas (EVIRA)
E. Sensitivity analysis for the Ecological Value Index of Roadless Areas (EVIRA)
F. Policy analyses: synergies and conflicts between conservation of roadless areas and conservation and sustainability agendas

A. Definition of roadless areas

We reviewed 282 scientific papers, out of which 58 publications provided information on the spatial influence of various road impacts and/or on the road-effect zone (Table S2). All studied impacts were documented within a distance of 1 km from the road, 39% were observed in the 1-2 km zone, and only 14% extended out to 5 km. Road effects that go beyond 50 km and to even 100 km are rarely documented; they refer to deforestation in relation to distance to main roads, not including other minor roads and paths that are necessary for forest clearings (Table S2). The 1-km buffer would therefore rather underestimate than overestimate the extension of areas impacted by roads. Still it represents a reasonable approach to excluding with high certainty those areas that are significantly affected by roads. We consider 1 km as the minimum value for road-effect zones at a global scale, taking into account landscape heterogeneity, as well as the wide range of road impacts across biomes and road categories. Consequently, we defined roadless areas as terrestrial areas not dissected by roads and low impacted by road effects (which are at least 1 km away from the nearest road).

B. Dataset and data accuracy

We used a data set of OpenStreetMap (11/2013) to create a global map of roadless areas. This data set is updated on a daily basis and can be freely downloaded. We purchased a pre-processed data set provided by Geofabrik (http://www.geofabrik.de/de/). The pre-processing did not change the road data, but instead provided a filtered data set that contained only road layers in shapefile format. OpenStreetMap is a volunteered geographic information project founded in Britain in 2004 *(16)*. It is one of the most cited, analyzed and commonly used platforms of this type and became one of the best alternative sources for geodata *(17, 18)*. The aim of OpenStreetMap is to produce and distribute free global geographic data *(19)*. The OpenStreetMap data set used in this research provides six main road categories. Examples of 'major roads' can be motorways and freeways (*category one*); 'minor roads' are categorized as small local roads, residential roads, etc. (*category two*). *Category three* is represented by 'highway links' (sliproads/ramps) that connect roads with each other. Service roads or roads for agricultural use are considered as 'very small roads' under *category four*. *Category five* is called 'path' and mainly used for horse riding and cycling, but also for small or off-road vehicles. *Category six* roads are 'unknown' types of roads. As all road categories have ecological impacts (Table S2), we included all of them in the analyses.

The CIA World Factbook estimated the road length to be 64-million km in 2013 *(20)*. The OpenStreetMap data set (2013) used in this research consists of 36-million km of roads. In contrast, the Global Roads Open Access Data Set (gROADS), published in 2013, contains 9.1-million km of roads (CIESIN 2013). The gROADS data set has been used in global studies on road impacts, in spite of containing less data than OpenStreetMap (e.g. *(7)*).

OpenStreetMap relies on the willingness of volunteers, both to contribute entries and to edit them for errors *(21)*. Therefore, the data are a crowd-sourced product with unknown data quality standards. However, a quality assessment of the OpenStreetMap data, including spatial data quality, evolution of street network, polygon geometry, comparison of user activity, development, positional accuracy, and completeness is available for different regions *(17, 22-28)*. Gröchenig et al. (2014) conducted a global evaluation of the mapping progress of OpenStreetMap history between 2006 and 2013 *(29)*. Their results state that external and internal factors significantly influence the mapping progress. Some of these factors are regional activity of the mapping community, data imports, and environmental disasters or other unforeseen events *(29)*. Demographic characteristics affect the mapping progress, and the quality of the data can vary significantly among countries (*17, 29)*.

A high number of road assessments were conducted in Europe *(30-34)*. Often, commercial or administrative data sets are used to compare and evaluate OpenStreetMap *(17)*. A study published in 2010 assessed the quality of OpenStreetMap for Germany *(32)*. Among its findings, the total length of roads was calculated as 1,204,213.69 km, whereas the road length data made available by TeleAtlas (an enterprise that provides digital maps, user content navigation, and location-based services) was 1,272,681.77 km. TeleAtlas focuses more on roads suitable for cars, whereas OpenStreetMap includes all road types *(32)*. In the case of the Brazilian Amazon it has been found that the road data from the Brazilian Institute for Geography and Statistics (IBGE) are more complete, including ca. 157,000 km of roads in contrast to ca. 114,000 km in our OpenStreetMap data set.

In areas of the tropics where land conversion is advanced, the road network may not be well reflected by OpenStreetMap. An extreme example of missing roads in the OpenStreetMap data set is Borneo. We carried out a comparative analysis of roads in the Sabah region, Malaysia, in northern Borneo. In areas considered to be roadless, closer inspection on the ground (in 2015) revealed extensive networks of vehicle tracks, for instance, throughout oil palm plantations. A similar result was found in forested areas impacted by logging roads. Indeed, cumulative data (1970-2010) compiled by the Center for International Forestry Research (CIFOR) indicate that there would be 37,498 km of logging roads in the region of Sabah alone. The 2013 OpenStreetMap data set (for Sabah created since 2009) used in this study comprises just 4,880 km, which is still more than the 2,937 km included in the road data set gROADS (1980-2010) that was the basis for other global road assessments (CIESIN 2013, *7*). Applying a 1-km buffer to each of the three road data sets for Sabah demonstrates that roadless areas are underestimated by the OpenStreetMap and the gROADS data set (Table S1). According to the gROADS data set (CIESIN 2013), 92% of Sabah is roadless. The OpenStreetMap data set shows that 91% of Sabah is roadless. In contrast, buffering the logging roads (CIFOR) reveals that only 40% of Sabah remains roadless. However, on the other hand, the CIFOR data set seems to overestimate existing logging roads. The CIFOR logging roads were mapped in four time intervals (1970, 1990, 2000 and 2010) by visual interpretation of satellite imagery. Analyzing the CIFOR logging roads with current Google Earth satellite images suggests that numerous roads have been overgrown by forest. The amount of logging roads that were either non-existent in 2010 or were <10 m wide (therefore not included in the CIFOR analysis) is high *(35)*. This simple exercise highlights the methodological problems to be overcome in future mapping. The three data sets can only be compared to a limited extent, since the roads have been mapped in different ways, time intervals and for different purposes. The gROADS data set (CIESIN 2013) focuses on roads between settlements. For Malaysia, gROADS is based on the Vector Smart Map Level 0 data. The CIFOR road data set does not include any other road category besides logging roads. In general, the three different road data sets (OpenStreetMap, gROADS, CIFOR) vary in length, location and type of roads, with OpenStreetMap being the data set with the largest length of roads at a global scale, and not limited to one type of roads (Table S1).

C. Data processing - Mapping of roadless areas and general processing

The global road data set was analyzed and processed for each continent, except for Antarctica and Greenland. All roads were buffered on both sides with a geodesic buffer of 1 km. Due to a very high number of vertices, all buffered roads were generalized with a "maximum offset tolerance" of 30 m, using the "Douglas-Peucker simplification algorithm" *(36)*. All analyses were conducted with ArcGIS 10.2. A road model tool was created with the ArcGIS model builder to facilitate the process. For the purpose of comparison, an alternative map of roadless areas was developed with a 5-km buffer to all roads (Fig. S2).

For area calculations, roadless areas were projected with the World Cylindrical Equal Area Projection. Spatial calculations and maps were made with ArcGIS Version 10.2. Protected area coverage of roadless areas was calculated based on IUCN categories of protected areas, including (a) IUCN categories Ia, Ib and II, and (b) other protected areas classified as IUCN categories III to VI (IUCN & UNEP-WCMC 2015). Protected area data sets for each country were downloaded and processed singularly instead of using the global protected area file due to inconsistencies in the global data set.

D. Data processing - Ecological Value Index of Roadless Areas (EVIRA)

There are manifold and partially contrasting approaches for defining the conservation values of given areas. Attempts at conservation priority setting have been classified as reactive and proactive *(37)*, some approaches focus on patterns rather than processes; however, in times of rapid environmental change, there are good arguments for especially targeting ecological functionality and biological viability *(9, 38)*. Therefore, we chose a functional priority-setting approach that is not based on merely anthropocentric values, such as use value or aesthetics, but comprises indicators that are defined in line with principles of modern ecosystem theory. In this context, we especially consider the capability of ecosystems to self-order and regulate abiotic and biotic conditions, which is greatly based on the capacity of uptaking and storing eco-exergy *(39, 40)*. Specifically, exergy has been used for analyzing and indicating ecosystem health *(41-46)*. As key attributes of ecosystem growth and development, Jørgensen (2006) *(42)* and Jørgensen et al. (2000) *(43)* proposed biomass, information and network as main growth forms of ecosystems.

To assess the conservation value of roadless areas, a corresponding additive index (Ecological Value Index of Roadless Areas, EVIRA) was created. Three indicators were chosen (for individual and more specific rationale of indicators see Table S5):

(1) Roadless area patch size: A larger roadless area patch size indicates less human disturbance, lower edge effects, higher populations of road-sensitive species, as well as higher ecological integrity and self-regulating capacity.

(2) Thiessen connectivity into all directions for roadless area patches: We describe connectivity (and degree of isolation), as the ratio between the size of a roadless area patch and its surrounding Thiessen polygon. A Thiessen (or Voronoi) polygon describes the area around a sample point or area where any position taken from inside the polygon is closer to the sample point/area than to any of the other sample points/areas *(47)*. To create Thiessen polygons Euclidean distance was calculated with the formula:

$$d(x,y) = \sqrt{\sum_{i=1}^{n}(x_i - y_i)^2}$$

The larger the Thiessen connectivity value, the closer neighboring roadless patches can be found. This is important for the integrity of landscape-scale processes (e.g., genetic exchange of metapopulations and endemics with narrow geographic ranges confined to roadless areas).

(3) Ecosystem Functionality Index *(9)*: This weighted, additive dimensionless index comprises vegetation density, tree height, carbon storage, species richness of vascular plants, plant functional richness and slope. Functionality is defined as "the state of ecosystems, characterized by inherent structures,

> ecological functions and dynamics, that provide ecosystems with both, the necessary efficiency and resilience to develop without abrupt change of system properties and geographical distribution, and allows for flexible response to external changes" *(9)*.

All indicators (Roadless area patch size, Thiessen connectivity, Ecosystem Functionality Index) were rasterized and adjusted in resolution and projection. A resolution of 0.002 (equally to 0.2 km) was chosen. ArcGIS 10.2 was used for projection, resolution and rasterization. All indicators were normalized between 0 and 100 and a weighted additive index was calculated using the software Insensa-GIS *(48)*. Thiessen connectivity into all directions and roadless areas patch size were weighted with 25%, whereas ecosystem functionality was weighted with 50%.

E. <u>Sensitivity analysis for the Ecological Value Index of Roadless Areas (EVIRA)</u>

Index construction always involves steps such as indicator selection and weighting. In order to transparently highlight the sensitivity of EVIRA to changes in these steps, we performed a statistical sensitivity analysis. Three different index versions were produced using jackknifing, ten of them using random weight variation within defined borders (connectivity into all directions and roadless area patch size 10-50%; ecosystem functionality index 30-70%) and one using equal weighting. Within the jackknifing procedure, three versions were created where each indicator was removed iteratively from the index calculation procedure. Overall 14 different index versions were created to perform the sensitivity analysis.

Pearson and Spearman rank correlation coefficients were calculated for the three indicators and EVIRA (Table S6). Significant and highly positive Spearman rank and Pearson correlation coefficients were found between the Ecosystem Functionality Index (EFI) (*9*) and EVIRA (Spearman r= 0.818; p<0.0001; Pearson r= 0.881; p<0.0001; Table S6). This is likely to be a consequence of the original weighting scheme of EVIRA, where EFI was given a weight double as high as the two other indicators. A high positive and significant Spearman rank correlation was also detected for roadless area patch size and EVIRA (Spearman r= 0.768; p<0.0001; Table S6). Therefore, EFI and roadless area patch size are the best single indicators for the final index output.

Mean values over all 14 index variations are shown in figure S6 with the highest values represented in blue and low values shown in orange. Similar to the original EVIRA, highest mean values are recorded for the Amazon, followed by the tundra and taiga of the northern and eastern lowlands of Siberia, as well as south-east Asian tropical rain forests.

The coefficient of variation was calculated over all 14 index variations to evaluate the variability of EVIRA (Fig. S9). Most parts of Australia show high levels of variation, as well as parts of Africa and central- and southwest Asia. The overall pattern is that regions with relatively high index values tend to have a lower coefficient of variation, whereas areas with high levels of variation tend to occur in regions with low index values. This results in a high confidence in the prediction of the ecological value, especially of those areas with high EVIRA values. A negative correlation coefficient between EVIRA and the coefficient of variation was detected (Spearman rank correlation: -0.97; Pearson correlation: -0.94). The volatility highlights the areas which were most frequently assigned a high index value (>70% of the maximum value) within the 14 different index variations

(Fig. S7). Very high readings were found for the sites with highest roadless area patch size as well as parts of Southeast Asia.

The proportion of area that changes its index value by less than 25%; between 25-50%; between 50-75%; and more than 75%, was explored for the equal weight method, and the three different index versions created by the jackknifing procedure (Fig. S8). Indicator selection seems to have a stronger effect on the output than the weighting scheme. More than half of the area changes its index value between 50 to 75% when connectivity into all directions was removed from the index, and 19% of the areas changed its index value by more than 75%. The exclusion of EFI showed that more than 60% of the area changed its index value between 25 and 50%. The removal of roadless area patch size (18% change in category 25-50%) and applying an equal weighting scheme (5% change in category 25-50%) did not change the index output significantly.

F. <u>Policy analyses: synergies and conflicts between conservation of roadless areas and conservation and sustainability agendas</u>

The "Aichi Biodiversity Targets" of the Convention on Biological Diversity (CBD) are part of the "Strategic Plan for Biodiversity 2011-2020" *(12)*. They circumscribe the United Nations' central agenda for the conservation of the Earth's diversity of life. They were adopted in October 2010 and comprise 20 targets that are grouped into five Strategic Goals. Seventeen "Sustainable Development Goals" (SDGs) have been defined within "Transforming our world: the 2030 Agenda for Sustainable Development" of the United Nations *(14)*, adopted in September 2015. They replace eight "Millennium Development

Goals" that were pursued from year 2000 to 2015 *(49)*. The SDGs are associated with 169 targets. Work on underlying indicators is ongoing; nevertheless, the latest report can provide direction for the interpretation of the goals and their respective targets *(50)*.

Specifically, our analyses of the global sustainability agendas aim at identifying potential synergies, conflicts and ambivalences between roadless areas conservation and the achievement of conservation and sustainability goals in the policy framework of the United Nations. In addition, these analyses indicate imminent conflicts among goals within the respective policy frameworks, particularly those concerning the global sustainability agenda. Furthermore, a considerable number of conservation and sustainability targets also were found to be ambivalent.

The calculation of conflict-synergy scores for the SDGs (Table S10) and the Aichi Strategic Goals (Table S11) is based on a simple index composed of individual scores attributed to all corresponding targets to which roadless areas are in some way applicable. We excluded the targets related to governance in general (marked by a combination of number and letter, e.g. "13.a") from the analysis, thus reducing the number from 169 to 126. The individual scores for targets can have three discrete values:

- -1 (indicated by blue color): conservation of roadless areas is in conflict with the achievement of the target;
- 0 (yellow): conservation of roadless areas has an ambivalent relationship with the achievement of the target; and

- 1 (green): conservation of roadless areas is in synergy with the achievement of the target.

Roadless areas do not relate to a number of targets; these targets are therefore excluded from the analysis (indicated by grey color). The conflict-synergy score for a goal is calculated as the mean of all values for corresponding targets. The scores can, thus, vary between -1 and +1. They are classified as follows:

- <-0.5 (indicated by blue color): conflicts with goal prevail;
- -0.5 to 0.5 (yellow): mixture of synergies and conflicts with goal; and
- >0.5 (green): synergies with goal prevail.

The conflict-synergy scores for goals are also visualized by the colors in the large boxes of Tables S10 and S11.

SUPPLEMENTARY FIGURES (S1-S11)

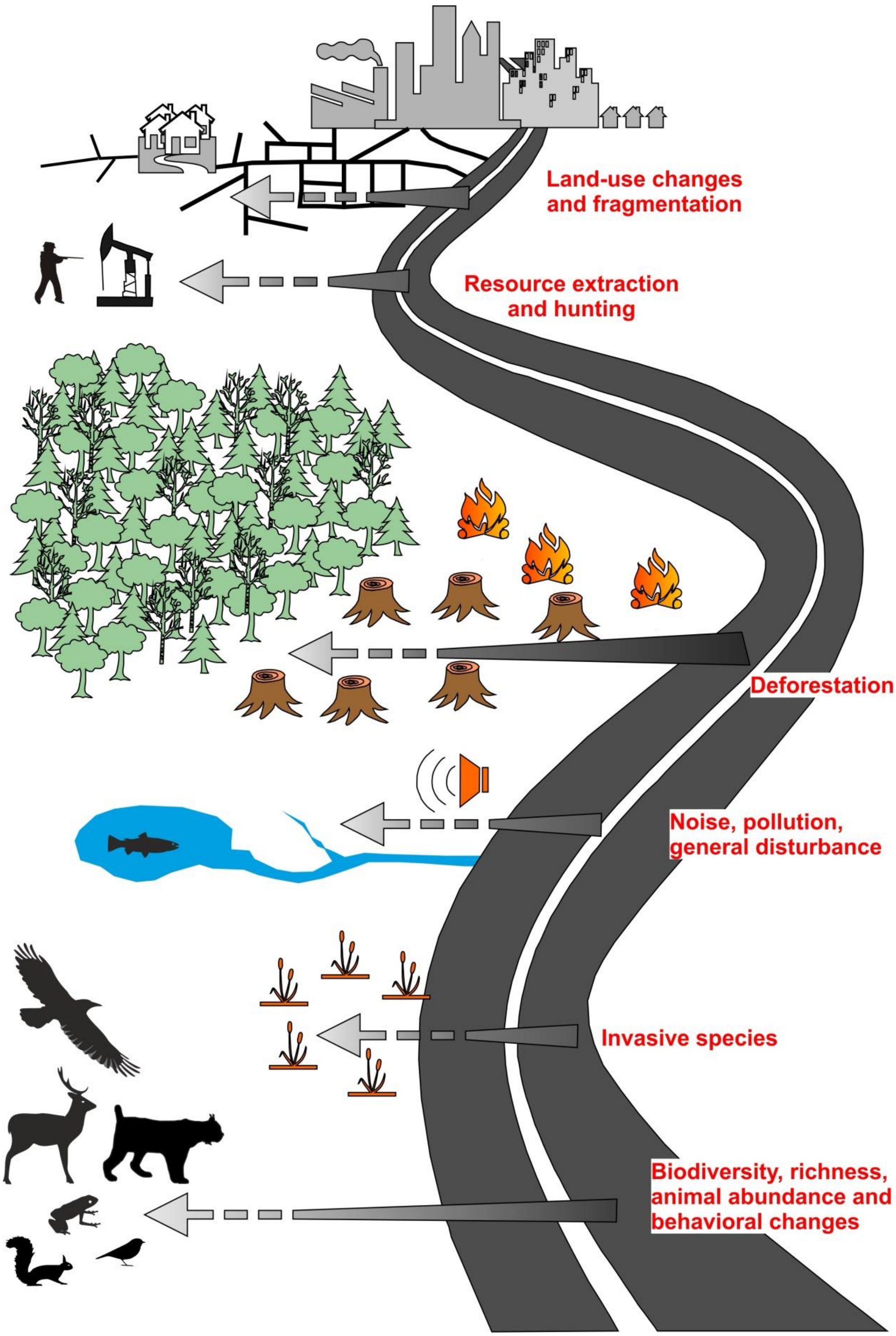

Fig. S1. Schematic representation of different categories of road impacts on biodiversity. These impacts decrease with the distance from the road. Road effects generally attenuate beyond one kilometer distance from the road (see literature review in table S2). One kilometer was therefore selected as a buffer to identify roadless areas as those areas relatively free from road disturbances.

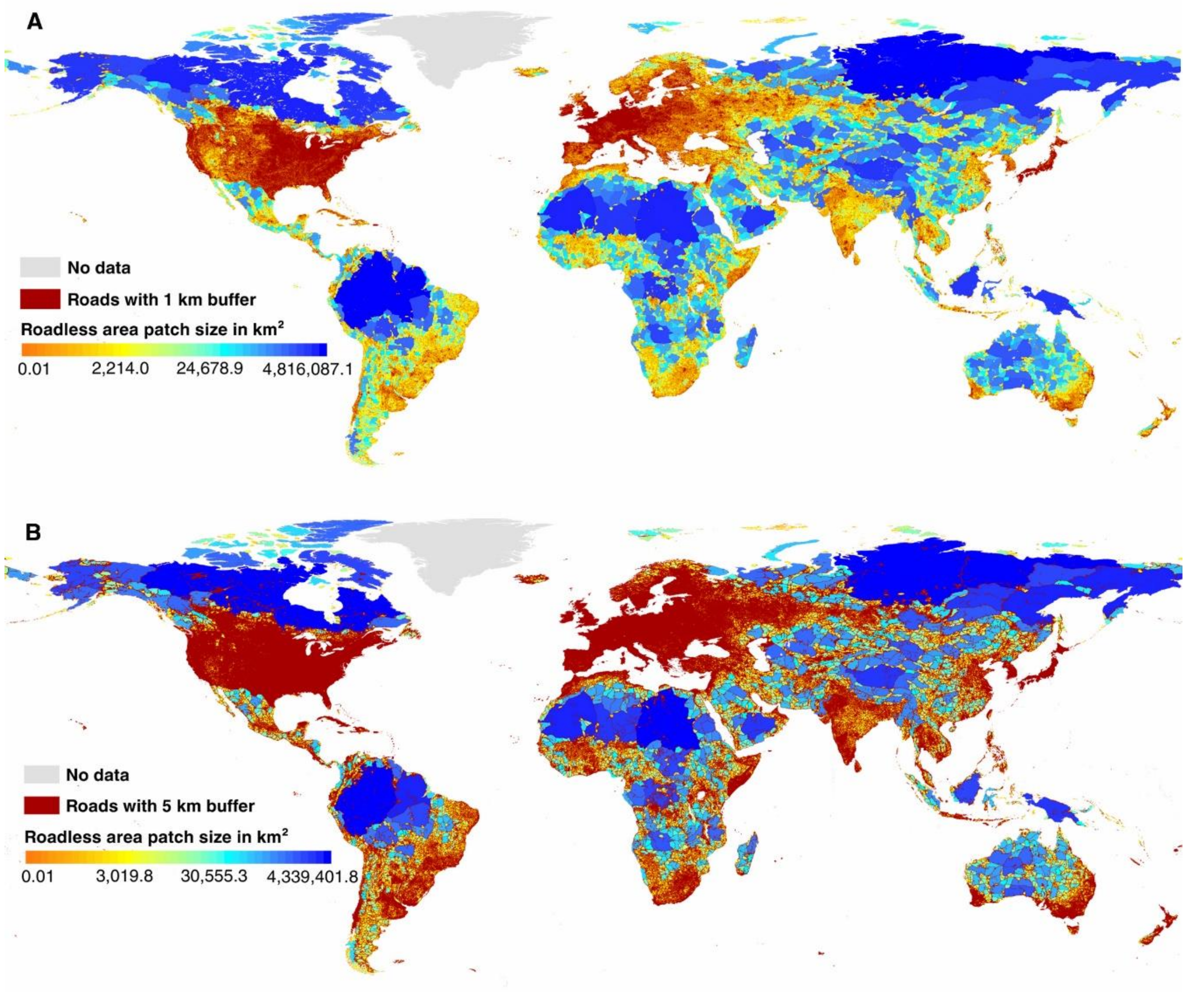

Fig. S2. The global distribution of roadless areas based on a (A) 1-km and a (B) 5-km buffer to all roads included in the OpenStreetMap data set (11/2013).

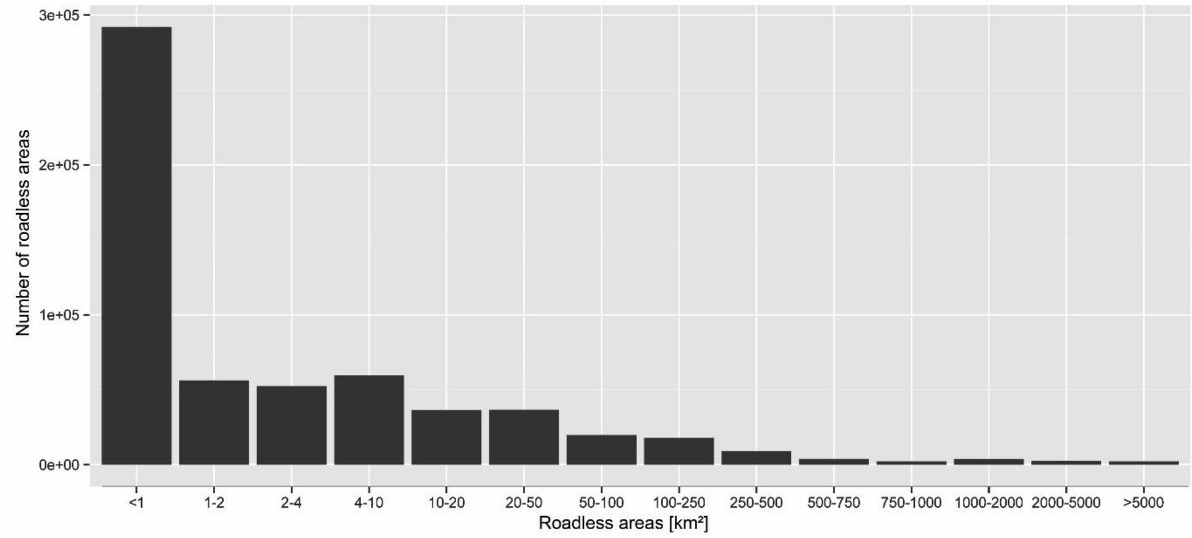

Fig. S3. Frequency of global roadless areas size classes based on 1-km buffer to all roads included in the OpenStreetMap data set (11/2013).

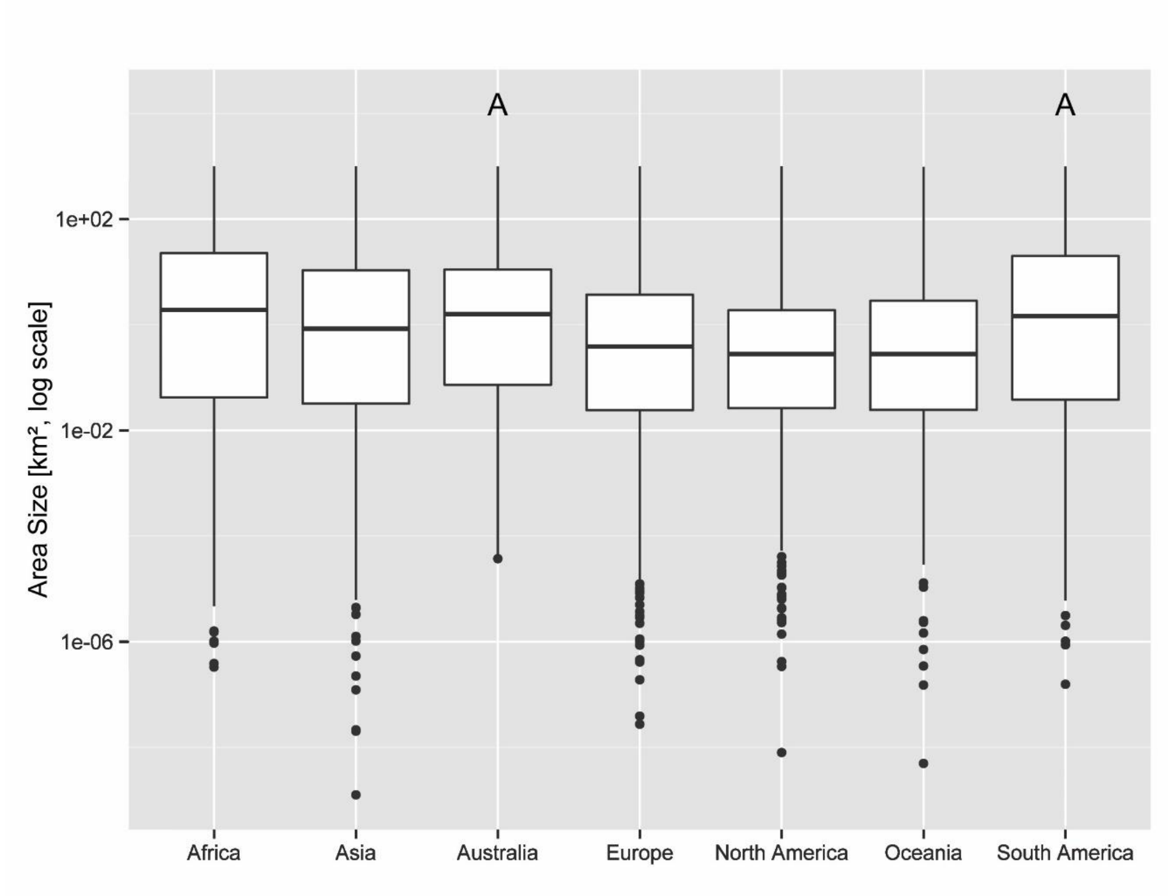

Fig. S4. Sizes of roadless areas across continents based on 1-km road buffer using the OpenStreetMap data set (11/2003) (Pairwise Wilcox test; “A” indicates that the corresponding distributions are not significantly different; p<0.001).

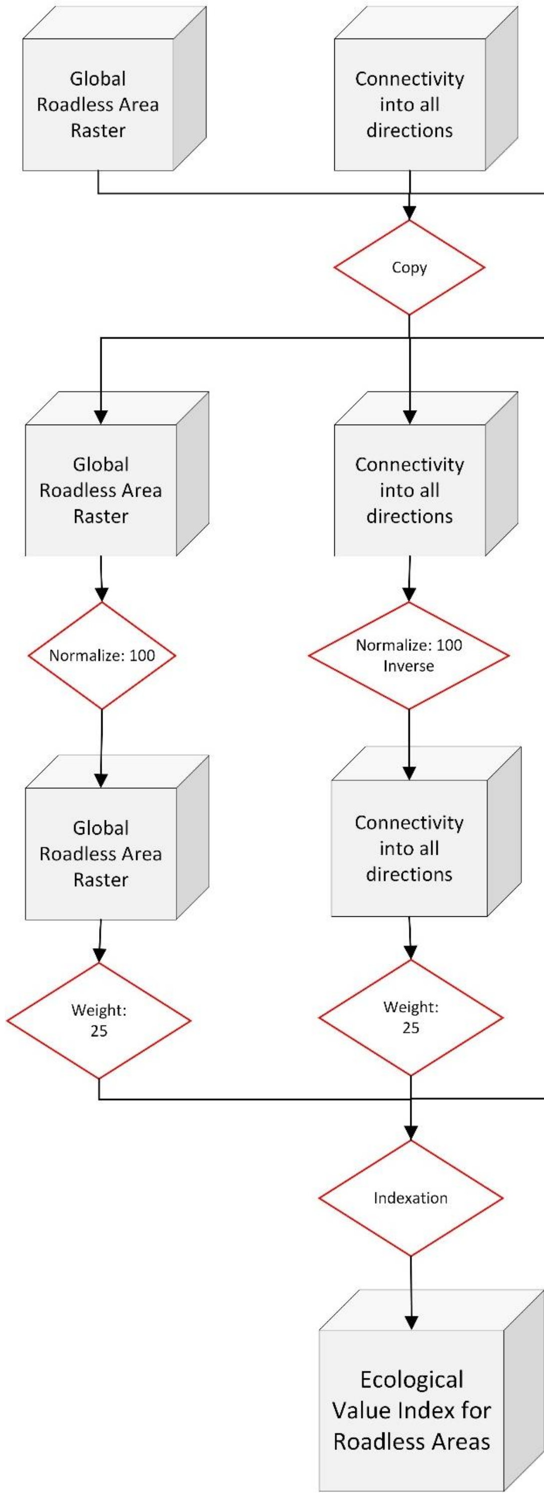

Fig. S5. Workflow of the indexation process for creating the *Ecological Value Index of Roadless Areas* (EVIRA).

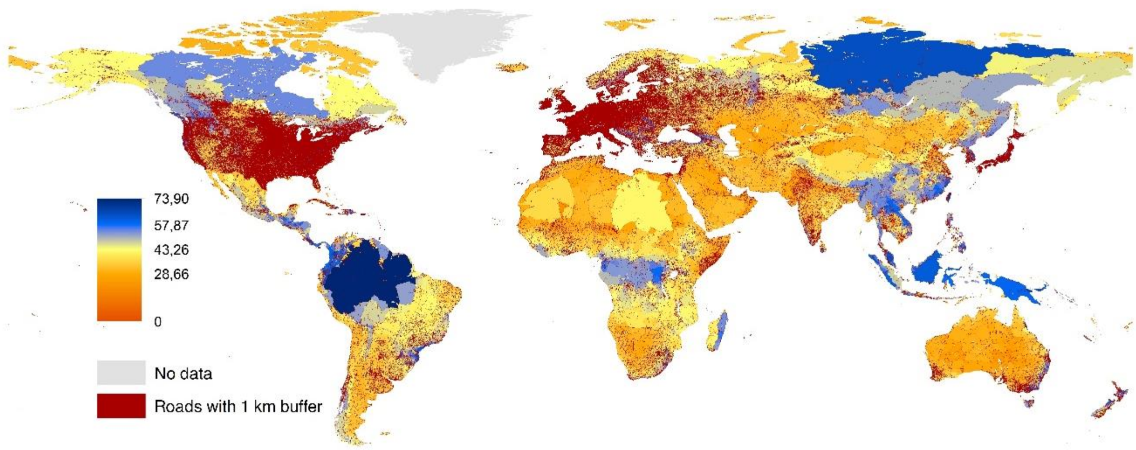

Fig. S6. Global map of mean values over 14 different index variations for the Ecological Value Index of Roadless Areas (EVIRA). Class breaks were calculated using the Jenks breaks algorithm.

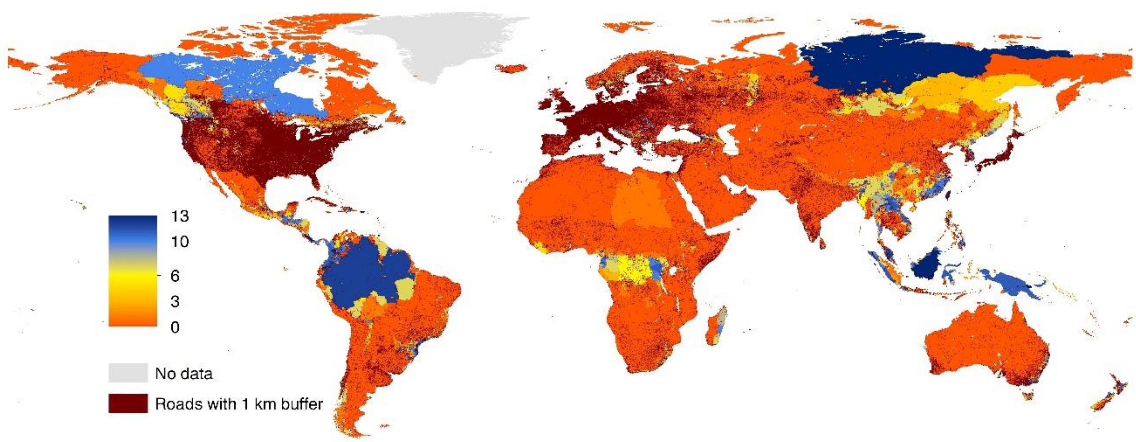

Fig. S7. Global map of volatility (frequency of that the value achieved at least 70% of the maximum index value) of the ecological value index of roadless areas (EVIRA) over all 14 index variations.

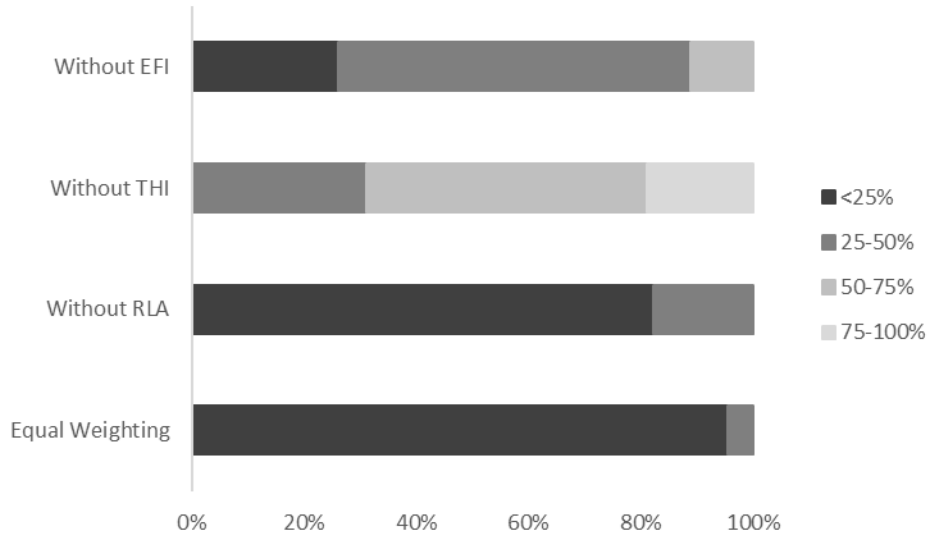

Fig. S8. Proportion of global area whose EVIRA value is changing < 25%, 25-50%, 50-75% and >75%, as shown by the sensitivity analysis. The three indicators making up the EVIRA index are the Ecosystem Functionality Index (EFI), the Thiessen connectivity into all directions (THI) and the Roadless area patch size (RLA).

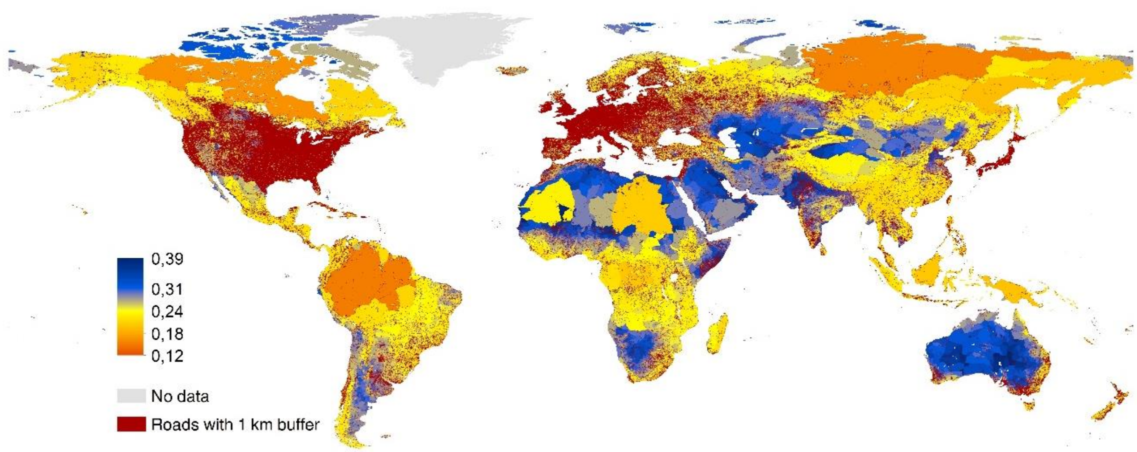

Fig. S9. Mean statistical sensitivity of the Ecological Value Index of Roadless Areas (EVIRA) as overall coefficient of variation of 14 index variations.

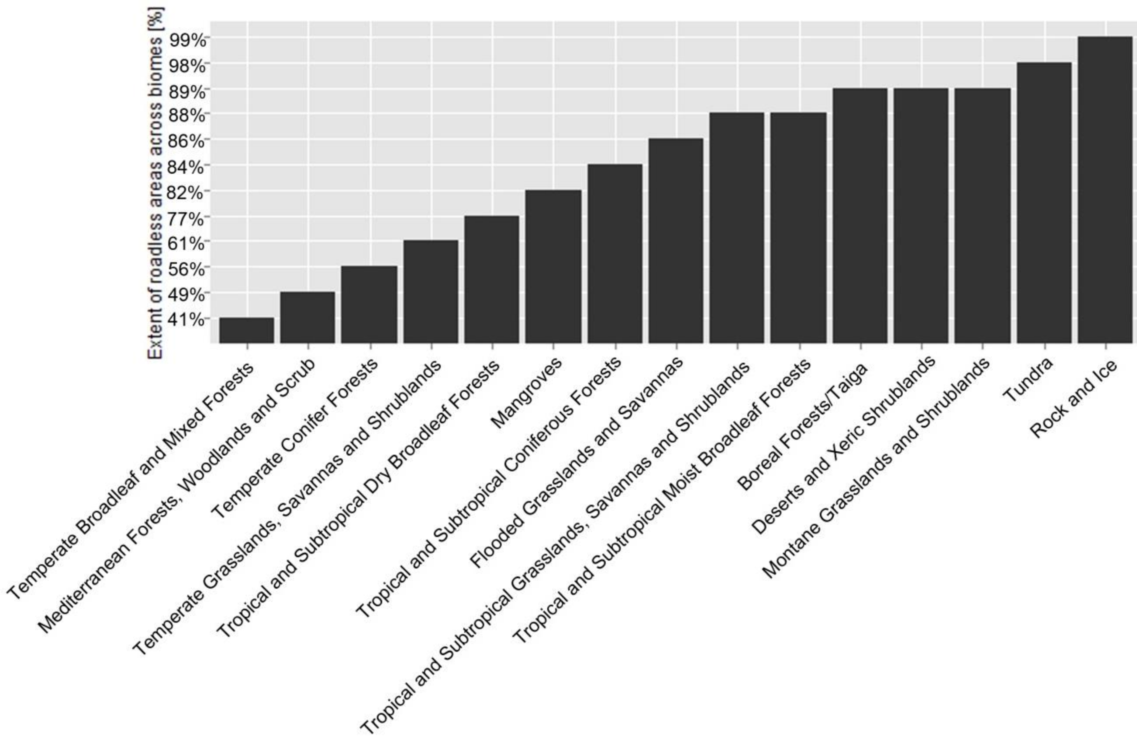

Fig. S10. Extent of roadless areas across biomes (without freshwater bodies, Antarctica and Greenland) according to classification by Olson et al. (2001) *(51)* and based on 1-km buffer to all roads included in the OpenStreetMap data set (11/2013).

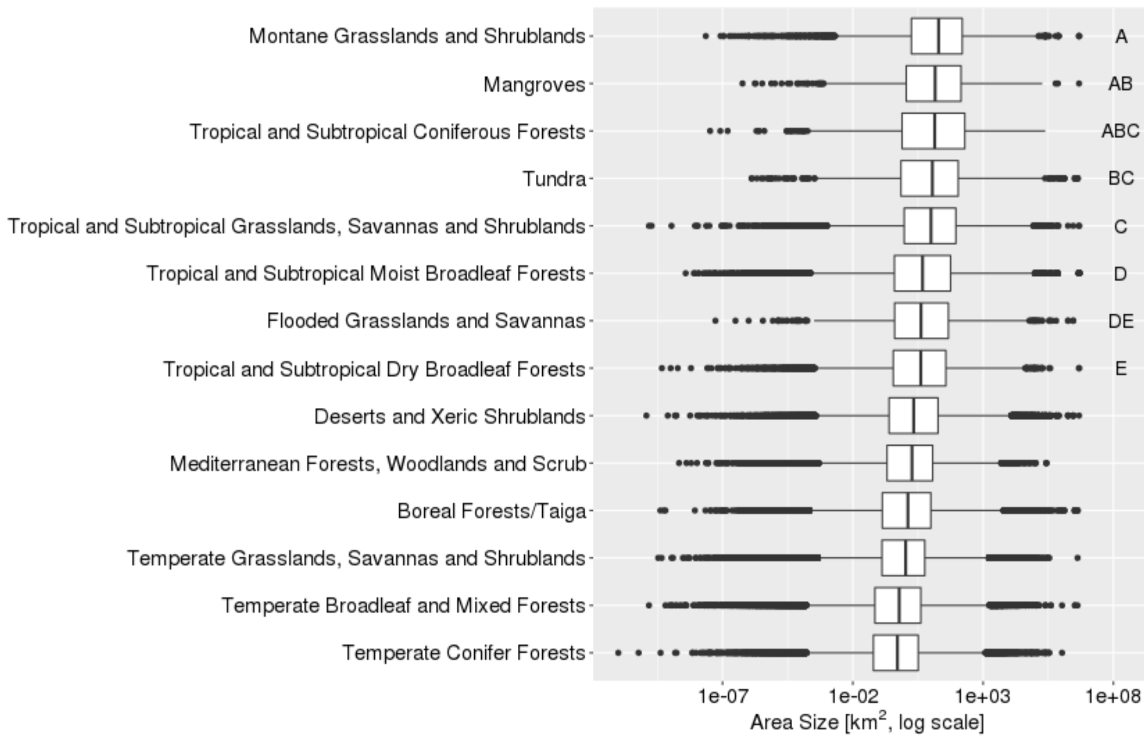

Fig. S11. Size distribution of roadless areas across different biome types assessed with a 1-km road buffer using the OpenStreetMap data set (11/2003) (Pairwise Wilcox test; if biomes share the same capital letters, then corresponding distributions are not significantly different; p<0.001).

SUPPLEMENTARY TABLES (Table S1-S11)

Table S1. Extent of 1-km-buffer roadless areas for Sabah, Malaysia, comparing three different road data sets (OpenStreetMap 11/2013, CIESIN 2013, CIFOR 2014).

	Roadless areas (km²)	**Roadless areas coverage (% of the territory of Sabah)**
Sabah total area	73,841.91	
Roadless areas using OSM data	66,944.69	91
Roadless areas using CIESIN data	68,271.54	92
Roadless areas using CIFOR data	29,700.56	40

Table S2. List of studies documenting or assuming road-effect zones or investigating the spatial influence of road effects. Studies are ordered according to the most important effect described (some studies dealt with more than one effect).

Road type or data	**Study system and location**	**Road effect tested**	**Effect description**	**Spatial range of influence of the road effect**	**Reference**
CHANGES IN ANIMAL ABUNDANCE, DENSITY AND POPULATION SIZE					
Highway, secondary, rural and cyclist road	Polders, farming areas, reclaimed marshland (Netherlands)	Changes in population density of four bird species	Population density increases with distance from the road for black-tailed godwit (*Limosa limosa*) and the lapwing (*Vanellus vanellus*), but not the other species	Up to 1,800 m	*(52)*
Highway	Willow coppices and shrubs (central Netherlands)	Density of territorial males of willow warblers (*Phylloscopus trochilus*)	Lower density of territorial males, lower presence of older males, 50% higher proportion of yearling males and 50% lower success of yearling males in the road zone Total annual output of males/ha 40% lower in the road zone	Road zone assumed as 200 m from the road; intermediate between 200-400 m, and control 400 m	*(53)*
Paved major roads with different traffic volume	Deciduous and coniferous woodland crossed by main roads (Netherlands)	Breeding density of woodland birds	Reduced density in 60% of the species adjacent to roads, due to noise	The maximum reduction of car noise at 200 m from the road The majority of the species (75%) showed maximum effect distances	*(54)*

				between 100 and 1,500 m For all species combined, the effect distances varied between: - 40-1,500 m and 70-2,800 m for roads with 10,000 and 60,000 cars/day, respectively, in deciduous woodland - 50-79 m and 100- 1,750 m for roads with 10,000 and 60,000 cars/day, respectively, in coniferous woodland	
Paved major roads with different traffic volume	Open moist grassland (N and W Netherlands)	Breeding densities of bird species, including waders	Most species had reduced density close to the road; this effect was very strong for the summed density of all species	For the density of all species combined, the disturbance distance was 120 m and 560 m for 5,000 and 50,000 cars/day, respectively. Among species, disturbance distance varied between 20-1,700 m at 5,000 cars/day, and 75- 3,530 m at 50,000 cars/day At 5,000 cars/day, 7 out of 12 species had an estimated population loss of 12-56% within 100 m of roads. At further distances, such reduction occurred in the black-tailed godwit (*Limosa limosa*, 22% in the 0-500 m zone), and the oystercatcher (*Haematopus ostralegus* 44% up to 500 m and 36% for 0-1,500 m zone). At 50,000 cars/day all species showed an estimated population loss of 40-74% within 100 m of the road and >10% at 0-500 m. Five species showed reductions of 14-44% up to 1,500 m	*(55)*
All roads	Rural area (Ontario, Canada)	Effect of traffic on population abundance of green frogs (*Rana clamitans*) and leopard frogs (*Rana pipiens*)	Negative effect of traffic density on leopard frog abundance (more vagile species), but not on green frog abundance	Leopard frog population density negatively affected by traffic density within a radius of 1.5 km	*(56)*
Highway	Desert (California, USA)	Tortoise activity and presence	Tortoise signs increasing with distance from the highway edge	Tortoise populations depressed in a zone extending at least 400 m from the road	*(57)*
Unpaved roads, mostly	Lowland tropical	Abundance of	Most species responded	Effects measures up to 1.2	*(58)*

from oil and logging companies	rainforest (SW Gabon)	mammal species	negatively to roads	km from the road	
Low-traffic road within forest	Deciduous forest (USA)	Change in abundance of salamander species	Reduction in salamander abundance	>35 m	*(59)*
Highway	Protected forest and commercial timberland (Adirondack Mountain, New York, USA)	Impact of road de-icing salts on the reproduction of adults and growth and survival of embryonic and larval of spotted salamander (*Ambystoma maculatum*) and wood frog (*Rana sylvatica*)	High concentration of salt reduced amphibian species survival close to the road (decline of embryo and larvae survival rate) A demographic model predicting population size decrease due to exposure to road salt (embryo and larva mortality effect); stronger effect closer to the road	Salt traveled up to 172 m from the highway into wetlands The negative effect of road salt on population sizes up to 200 m	*(60)*
Highway	Desert (Utah, USA)	Abundance and density of small mammals	No clear abundance, density, or diversity effects relative to distance from the road Species-specific response	No road-effect zone measured up to 400 and 600 m from the road in each of the two study years	*(61)*
All road types and also other infrastructure	Various; meta-analysis of 49 studies on 234 mammal and bird species	Road avoidance and reduced population density of birds and mammals	Mammal and bird population densities declined with their proximity to infrastructure Stronger avoidance in open areas compared to forested areas Habitat- and species-specific response	Up to about 1 km for birds, and up to about 5 km for mammal populations	*(62)*
Paved highway	Boreal forest (Canada)	Population density of brook charr (*Salvelinus fontinalis*) in streams	Population density differed markedly between upstream and downstream sites near highway crossings (of intermediate and low passability)	Up to 800 m from highway	*(63)*
Phantom road	Fir forest and cherry bushes (Idaho, USA)	Simulated traffic noise effect on bird abundance	Serious (25%) decline in bird abundance and almost complete avoidance by some species between noise-on and noise-off periods along the phantom road; such effect was not detected at control sites	Control sites at ca 800 m	*(64)*
Highway	Mountainous area with shrub-steppe vegetation (Ghamishloo Wildlife Refuge, Iran)	Loss of suitable habitat and disruption of the distribution pattern of two ungulate species, the goitered gazelle (*Gazella subgutturosa subgutturosa*) and the wild sheep	51% and 10% of high quality habitat unavailable for gazelle and sheep, respectively, due to road construction Presence points increased with road distance	Large increase in presence at > 3km from the road	*(65)*

		(*Ovis orientalis isphahanica*)			
Highways and national roads	Mediterranean agricultural landscape and cork oak woodland (Alentejo, Portugal)	Likelihood of owl species (barn owls *Tyto alba*, tawny owls *Strix aluco* and little owls *Athene noctua*) occurrence	Higher probability of owl occurrence at longer distance from major roads, particularly for barn owl	Owl presence occurred at further distances (1,591 ± SD 960 m) than absences (1,097 ± SD 826 m)	*(66)*
Paved interstate and county roads	Desert (Mojave, California, USA)	Signs of Mojave Desert tortoise presence (*Gopherus agassizii*)	Tortoise signs increased significantly with distance from roads	Reductions in signs extended farther from the high-traffic interstate than from the smaller, lower-traffic county roads (306 m versus 230 m)	*(67)*
Wide paved and minor unpaved roads	Mediterranean scrubland, dunes and wetlands (Doñana Biosphere Reserve, S Spain)	Presence probability of two ungulates, red deer (*Cervus elaphus*) and wild boar (*Sus scrofa*)	Presence probabilities for both species increased with the distance to the nearest road, in most cases were unpaved roads with negligible traffic volume	At 180 m from the nearest road, wild boar presence probability was lower than 0.2, and for red deer was lower than 0.7	*(68)*
MODIFICATION OF ANIMAL BEHAVIOR					
Highway	Willow coppices and shrubs (central Netherlands)	Breeding dispersal of male willow warblers (*Phylloscopus trochilus*)	Higher proportion of yearlings dispersing and longer dispersal distance in the road-zone	Road zone assumed as 200 m from the road; intermediate between 200-400 m, and control 400 m	*(69)*
Highway and major railroad line	Mountain areas covered mostly with mixed coniferous forest, valleys and prairies (Montana, USA)	Movements of grizzly bears (*Ursus arctos*)	Highway crossing frequency declined exponentially with increasing traffic volume Avoidance of areas close to the highway	Bears strongly avoided areas within 500 m of the highway (asymptote within the 500-600 m category)	*(70)*
Roads in rural areas	Steppe (Patagonia, Argentina)	Flying and feeding behavior of scavenger species	Flying activity and carcass detection was greater near roads (500 m buffer) Andean condors (*Vultur gryphus*) and black-chested buzzard-eagles (*Geranoaetus melanoleucus*) fed far from roads, while other species fed close to roads	Optimal distance for feeding activities for condors and eagles was 3,110 and 10,460 m from the road, respectively, and for the other species, from 218 to 365 m	*(71)*
Paved and unpaved roads	Steppe (Patagonia, Argentina)	Andean condor (*Vultur gryphus*) behavior at carcasses	In the patches far from roads many more condors came to feed, the average time spent per individual was longer, the proportion of time spent vigilant was lower, and the amount of food left uneaten on the carcasses was lower	Up to 350 m	*(72)*

Two-lane roads	Arid shrublands and grasslands (California, USA)	Changes in survival, reproduction, space use, den-site selection, prey availability, and diet of San Joaquin kit foxes (*Vulpes macrotis mutica*)	No effects of the distance to the road on survival, reproduction, litter size, space-use patterns and diet	No effects from 0 m to > 1,760 m from the road	*(73)*
Several types, from highways to unpaved roads	Lentic water bodies including ponds, lakes, dams, and quiet pools within streams (S Victoria, Australia)	Traffic noise effect on the pitch of advertisement calls in two species of frogs, the southern brown tree frog (*Litoria ewingii*) and the common eastern froglet (*Crinia signifera*)	Tree frogs call at a higher pitch in traffic noise and shift the call frequency	Maximum noise at 40 m from highway	*(74)*
Paved roads	Various, review of 25 studies on 13 raptor species	Raptor nest location	Meta-analysis showed an overall positive impact on the displacement of nests from roads Big raptors nesting in trees exhibited greater displacement distances from nests to roads than big raptors nesting in cliffs Distance from nests to roads increase 20–30% compared to control random points	The absolute magnitude of the displacement distance of raptor nests ranged between 200 and 800 m from the road, and 1,400 m for tree nesting raptors of big size, such as large eagles and vultures	*(75)*
Highway and railway line	Mixed woodland (Buunderkamp, Netherlands)	Traffic noise and effects on vocal activity and reproductive success of great tits (*Parus major*)	Traffic noise strongly decreased with distance from the motorway and varied with the time of day, season and weather conditions Noise levels affected negatively the reproductive success of great tits (smaller clutches and fewer fledged chicks in noisier areas)	Average drop of 20 dB SPL in sound levels over less than 500 m from the road Over 400 m from the motorway, mainly bird vocal activity influenced variation in sound levels in the 4 kHz band	*(76)*
Highway	Road verges, bushes, open fields, intermittent trees, woodland (UK)	Bat activity and diversity	Total bat activity, the number of species and the activity of *Pipistrellus pipistrellus* (the most abundant species) were all positively correlated with distance from the road	Activity and diversity increased up to 1.6 km either side of the road	*(77)*
Several road types (paved roads, gravel roads, unimproved roads, truck trails and ATV trails)	Montane ecosystem (Rocky Mountains, Canada)	Alteration of red deer (*Cervus elaphus*) behavior	Deer close to roads decreased their feeding time and increased vigilance and time spent travelling More evident when traffic surpasses 12 vehicles per day	Switch into a more-alert behavior closer than 500 m to roads with more than 12 vehicles/day Twice longer foraging bouts, 20% increase in feeding time, 23% vigilance decrease and 10% decrease in travelling	*(78)*

				time in deer >1 km from roads	
Forest and main roads	Fir–beech forests (Dinaric Mountains, Slovenia)	Home-range size of red deer (*Cervus elaphus*)	Home-range size increased as the distance of main roads from the edge of the home range increased	Home range stabilizes at ca 1,800 m from the road	*(79)*
Highway and dirt roads	Tropical forest in metropolitan area (SE Brazil)	Scavenger removal of experimentally-placed carcasses	High carcass removal for both road categories, with a peak during the day on the highway and at night on dirt roads	Road-effect zone as assumption: >1 km from the highway there is no effect of highway on the carcass removal rate in dirt roads	*(80)*
Forest roads	Scrublands and oak and mixed forests, and portions of natural grasslands, and agricultural areas (central and northern Greece)	Rendezvous site selection by wolves (*Canis lupus*)	Rendezvous sites were located away from forest roads (most important factor at home-range scale)	Wolves selected rendezvous sites farther from forest roads (mean= 435 m, range=73–1,614 m)	*(81)*
Paved and unpaved roads for visitors use	Open grasslands, bush, savanna and woodlands (Kruger National Park, South Africa)	Behavioral response and local spatial distribution of impala (*Aepyceros melampus*)	Impalas change their local spatial distribution near paved and well-traveled roads; unpaved roads largely unaffected their local distribution Greater tolerance distances on paved roads compared to unpaved roads. More flight response in unpaved roads Few flight response (19.5%); habituation may exist	Mean flight distance from the road 30.5 m (range 0–154) vs 35.0 m (range 0–215) for those animals that did not respond. Animals avoid close proximity (first 10 m) to paved roads	*(82)*
REDUCTION OF SPECIES RICHNESS AND DIVERSITY					
Two-lane roads	Mosaic of forest, shrubland and pastures, among 12 cities and close to cities (NW Madrid, Spain)	Abundance and species richness patterns of the native avifauna in fragmented landscape	Total number of bird species, total bird abundance and number of threatened species was negatively influenced by the distance to the nearby roads The abundance of urban-exploiter bird species increased closer to roads	In general, significant threshold distances averaged 300 m for roads, but varied among parameters Mean species richness was lowest <110 m from the road and highest >1,030 m Number of threatened species decreased <400 m from road Highest bird abundance at 290-540 m from the road in deciduous forest areas Abundance of urban exploiters increased if roads <510 m	*(83)*
Paved roads	Wetlands (Southern	Richness of four different wetland	Plant, bird, and herptile species richness diminishes	Strongest relationships at distances up to 1,000 to	*(84)*

	Ontario, Canada)	taxa (birds, mammals, herptiles, and plants)	with increasing density of paved roads on adjacent lands	2,000 m from the wetland edge Critical distance for plants is between 1 and 2 km from the wetland edge; for birds, between 0.5 and 1 km, and for herptiles and mammals at least 2 km	
Unpaved forest roads	Forest (S Appalachian Mountains, Tennessee, USA)	Abundance and richness of the macroinvertebrate fauna of the soil and leaf-litter depth	Reduced both the abundance and the richness of the macroinvertebrate soil fauna and the depth of the leaf-litter	Effects on faunal abundance and leaf-litter depth up to 100 m into the forest (max distance tested), whereas persists to 15 m	*(85)*
Unpaved forest roads	Temperate deciduous forest (USA)	Change in the distributions of understory plants, and site variables (species cover, canopy cover, litter depth and cover, and bare ground)	Richness and diversity of native species were lower on roadsides Exotic species were most prevalent near roads Roads created a disturbance corridor that affected site variables	Native species richness back to normal levels after 5 m distance Prevalence of exotic species and effects on site variables up to 15 m	*(86)*
Highways (plus other anthropogenic barriers)	Desert regions (California, USA)	Genetic diversity in metapopulation of desert bighorn sheep (*Ovis canadensis nelson*)	Reduction in the relative gene flow among study populations Decline in genetic diversity at a rate of 0.4% per year	Barrier effect distance (at which relative gene flow decrease equivalently) estimated at c. 40 km	*(87)*
Several road types (highway, paved rural road, unpaved dirt road)	Second-growth forest (Orange County, New York, USA)	Diversity, abundance and species density of carrion beetles	No consistent effects of distance from road on the diversity, abundance or species density of beetles across road types Forests near highways and paved rural roads were less diverse than near dirt roads	No effect up to 120 m from the roads (suggestion that road effect can permeate further)	*(88)*
Highway	Rural area (Ontario, Canada)	Anuran species richness and relative abundance for seven species	Species richness and abundance declined closer to the road Suggestion that new roads should be at least 500 m from wetlands (conservative estimate of the road-effect zone for species richness), but greater buffer distances recommended (at least 3,000 m for leopard frogs *Rana pipiens*)	Road-effect zones of 250–1,000 m for four of seven species and species richness, and well beyond 1,000 m for two species. Breakpoint at approximately 450-800 m from the highway for species richness; 200-300 m for the spring peeper (*Pseudacris crucifer*), American toad (*Bufo americanus*), and gray treefrog (*Hyla versicolor*); 600–1,000 m for the wood frog (*Rana sylvatica*); and 1,100 to 2,400 m for the chorus frog (*Pseudacris triseriata*)	*(89)*

High-traffic paved roads	Boreal forest (Canada)	Change in breeding bird occurrence	Bird species richness increased with increasing distance from roads Traffic noise declined with distance from the roads	Bird species richness reached a maximum at about 350 m from the road Traffic noise reached a minimum at about 450 m from the roads	*(90)*
Low-traffic unpaved roads	Tropical rainforest (Amazon, Ecuador)	Change in species richness and diversity of amphibians, butterflies and birds	Amphibian richness and understory bird richness and diversity decreased near roads Butterfly and overall diurnal bird richness increased near roads Taxon-specific response to roads	Up to 200 m from the road for butterflies, up to 250 m for amphibians and up to 350 m for birds	*(91)*
PROMOTION OF INVASIVE SPECIES					
Paved roads	Grasslands (California, USA)	Native and exotic plant diversity	In non-serpentine grasslands the percentage cover by native species, the percentage of species that were native, and the number of native grass species increased with distance from roads, while the cover by exotic species and number of exotic forb species decreased No effect of road proximity in serpentine grasslands	Native cover was greatest in sites >1,000 m from roads (23%) and least in sites 10 m from roads (9%) Percentage of species that were native was significantly greatest in sites >1,000 m from roads (44%) and least in those 10 m from roads (32%)	*(92)*
Paved roads	Grasslands (California, USA)	Survival and biomass of the invasive plant yellow starthistle (*Centaurea solstitialis*)	In non-serpentine grasslands, *Centaurea* survival and biomass was greater in sites closer to roads No effect of road proximity on the performance of planted *Centaurea* on serpentine soil	Survival and biomass greater in near (10 m) than in distant (>1,000 m) plots	*(93)*
All types, from highways to dirt roads, typically two-lane dirt and paved roads	Mature sugar maple-dominated forests (W Great Lakes, Minnesota and Wisconsin, USA)	Extent and patterns of earthworm invasion	Distance to the nearest road was the best predictor of earthworm invasion in Wisconsin Negative relationship between the distance to the nearest road and the presence of four taxonomic groups, except *Dendrobaena* which had positive	The invasion of the *Lumbricus–Aporrectodea* assemblage generally extends nearly 1,200 m from roads. The probability of occurrence does not decline below 50% until 470-930 m, and to 5% until the nearest road is > 1,300 m away Probability of finding *Dendrobaena* alone increases with road distance crossing 50% at >1,540 m.	*(94)*
Paved and forest roads	Deciduous forest (Maryland, USA)	Presence and percent cover of invasive plant species	More invasive species close to roads; sites containing three or more invasive species observed along paved roads Spread rates are higher in roadsides; roadside populations occupied a larger patches and expand more	Effects measured up to 150 m from the road; the range of influence is greater following the spread of the species	*(95)*

			rapidly		
High, medium and low traffic roads	Dry deciduous forest (India)	Presence of invasive plants	Increase in the presence of invasive plant species near roads, especially in medium and high traffic roads	Up to 100 m (not measured further)	*(96)*
Primary roads	Terrestrial, freshwater and marine ecosystems (NW Europe, encompassing Great Britain, France, Netherlands and Belgium)	Distribution of invasive species (72, including 17 terrestrial plants, 19 terrestrial animals, 17 freshwater and 19 marine organisms)	Roads promote the dispersal of non-native species Proximity to roads was a particularly important driver for plant species distribution	Maximum probability of invasion of two plants, the Kudzu (*Pueraria lobata montana*) and Kahili ginger (*Hedychium gardnerianum*) within 2 km from roads	*(97)*
INDUCING DEFORESTATION					
Highways	Tropical rainforest (Amazon, Brazil)	Deforestation through forest conversion to crops, pastures and secondary forest	Deforestation has claimed 29-58% of the forests within 50 km of paved roads	More than two-thirds of Amazon deforestation within 50 km of major paved highways	*(98)*
Highways and unpaved roads	Tropical rainforest and adjoining woodlands and savannas (Amazon, Brazil)	Deforestation	Proximity to roads, particularly to highways, increased deforestation	Deforestation rose mostly sharply within 50-100 km of highways and within 25-50 km of unpaved roads	*(99)*
Paved and unpaved roads	Tropical rainforest (Amazon, Brazil)	Deforestation spillover	Deforestation rises in sites that lack roads but are in the same county as site with a new paved or unpaved road	100 km	*(100)*
State and federal roads, some private roads	Tropical rainforest (Amazon, Brazil)	Deforestation fires (measured by hot pixels)	Exponential declines in hot pixel frequency with increasing distance from roads Fewer deforestation fires within protected areas than outside	Almost 90% fires were ≤10 km from roads	*(101)*
Paved and unpaved roads	Tropical rainforest (Southern Amazon, Peru, Brazil, Bolivia)	Deforestation	Deforestation rates drop with distance from major roads, although the distance before this drop off appears to relate to degree of road paving at regional level	45 km for roads where paving is complete; 18 km where paving is underway	*(102)*
Highway	Cerrado Savannas (Brazil)	Deforestation and habitat degradation	Deforestation increases closer to the roads, with pasture growing near the road, and forest cover growing further away	32.6% loss of Cerrado up to 9 km from the highway	*(103)*
Official and unofficial roads	Tropical rainforest (Amazon,	Deforestation	Deforestation was much higher near roads Protected areas near roads had	Nearly 95% of all deforestation occurred within 5.5 km of roads	*(104)*

	Brazil)		lower deforestation than did unprotected areas near roads	Highways begin to have a rapidly diminishing influence only at 32 km	
CHANGE OF LANDSCAPE PATTERNS AND FRAGMENTATION					
All road network, mainly composed of minor roads	17 townships across three ecoregions of forested landscapes (n. Wisconsin, USA)	Changes in landscape patterns and road density in a six-decade study period	Substantial changes in landscape patterns Road density doubled and the immediate area affected by roads increase twofold (5% to 10%). Reduction of median, mean and largest roadless patch size by a factor of four. Increases in housing density and fragmentation	Road-effect zone as assumption: 15 m	*(105)*
FACILITATION OF RESOURCE EXTRACTION AND HUNTING					
Road for oil extraction and access from rivers	Amazon Basin (Yasuní Biosphere Reserve, Ecuador)	Probability of hunting by the Waorani indigenous group	Spatial extent of hunting doubled in the presence of road, and include remote areas	Mean distance walked from a point of access (road, river) to a kill site was 1.36 km (SD=1.18), and the maximum distance was 7 km (99% records <5 km)	*(106)*
NOISE INCREASE					
Busy roads (and other sources of noise)	Various (review paper)	Effect of noise (sound pressure level) on response curve of species occupancy (general model)	Spatial propagation of elevated noise levels from a point source (such as a single car, which decays at a spreading loss of 6 dB or more per doubling of distance, line sources (such as a busy highway) lose only 3 dB per doubling of distance	The sound pressure level of noise decreases with increasing distance but may not reach "baseline" ambient levels until ~1 km away (this distance will vary depending on noise source and the environment)	*(107)*
VARIOUS					
Highway	Suburban landscape, including swamps, streams, wetlands, deciduous forest, open-fields, residential areas (Massachusetts, USA)	Alteration of streams, wetland drainage, road salt reaching water bodies, invasion by exotic species, changes in habitat and movement patterns of large mammals such as moose *Alces alces* and deer *Odocoileus virginianus*, forest and grassland birds, and amphibians	The effects of all factors extended >100 m from road. Moose corridors, road, avoidance by grassland birds and road salt extended >1 km	The road-effect zone averages approximately 600 m wide and is asymmetric	*(108)*
Highways, secondary and	Various, all	Estimation of the percentage of	One-fifth of the U.S. land area is ecologically affected by	Road-effect zone as	*(109)*

primary roads	USA	land ecologically affected by the public road system	public roads system	assumption: primary roads (10,000 vehicles/day): 305 m in woodland and 365 m in grassland primary roads (50,000 vehicles/day): 810 m in natural ecosystems in urban areas secondary roads: 200 m	

Table S3. Extent and amount of roadless areas (5-km-buffer) per continent using the OpenStreetMap data set (11/2003) (without Antarctica, Greenland, and freshwater bodies).

	Asia	**Africa**	**North America**	**South America**	**Europe**	**Australia**	**Oceania**	**Global land**
Total area (million km²)	44.32	29.70	21.51	17.64	9.75	7.64	0.43	130.00
Total roadless area cover (million km²)	28.62	19.36	9.88	11.09	1.30	5.09	0.11	75.45
Percentage of roadless coverage (%)	64.58	65.19	45.93	62.89	13.33	66.62	25.58	58.04

Table S4. Extent and amount of roadless areas (1-km buffer) per continent using the OpenStreetMap data set (11/2003) (without Antarctica and Greenland, and freshwater bodies).

	Asia	Africa	North America	South America	Europe	Australia	Oceania	Global land
Total area (million km²)	44.32	29.70	21.51	17.64	9.75	7.64	0.43	130.00
Total roadless area cover (million km²)	38.83	26.53	13.20	15.52	4.06	6.75	0.27	105.16
Percent roadless cover	87.60	89.30	61.39	88.00	41.64	88.26	63.87	80.28
Mean roadless area patch size (km²)	308.69	522.51	59.69	418.07	47.85	248.58	47.85	176.94
Maximum roadless patch size (million km²)	4.23	2.88	3.33	4.82	0.24	0.27	0.03	4.82
Median roadless patch size (km²)	2.85	6.75	0.48	4.81	0.85	2.98	0.84	1.07
Total no. roadless patches	101,992	50,770	221,197	37,124	153,323	24,216	5,691	594,312
No. roadless patches >1 km²	63,555	36,223	86,112	24,817	73,148	15,673	2,699	302,227
No. roadless patches >5 km²	43,854	27,237	36,787	18,420	40,268	10,178	1,463	178,207
No. roadless patches >10 km²	35,274	22,864	23,502	15,431	28,363	7,782	1,073	134,289
No. roadless patches >50 km²	18,356	12,992	7,609	9,189	9,561	3,223	453	61,383
No. roadless patches >100 km²	13,124	9,505	4,580	6,893	5,210	2,055	295	41,662
No. roadless patches >1000 km²	3,077	2,187	769	1,653	432	539	49	8,706

Table S5. Rationale of indicators used for *Ecological Value Index of Roadless Areas* (EVIRA).

Indicators	Rationale	Description
Roadless area patch size	Large roadless areas provide a much wider range of ecological benefits than smaller ones where road edge effects impact a larger share of the roadless patch (see Table S2).	Habitat fragmentation and corresponding negative environmental changes have been extensively treated in many studies (a comprehensive overview is given by Bennett et al. (2010) *(110)*. The impacts do not just relate to gene flow, population viability and loss of (less dispersive) species in habitat fragments, but also to ecosystem functioning. For example, there is certain evidence related to nutrient cycling, dung removal, pollination, and seed dispersal *(111)*. "The impacts of fragmentation on ecosystem functioning are often exacerbated by synergistic effects such as interactions with the matrix and increased hunting pressure in fragmented forests" *(111)*. There is growing evidence that certain species avoid areas with even minimal anthropogenic disturbance *(112, 113)*, which is another argument for conservation of large roadless areas. Especially in tropical regions, many species exist at rather low population densities, are seasonal migrants (often across different altitudinal belts) following scarce resources, or otherwise require large habitats for maintaining viable populations *(114-116)*.
Thiessen connectivity into all directions for roadless area patches	The larger the Thiessen connectivity value, the closer neighboring roadless patches can be found. This is important for the integrity of ecological landscape-scale processes (e.g., genetic exchange of populations confined to roadless areas).	Roaded forest ecosystems, for instance, are far more vulnerable than intact ones to predatory logging, wildfires, illegal mining, exotic species invasions, and other anthropogenic threats *(7, 114)*.
Ecosystem Functionality Index comprising the following sub-indicators:	Ecosystem Functionality is defined as the state of ecosystems, characterized by inherent structures, ecological functions and dynamics, that provide ecosystems with both, the necessary efficiency and resilience to develop without abrupt change of system properties and geographical distribution, and allows for flexible response to external changes.	This Ecosystem Functionality Index has been published by Freudenberger et al. (2012a) *(38)*.
- Vegetation density	Vegetation density is an indicator for biomass and the ecosystems' ability to dissipate incoming solar energy. Furthermore, a higher number of primary producers increase the capture of solar energy thereby improving ecosystem functionality.	Rationale from Freudenberger et al. (2012a, b) *(9, 38)*. Further references and sources provided in the corresponding methods sections.
- Tree height	Tree height is used as an indicator for biomass as well as structural complexity of an ecosystem. Old-growth forest conditions and complex vegetation stratification including foliage layering is dependent on tree height, thereby enhancing biodiversity and ecosystem functioning. Furthermore, it plays an important part in the absorption of solar radiation and in moderating microclimatic conditions.	Rationale from Freudenberger et al. (2012a, b) *(9, 38)*. Further references and sources provided in the corresponding methods sections.
- Carbon storage	Carbon storage is considered as an indicator for biomass and the ability of ecosystems to dissipate incoming solar energy. Areas with	Rationale from Freudenberger et al. (2012a, b) *(9, 38)*. Further references provided in the corresponding methods sections.

	higher carbon storage are also characterized by more intensive interactions with the atmosphere and higher regulating capacity.	
- Species richness of vascular plants	Species richness is considered to represent functional and structural redundancy, which is relevant for the resistance and resilience of ecosystems to e.g. climate change. Additionally, species richness is also associated with complex trophic structure and higher cycling rates of biomass, energy and information.	Rationale from Freudenberger et al. (2012a, b) *(9, 38)*. Further references and sources provided in the corresponding methods sections.
- Plant functional richness	Plant functional richness is an indicator derived from modelling survival probabilities of different plant functional types under climate change. Ecosystems with higher functional species richness are more likely to adapt to environmental change and therefore increase the adaptive capacity of an ecosystem.	Rationale from Freudenberger et al. (2012a, b) *(9, 38)*. Further references and sources provided in the corresponding methods sections.
- Slope	Topographical heterogeneity is connected to habitat diversity and species richness. At macro-scale habitat diversity increases along altitudinal gradients. Geographical barriers increase opportunities for allopatric speciation, and contribute to the genetic information that is stored within an ecosystem.	Rationale from Freudenberger et al. (2012a, b) *(9, 38)*. Further references and sources provided in the corresponding methods sections.

Table S6. Pearson (dark grey) and Spearman rank (light grey) correlation coefficient matrix for the three indicators of the ecological value index for roadless areas (EVIRA). All correlation coefficients are highly significant with $p<0.0001$. Correlation coefficients with values higher than 0.7 are displayed in bold.

	Ecological value index of roadless areas (EVIRA)	Roadless area patch size	Thiessen connectivity into all directions	Ecosystem functionality index (EFI)
Ecological value index of roadless areas (EVIRA)	1.000	**0.768**	-0.005	**0.818**
Roadless area patch size	0.488	1.000	-0.006	0.260
Thiessen connectivity into all directions	-0.272	-0.875	1.000	-0.002
Ecosystem functionality index (EFI)	**0.881**	0.155	0.048	1.000

Table S7. Distribution of roadless areas (1-km buffer) across anthromes (km^2) (according to Ellis et al. 2010 (*10*); analysis based on OpenStreetMap data set 11/2013).

Anthrome classes	South America	Central and North America	Europe	Asia	Africa	Australia	Oceania	Global	Share of global roadless areas (%)
Urban	4,007	4,387	2,374	32,332	9,058	706	263	53,129	0.05
Mixed settlements	18,372	18,749	5,295	233,664	93,038	1,070	1,556	371,746	0.36
Rice villages		444		1,561,288	358			1,562,090	1.50
Irrigated villages	9,099	18,415	8,092	917,304	31,193			984,105	0.94
Rainfed villages	48,983	70,791	48,853	1,307,198	514,561		85	1,990,474	1.91
Pastoral villages	67,829	16,127	1,748	233,641	195,302			514,649	0.49
Residential irrigated croplands	34,121	50,856	52,030	401,213	47,493	497	191	586,40	0.56
Residential rainfed croplands	453,081	324,541	779,233	2,209,022	1,853,242	7,405	6,051	5,632,575	5.39
Populated croplands	567,180	302,940	531,100	1,484,977	606,286	70,433	15,408	3,578,326	3.43
Remote croplands	161,957	345,517	21,507	360,306	135,530	391,144	7,822	1,423,783	1.36
Residential rangelands	1,252,057	177,381	62,984	1,404,975	3,314,670	9,205	2,844	6,224,116	5.96

Populated rangelands	2,800,656	572,493	261,741	3,430,646	4,634,380	67,350	27,188	11,794,455	11.29
Remote rangelands	2,214,349	737,996	94,936	5,999,912	2,294,862	6,047,983	76,368	17,466,406	16.72
Residential woodlands	230,507	141,898	106,706	1,322,994	1,343,634	4,246	20,478	3,170,464	3.04
Populated woodlands	1,464,277	490,479	709,214	2,397,132	2,134,048	29,333	60,523	7,285,006	6.97
Remote woodlands	2,182,821	485,807	201,057	1,241,981	448,189	29,731	27,679	4,617,265	4.42
Inhabited treeless and barren lands	781,593	248,646	49,804	2,183,217	1,665,865	508	1,056	4,930,688	4.72
Wild woodlands	2,710,257	5,929,872	829,528	7,534,326	332,290	71,611	17,868	17,425,751	16.68
Wild treeless and barren lands	484,370	2,976,033	171,235	4,345,674	6,858,975	1,771	444	14,838,501	14.21

Table S8. Protection status of roadless areas (1-km buffer) per continent (without Antarctica, Greenland, and large freshwater bodies) based on WDPA 2014 and OpenStreetMap (11/2003).

	Asia	Africa	North America	South America	Europe	Australia	Oceania	Global land
Protected areas cover (all categories) (km^2)	4,977,721	4,112,914	2,646,754	4,087,773	1,510,183	1,196,688	93,123	18,625,157
Protected area cover (%)	11.2	13.8	12.3	23.2	15.5	15.7	21.8	14.2
Roadless areas in IUCN categories (km^2)	3,989,458	2,056,657	2,146,627	2,364,065	410,437	1,074,445	72,177	12,113,866
Percent IUCN coverage of roadless areas	9.0	6.9	10.0	13.4	4.2	14.1	17.0	9.3
Strictly protected areas (IUCN I & II) (km^2)	1,029,356	1,028,218	1,511,100	997,502	272,877	589,763	33,848	5,462,664
Strictly protected areas (IUCN I & II) (%)	2.3	3.5	7.0	5.7	2.8	7.7	7.9	4.2
Roadless areas in strictly protected areas (IUCN I & II) (km^2)	966.322	969.151	1.370.853	974.208	180.903	525.068	28.492	5.014.999

Roadless areas strictly protected (IUCN I & II) (%)	2.2	3.3	6.4	5.5	1.9	6.9	6.7	3.8
Protected areas (IUCN III-VI) (km²)	3,215,796	1,194,583,55	1,006,467,51	1,450,552,58	701,944,89	581,476,89	54,291,11	8,205,112,91
Protected areas (IUCN III-VI) (%)	7.3	4.0	4.7	8.2	7.2	7.6	12.7	6.3
Roadless areas in protected areas (IUCN III-VI) (km²)	3,023,136	1,087,506	775,773	1,389,857	229,534	549,377	43,683	7,098,867
Roadless areas in protected areas (IUCN III-VI) (%)	6.8	3.7	3.7	7.9	2.3	7.2	10.2	5.4

Table S9. Extent and coverage of roadless areas of 1-km buffer under strict protection (IUCN I-II) category, according to their Ecological Value Index of Roadless Areas (EVIRA) using the OpenStreetMap data set (11/2003).

EVIRA values	North America (km^2)	South America (km^2)	Asia (km^2)	Africa (km^2)	Europe (km^2)	Australia (km^2)	Oceania (km^2)	Global (km^2)
0 - 13	0	0	0	0	0	0	0	0
14 - 28	109,7	8,0	5,430	6,092	1,700	50,525	2.2	63,868
29 - 33	86,441	9,367	98,425	269,842	2,042	274,650	855	741,622
34 - 37	81,286	20,640	108,467	201,490	13,496	82,089	36	507,500
38 - 42	75,476	45,810	81,685	240,560	44,801	29,444	106	517,883
43 - 47	454,357	64,917	100,975	85,371	66,762	23,597	417	796,396
48 - 53	204,952	151,089	173,866	50,750	40,796	11,856	15,446	648,755
54 - 58	444,939	132,629	147,985	88,619	7,878	34,984	8,074	865,107
59 - 64	17,582	31,144	105,544	25,579	2,437	16,871	3,466	202,623
65 - 80	3,617	518,198	143,008	0.0	227	82	0.3	665,132
Sum	**1,368,760**	**973,802**	**965,384**	**968,299**	**180,140**	**524,099**	**28,401**	**5,008,886**

Table S10. Synergies and conflicts between conservation of roadless areas and the United Nations' Sustainable Development Goals (SDGs) and their corresponding targets. Left column: Assessment of goals (large boxes): grey: at most weak synergies and conflicts with goal, blue: conflicts with goal prevail, yellow: mixture of synergies and conflicts with goal, green: synergies with goal prevail. Assessment of targets (insert boxes): grey: not applicable, blue: conflict, yellow: ambivalent relationship, green: synergy. Numbers in italics: target numbers. Bold number at bottom: conflict-synergy score of goals. →: reference to target(s).

Sustainable Development Goals and targets	**Brief analysis of synergies and conflicts between conservation of roadless areas and Sustainable Development Goal targets**
Goal 1. End poverty in all its forms everywhere *1* *2* *3* *4* *5* **-0,5** Compare AICHI BIODIVERSITY TARGETS 2, 14.	**Synergies:** The SDGs explicitly acknowledge the importance of integrating ecosystem and biodiversity values into poverty reduction strategies and accounts (compare to 15.9). In remote areas inhabited by indigenous or traditional people in the developing world, where governance is weak, road development may trigger uncontrolled frontier expansion and associated poverty. In the Amazon, frontier expansion through road construction has fostered large-scale economic activities (e.g. oil extraction, livestock and soy production), but often at the expense of the local communities. Road development in the region is associated to dire conflicts over land and natural resources *(117, 118)*. A better planning of the road development process and a prioritization of roadless areas for conservation purposes can help to reduce risks related to poverty (→ targets 1.1, 1.2, 1.4). In the Amazon, for instance, a more sensitive proposed development strategy should focus on strengthening governance in areas where roads have been established for a long time (and human population is relatively large and human development indices are low), while leaving more remote areas roadless or with roads unpaved *(119)*. Functional ecosystems, as they exist in roadless areas, effectively reduce human exposure to environmental shocks and disasters, including climate-related extreme events (such as floods: e.g., *(120)*, water scarcity: e.g., *(121)*, compare goal 6, fires: e.g., *(122);* → target 1.5). It is of great importance to maintain ecosystem functionality on the landscape scale, e.g. by prioritizing conservation of roadless areas around the headwaters of rivers against extreme fluctuations in run-off along the densely populated and intensively managed tailwater.
	Conflicts: Poverty often is related to the lack of access to markets and employment options (compare goal 8), health (compare goal 3) and education infrastructure (compare goal 4; *(123-126)*). Case studies have shown how roads significantly reduce poverty and increase consumption growth (→ targets 1.1, 1.2, 1.4; *(127-129)*). Reduced mobility also hampers organizational capacities, especially in remote rural areas, where it is difficult for poor people to meet and coordinate activities. In general, poor people will ask for better roads and mobility. Goal 9 explicitly addresses the relevance of infrastructure (see below). The conservation of roadless areas seems to represent a serious conflict and obstacle to development – if this development is thought along conventional lines and without exploring more sustainable alternatives for providing mobility.

Goal 2. End hunger, achieve food security and improved nutrition and promote sustainable agriculture *1* *2* *3* *4* *5* **-0,6** Compare AICHI BIODIVERSITY TARGETS 7, 8, 14.	**Synergies:** In remote regions, as they are found in parts of the western Amazon forests, the subsistence of many indigenous communities depends on forest products. However, new roads built into previously remote areas of low human population density have often triggered conversion of forest to croplands and pastures *(130)* and unsustainable exploitation of wildlife that can then be marketed easily as bushmeat in cities. Bushmeat can thus become scarce for residents who rely on this protein source *(131, 132)*. Functional ecosystems, as they exist in roadless areas, effectively reduce human exposure to environmental shocks and disasters, including climate-related extreme events (→ target 2.4; compare goal 1).
	Conflicts: At many places of the world, undernourishment increases with distance from roads and with it from markets and health services, among others *(133)*. Hunger can also be promoted by limited options for reaching poor rural people with food aid and development assistance (*(134)*; → targets 2.1-2.3, 2.5; compare goals 1, 3, 4, 6, 9).
Goal 3. Ensure healthy lives and promote well-being for all at all ages *1* *2* *3* *4* *5* *6* *7* *8* *9* **-0,1** Compare AICHI BIODIVERSITY TARGETS 6, 14.	**Synergies:** In general, roadless areas guarantee high ecosystem functionality (compare goal 1) and with it a variety of ecosystem services that are fundamental to people's health. Among others, tropical forest-dwelling indigenous communities use a variety of medicinal forest plants that can become scarce in the course of road construction and subsequent deforestation *(135)*. Roadless areas exclude deaths and injuries from road traffic accidents (→ target 3.6) as well road and traffic-related hazardous chemicals and air, water and soil pollution and contamination (*(136, 137)*; → target 3.9; compare goal 6). Road development in the Amazon and Indonesia has been shown to be associated with the spread of diseases (*(117);* → target 3.3). Abrupt contact with modern life-styles via new roads increases the vulnerability of formerly remote human populations to drug abuse and alcohol consumption (*(138);* → target 3.5).
	Conflicts: Remote rural populations mostly have reduced access to health care and medical assistance (*(133);* → targets 3.1, 3.2, 3.4, 3.7, 3.8).
Goal 4. Ensure inclusive and equitable quality education and promote lifelong learning opportunities for all *1* *2* *3* *4* *5* *6* *7* **-0,9** Compare AICHI BIODIVERSITY TARGETS 1.	**Synergies:** Experiencing wilderness has become an important element of education. While roadless areas are less accessible by motorized ways, they provide opportunities for this kind of education (*(139)* compare goal 8: nature tourism).
	Conflicts: With increasing distance from roads, access to "quality" education becomes more difficult. Among others, remote rural populations often lack literacy in the use of emerging technological devices (computers, internet etc., *(140);* → targets 4.1-4.7).
Goal 5. Achieve gender equality and empower all women and girls	**Synergies and conflicts: -**
Goal 6. Ensure availability and sustainable management of water and sanitation for all *1* *2* *3*	**Synergies:** Roads significantly harm the integrity and functionality of ecosystems and several services they provide to people (compare goal 1). Roads (including their construction) and traffic have been known for a long time as a source for water pollution (*(141);* → targets 6.1, 6.3, 6.5, 6.6).

4 5 6 **0,4** Compare AICHI BIODIVERSITY TARGETS 6, 8, 14.	**Conflicts:** In general, remote rural populations often have reduced access to technology, infrastructural development and assistance. It is cost-efficient, and practical for maintenance, to install water and sewer systems in the course of road construction (→ targets 6.1, 6.2).
Goal 7. Ensure access to affordable, reliable, sustainable and modern energy for all 1 2 3 **-0,5** Compare AICHI BIODIVERSITY TARGETS 7, 8, 14.	**Synergies:** none. **Conflicts:** In general, remote rural populations often have reduced access to technology and infrastructural development (compare goal 6). Electric wires can relatively easily be installed and maintained along roads (→ targets 7.1, 7.2). However, small-scale renewable (solar, wind) energy plants can be an alternative with additional advantages (low cost, energy autonomy; → target 7.2).
Goal 8. Promote sustained, inclusive and sustainable economic growth, full and productive employment and decent work for all 1 2 3 4 5 6 7 8 9 10 **-0,3** Compare AICHI BIODIVERSITY TARGETS 2.	**Synergies:** Roadless areas can contribute substantially to slowing down environmental degradation (→ target 8.4; compare goal 15, 13). In addition, certain micro- and small enterprises can arise in spite of relatively great distances from roads (→ target 8.3) – or even depend on remoteness (nature tourism, e.g., *(142);* → target 8.9). It has been shown for the Amazon region that road development is associated with slave labor (*(118);* → target 8.7). Facilitated access to markets by roads may not always improve the income levels of poor people, as they will not be able to afford goods such as cars and petrol. **Conflicts:** Ease of mobility of people and goods promotes economic productivity and growth (*(143)*; → targets 8.1, 8.2; compare goals 9, 1). Young people of remote rural areas mostly have reduced access to good education and training opportunities (compare goal 4) and subsequently lower chances on the labor market (*(144);* → target 8.6).
Goal 9. Build resilient infrastructure, promote inclusive and sustainable industrialization and foster innovation 1 2 3 4 5 **-0,3** Compare AICHI BIODIVERSITY TARGET 2.	**Synergies:** Upgrades of roads in the existing network can be a cost-efficient and environmentally less problematic alternative to building new roads (*(4);* → target 9.4). **Conflicts:** Economic development, especially in developing economies or those in transition, depends on an effective road network (*(143)*; → targets 9.1, 9.2; compare goals 8, 1).
Goal 10. Reduce inequality within and among countries 1 2 3 4 5 6 7 **-0,7**	**Synergies:** none. **Conflicts:** Modern road traffic has increased the mobility of people and goods, but comes with an increased risk of accidents (*(145);* → target 10.7). Roads have a variety of homogenizing effects - in terms of biological diversity (e.g., aided dispersal of invasive species: *(146)*, culturally (*(147);* → target 10.2) etc. Economically, road building provides poor rural societies a better access to economic dynamics and is thus a standard element of economic development strategies (*(143)*; → target 10.1; compare goals 9, 8, 1).

Goal 11. Make cities and human settlements inclusive, safe, resilient and sustainable *1* *2* *3* *4* *5* *6* *7* **0,5** Compare AICHI BIODIVERSITY TARGET 14.	**Synergies:** "Indigenous peoples in voluntary isolation" request participation in road and human settlement planning and want to be exempted from any such development *(117)*. Targeting roadless areas will help concentrate development in urban areas and their immediate surroundings (*(105);* → target 11.3). Failing to do so regularly results in "contagious development", i.e., unleashing a positive feedback of road construction and intensive land-use in a formerly road-free landscape *(4, 7)*. Remote areas, which provide vital ecosystem services to cities, can thus be kept functioning (→ target 11.5; compare goal 13, 1, 2). The status of natural heritage sites ("Criteria for the assessment of Outstanding Universal Value": vii, ix and x; *(148)*) is vitally coupled with remoteness (→ target 11.4). **Conflicts:** Further road construction may be deemed necessary to provide convenient access to public transport for a larger part of the population. However, people in remote rural regions may not be able to pay for public transport (*(149);* → target 11.2).
Goal 12. Ensure sustainable consumption and production patterns *1* *2* *3* *4* *5* *6* *7* *8* **1,0** Compare AICHI BIODIVERSITY TARGET 4.	**Synergies:** Road construction and maintenance consume significant amounts of material (and energy) and thus enlarge the national and per capita material footprint (*(150);* → target 12.2). Including roadless and other important areas for biodiversity and ecosystem services for people would make sustainability reports of companies *(151)* more diagnostic and could thus provide guidance for the adoption of sustainable practices (→ target 12.6). **Conflicts:** none.
Goal 13. Take urgent action to combat climate change and its impacts *1* *2* *3* **1,0** Compare AICHI BIODIVERSITY TARGETS 15, 10, 14.	**Synergies:** Functional ecosystems, as they exist in roadless areas, strengthen the resilience and adaptive capacity of human societies to climate-related hazards and natural disasters (→ target 13.1; compare goals 1-3). Roadless areas conservation would thus form a meaningful element of policies, strategies and planning for climate change adaptation (*(2);* → target 13.2). Road construction and maintenance (with cement production being a relevant source of greenhouse gas emissions *(152)*) as well as traffic *(153)* also contribute large shares to overall greenhouse gas emissions. Policies, strategies and planning for climate change mitigation should therefore strive to reduce these activities to the lowest level possible (→ target 13.2). **Conflicts:** none.
Goal 14. Conserve and sustainably use the oceans, seas and marine resources for sustainable development *1* *2* *3* *4* *5* *6* *7* **1,0** Compare AICHI BIODIVERSITY TARGET 6.	**Synergies:** Considerable river sediment loads can result from road construction and erosion along roads *(121)*. Runoff from subsequent development, such as logging in mountain areas *(154)*, or agriculture, can also impact rivers and, finally, estuaries and near-coast marine waters (→ target 14.1). **Conflicts:** none.

Goal 15. Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss *1* *2* *3* *4* *5* *6* *7* *8* *9* **1,0** Compare AICHI BIODIVERSITY TARGETS 5, 11, 15, 12, 10.	**Synergies:** The conservation of roadless areas represents an effective and inexpensive means to conserving terrestrial and inland freshwater biodiversity and ecosystem services (*(2, 4);* → targets 15.1, 15.4, 15.5, 15.7, 15.8). This includes halting deforestation (*(98);* → targets 15.2) and combating desertification (*(155);* → targets 15.3). The inclusion of roadless areas would be a meaningful contribution to integrating ecosystem and biodiversity values into national and local planning as well as development processes, as is already the case in the United States of America and Germany (*(2, 4);* → targets 15.9). The present study demonstrates roadless areas are a tangible and transparent indicator for environmental accounting (→ target 15.9). **Conflicts:** none.
Goal 16. Promote peaceful and inclusive societies for sustainable development, provide access to justice for all and build effective, accountable and inclusive institutions at all levels *1* *2* *3* *4* *5* *6* *7* *8* *9* **1,0**	**Synergies:** Road development in the Brazilian Amazon is associated with an increase in homicide rate (*(118)*; → target 16.1). **Conflicts:** none.
Goal 17. Strengthen the means of implementation and revitalize the global partnership for sustainable development *1* *2* *3* *4* *5* *6* *7* *8* *9* *10* *11* *12* *13* *14* *15* *16* *17* *18* *19* **-1,0**	**Synergies:** none. **Conflicts:** Roads connect national economies (compare goal 8) and thus facilitate import-export traffic across borders (→ target 17.11), especially for landlocked regions or countries *(156)*.

Table S11. Synergies and conflicts between conservation of roadless areas and the United Nations' Aichi Strategic Goals and Biodiversity Targets. The color scheme indicates the level of synergy or conflict of goals and targets with roadless areas conservation (green: synergies prevail; grey: not applicable; yellow: ambivalent relationship). The numbers in insert boxes represent the conflict-synergy score of goals.

Aichi Strategic Goals and Biodiversity Targets	Brief analysis of synergies and conflicts between conservation of roadless areas and Aichi Biodiversity Targets
Strategic Goal A: Address the underlying causes of biodiversity loss by mainstreaming biodiversity across government and society $\boxed{0{,}5}$	
Target 1. By 2020, at the latest, people are aware of the values of biodiversity and the steps they can take to conserve and use it sustainably. Compare Sustainable Development Goal 4.	On the one hand, pristine ecosystems, such as they occur in roadless areas, are key for effective biodiversity conservation *(2)*. In agreement with modern concepts of sustainable land use, such as in biosphere reserves, these ecosystems are an essential element of sustainable use of the overall landscape *(157)*. Remote roadless areas provide opportunities for learning about natural ecosystems, i.e., wilderness (see goals B and C). On the other hand, roadless areas reduce accessibility of nature in general, thus making it more difficult to value biodiversity emotionally.
Target 2. By 2020, at the latest, biodiversity values have been integrated into national and local development and poverty reduction strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems. Compare Sustainable Development Goals 9, 8, 1.	While road infrastructure is related to economic growth and poverty alleviation *(158, 159)*, it has a crucial impact on biodiversity loss (see goal C), which in turn is directly linked with poverty aggravation *(160, 161)*. In remote areas inhabited mostly by indigenous or traditional people, road development may increase the spread of diseases, trigger conflicts over land and natural resources, and disrupt the traditional modes of production (which then have to compete with the global market), ultimately pushing these people towards poverty *(117, 162)*. The role of road development on poverty alleviation is hence conflicting, which calls for a better planning integrating roadless areas prioritization for biodiversity maintenance towards poverty alleviation.
Target 3. By 2020, at the latest, incentives, including subsidies, harmful to biodiversity are eliminated, phased out or reformed in order to minimize or avoid negative impacts, and positive incentives for the conservation and sustainable use of biodiversity are developed and applied, consistent and in harmony with the Convention and other relevant international obligations, taking into account national socio economic conditions.	Road transport receives between one- and two-thirds of worldwide conventional subsidies that are harmful in the long run to both the economy and the environment *(163)*. Road transport sector figures among the five most prominent sectors receiving such perverse subsidies *(164)*. An outstanding example refers to road infrastructure subsidies in the Amazon that have led to cattle ranching, extensive deforestation and biodiversity loss *(165)*. Alternatively, the integration of roadless areas into governmental policies could help in reducing and eliminating a substantial part of the harmful subsidies for the road transport sector.
Target 4. By 2020, at the latest, Governments, business and stakeholders at all levels have taken steps to achieve or have implemented plans for sustainable production and consumption and have kept the impacts of use of natural resources well within safe ecological limits. Compare Sustainable Development Goal 12.	Roadless areas, and relatively undisturbed areas in general, are of high resilience and ecosystem functionality *(2)*. Conserving these areas therefore contributes to maximizing ecosystem functionality of the wider landscape - they are an essential element of its sustainable use (compare targets 1, 7).
Strategic Goal B: Reduce the direct pressures on biodiversity and promote sustainable use $\boxed{0{,}8}$	
Target 5. By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced. Compare Sustainable Development Goal 15.	Road development is a major driver of habitat loss and fragmentation *(166)*. Roads act as barriers for species *(167)* and deforestation has been shown to increase along roads [*(98)*, Table S2]. Conserving roadless areas therefore directly helps to achieve this target.
Target 6. By 2020 all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem based approaches, so that overfishing is avoided, recovery plans and measures are in place for all depleted species, fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems and the impacts of fisheries on stocks, species and ecosystems are within safe	Roads facilitate the accessibility to remote terrestrial or freshwater ecosystems and increase the efficiency of natural resources exploitation and exportation, which are often depleted above their safe ecological limits *(1)*. For instance, a single road construction has been reported to have severe effect to a lake trout population, due to improved access for fishermen *(168)*. In addition, roads, their construction and traffic emit water pollutants *(137, 141)*. Similarly, road construction and roads can produce large sediment loads in rivers, particularly detrimental in wetlands and mountain areas.

ecological limits. Compare Sustainable Development Goals 14, 6, 3.	Roads also open up a landscape for logging and agriculture, and resulting runoff equally enters rivers *(154)*. Large part of this sediment ends up in estuaries and coastal waters.
Target 7. By 2020 areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity. Compare Sustainable Development Goal 2.	On one side, roadless areas exclude certain types of local development and even sustainable land use. And to keep up with demand for natural resources, any additional roadless area may require the intensification of land use in developed areas. On the other side, conservation of functional ecosystems, as they are still found in roadless areas, is essential for the larger landscape to stay functional. From this perspective, the remaining roadless areas can be seen as key elements of sustainably managed landscapes (compare targets 1, 4, 8).
Target 8. By 2020, pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity. Compare Sustainable Development Goals 6, 2.	Agricultural intensification might be necessary to make up for setting aside roadless areas (compare target 7). This might lead to increased use of fertilizers and pollution. It should be noted, however, that in many developing countries in particular there is a large amount of degraded land that can be restored and replace set-asides. However, conservation of roadless areas as relatively pristine ecosystems are a cost-efficient way of maximizing the provisioning of regulating ecosystem services such as nutrient uptake and water purification *(121)*.
Target 9. By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment.	Road density is a strong correlate of spatial patterns in biological invasions *(146)*. Limiting road development in roadless areas can, therefore, help to directly reduce the spread of invasive species (Table S2).
Target 10. By 2015, the multiple anthropogenic pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning. Compare Sustainable Development Goals 13, 15.	Roadless areas often represent areas with large carbon pools and sequestration potential. Furthermore, they represent areas of high ecosystem functionality important for climate regulation and long-term climate change adaptation. The conservation of roadless areas, thus, helps to mitigate and adapt to the impacts of climate change *(2, 4)*. Regarding marine ecosystems in particular, roadless areas prevent road-related sediment and agricultural runoff from impacting near-shore waters (compare target 6).
Strategic Goal C: To improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity **0,3**	
Target 11. By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes. Compare Sustainable Development Goal 15.	The conservation of roadless areas directly contributes to the conservation of valuable terrestrial ecosystems for biodiversity conservation. These areas also typically provide a wide array of ecosystem services, especially regulating services, and do this in large quantities. Furthermore, the conservation of these unfragmented and pristine areas directly contributes to the target of increasing connectivity. Conservation of the functionality of the watershed is highly dependent on the preservation of vegetation cover *(169)*, which benefits from conservation of roadless areas.
Target 12. By 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained. Compare Sustainable Development Goal 15.	Threatened species typical of anthropogenically disturbed ecosystems, such as old cultural landscapes in Europe and elsewhere, depend on certain semi-intensive, often historical, land use regimes (*170*). Therefore, in human-modified landscapes, the conservation of roadless areas in cases may be found little useful, or even counterproductive, to the target of improving the conservation status of some species. At the same time, other species (e.g., some amphibians) may experience reduced mortality in the absence of roads. After all, most threatened species are endangered by man-made loss of pristine ecosystems *(171)*. Roadless areas can retain populations of threatened species, supporting the native flora and fauna and buffering changes in the environmental conditions. Roadless areas which are large enough to host source populations can then serve as the origin for recolonization of areas where threatened species had disappeared *(172)*.
Target 13. By 2020, the genetic diversity of cultivated plants and farmed and domesticated animals and of wild relatives, including other socio-economically as well as culturally valuable species, is maintained, and strategies have been developed and implemented for minimizing genetic erosion and safeguarding their genetic diversity.	For one thing, on-farm conservation and use of cultivated species often requires the application of relatively extensive agricultural practices *(173)*. This could lead to competition for area between the conservation of roadless areas and more extensive agricultural practices for the preservation of the diversity of cultivated plants and animals. Then again, wild relatives of domesticated plant and animal species can often only be found in pristine natural areas *(174)*.
Strategic Goal D: Enhance the benefits to all from biodiversity and ecosystem services **1,0**	
Target 14. By 2020, ecosystems that provide essential services, including services related to water, and	Functional ecosystems, as they exist in roadless areas, provide large quantities of many ecosystem services, especially of regulating services.

contribute to health, livelihoods and well-being, are restored and safeguarded, taking into account the needs of women, indigenous and local communities, and the poor and vulnerable. Compare Sustainable Development Goals 6, 11, 1, 2, 3, 13.	They effectively reduce human exposure to extreme environmental events [e.g., fires, *(122)*]. Remote areas are often also of high value especially to indigenous and traditional people *(117)*. Remote areas also provide vital ecosystem services to poor city dwellers, such as purification and stable provisioning of water *(121)*.
Target 15. By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification. Compare Sustainable Development Goals 15, 13.	Roadless areas comprise relatively little disturbed areas. Many of these harbor large carbon pools and sinks, e.g., peatlands and intact forests in tropical and boreal regions *(175)*. Furthermore, they provide many regulating ecosystem services and high ecosystem functionality and are, therefore, crucial for ecosystem-based adaptation to climate change (see above targets 1, 4, 7). They also provide a natural buffer against increasing desertification through maintenance of vegetation cover *(155)*.
Target 16. By 2015, the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization is in force and operational, consistent with national legislation.	
Strategic Goal E: Enhance implementation through participatory planning, knowledge management and capacity building **1,0**	
Target 17. By 2015 each Party has developed, adopted as a policy instrument, and has commenced implementing an effective, participatory and updated national biodiversity strategy and action plan.	
Target 18. By 2020, the traditional knowledge, innovations and practices of indigenous and local communities relevant for the conservation and sustainable use of biodiversity, and their customary use of biological resources, are respected, subject to national legislation and relevant international obligations, and fully integrated and reflected in the implementation of the Convention with the full and effective participation of indigenous and local communities, at all relevant levels.	Indigenous communities are most vulnerable to the impacts of road development. Road construction in former roadless areas can cause traditional environmental knowledge loss and even a depopulation of indigenous communities *(176)*. Indigenous people may lose their land *(177)*, or use it less after road construction *(178)*, benefit less from biological resources and face an alteration of traditional roles and practices *(179)*.
Target 19. By 2020, knowledge, the science base and technologies relating to biodiversity, its values, functioning, status and trends, and the consequences of its loss, are improved, widely shared and transferred, and applied.	Natural ecosystems, as they still exist in remote roadless areas, are unique learning sites not only for education (see above target 1). They also provide important insights into ecosystem properties and processes such as biomass stocks, ecological dynamics, or resistance and resilience to natural disturbances *(180)*.
Target 20. By 2020, at the latest, the mobilization of financial resources for effectively implementing the Strategic Plan for Biodiversity 2011-2020 from all sources, and in accordance with the consolidated and agreed process in the Strategy for Resource Mobilization, should increase substantially from the current levels. This target will be subject to changes contingent to resource needs assessments to be developed and reported by Parties.	

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SYNTHESIS & INTEGRATION

Spotted Owls and forest fire: a systematic review and meta-analysis of the evidence

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Abstract. Forest and Spotted Owl management documents often state that severe wildfire is a cause of recent declines in populations of Spotted Owls and that mixed-severity fires (5–70% of burned area in high-severity patches with >75% mortality of dominant vegetation) pose a primary threat to Spotted Owl population viability. This systematic review and meta-analysis summarize all available scientific literature on the effects of wildfire on Spotted Owl demography and ecology from studies using empirical data to answer the question: How does fire, especially recent mixed-severity fires with representative patches of high-severity burn within their home ranges, affect Spotted Owl foraging habitat selection, demography, and site occupancy parameters? Fifteen papers reported 50 effects from fire that could be differentiated from post-fire logging. Meta-analysis of mean standardized effects (Hedge's *d*) found only one parameter was significantly different from zero, a significant positive foraging habitat selection for low-severity burned forest. Multi-level mixed-effects meta-regressions (hierarchical models) of Hedge's *d* against percent of study area burned at high severity and time since fire found the following: a negative correlation of occupancy with time since fire; a positive effect on recruitment immediately after the fire, with the effect diminishing with time since fire; reproduction was positively correlated with the percent of high-severity fire in owl territories; and positive selection for foraging in low- and moderate-severity burned forest, with high-severity burned forest used in proportion to its availability, but not avoided. Meta-analysis of variation found significantly greater variation in parameters from burned sites relative to unburned, with specifically higher variation in estimates of occupancy, demography, and survival, and lower variation in estimates of selection probability for foraging habitat in low-severity burned forest. Spotted Owls were usually not significantly affected by mixed-severity fire, as 83% of all studies and 60% of all effects found no significant impact of fire on mean owl parameters. Contrary to current perceptions and recovery efforts for the Spotted Owl, mixed-severity fire does not appear to be a serious threat to owl populations; rather, wildfire has arguably more benefits than costs for Spotted Owls.

Key words: adaptive management; evidence-based decision making; meta-analysis; mixed-severity fire; Spotted Owls; *Strix occidentalis;* systematic review; wildfire.

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Introduction

Wildfires are major natural disturbances in forests of the western United States, and native plants and animals in this region have been coexisting with fire for thousands of years of their evolutionary history (Pierce et al. 2004, Power et al. 2008, Marlon et al. 2012). Western forest fires typically burn as mixed-severity fires with each fire resulting in a mosaic of different vegetation burn severities, including substantial patches (range, 5–70% of burned area; mean, 22%) of high-severity fire (Beaty and Taylor 2001, Hessburg et al. 2007, Whitlock et al. 2008, Williams and Baker 2012, Odion et al. 2014*a*, Baker 2015*a*). High-severity fire (high vegetation burn severity) kills most or all of the dominant vegetation in a stand (>75% mortality; Hanson et al. 2009, Baker 2015*a*, *b*) and creates complex early seral forests, where standing dead trees, fallen logs, shrubs, tree seedlings, and herbaceous plants comprise the structure (Swanson et al. 2011, DellaSala et al. 2014). Post-fire vegetation processes (i.e., succession) then commence according to the pre-fire vegetation, local wildfire processes, propagules from outside the disturbance, and the dynamic biotic and abiotic conditions at the site (Gutsell and Johnson 2006, Johnson and Miyanishi 2006, Mori 2011).

Spotted Owls (*Strix occidentalis*) occur in western U.S. forests and have been intensively studied since the 1970s (Fig. 1). The species is strongly associated with mature and old-growth (i.e., late-successional) conifer and mixed conifer–hardwood forests with thick overhead canopy and many large live and dead trees and fallen logs (Gutiérrez et al. 1995). Its association with older forests has made the Spotted Owl an important umbrella indicator species for public lands management (Noon and Franklin 2002). The scientific literature has established that the optimal habitat for Spotted Owl nesting, roosting, and foraging is provided by conifer and mixed conifer–hardwood forests dominated by medium (30–60 cm) and large (>61 cm) trees with medium (50–70%) to high (>70%) canopy cover (Gutiérrez et al. 1995). The populations of all three subspecies have declined due to widespread historical and ongoing habitat loss, primarily from logging mature and old-growth forests favored by the owls for nesting and roosting (Seamans et al. 2002, Forsman et al. 2011, USFWS 2011, 2012, Conner et al. 2013, Tempel and Gutiérrez 2013, Dugger et al. 2016).

Research on Spotted Owl in fire-affected landscapes did not begin until the early 2000s, and much of what scientists previously understood about habitat associations of Spotted Owl was derived from studies in forests that had generally not experienced recent fire, and where the non-suitable owl habitat was a result of logging (Gutiérrez et al. 1992, Franklin et al. 2000, Seamans et al. 2002, Blakesley et al. 2005, Seamans and Gutiérrez 2007, Forsman et al. 2011, Tempel et al. 2014). Because Spotted Owls are associated with dense, late-successional forests, it has often been assumed that fires that burn at high severity are analogous to clear-cut logging and have a negative effect on population viability. It has become widely believed among wildlife management professionals that severe wildfire is a contributing cause of recent Spotted Owl population declines (USFWS 2011, 2012, 2017), and many land managers believe that forest fires currently pose the greatest risk to owl habitat and are a primary threat to population viability (Davis et al. 2016, Gutiérrez et al. 2017). These beliefs result in fuel-reduction logging projects in Spotted Owl habitat (USDA 2012, 2018) which the USDA Forest Service and US Fish and Wildlife Service state are actions consistent with Spotted Owl recovery (USDA 2012, 2018, Gutiérrez et al. 2017, USFWS 2017). Narrative literature reviews have attempted to summarize the effects of fire on Spotted Owl (Bond 2016, Gutiérrez et al. 2017), but evidence-based conservation decisions should be based upon systematic, transparent reviews of primary literature with quantitative meta-analysis of effects (Sutherland et al. 2004, Pullin and Stewart 2006, Pullin and Knight 2009, Koricheva et al. 2013).

The following systematic review and meta-analysis summarize all available published scientific literature on the effects of wildfire on aspects of Spotted Owl demography (survival, recruitment, and reproduction), site occupancy, and habitat selection, from studies using empirical data to answer the question: How does fire, especially mixed-severity fire with substantial patches of high-severity fire within their home ranges, affect Spotted Owl demography, site occupancy, and habitat selection in the first few post-fire years?

Methods

Literature search

I conducted a systematic review of the primary scientific literature and used meta-analyses and meta-regression to examine the evidence for the direct effects of wildfire on Spotted Owl demography, site occupancy, and habitat selection. My subject was Spotted Owls; the intervention was

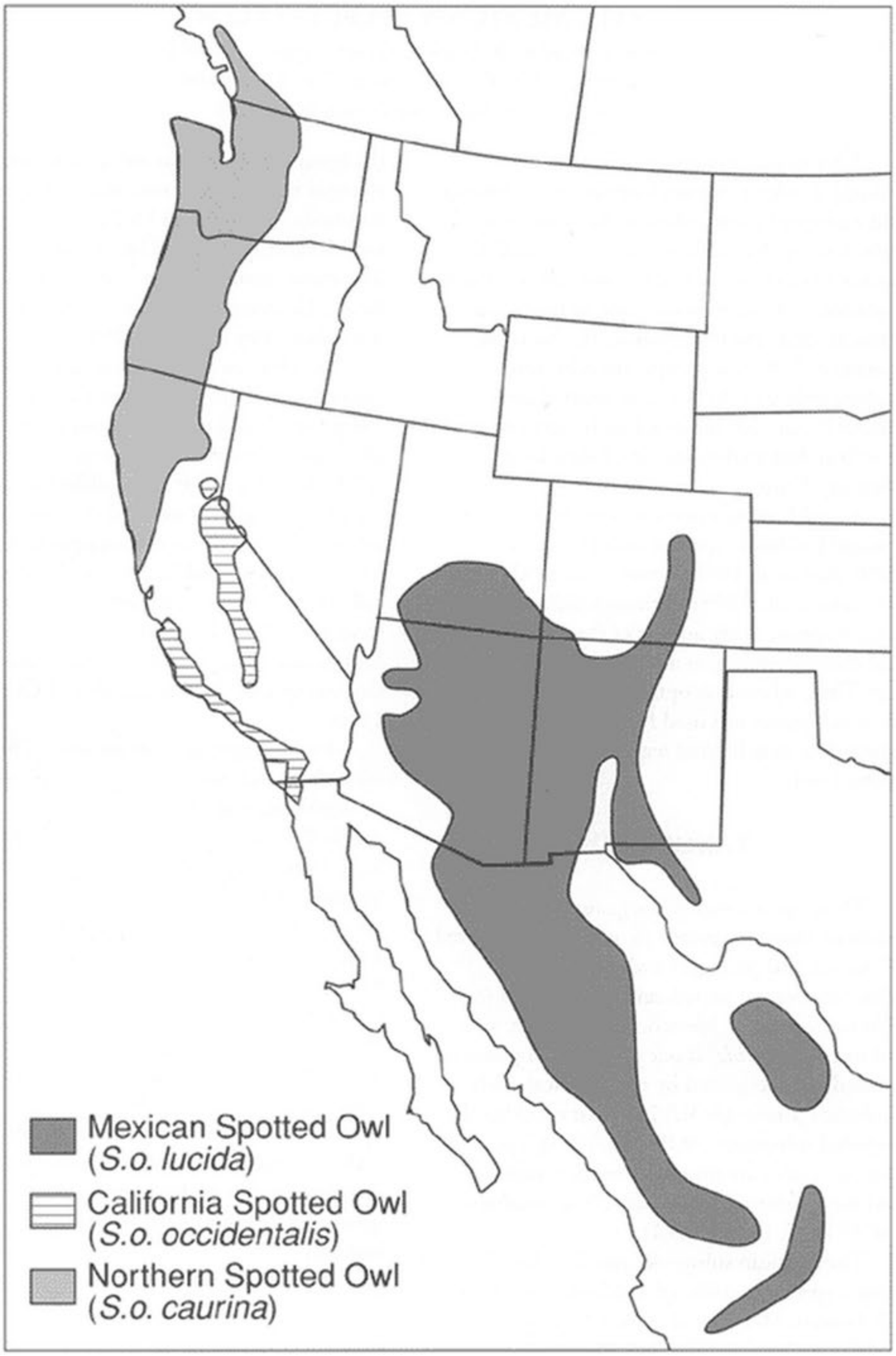

Fig. 1. Range map for the three subspecies of the Spotted Owl (*Strix occidentalis*).

wildfire; the outcomes were change or difference in estimates of demography, site occupancy, and habitat selection probabilities; and the comparator was pre-fire estimates or control estimates from unburned areas (Pullin and Stewart 2006). I searched the following electronic databases on 1 April 2018: Agricola, BIOSIS Previews, ISI Web of Science, and Google Scholar. Search terms were as follows: spotted AND owl AND *fire, Strix AND occidentalis AND *fire. My search included papers published in any year.

I used a threefold filtering process for accepting studies into the final systematic review. Initially, I filtered all articles by title and removed any obviously irrelevant material from the list of articles found in the search. Subsequently, I examined the abstracts of the remaining studies with regard to possible relevance to the systematic review question, using inclusion criteria based on the subject matter and the presentation of empirical data. I accepted articles for viewing at full text if I determined that they may contain information pertinent to the review question or if the abstract was ambiguous and did not allow inferences to be drawn about the content of the article. Finally, I read all remaining studies at full text and either rejected or accepted into the final review based upon subject matter (Pullin and Stewart 2006, Koricheva et al. 2013). Studies that only modeled effects of simulated fires on Spotted Owl habitat and demography were not considered here.

Because post-fire logging often occurred, I also recorded effects of this disturbance where they were reported. I believe all studies in the final review were generally comparable because time since fire and percent of high-severity burn were similar among studies (Tables 1, 2), and the high number of non-significant results reported indicates little to no publication bias exists in this topic (Tables 1, 2; Appendix S1: Fig. S1). I considered the basic sampling unit of all studies to be the central core of the owl breeding-season territory (~400 ha, or a circle with radius 1.1 km centered on the nest or roost stand) because this is the spatial and temporal scale for sampling used in almost all Spotted Owl studies. In contrast, Spotted Owl year-round home ranges vary according to latitude and dominant vegetation, but range from 300 to 11,000 ha, or circles with radius 1.0–5.9 km (Zabel et al. 1992). I considered forest fires to affect the landscape scale (~10,000 ha/decade), but that fires would affect numerous individual owl breeding-season territories (1200 ha) and year-round home ranges (300–19,000 ha) in various ways.

Meta-analyses and meta-regression

I evaluated all final review papers and included all papers where effects of fire were reported and could be differentiated from other disturbances such as post-fire logging. I extracted evidence by reading every paper and tabulating all quantified results from text, tables, and figures (Table 1). I noted the mean ($\bar{x}$) and variation (SD) of burned and unburned groups for all significant and non-significant parameters, the parameters being estimated, sample sizes (n = number of owl breeding sites in burned and unburned groups), amount of high-severity fire in the total fire perimeter and/or within the owl territory core areas examined, time since fire (years), amount of post-fire logging that occurred, subspecies (California = *Strix occidentalis occidentalis*, Mexican = *Strix occidentalis lucida*, or northern = *Strix occidentalis caurina*), and whether the result was statistically significant (as defined in each paper).

I conducted all analyses in R 3.3.1 (www.r-project.org). For meta-analysis, I noted or calculated the mean, variance (SD), and sample size for burned (treatment) and unburned (control) groups. I calculated raw effect sizes as mean differences ($\bar{x}_{\text{burned}} - \bar{x}_{\text{control}}$) and signs (positive or negative) for all reported effects, regardless of their statistical significance. Most papers reported effect sizes as probabilities (occupancy, survival, and foraging habitat selection) so raw effect sizes were scaled between negative and positive one with a mean of zero, making comparison among studies easy. When papers reported multiple effects (e.g., occupancy and reproduction, or survival and recruitment), I recorded each effect individually. Where papers did not report any effect size for a parameter determined to have no significant effects from fire, I included a zero to represent the presence of no significant effect and to avoid a significance bias in the meta-analysis. I stratified data by subspecies (California, Mexican, or northern) and parameter type according to whether the study estimated site occupancy, foraging habitat selection (substratified into selection for low-, moderate-, and high-severity burned forest), and demographic rates (substratified into survival, reproduction, and recruitment). I performed meta-analyses on parameters for which ≥4 estimates existed from ≥4 different fires.

I used three quantitative methods for evaluating the evidence (Koricheva et al. 2013): a random-effects meta-analysis of mean effect sizes as

Table 1. Summary of systematic review of studies examining effects of fire on Spotted Owls.

No.	Ref.	Sample size	HOD	Time since fire	Context	Fire effects (* = statistically significant, NS = non-significant)	Fire	Any effect	Signif. effect	Post-fire logging
1	Bond et al. (2002)	21 owls in 11 burned sites	OD	1 yr post-fire	No effect on survival, site fidelity, mate fidelity, or reproduction. 50% of territories burned 36–88% high severity, 50% burned mostly low –moderate severity, unknown amount of post-fire logging	No significant effects. (3% higher survival NS, 1% lower site fidelity [occupancy] NS, 26% higher repro NS)	0/+/−	+0.032 −0.013 +0.259	na	na
2	Jenness et al. (2004)	33 burned and 31 unburned breeding sites	OD	1-yr study, 1–4 yr post-fire	No effect on occupancy from fire or amount of high-severity fire. No effect on reproduction. 55% of burned territories area burned, 18% at high severity, unknown amount of post-fire logging	No significant effects from fire. (14% lower occupancy NS, 7% lower repro in burn NS)	0/−	−0.14 −0.07	na	na
3	Bond et al. (2009)	Seven radioed owls from four burned sites	H	1-yr study, 4 yr post-fire	Owls preferred burned forest for foraging, especially high-severity burned forest. Owls preferred roost sites burned at low severity and avoided unburned sites and sites burned at moderate and high severity. 69% of foraging area burned, 13% at high severity, <3% post-fire logging	Positive effect from fire on foraging habitat selection (+42%, +42% +33%*), negative and positive effect of fire on roosting nesting habitat selection (+29%, −13%, −28%*)	+/−	+0.33 +0.42 +0.42 +0.29 −0.13 −0.28	+0.33 +0.42 +0.42 +0.29 −0.13 −0.28	na
4	Bond et al. (2010)	Five radioed owls in occupied burned sites	H	1-yr study, 4 yr post-fire	Three of five owls occupied burned forest over winter	No significant effects, perhaps some positive effect	0/+	na	na	na
5	Clark et al. (2011)	11 radioed owls in burned and post-fire logged sites, 12 in unburned sites	D	2-yr study, 3–4 yr post logging	No effects on survival. Reduced survival in salvage-logged areas relative to owls in unburned forest. 14% high severity, 21% post-fire logged	Negative survival effect from combined effects of fire and post-fire logging (−0.07 NS)	?	na	na	−0.07
6	Roberts et al. (2011)	16 burned and 16 unburned survey areas	O	1-yr study, 2–14 yr post-fire	No effect of fire on survey area occupancy. 14% of survey area burned at high severity, little to no post-fire logging	No significant effect from fire. Possible negative effect from basal area and canopy cover model (−26% lower occupancy in burned survey area NS)	0/−	−0.260	na	na

(Table 1. *Continued*)

No.	Ref.	Sample size	HOD	Time since fire	Context	Fire effects (* = statistically significant, NS = non-significant)	Fire	Any effect	Signif. effect	Post-fire logging
7	Lee et al. (2012)	41 burned and 145 unburned breeding sites	O	11-yr study, 1–7 yr post-fire from six large fires	No effect on occupancy probability. 32% high severity. Unknown amount of post-fire logging	No significant effect from fire, perhaps a slightly positive effect (4% higher occupancy in burned sites NS)	0/+	+0.041	na	na
8	Bond et al. (2013)	Seven radioed owls	H	1-yr study, 4 yr post-fire	Owls in burned forest have same size or smaller home ranges than owls in unburned forest. 69% of foraging area burned, 13% at high severity, 3% post-fire logging	No significant effect from fire, possible positive effect (HR size 12% smaller in burned area NS)	0/+	+0.12	na	na
9	Clark et al. (2013)	40 burned and salvage-logged sites and 103 unburned sites	O	13-yr study, 1–4 yr post-fire	Lower site occupancy on salvage-logged sites relative to unburned sites. 11% high severity, 13% post-fire logged	Negative effect on occupancy from combined fire and post-fire logging (−0.39*)	?	na	na	−0.39
10	Lee et al. (2013)	71 burned and 97 unburned breeding sites, post-fire logging on 21 of the burned sites	O	8-yr study, 1–8 yr post-fire	No effects from fire or logging. Burned site occupancy 17% (10% for pairs) lower than unburned sites. Post-fire logged sites occupancy 5% lower than unlogged burned sites. 23% high severity in burned sites, 59% logged in post-fire logged sites	No significant effect from fire, negative effect (17% lower any occupancy, 10% lower pair occupancy in burn NS) Same data as ref. no. 14	0/−	−0.171 −0.107	na	−0.05
11	Ganey et al. (2014)	Four radioed owls	H	1-yr study, 4–6 yr post-fire	Owls moved to burned forest over winter. Burned wintering sites had 2–6 times more prey biomass relative to unburned core areas. 21% high severity, unknown amount of post-fire logged	Positive effect from fire	+	na	na	na
12	Tempel et al. (2014)	12 burned, 62 unburned sites	DO	20-yr study of survival and reproduction, 6-yr study of occupancy.	No effect on survival, reproduction, or site extinction. Reported a negative effect of fire on colonization rate, but colonization parameter was faulty due to low sample size and zero colonization events. Unknown amount of high-severity fire, unknown amount of post-fire logging	No significant effect from fire. Possible negative effect from fire (6% lower occupancy when fire frequency doubled in simulations that assumed zero post-fire colonizations)	0/−	0 0 −0.060	−0.060	na

(Table 1. *Continued*)

No.	Ref.	Sample size	HOD	Time since fire	Context	Fire effects (* = statistically significant, NS = non-significant)	Fire	Any effect	Signif. effect	Post-fire logging
13	Lee and Bond (2015*a*)	45 burned breeding sites	O	Rim Fire, 1-yr study, 1 yr post-fire	Higher burned-site occupancy rates than any published unburned area. 100% high-severity fire in territory surrounding nest and roost sites reduced single owl occupancy probability 5% relative to sites with 0% high severity. Amount of high-severity fire did not affect occupancy by pairs of owls. In fire perimeter: 37% high severity, no post-fire logging	Positive (17% higher occupancy rates*). Small negative effect on site occupancy (3% lower occupancy in burn*). No significant effect on pair occupancy	+/0	+0.175 −0.04 0	+0.175	na
14	Lee and Bond (2015*b*)	71 burned and 97 unburned breeding sites, post-fire logging on 21 of the burned sites	OD	8-yr study, 1–8 yr post-fire	Occupancy of high-quality sites (previously reproductive) that burned was 2% lower than unburned sites. Occupancy of high-quality sites that were post-fire logged was 3% lower. Occupancy of low-quality sites (previously non-reproductive) was 19% lower in burned vs. unburned sites and 26% lower after post-fire logging. Fire did not affect reproduction. 23% high severity in burned sites, 59% logged in post-fire logged sites	Negative effect on site occupancy (2% and 19% lower*), No significant effect on reproduction	−/0	−0.02 −0.19 0	−0.02 −0.19	−0.03 −0.26
15	Bond et al. (2016)	Eight radioed owls in five sites	H	2-yr study, 3–4 yr post-fire	Owls used forests burned at all severities in proportion to their availability, with the exception of significant selection for moderately burned forest farther from core areas. 23% high severity, <5% post-fire logging	No significant effect from fire (3% lower probability of use in high-severity burn NS), some positive effect (15% higher probability of use of low-severity burn NS, 10% higher probability of use in moderate-severity burned forest NS, 3% higher probability of use of moderate severity away from the core*)	0/+	−0.03 +0.15 +0.10	+0.033	na

(Table 1. *Continued*)

No.	Ref.	Sample size	HOD	Time since fire	Context	Fire effects (* = statistically significant, NS = non-significant)	Fire	Any effect	Signif. effect	Post-fire logging
16	Comfort et al. (2016)	23 radioed owls in post-fire logged area	H	2-yr study, 3–4 yr post logging	Scale-dependent effects of logging (+/−). Owls selected a moderate amount of hard edges around logged stands. 14% high severity, 21% post-fire logged	Positive and negative effect from post-fire logging created edges	?	na	na	+/−
17	Jones et al. (2016)	30 burned sites, 15 unburned sites, nine radioed owls in seven sites	OH	23-yr study, 1 yr post-fire	Negative effects from high-severity fire. Positive effect of low- to moderate-severity fire. 64% high-severity burn, 2% post-fire logging	>50% high-severity burned sites had lower occupancy (−0.49*), <50% high-severity burned sites had higher occupancy (+0.07 NS). High-severity burned habitat was avoided (−0.307*), low-tomoderate-severity burn was preferred (+0.04 NS)	+/−	+0.070 −0.490 −0.307 +0.04	−0.490 −0.307 +0.04	na
18	Tempel et al. (2016)	43 burned sites and 232 unburned sites in four study areas	O	19-yr study, examined 3-yr post-fire effects	No effects of fire. One study area had positive effect of fire. Lower site extinction probability correlated with proportion of site where wildfire reduced canopy >10%. 1% of all territories burned, unknown amount of post-fire logging	No significant effect from fire, some positive effect (1% lower extinction rate in burned sites NS)	0/+	+0.003 0 0 0	na	na
19	Eyes et al. (2017)	13 radioed owls in eight sites (14 owl-year data sets)	H	3-yr study, 1–14 yr post-fire	No effect of fire on foraging habitat selection, owls foraged in all burn severities in proportion to their availability. 6% high severity, little to no post-fire logging	No significant effect from fire. Possibly negative effect (6% lower probability of use for highest burn severity NS; 3% lower use of moderate severity NS)	0/−	−0.06 −0.03	na	na
20	Rockweit et al. (2017)	193 burned and 386 unburned encounter histories from 28 burned (8, 2, 4, 14) and 70 unburned sites	D	26-yr study, 4–26 yr post-fire	Four fires had different effects. Generally, fires reduced survival and increased recruitment. 10%, 12%, 16%, and 48% high severity, no post-fire logging reported	Two fires had no significant effects on survival or recruitment. Two fires had reduced survival (−0.17 and −0.30*), one had increased recruitment (+0.22*)	0/+/−	−0.03 −0.10 −0.17 −0.30 +0.01 +0.02 +0.04 +0.22	−0.17 −0.30 +0.22	na

(Table 1. *Continued*)

No.	Ref.	Sample size	HOD	Time since fire	Context	Fire effects (* = statistically significant, NS = non-significant)	Fire	Any effect	Signif. effect	Post-fire logging
21	Hanson et al. (2018)	54 burned sites in eight fires that were occupied immediately before fire, before–after comparison	O	14-yr study, 1 yr post-fire	Eight large fires (4 included in Tempel et al. 2016). Four groups: 20–49% and 50–80% high-severity fire; and <5% and ≥5% post-fire logging within 1500 m of site center. Mean 63% high severity in core areas, mean 17% logged if ≥5% of core was post-fire logged Compared burned site occupancy with unburned occupancy from Tempel et al. (2016)	No significant effect from fire, significant negative effect of post-fire logging (3% reduction in occupancy if 50–80% of core burned high-severity fire NS, 52% reduction in occupancy from ≥5% post-fire logging*)	0/-	−0.017 −0.013	na	−0.52

Notes: HOD indicates habitat selection (H), occupancy (O), or demographic (D) parameters were estimated. A question mark (?) indicates confounded fire and post-fire logging effects, so fire effects could not be estimated.

the standardized difference in means (Hedge's *d*; Hedges and Olkin 1985); multi-level linear mixed-effects models (hierarchical models) meta-regression of time since fire and percent of high-severity fire in the study area as covariates to explain heterogeneity in mean effect sizes (Hedges and Vevea 1998, Nakagawa and Santos 2012); and a random-effects meta-analysis of variation to examine differences in parameter variances due to fire with effect sizes as the natural logarithm of the ratio between the coefficients of variation (lnCVR; Nakagawa et al. 2015). For analyses, I used the metafor package of R (Viechtbauer 2010) and used function metacont for random-effects meta-analyses, function rma.mv for multi-level linear mixed-effects model meta-regression, and function rma for random-effects meta-analysis of variation (Viechtbauer 2010). Study within geographic area was included as multi-level random effects to properly estimate study site- and region-specific variation and to account for repeated measurements (pseudo-replication) within a study or region. Regions were defined as Sierra Nevada, southern California, national parks, not California, and the Eldorado density study area (because several studies used data from there).

I used all three methods at three levels: on all parameters, on three main groups of parameters (occupancy, foraging habitat selection, and demography), and on subgroups of habitat selection (for low-, moderate-, and high-severity burned forest) and demography (survival, reproduction, and recruitment). In meta-analyses, I used z tests to determine if effects were significantly different from zero (95% confidence interval excluded zero). In meta-regression, z tests determined whether intercepts or slope coefficients were significantly different from zero. I quantified heterogeneity among effects as Cochran's Q (Hedges and Olkin 1985) and I^2 (Higgins and Thompson 2002). I used a funnel plot and the rank correlation test (Kendall's τ) to assess publication bias (Begg and Mazumdar 1994).

Results

Literature search

I found 21 papers reporting empirical evidence relevant to direct fire effects on owls (Table 1). Three papers presented data from a study area which was extensively logged post-fire and results did not discriminate between effects of fire and post-fire logging (Clark et al. 2011, 2013, Comfort et al. 2016), so these three papers were not included in meta-analyses with the meta-analysis set of papers that were not confounded

Table 2. Summary statistics for published effects of mixed-severity fire on Spotted Owls (*Strix occidentalis*) 1987–2018 used in meta-analysis.

Ref no.	Study	Subspecies	Region	Parameter	*n* burned	*n* unburned	Raw effect size (mean difference)	Significant (in study)	Time since fire (yr)	Percentage of high-severity fire in burned territories
1	Bond (2002)	CNM	NotCal	Occupancy	18	100	−0.013	na	1	30
1	Bond (2002)	CNM	NotCal	Reproduction	7	100	0.259	na	1	30
1	Bond (2002)	CNM	NotCal	Survival	21	100	0.032	na	1	30
2	Jenness (2004)	M	NotCal	Occupancy	33	31	−0.14	na	2.5	16
2	Jenness (2004)	M	NotCal	Reproduction	33	31	−0.07	na	2.5	16
3	Bond (2009)	C	SN	Foraging High	7	7†	0.42	0.42	4	13
3	Bond (2009)	C	SN	Foraging Low	7	7†	0.33	0.33	4	13
3	Bond (2009)	C	SN	Foraging Mod	7	7†	0.42	0.42	4	13
6	Roberts (2011)	C	NP	Occupancy	16	16	−0.26	na	8	12
7	Lee (2012)	C	SN	Occupancy	41	145	0.041	na	4	32
10	Lee (2013)	C	SoCal	Occupancy	71	97	−0.171	na	4.5	23
10	Lee (2013)	C	SoCal	Occupancy	71	97	−0.107	na	4.5	23
12	Tempel (2014)	C	Eldorado	Occupancy	12	62	−0.06	−0.06	3	23‡
12	Tempel (2014)	C	Eldorado	Reproduction	12	62	0	na	3	23‡
12	Tempel (2014)	C	Eldorado	Survival	12	62	0	na	3	23‡
13	Lee (2015*a*)	C	SN	Occupancy	45	45	−0.04	na	1	37
13	Lee (2015*a*)	C	SN	Occupancy	45	45	0	na	1	37
13	Lee (2015*a*)	C	SN	Occupancy	45	145	0.175	0.175	1	37
14	Lee (2015*b*)	C	SoCal	Occupancy	71	97	−0.19	−0.19	4.5	23
14	Lee (2015*b*)	C	SoCal	Occupancy	71	97	−0.02	−0.02	4.5	23
14	Lee (2015*b*)	C	SoCal	Reproduction	71	97	0	na	4.5	23
15	Bond (2016)	C	SoCal	Foraging High	8	8†	−0.093	na	3.5	15
15	Bond (2016)	C	SoCal	Foraging High	8	8†	−0.035	na	3.5	16
15	Bond (2016)	C	SoCal	Foraging High	8	8†	0.092	na	3.5	9
15	Bond (2016)	C	SoCal	Foraging Low	8	8†	0.115	na	3.5	15
15	Bond (2016)	C	SoCal	Foraging Low	8	8†	0.167	na	3.5	9
15	Bond (2016)	C	SoCal	Foraging Low	8	8†	0.169	na	3.5	16
15	Bond (2016)	C	SoCal	Foraging Mod	8	8†	−0.042	na	3.5	15
15	Bond (2016)	C	SoCal	Foraging Mod	8	8†	0.033	0.033	3.5	16
15	Bond (2016)	C	SoCal	Foraging Mod	8	8†	0.102	na	3.5	9
17	Jones (2016)	C	Eldorado	Foraging High	9	9†	−0.307	−0.307	1	19

(Table 2. *Continued*)

Ref no.	Study	Subspecies	Region	Parameter	n burned	n unburned	Raw effect size (mean difference)	Significant (in study)	Time since fire (yr)	Percentage of high-severity fire in burned territories
17	Jones (2016)	C	Eldorado	Foraging Mod	9	9†	0.04	+0.04	1	19
17	Jones (2016)	C	Eldorado	Occupancy	14	15	−0.490	−0.490	1	64
17	Jones (2016)	C	Eldorado	Occupancy	16	15	0.07	na	1	19
18	Tempel (2016)	C	SN	Occupancy	12	78	0	na	4	23‡
18	Tempel (2016)	C	Eldorado	Occupancy	14	60	0	na	4	23‡
18	Tempel (2016)	C	SN	Occupancy	3	63	0	na	4	23‡
18	Tempel (2016)	C	NP	Occupancy	14	31	0.003	0.003	4	23‡
19	Eyes (2017)	C	SN	Foraging High	13	13†	−0.06	−0.06	7	6
19	Eyes (2017)	C	SN	Foraging Mod	13	13†	−0.03	−0.03	7	6
20	Rockweit (2017)	N	NotCal	Recruitment	8	8	0.01	na	12.5	10
20	Rockweit (2017)	N	NotCal	Recruitment	2	2	0.02	na	6.5	16
20	Rockweit (2017)	N	NotCal	Recruitment	4	4	0.04	na	4	48
20	Rockweit (2017)	N	NotCal	Recruitment	14	14	0.22	0.22	2	12
20	Rockweit (2017)	N	NotCal	Survival	4	4	−0.30	−0.3	4	48
20	Rockweit (2017)	N	NotCal	Survival	14	14	−0.17	−0.17	2	12
20	Rockweit (2017)	N	NotCal	Survival	2	2	−0.10	na	6.5	16
20	Rockweit (2017)	N	NotCal	Survival	8	8	−0.03	na	12.5	10
21	Hanson (2018)	C	SN	Occupancy	13	201	−0.017	−0.017	1	63
21	Hanson (2018)	C	SN	Occupancy	15	201	0.013	0.013	1	35

Notes: Study indicates first author and year. Subspecies are C, California (*Strix occidentalis occidentalis*); N, northern (*Strix occidentalis caurina*); M, Mexican (*Strix occidentalis lucida*); CNM, study included all subspecies. Regions are SN, Sierra Nevada, California (except El Dorado study area and national parks); SoCal, southern California; Eldorado, El Dorado study area in Sierra Nevada, California; NotCal, not California Spotted Owl subspecies; NP, national parks. Parameters: habitat selection (foraging or roosting) in low-, moderate-, (mod) or high-severity burned forest; occupancy, recruitment, reproduction, and survival. Sample sizes (n) are number of breeding site territories burned and unburned. Raw mean effect size is $\bar{x}_{\text{burned}} - \bar{x}_{\text{control}}$, significant repeats effects that the individual study determined was statistically significant. Time since fire is the median number of years between the fire and the parameter estimate(s). Percent high-severity fire in burned study territories is the mean relevant to the estimate, or the grand mean if percentage of high severity was not reported (see ‡).

† Habitat selection occurred within territories that contained a mosaic of burn severities and unburned forest.

‡ Percent high-severity fire was not reported for burned territories only for all territories burned and unburned, so the grand mean of reported percentages was used.

by extensive post-fire logging (Table 2). All 21 papers are summarized in Appendix S1.

Fifteen of the 18 papers in the meta-analysis set reported evidence explicitly pertaining to mixed-severity wildfires that burned during the past few decades and which included proportions of high-severity burn characteristic of this fire regime, while three reported evidence from an undifferentiated mix of wildfire and prescribed fires. The studies reported varying amounts of high-severity fire, a defining feature of mixed-severity fires, and the burn severity type that is most responsible for vegetation changes in wildfires, with an overall mean percent of high-severity fire of 26% (standard error [SE] = 3.6, range 6–64) within the study area. Because almost all the studies in this review reported on effects from recent wildfires (all fires burned in the past 30 yr, mean time since fire = 4 yr, SE = 1.1, range 1–26), the reported effects are representative of natural mixed-severity fires as they burned through currently existing forest structure, fire regime, and climate conditions. Papers reported effects of fire on site occupancy (11), foraging habitat selection (4), reproduction (4), apparent survival (3), overwinter roosting habitat selection (2), site fidelity (1), mate fidelity (1), breeding-season nesting and roosting habitat selection (1), home-range size (1), and recruitment (1). Sample sizes measured as number of burned sites were variable among studies (demography CV = 122%, site occupancy CV = 56%, and habitat selection CV = 24%).

Meta-analyses

Meta-analysis of 50 reported effects on occupancy, foraging habitat selection, and demographic rates found effect sizes and signs were variable (Table 2 and Fig. 2), with high heterogeneity among effects ($Q = 1091$, df = 51, $P < 0.0001$; $I^2 = 95.3\%$). Funnel plot (Appendix S1: Fig. S1) and rank correlation test (Kendall's $\tau = 0.108$, $P = 0.27$) showed no publication bias or unusual heterogeneity. Sample sizes (n = number of reported effects) were variable among parameter types (Fig. 3). The number of reported effects were occupancy = 20; demography = 14; and foraging habitat selection = 16. The number of reported effects by demography subtype were survival = 6; reproduction = 4; and recruitment = 4. The number of reported effects by habitat selection subtype were low-severity burned forest = 4; moderate-severity burned forest = 6; and high-severity burned forest = 6.

The mixed-effects model meta-analysis of fire effects on Spotted Owl parameters grouped by type (occupancy, demography, and foraging habitat selection), and subtypes of demography (survival, reproduction, and recruitment) or foraging habitat selection (selection for low-, moderate-, and high-severity burned forest), found mixed-severity fire has generally no significant effect on Spotted Owls (Fig. 3a). Mean overall raw effect size was positive (+0.001), but weighted mean Hedge's d from the random-effects model was not significantly different from zero (Fig. 3a, 95% confidence interval included zero). Mean raw effect sizes were negative for occupancy (−0.060), demography (−0.006), and survival (−0.095), but no Hedge's d value for these three negative effects was significantly different from zero (Fig. 3a). Mean raw effect sizes were positive for reproduction (+0.047), recruitment (+0.073), foraging habitat selection (+0.083), selection of high-severity (+0.004), moderate-severity (+0.087), and low-severity burned forest (+0.195), but Hedge's d values were not significantly different from zero for any of these positive effects, except for significant selection of low-severity burned forest (Fig. 3a).

Variation was generally higher among parameter estimates from burned areas compared to estimates from unburned areas (mean $CV_{burned} - CV_{unburned} = 23\%$; range 4–57%). The mixed-effects meta-analysis of variation in fire effects on Spotted Owl parameters (lnCVR) found mixed-severity fire resulted in significantly higher variation in parameter estimates in all parameters and in occupancy, demography, and survival (Fig. 3b). There was significantly lower variation in estimates of foraging habitat selection probability for low-severity burned forest (Fig. 3b).

Meta-regression

Meta-regression of all standardized mean effects found significant effect of time since fire (Table 3), and a nearly significant effect of percent high-severity burn in territory cores (Table 3), so those effects were included in parameter-specific meta-regressions. Subspecies was not a significant factor (Table 3), so effects from different subspecies were pooled in subsequent parameter-specific analyses.

Meta-regression of occupancy probability found no significant immediate effect of fire on occupancy (intercept not significantly different from zero; Table 4). There was a significant negative effect of time since fire (Fig. 4, Table 4), but no effect of percent high-severity fire in study territories (Table 4). The negative effect of time since fire was sensitive to one study (Roberts et al. 2011), and when that study was omitted, the effect disappeared.

Meta-regression of demographic parameters found a significant positive effect on recruitment immediately after the fire (intercept significantly different from zero), but the effect diminished

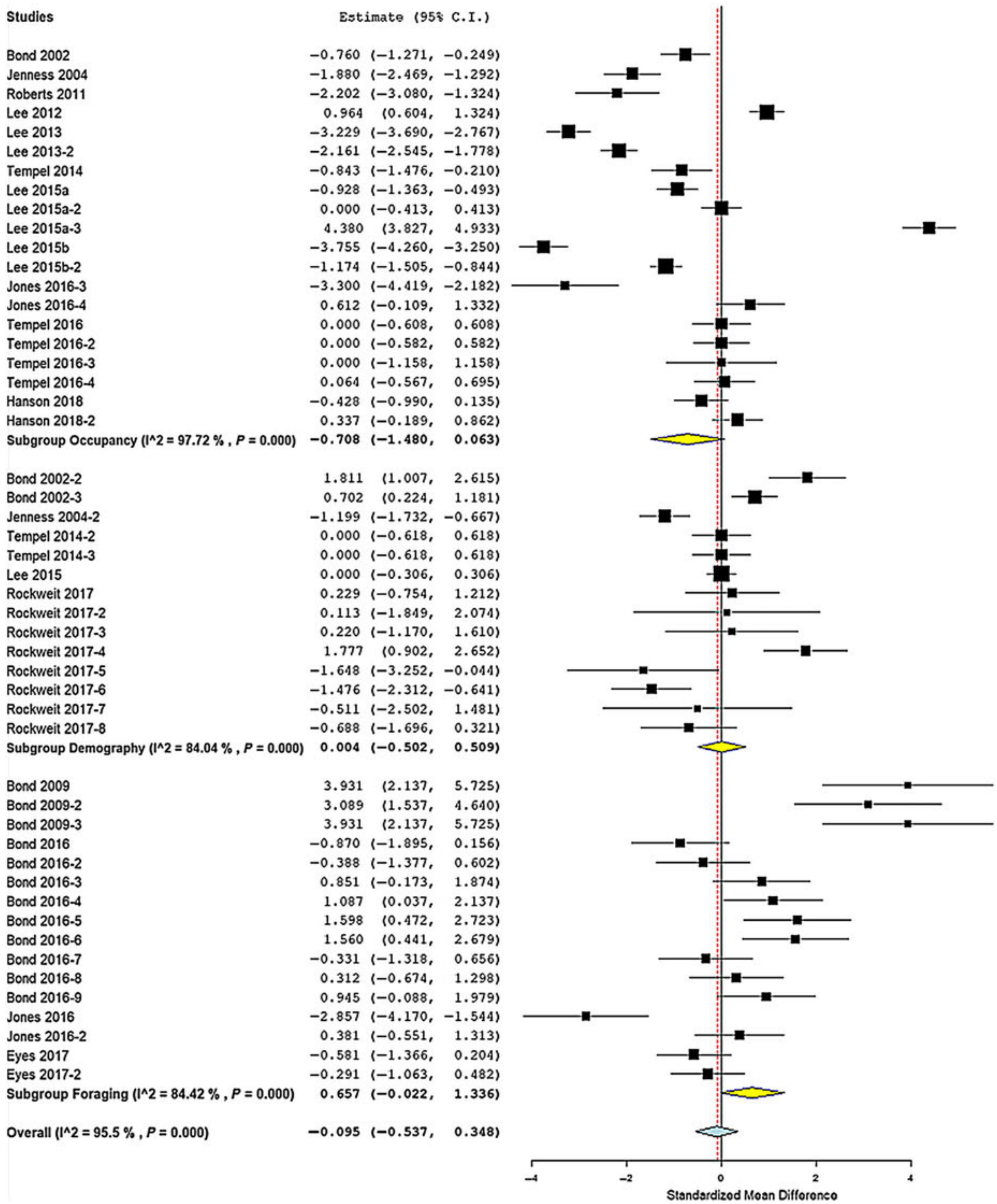

Fig. 2. Forest plot of effect sizes for 50 Spotted Owl (*Strix occidentalis*) parameters (grouped into occupancy, demography, and foraging habitat selection) affected by mixed-severity wildfire as standardized mean difference (Hedge's *d*) between burned and unburned samples. Studies and parameters are listed in Table 2.

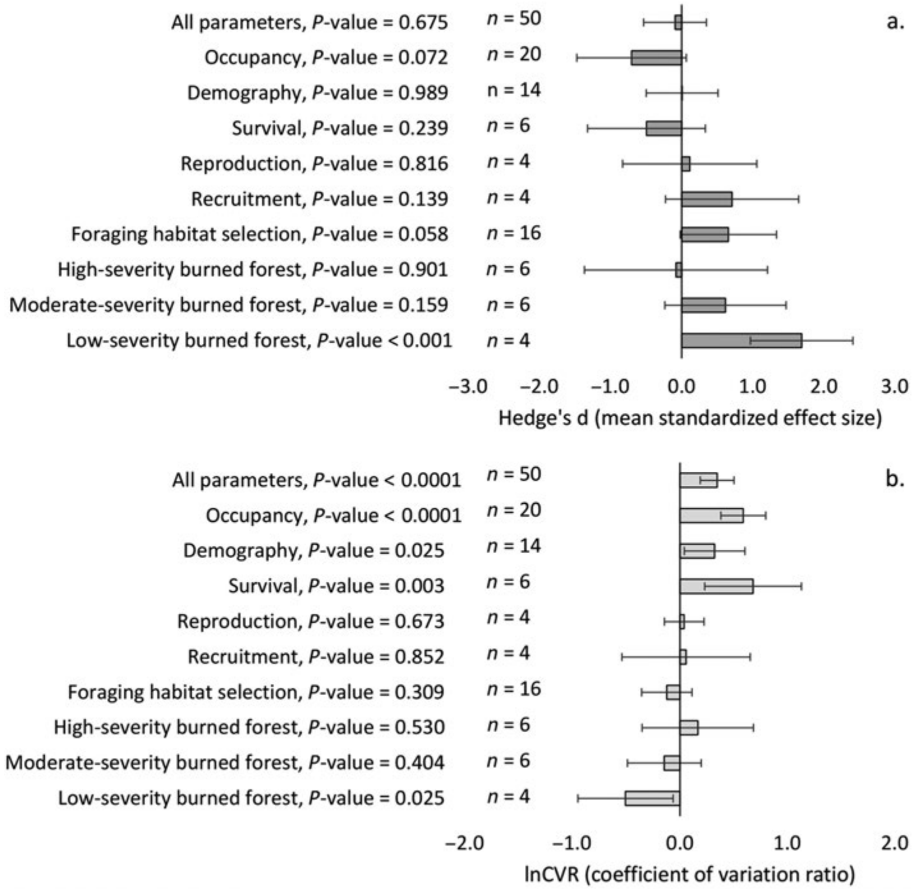

Fig. 3. Results of mixed-effects meta-analyses of mixed-severity fire effects (n = 50 effects from 21 studies) on Spotted Owl (*Strix occidentalis*) parameters grouped by type (occupancy, demography, and foraging habitat selection) and subtype of demography (survival, reproduction, and recruitment), or habitat selection (selection for low-, moderate-, and high-severity burned forest). (a) Hedge's d is standardized mean effect size, and error bars are 95% confidence intervals. The only significant effect (95% confidence intervals excluded zero) was a positive effect of habitat selection for low-severity burned forest. (b) lnCVR is the natural logarithm of the ratio between the coefficients of variation, a measure of differences in variation of parameter estimates between burned and unburned areas. Mixed-severity fire resulted in significantly higher variation in parameter estimates in all parameters, occupancy, demography, and survival, and significantly lower variation in habitat selection for low-severity burned forest.

with time since fire (Fig. 5, Table 4). Reproduction intercept was not significantly different from recruitment (Table 4), and not significantly different from zero ($z = -0.218$, $P = 0.86$), but reproduction was significantly positively correlated with the percent of high-severity fire in owl territories (Fig. 5, Table 4). Survival was significantly lower than recruitment (Table 4), but survival intercept was not significantly different from zero ($z = -0.052$, $P = 0.97$). There were no significant survival effects of time since fire or percent of high-severity fire (Table 4).

Meta-regression of foraging habitat selection parameters found a significant positive selection for low- and moderate-severity burned forest, with high-severity burned forest used in proportion to its availability, but not avoided (Fig. 5, Table 4). Time since fire did not affect foraging habitat selection during the period covered by the studies I examined (up to 7 yr), and the

Table 3. Results from multivariate mixed-effects meta-regression model of mixed-severity fire effects (n = 50 effects from 21 studies) on Spotted Owl (*Strix occidentalis*) parameters related to occupancy, demography, and foraging habitat selection.

Covariates	β	SE	z	P
Intercept (California subspecies)	1.601	1.070	1.497	0.134
Time since fire	−0.199	0.099	−2.017	**0.044**
Percentage of area high-severity fire in study territories	−0.044	0.023	−1.866	0.062
Mix of California, northern, Mexican subspecies	0.467	1.592	0.294	0.769
Mexican subspecies	−1.947	1.608	−1.211	0.226
Northern subspecies	0.360	1.571	0.229	0.819

Notes: SE, standard error. Time since fire was significant, and percent high-severity burn in territory cores was nearly significant, so those effects were included in parameter-specific meta-regressions. Subspecies was not a significant factor, so effects from different subspecies were pooled in subsequent parameter-specific analyses. Bold values are significant at alpha = 0.05.

amount of high-severity fire did not affect habitat selection overall (Table 4).

Post-fire logging had negative effects on Spotted Owls in 100% of the papers that examined this disturbance and where effects from fire and post-fire logging could be differentiated, with large effect sizes (−0.18 occupancy, −0.07 survival).

Discussion

This systematic review and summary of effects from the primary literature indicated Spotted Owls are usually not significantly affected by mixed-severity fire as 83% of all studies and 60% of all effects found no significant impact of fire on owl parameters. Meta-analysis of mean effects found no significant effects of fire on owls, except a positive effect on foraging habitat selection for low-severity burned forest. Meta-regression indicated significant positive effects in recruitment, reproduction, and foraging habitat selection for low- and moderate-severity burned forest. Meta-regression found a significant negative effect of time since fire on occupancy probability. Meta-analysis of variation found mixed-severity fire resulted in greater parameter variation overall, and specifically in occupancy, demography, and survival, and significantly less variation in foraging habitat selection for low-severity burned forest.

Table 4. Table of model coefficients from multi-level linear mixed-effects model meta-regression for effects of mixed-severity fire on Spotted Owls 1987–2018.

Coefficient	β	SE	z	P
Occupancy				
Intercept	1.854	1.115	1.662	0.096
Time since fire	−0.512	0.216	−2.375	**0.018**
Percentage of area high-severity fire in study territories	−0.036	0.022	−1.645	0.100
Demography				
Intercept (Recruitment)	2.328	1.152	2.021	**0.043**
Time since fire (Recruitment)	−0.153	0.065	−2.347	**0.019**
Percentage of area high-severity fire in study territories	−0.032	0.022	−1.466	0.143
Reproduction	−6.479	3.337	−1.942	0.052
Survival	−2.558	1.206	−2.121	**0.034**
Time since fire (reproduction)	0.034	0.422	0.081	0.936
Time since fire (survival)	0.101	0.112	0.900	0.368
Percentage of area high-severity fire (reproduction)	0.234	0.109	2.142	**0.032**
Percentage of area high-severity fire (survival)	0.031	0.033	0.924	0.356
Foraging habitat selection				
Intercept (High severity)	1.167	2.926	0.399	0.690
Time since fire	−0.061	0.529	−0.115	0.908
Percentage of area high-severity fire in study territories	−0.084	0.068	−1.236	0.216
Low severity	1.936	0.732	2.644	**0.008**
Moderate severity	0.777	0.321	2.416	**0.016**

Note: SE, standard error. Bold values are significant at alpha = 0.05.

These results represent Spotted Owl responses to mixed-severity wildfires that burned within the past 30 yr with representative proportions of high-severity fire in a landscape mosaic. Additionally, because most of the studies in this review reported on effects from wildfire, rather than prescribed fire, the fires and their effects are representative of wildfires as they burned through currently existing forest structure, fire regime, and climate conditions. Several studies have reported that fires during the past few decades have been larger and more severe than the historical mean (Miller and Safford 2012, 2017, Mallek et al. 2013, Steel et al. 2015), but others have disputed this

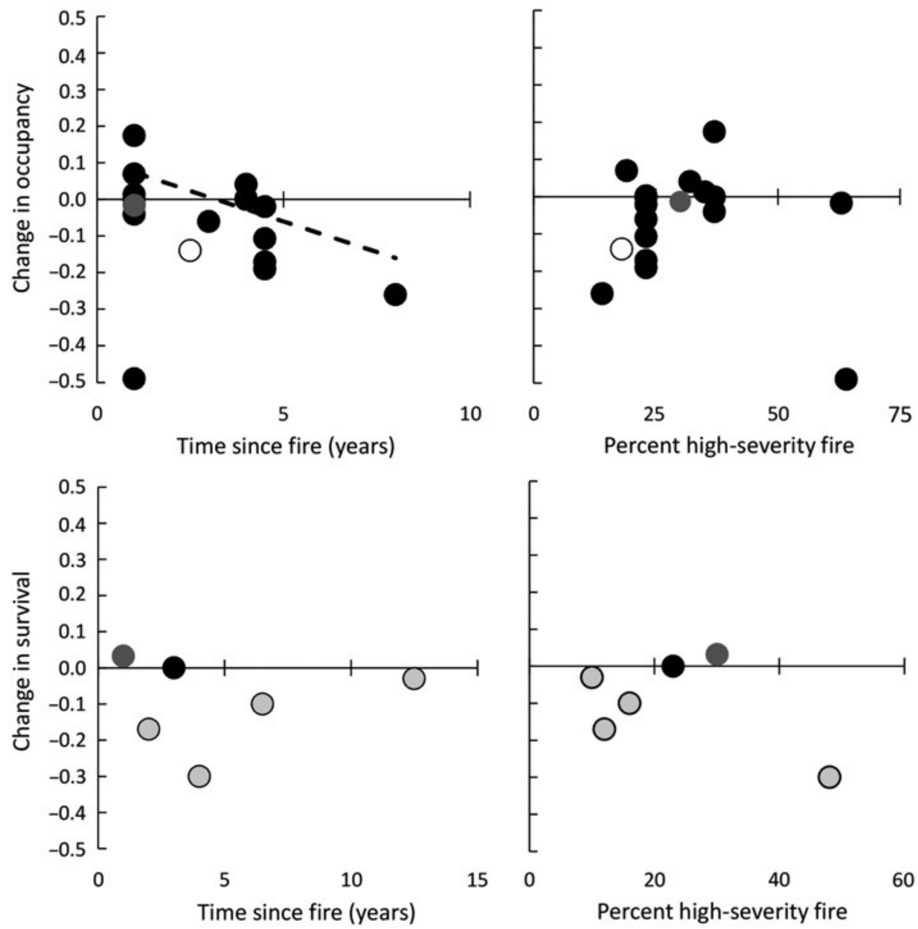

Fig. 4. Results of multi-level linear mixed-effects models (hierarchical models) meta-regression of time since fire and percent of high-severity fire in the study area as covariates to explain heterogeneity in effect sizes from mixed-severity fire on Spotted Owl (*Strix occidentalis*) parameters of breeding site occupancy and survival. The only significant effect was a reduction in occupancy with increasing time since fire, but the effect was sensitive to one study. Symbols indicate subspecies: filled black circles, California; white circles with black outline, Mexican; light gray circles with black outline, northern; and dark gray circles, all three subspecies.

point (Odion and Hanson 2006, Hanson et al. 2009, Odion et al. 2014*a*, Baker 2015*a*). Regardless of what is correct about trends in fire severity, Spotted Owls appear fairly resistant and/or resilient to effects from recent hot, large fires, wherever these fires fall in the long-term range of variability for size and amount of high-severity burn. This is corroborated by the meta-regressions that explicitly quantified the relationship between amount of high-severity fire and Spotted Owl parameters and found only a positive significant correlation (reproduction). My finding of no significant negative relationships between amount of high-severity fire and Spotted Owl parameters demonstrates that large high-severity fire patches, including territories that burn 100% at high severity as was seen in sites within several of the studies in this review, do not have unequivocally negative outcomes for Spotted Owls.

Contrary to current perceptions, recovery efforts, and forest management projects for the Spotted Owl (USFWS 2011, 2012, 2017, USDA 2012, 2018, Gutiérrez et al. 2017) mixed-severity fire as it has been burning in recent decades does not appear to be an immediate, dire threat to owl populations that require landscape-level fuel-reduction

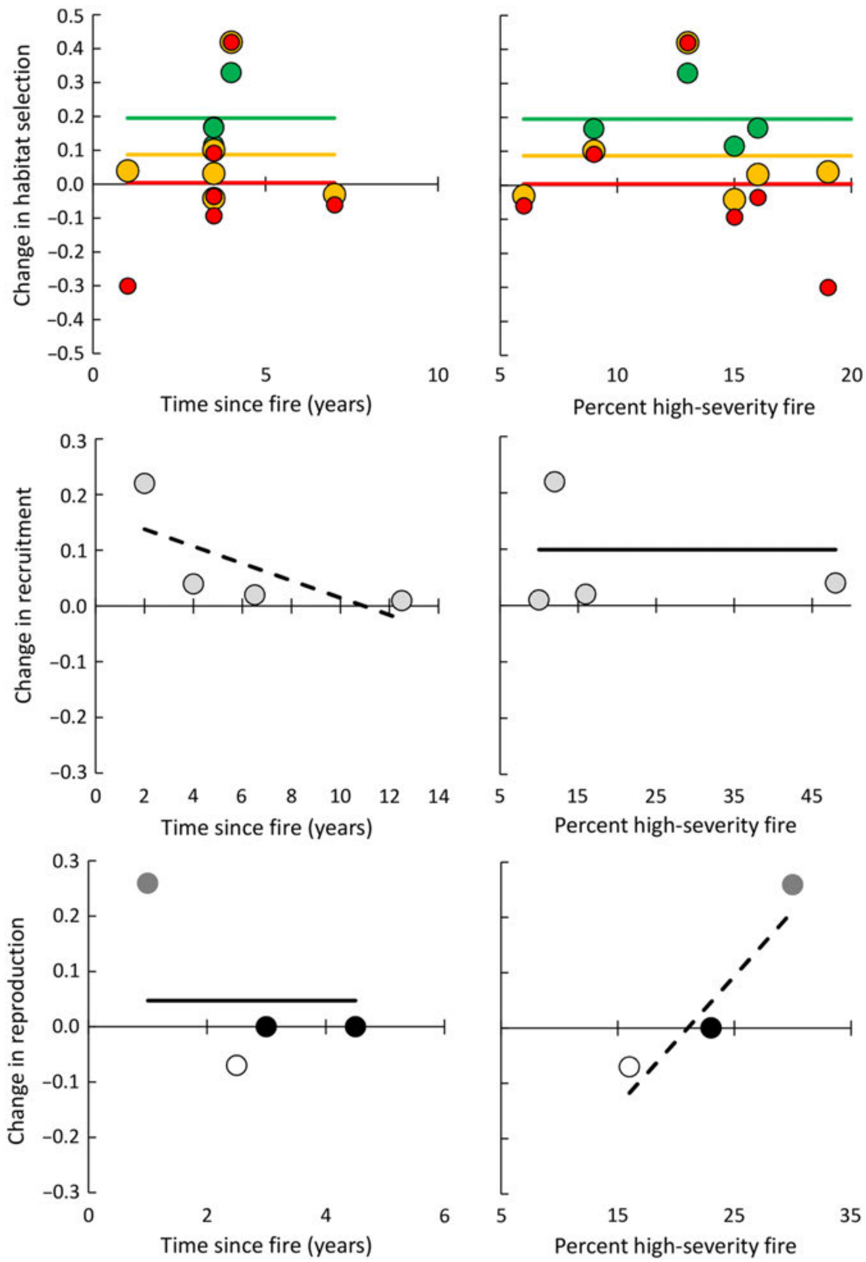

Fig. 5. Results of multi-level linear mixed-effects models (hierarchical models) meta-regression of time since fire and percent of high-severity fire in the study area as covariates to explain heterogeneity in effect sizes from mixed-severity fire on Spotted Owl (*Strix occidentalis*) parameters of foraging habitat selection, recruitment, and reproduction. Significant effects included positive selection for low- and moderate-severity burned forest for foraging, increased recruitment immediately post-fire that diminished with increasing time since fire, and increased reproduction with a positive correlation with amount of high-severity fire. In top two panels, all studies were California subspecies, and colors indicate forest in different burn severity categories: green, low severity; orange, moderate severity; red, high severity. In bottom four panels, symbols indicate subspecies: filled black circles, California; white circles with black outline, Mexican; light gray circles with black outline, northern; and dark gray circles, all three subspecies.

treatments to mitigate fire severity. Empirical studies reviewed here demonstrated that wildfires can generally have no significant effect, but effects can include improved foraging habitat, reduced site occupancy, and improved demographic rates. Most territories occupied by reproductive Spotted Owl pairs that burn remain occupied and reproductive at the same rates as sites that did not experience recent fire, regardless of the amount of high-severity fire in core nesting and roosting areas.

To place my results into perspective, mixed-severity fire typically affects (≥50% vegetation basal area mortality) a very small portion (0.02–0.50%) of Spotted Owl nesting and roosting habitat per year (Odion et al. 2014*b*, Baker 2015*b*, Stephens et al. 2016). Breeding sites that experienced a typical mixed-severity burn mosaic can be expected to have occupancy probability reduced by −0.06 on average. A 0.06 decline in occupancy is less than typical annual declines in occupancy rates observed in the Sierra Nevada in the absence of large fires (Jones et al. 2016: Fig. 3f). In comparison, post-fire logging caused a mean occupancy probability reduction of −0.18.

Post-fire logging is likely to be partially responsible for some of the negative effects attributed to high-severity fire in the studies reviewed here (Tempel et al. 2014, Jones et al. 2016, Rockweit et al. 2017, Hanson et al. 2018). Because Spotted Owl studies typically characterize territory vegetation only in the breeding core area within 1.1 km of the nest, these studies ignore habitat changes and alterations in the year-round home-range area that can extend up to 5.9 km from the nest (Zabel et al. 1992). Spotted Owl habitat protections have generally not included areas beyond 1 km from the nest, a management policy that has not contributed to population recovery.

Complex early seral forests created by fire differ from post-fire salvage-logged forests in that dead trees remain on-site, providing perching sites for hunting owls as well as food sources and shelter for numerous wildlife species (Hutto 2006, Swanson et al. 2011, DellaSala et al. 2014). Longitudinal studies also indicated that burned breeding sites where owls were not detected immediately after fire were often recolonized later (Lee et al. 2012, 2013, Tempel et al. 2016), and this review shows burned forest habitat is used for foraging, demonstrating the mistake of concluding severely burned sites or habitats are lost to Spotted Owls or require restoration (Davis et al. 2016). A recent global meta-analysis found post-fire logging is generally not consistent with ecological management objectives (Thorn et al. 2018).

This review on fire and Spotted Owls forms one portion of the evidence base for data-driven forest management. A recent systematic review of thinning and fire found 56 studies addressing fuel treatment effectiveness in real (not simulated) wildfires from eight states in the western United States (Kalies and Kent 2016). There was general agreement that thin + burn treatments (thinning immediately followed by burning) had some positive effects in terms of reducing fire severity, while treatments by burning or thinning alone were less effective or ineffective (Kalies and Kent 2016). There is also evidence that doing nothing can achieve many forest restoration goals related to age structure and fuels' density (Zachmann et al. 2018). Additional systematic reviews are needed to examine (1) the quantifiable risk of fire to Spotted Owl habitat, as there are disparate lines of evidence regarding whether fire is impeding the recovery of late-seral-stage forests; and (2) the impacts of fuel treatments on Spotted Owl demography and site occupancy. Thinning immediately followed by burning to reduce wildfire risk may or may not have adverse effects on Spotted Owls (Franklin et al. 2000, Dugger et al. 2005, Tempel et al. 2014, 2016, Odion et al. 2014*b*), but the evidence presented here indicates fire itself has arguably more benefits than costs to the species and thus suggests thinning is not necessary.

The results presented here should serve to guide management decisions, but also should be understood as limited by the available data. The sample sizes of number of estimated effects from mixed-severity fire on survival and recruitment were small and limited mainly to the northern subspecies. There were also very few studies from the Mexican subspecies. A few studies presented effect sizes that were influential on results, especially meta-regression results (Roberts et al. 2011), so studies examining longer times since fire are needed. We encourage future studies to increase sample sizes of each parameter and to provide a more balanced sample of studies from all subspecies, and over longer time frames.

Management Implications

The preponderance of evidence presented here shows mixed-severity forest fires, as they have burned through Spotted Owl habitat in recent decades under current forest structural, fire regime, and climate conditions, have no significant negative effects on Spotted Owl foraging habitat selection, or demography, and have significant positive effects on foraging habitat selection, recruitment, and reproduction. Forest fire does not appear to be a serious threat to owl populations and likely imparts more benefits than costs for Spotted Owls; therefore, fuel-reduction treatments intended to mitigate fire severity in Spotted Owl habitat are unnecessary. These findings should inform revisions to planning documents to consider burned forest, including large patches of high-severity burned forest, as useful habitat that imparts significant benefits to Spotted Owls. Forest and wildlife planning documents promote a diverse mosaic of heterogeneous tree densities and ages (USFWS 2017, USDA 2018), the very conditions created by mixed-severity wildfire, and it follows that heterogeneous post-fire structure would lead to greater variation in some Spotted Owl parameters, as was observed in the meta-analysis of variation. Planning documents (USFWS 2011, 2012, 2017, Gutiérrez et al. 2017, USDA 2018) claiming that forest fires currently pose the greatest risk to owl habitat and are a primary threat to population viability appear outdated in light of this review.

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Supporting Information

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Newly discovered landscape traps produce regime shifts in wet forests

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We describe the "landscape trap" concept, whereby entire landscapes are shifted into, and then maintained (trapped) in, a highly compromised structural and functional state as the result of multiple temporal and spatial feedbacks between human and natural disturbance regimes. The landscape trap concept builds on ideas like stable alternative states and other relevant concepts, but it substantively expands the conceptual thinking in a number of unique ways. In this paper, we (*i*) review the literature to develop the concept of landscape traps, including their general features; (*ii*) provide a case study as an example of a landscape trap from the mountain ash (*Eucalyptus regnans*) forests of southeastern Australia; (*iii*) suggest how landscape traps can be detected before they are irrevocably established; and (*iv*) present evidence of the generality of landscape traps in different ecosystems worldwide.

altered ecosystem processes | old growth

In many environments worldwide, key drivers of ecosystem change interact and reinforce one another to trigger cascades of ecosystem modification that are difficult or impossible to reverse (1–3). These cascades are often referred to as regime shifts (4–6). Examples of significant regime shifts include overfishing and trophic cascades in marine predator–prey systems (7) and human disturbance-driven losses of detritivore populations and subsequent changes in the decomposition of organic matter (8). Regime shifts are almost always identified in retrospect, making it difficult to know how to avoid them in advance and problematic to reverse their effects. Therefore, understanding of the mechanistic processes by which regime shifts occur may provide opportunities to change resource management and avoid irreversible and undesirable ecological changes.

In this paper, we describe the "landscape trap" concept, of which the outcome is a regime shift triggered by a series of feedback processes resulting from interacting natural and anthropogenic disturbances. We define a landscape trap as that wherein entire landscapes are shifted into a state in which major functional and ecological attributes are compromised. These shifts in a landscape lead to feedback processes that either maintain an ecosystem in a compromised state or push it into a further regime shift in which an entirely new type of vegetation cover develops. Landscape traps are large-scale ecological phenomena that arise through a combination of altered spatial characteristics of a landscape coupled with synergistic interactions among multiple human and natural disturbances. Thus, changes in the frequency and spatial contagion of large-scale disturbances are the key interacting factors driving entire landscapes into an undesirable and potentially irreversible state (i.e., landscape trap). We demonstrate the concept with examples involving spatial and temporal feedback between logging and fire in forest ecosystems and also provide examples of landscape traps in other environments.

Like other kinds of ecological traps, the landscape trap concept shares characteristics like shifts between alternative stable states and multiple feedback processes (9). However, a focus at a landscape scale and on temporal and spatial changes in disturbances sets the landscape trap concept apart from other kinds of ecological traps and regime shifts, such as population traps and extinction vortices in small populations of animals (10) and elevated rates of animal species loss below threshold levels of native vegetation cover (11).

To the best of our collective knowledge, the landscape trap concept has not been previously reported, yet we argue that landscape traps may be more prevalent in ecosystems around the world than currently recognized. Common ingredients contributing to landscape traps are (*i*) overharvesting of natural resources in a landscape; (*ii*) climate change effects on species' life histories and/or the frequency and severity of ecological disturbances; (*iii*) major changes in the spatial characteristics of landscapes; (*iv*) feedback loops between the changed environmental conditions and other major stressors; and (*v*) severely impaired ecological functions of a landscape in an altered state, such as, for example, reduced populations of species and habitat suitability, reduced carbon storage, and reduced water and timber production. The interaction of these factors is shown in a conceptual model in Fig. 1.

We suggest that landscape traps exist in many ecosystems. For example, logged tropical rainforests in parts of Asia have become more fire-prone (12). Postfire salvage logging in some of these rainforests, in turn, changes the vegetation composition toward more fire-prone grassland taxa. Additional fire further degrades fire-sensitive remnant rainforest, eventually leading to a regime shift to exotic fire-promoting grasslands, limiting opportunities for the vegetation to revert to tropical rainforest (13). Such kinds of interrelationships between logging and altered fire regimes are widespread in tropical rainforests in many other parts of the world, including South America and Africa (14), as are relationships between logging and exotic fire-prone grasses (15).

Temperate forests are not immune to such traps. In moist temperate forests of western North America, logging-related alterations in stand structure increase the risk for both occurrence and severity of subsequent wildfires through changes in fuel types and conditions (16, 17). High-severity wildfires kill young trees planted following previous logging operations. This necessitates reforestation efforts, but these young stands are susceptible to being killed in subsequent recurring high-severity fires (16). Similar kinds of relationships between logging regimes and altered fire regimes have been reported in a range of forest types elsewhere around the world (reviewed in 18).

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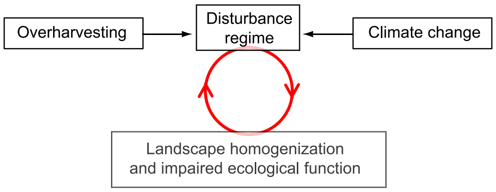

Fig. 1. Conceptual model of a landscape trap. The trap results from the reinforcing feedback loop shown in red.

Results and Discussion

Specific Example of a Landscape Trap: Mountain Ash Forests of Victoria, Southeastern Australia. The specific example of a landscape trap that we present comes from the mountain ash (*Eucalyptus regnans*) forests of southeastern Australia in the central highlands of Victoria. The likely regime shift is from landscapes dominated by old-growth forests that are 200–450 y of age to those dominated by young fire-prone forests that do not survive to become old growth. Evidence comes from new spatial information following massive wildfires in 2009, perhaps the most economically destructive in Australian history (19), coupled with understanding that has emerged from 28 y of extensive field information and associated data analyses in mountain ash forests (20).

The central highlands of Victoria support ∼121,000 ha of mountain ash forest. These are spectacular forests with old-growth trees reaching 90 m or more in height (14). Mountain ash forests persist only within a particular fire regime (*sensu* 21). Before European settlement over 150 y ago, the fire regime was infrequent severe wildfire that occurred in late summer (22). Young seedlings germinate from seed released from the crowns of burned mature trees to produce a new even-aged stand (20). Wildfires may be stand-replacing, because the young trees regenerating after fire belong to a single age cohort (23). When the interval between stand-replacing disturbances is less than 20–30 y (which is the period required for trees to reach sexual maturity and begin producing seed) (24), stands of mountain ash forest will be replaced by other species, particularly wattle (*Acacia* spp.) (20).

In the past century, a new disturbance regime (logging) has been added to the previous natural fire regime. Large areas of mountain ash have been subject to timber and pulpwood harvesting (Fig. 2). In the past 40 y, the traditional method of logging has been clear-cutting, in which all merchantable trees within a 15- to 40-ha area are cut in a single operation (25). Following clear-cutting, logging debris is burned to create a bed of ashes in which the regeneration of a new eucalypt stand takes place, often by artificial reseeding. The vast majority of mountain ash landscapes have become dominated by large areas of regrowth forest with small areas of old forest embedded within them. Old-growth mountain ash forest (*sensu* 20) typically covers less than 3% of the majority of the 3,000- to 6,000-ha wood production forest blocks in the central highlands; however, in some cases, it is less than 1% (20). Indeed, following more than a century of logging and wildfires in 1926, 1932, 1939, 1983, and, most recently, 2009, ∼1.1% of the entire mountain ash forest estate is now in an old-growth stage. This landscape is in stark contrast to mountain ash landscapes 100–150 y ago, which historical accounts (e.g., 26), coupled with stand reconstruction work relating to tree age and stem diameters of large dead (snag) trees remaining within young stands (27), suggest were dominated by large areas of old growth, possibly as high as 60–80% total cover in the central highlands of Victoria (20) (Fig. 2).

Development of a Landscape Fire-Trap in Mountain Ash Forests. The interacting effects of wildfire, logging, and the combination of wildfire and logging (i.e., salvage logging) (*sensu* 28) are creating a previously unrecognized landscape trap in which the disturbance dynamics of "trapped" mountain ash forest landscapes are markedly different from those before European settlement (Figs. S1 and S2). The core process underlying this landscape trap is a positive feedback loop between fire frequency/severity and a reduction in forest age at the stand and landscape levels, leading to an increased risk for dense young regenerating stands repeatedly reburning before they reach a more mature state (Fig. 3). The landscape trap will potentially create irreversible changes in disturbance dynamics, forest cover, landscape pattern, and vegetation structure, and thereby lead to a major regime shift or alternative state. We explain below the evidence for the positive feedback process that underpins this landscape trap (Fig. S2) and discuss why it is historically unprecedented and why it is beginning to dominate the contemporary landscape.

Positive feedback loop between reduced forest stand age and fire. Young stands of mountain ash forest are created by natural regeneration following wildfire. Detailed on-site measurements following the 2009 wildfires have revealed that young forest burns at higher severity than mature forest. We suggest this is for four key reasons:

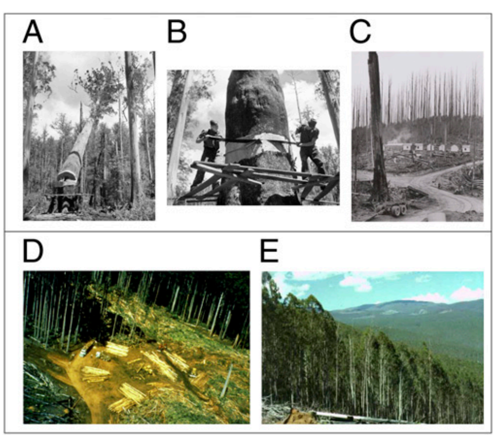

Fig. 2. Photo montage showing historical logging in extensive stands of old-growth forest (*A*–*C*) and extensive clear-cut areas of forest cut in the past 10 y (*D* and *E*) in the mountain ash forest in the central highlands of Victoria. (Photos courtesy of National Archives of Australia, State Library of Victoria and D.B.L.)

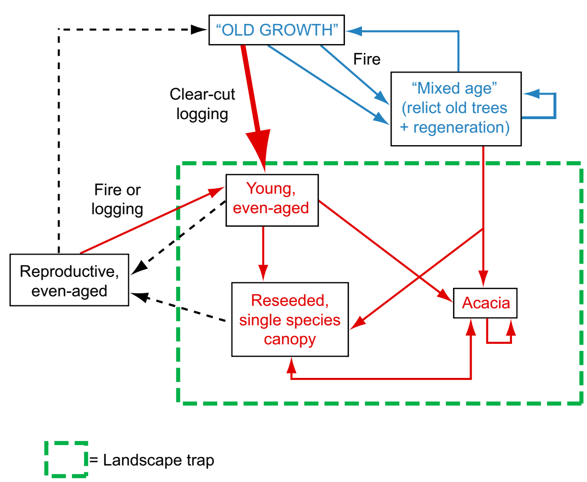

Fig. 3. Development of a landscape trap in the mountain ash forests of the central highlands of Victoria.

i) Young regenerating stands of mountain ash trees are characterized by densely spaced regrowth saplings. There can be several million eucalypt seedlings per hectare soon after a fire or logging. Through processes of rapid natural self-thinning, this declines to ∼400 stems per hectare at 40 y and 40–80 stems per hectare in mature forest after 150–200 y (29). The marked reduction in the number of stems per unit area over time is primarily attributable to competition-derived death and collapse of small suppressed pole and sapling trees, which add greatly to the density of the vegetation in young regrowing forests but do not generally occur in mature and old-growth mountain ash forests (30). Densely spaced stands of regrowth saplings, coupled with the subsequent natural processes of rapid self-thinning that characterize the early stages of stand regeneration in mountain ash forests, create significantly more fine and medium fuels than in old forests (31).

ii) The closely spaced crowns in densely stocked young stands are readily susceptible to carrying a crown fire. This is in contrast to old-growth stands, which are characterized by large relatively well-spaced trees with open crowns and small lateral subcrowns (24).

iii) Trees in young stands are shorter than those in old-growth stands. The flame height needed to scorch or consume the canopy in young stands is therefore significantly lower than in old-growth stands (22).

iv) Young forests support significantly smaller diameter logs on the ground than old-growth stands (32). Such smaller diameter logs support significantly less dense and luxuriant moss mats than larger diameter fallen trees. Moss mats hold large amounts of water (1,100% of dry weight) (33); they play a significant role in moisture retention within logs, and thereby may reduce the risk for burning.

Why has this positive feedback loop not occurred historically? Before European settlement, frequent, widespread, high-severity wildfires in mountain ash forests would have been suppressed by a combination of extended periods of wet climatic conditions and the absence of the intensive human disturbances resulting from clear-cut logging. This favored a negative feedback loop between forest age and fire, enabling young forest to mature into a less fire-prone state that was not conducive to widespread high-severity wildfire (Fig. S1).

Why is this positive feedback loop now beginning to develop? Two major changes have occurred relatively recently to favor the positive feedback loop: reduced forest age in mountain ash forests and increased fire frequency (Fig. 3 and Figs. S1 and S2). First, there has been a 25% reduction in rainfall in southeastern Australia over the past few decades (34). Second, logging has converted more than 90% of formerly old forest to young regenerating stands. Young forest resulting from clear-cut logging has two added elements of fire proneness: (*i*) fine fuels created by logging operations are added to those from the collapse of small-diameter stems and shedding of branches during natural self-thinning and self-pruning processes in densely stocked regenerating stands, and (*ii*) the spatial pattern of stand age classes in mountain ash landscapes has been altered, with an increased prevalence of young densely stocked forest and a significantly reduced area of (mesic) old-growth forest. This, in turn, has increased the fire contagion in the landscape.

Codes of logging practice and the practical logistics of harvesting operations mean that clear-cutting is applied to flatter and more accessible parts of mountain ash landscapes. However, these places are also where old-growth stands were formerly most likely to occur. Evidence for this comes from work in closed-water catchments of the central highlands of Victoria, where there were no confounding effects of past and present human disturbances that would have otherwise obscured key spatial patterns of forest age classes (22). Before the 2009 wildfires, old growth mountain ash occupied a subset of the overall environmental domain of mountain ash per se, typically within a narrow band of mesic sites rather than ridges or steep slopes. This environmental domain was not only favorable for tree growth but interacted with spatial differences in natural disturbance regimes (35). Mesic sites support taller trees. They are also places where both the fire frequency and the intensity of past wildfires were attenuated (22). Former areas of old-growth forest on flat terrain have now been converted to young regenerating stands and are spatially connected to young burned or logged forest on midslopes and ridges. Importantly, the more widespread that young logged and regenerated forest becomes, the greater is the risk for increasing spatial contagion in the spread of wildfire through landscapes (31), because moist remnant areas that would have slowed or halted the spread of fire (and formerly supported old forest) have been converted to young forest. Spatial contagion in recurrent high-severity fire may therefore reinforce a pattern of increasing homogeneity in the cover of young forest in a landscape (Fig. S2). This pattern occurs because some areas of fire refugia (e.g., flat plateau, deep south-facing valley floors) that were traditionally characterized by a long absence of fire (particularly high-severity fire) and supported stands of multiaged forest or old-growth forest (35) become more susceptible to being burned by high-severity conflagrations that spread from adjacent more flammable logged and young regenerating areas (Figs. S1 and S2). Notably, although natural disturbance regimes often increase heterogeneity in many landscapes (36), the opposite frequently occurs in areas subject to landscape trap phenomena, in which the combination of human and natural disturbance regimes can lead to increased landscape homogeneity.

Research in moist forests around the world suggests that other factors associated with logging may increase susceptibility of young regenerating forests to being burned or reburning at high severity. For example, the large quantities of logging slash created by harvesting operations can sustain fires for longer than fuels in unlogged forest (12). Similarly, lightning strike ignition is more likely to occur in harvested stands because of increased fine fuels resulting from logging slash, and this effect may remain for 10–30 y following logging (37). Finally, the removal of trees by logging creates microclimatic conditions that lead to increased drying of understory vegetation and the forest floor, and a correspondingly elevated fire risk (38).

Once a mountain ash forest landscape is dominated by widespread areas of young fire-prone forest, the elevated risk for high-severity spatially contagious fire decreases the probability that the landscape can return to its former mature state, particularly under the drier and warmer conditions associated with climate change. Hence, the dynamics of trapped mountain ash forest landscapes are different from those in the past (>100 y ago) (Fig. 3 and Figs. S1 and S2). The current set of interacting disturbance regimes of fire, logging, and postfire (salvage) logging did not exist before European settlement. Importantly, there is a major asymmetry in the period during which mountain ash forest ecosystems have coevolved with natural disturbances (>20 million y) compared with the 20–100 y during which the interacting human and natural disturbance regimes have produced a landscape trap.

End point: Regime shift? The positive feedback cycle of widespread young regenerating stands and frequent high-severity wildfire means that either extensive areas of trapped young mountain ash forest will be maintained or a further regime shift will occur in which a new type of vegetation cover develops, particularly wattle (*Acacia* spp.) (Fig. 3 and Figs. S1 and S2). Once mountain ash has been eliminated from an extensive area, it recolonizes slowly because the seed released from the crowns of burned mature trees disperses ∼1.5–2.0 crown heights from a source

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tree and successful regeneration (fire) events may occur every 30–400 y. Therefore, the regeneration niche, which is a key part of the life cycle of mountain ash (39), is maladapted to the altered landscape conditions and altered fire regime created by recurrent logging and wildfire. Recurrent high-frequency wildfire may result in repeatedly burned areas that were formerly dominated by mountain ash being colonized by other eucalypt species that do not depend on seedling regeneration but, instead, recover after wildfire via strategies like epicormic resprouting [e.g., shining gum (*Eucalyptus nitens*), messmate (*E. obliqua*)].

Irrespective of whether mountain ash forest landscapes remain trapped as widespread, young, fire-prone stands or undergo a regime shift to extensive areas dominated by *Acacia* spp. and other species, such changes will result in significant impairment of ecological functions like carbon storage, water production (40, 41), and biodiversity conservation. For example, neither young small-diameter mountain ash trees nor *Acacia* spp. support the cavities that are crucial nesting and denning sites for many species of animals. They also lack critical structural features, such as extensive bark streamers, that are key foraging microhabitats for wildlife (42). These changes in vegetation structure are likely to lead to irreversible losses in habitat suitability for ∼40 species of vertebrates in mountain ash forests that are dependent on large 120- to 150+-y-old trees with hollows.

Avoiding a Landscape Trap in Mountain Ash Forests of Victoria. Three important strategies are needed to reduce the problems created by the landscape trap in the mountain ash forests of Victoria. First, large (>1,000 ha) areas of currently unburned forest need to be retained, wherein the number of anthropogenic stressors is reduced. The area of green forest was reduced dramatically by the 2009 wildfires; hence, relative biodiversity, carbon storage, and water production values of remaining unburned forest have increased. However, such uncommon areas of unlogged forest are increasingly sought after for timber and pulpwood harvesting because (*i*) they are among the declining number of places suitable for cutting as a consequence of past fires and past (prefire) logging operations, (*ii*) there are legislated guarantees to provide logging contractors with forest to cut for timber and pulpwood (43), and (*iii*) cutting burnt forest (i.e., salvage logging) has major negative environmental impacts and long-term effects on forest recovery and forest biodiversity (28). Targeting limited remaining areas of unburned forest for logging depletes the overall amount of these forests, with long-term economic implications for harvest contractors. Increased logging pressure on green areas has other ecological implications: Remaining areas of green forest are important refugia for biodiversity following wildfires and are critical for underpinning postfire ecological recovery (32). Legislative and other impediments to reducing harvest levels highlight the existence of management and socioeconomic traps within landscape traps, and these need serious and timely review.

A second strategy to avoid the development of a landscape trap in the now highly fire-prone mountain ash landscapes of Victoria is to recalculate the sustained yield to accommodate future losses of timber resulting from the inevitable burning of some parts of forest landscapes. This strategy has the advantage of not overcommitting remaining unlogged green forest in the event of wildfires, thereby resulting in more conservative management of natural resources and more explicit recognition of the uncertainty created by major natural disturbances.

Given the extent of recently burned forest in Victoria, a third important strategy to reduce the risks for development of a landscape trap is to try to limit the amount of future fire. Although mountain ash trees are dependent on fire to promote regeneration, fires have been extensive in the past 25–100 y; another fire in the coming 20 y within currently young regenerating stands is likely to lead to a major regime shift (Fig. 3). Reducing the amount of fire in mountain ash forests is a significant challenge. Broad-area prescribed burning is not a viable management option because high levels of moisture in the vegetation and large quantities of biomass make planned fires extremely difficult to control (20). However, prescribed burning as part of a regime of fire can be an appropriate management option in drier forest types that are adjacent to mountain ash forests. Carefully applied strategic burning in such drier environments may help to reduce the extent of spatial contagion in wildfire that occurs in these areas and, in turn, reduce the risk for adjacent stands of mountain ash forest being burned (44).

Examples of Landscape Traps in Ecosystems Other Than Forests. We contend that landscape traps may be prevalent in many ecosystems. For example, climate change and overfishing have facilitated the conversion of subtidal kelp (*Macrocystis pyrifera*) forests in Tasmanian coastal waters to "barrens" habitat resulting from overgrazing by the sea urchin *Centrostephanus rodgersii*. Ocean warming and altered circulation patterns have enabled the poleward spread of this sea urchin (45), and overfishing of predators, such as the southern rock lobster (*Jasus edwardsii*), has enabled *C. rodgersii* to establish high-population density barrens that result in the loss of biodiversity and a reduction in the productivity of fisheries and contribute to the decline of such predators as *J. edwardsii* (46). Aquatic environments where water quality can be radically altered by nutrient inputs from human activities (e.g., 47) also are susceptible to the development of landscape traps.

Grazing on public lands in the western United States has been blamed for reducing biodiversity and, together with exotic weeds, may have led these grassland ecosystems into a landscape trap that produces a plant community from which there is no going back (48). Livestock grazing in western United States may have reduced the abundance of preferred plant species while subjecting the soil to weed invasion, such that large areas are now degraded rangelands in the same manner illustrated in eastern Australia by the "woody weed" problem in semiarid woodlands (49). Introduced grasses, such as cheatgrass (*Bromus tectorum*), can similarly move grassland communities in the intermountain western United States into a regime change that is nearly impossible to reverse (50, 51). A lack of reversible change may be best illustrated by landscape traps in regions heavily impacted by disturbances like mountaintop mining (52).

Concluding Comments

We suggest that strategies and management interventions are needed to reduce the probability of landscape traps developing (Fig. 4). One approach is to recognize that landscape traps can exist and identify the suite of spatial and temporal characteristics that can combine to give rise to them, including (*i*) exploitation of the natural resources in a landscape through unsustainable levels of harvesting; (*ii*) alteration in the spatial characteristics of landscapes, including modifications to the frequency and severity of ecological disturbances; (*iii*) feedbacks between altered environmental conditions and other major anthropogenic stressors;

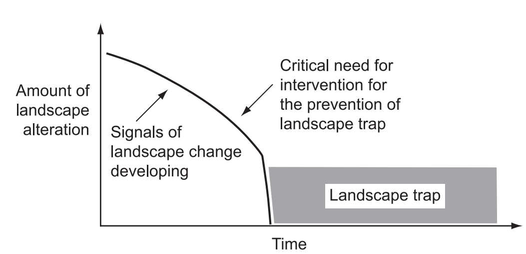

Fig. 4. Conceptual model highlighting signals and interventions required to reverse the development of a landscape trap.

and (*iv*) severely impaired landscape processes and functions. A second approach is to limit the number of anthropogenic stressors in landscapes and reduce the potential for negative interactions among multiple stressors. This may equate to a more conservative approach to the harvesting of natural resources or, in other cases, application of management strategies that reduce feedbacks (e.g., fuel reduction through prescribed burning). Sustained yields of natural resources also may need to be rapidly reassessed following catastrophic events to avoid overcommitting remaining intact areas and further increasing the risk for creating a landscape trap.

We suggest that the need for proactive management to prevent the development of landscape traps is critical, given that (*i*) landscape traps might be at increased risk for development in response to significant "events" like major natural disturbances, which are likely to become more frequent, more severe, or both under rapid climate change in many regions (e.g., 53, 54), and (*ii*) marked asymmetry exists between the rapidity with which landscape traps may develop and the prolonged time scales (hundreds to thousands of years) that characterize natural ecological processes and natural disturbance regimes.

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CONSERVATION ECOLOGY

Report

USDA Forest Service Roadless Areas: Potential Biodiversity Conservation Reserves

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ABSTRACT. In January 2001, approximately 23 x 10^6 ha of land in the U.S. National Forest System were slated to remain roadless and protected from timber extraction under the *Final Roadless Conservation Rule*. We examined the potential contributions of these areas to the conservation of biodiversity. Using GIS, we analyzed the concordance of inventoried roadless areas (IRAs) with ecoregion-scale biological importance and endangered and imperiled species distributions on a scale of 1:24,000. We found that more than 25% of IRAs are located in globally or regionally outstanding ecoregions and that 77% of inventoried roadless areas have the potential to conserve threatened, endangered, or imperiled species. IRAs would increase the conservation reserve network containing these species by 156%. We further illustrate the conservation potential of IRAs by highlighting their contribution to the conservation of the grizzly bear (*Ursos arctos*), a wide-ranging carnivore. The area created by the addition of IRAs to the existing system of conservation reserves shows a strong concordance with grizzly bear recovery zones and habitat range. Based on these findings, we conclude that IRAs belonging to the U.S. Forest Service are one of the most important biotic areas in the nation, and that their status as roadless areas could have lasting and far-reaching effects for biodiversity conservation.

INTRODUCTION

In January 2001, the Clinton administration promulgated its *Roadless Area Conservation Rule,* which states that 237,000 km^2 of inventoried roadless areas (IRAs) within the U.S. National Forest System will remain roadless and protected from timber extraction (USDA Forest Service 2000). These lands represent 31% of the National Forest System or 2.5% of the total U.S. land base (DeVelice and Martin 2001). They would increase the amount of strictly protected land area in the United States in IUCN categories I–III from 4.8 to 8.5%. Beyond these most basic statistics, few studies have analyzed the potential contribution of IRAs to biodiversity conservation (Martin et al. 2000, DeVelice and Martin 2001).

DeVelice and Martin (2001) assessed the extent to which IRAs could contribute to building a representative network of conservation reserves in the United States. Using ecoregions as their unit of analysis (Ricketts et al. 1999), they found that IRAs could potentially expand ecoregional representation, increase the area of reserves at lower elevations, and increase the size of conservation areas to provide refuge for wide-ranging species. However, in their assessment they did not evaluate the contribution of IRAs toward the conservation of biodiversity and populations of specifically threatened, endangered, or imperiled species.

The lands belonging to the USDA Forest Service contain more than 80% of mammal and reptile species and more than 90% of the bird, amphibian, and fish species in the United States, including many that have been extirpated from large portions of their presettlement ranges (USDA Forest Service 1997). According to the NatureServe database, more than 1400 of these species have been designated as threatened and endangered (TE) species under the *Endangered Species Act* (ESA). The *Forest Service Roadless Area Final Environmental Impact Statement* identified approximately 400 TE or proposed species found on USDA Forest Service land and an estimated 220 (55%) that are directly or indirectly associated with IRAs (USDA Forest Service 2000). IRAs provide or influence designated critical habitat for at least 30 of these species (USDA Forest Service 2000).

However, the ESA list is not a complete listing of imperiled species. There are numerous species that are globally rare or threatened with extinction but for

[1]World Wildlife Fund; [2]NatureServe; [3]Pinchot Institute

various reasons do not appear on the ESA TE species list. Many of these species also occur on USDA Forest Service land. To fill this gap, we supplemented the TE species list with species categorized as critically imperiled or imperiled according to NatureServe's central database.

The objective of this paper is to assess three critical questions associated with IRAs:

> Is there a high concordance between IRAs and ecoregions of particular biodiversity values?
>
> Do IRAs overlap with threatened, endangered, or imperiled species?
>
> Is there potential for IRAs to assist in the conservation of wide-ranging species, such as the threatened grizzly bear (*Ursos arctos horribilis*), in the conterminous United States?

METHODS

We obtained the spatial coverages of the inventoried road areas (IRAs) in vector format from the roadless area conservation Web site (Table 1).

Fig. 1. Overlap of USDA Forest Service inventoried roadless areas (IRAs) with ecoregions that contain USDA Forest Service lands. The bold line indicates the separation of IRAs into three geographic regions: east, west, and Alaska.

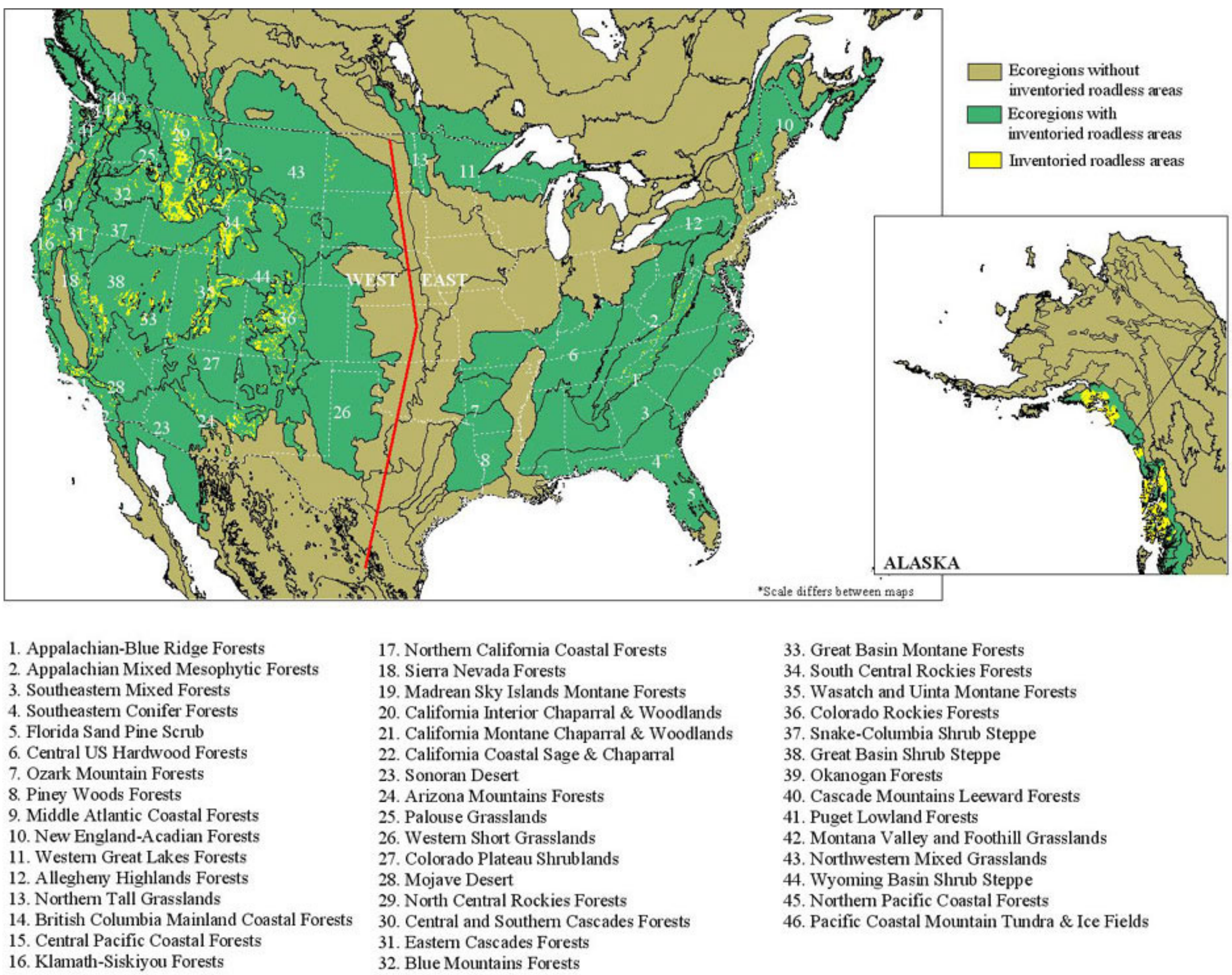


Table 1. Data sources. All data web data sources were accessed in February 2001.

Database name	Source
USDA Forest Service roadless area database	http://roadless.fs.fed.us/documents/feis/data/gis/coverages/index.shtml
World Wildlife Fund ecoregions database	Ricketts et al. 1999
NatureServe central databases	NatureServe
Protected areas database	Conservation Biology Institute and World Wildlife Fund
Grizzly bear recovery area boundaries	U.S. Fish and Wildlife Service and University of Montana

Ecoregions

As seen in Fig. 1 and Table 1, we evaluated the potential benefit of IRAs for biodiversity conservation using the ecoregions and biological importance rankings provided in Ricketts et al. (1999). Using ArcView 3.2, we combined the IRAs and ecoregion coverages, both in vector format. To facilitate interpretation, we separated our analysis into three geographic regions, i.e., the eastern United States, the western United States, and Alaska, following the methodology used by DeVelice and Martin (2001).

Ricketts et al. (1999:7) defined an ecoregion as " ... a relatively large area of land or water that contains a geographically distinct assemblage of natural communities." Ecoregions were selected as the units of analysis because they integrate ecological, biological, and geographic considerations into land-use decision making and are being used to establish priorities for large-scale conservation efforts (Omernik 1995*a*,*b*, Ricketts et al. 1999, Groves et al. 2002). Where ecoregions extend into either Canada or Mexico, we included only those portions within U.S. boundaries for all analyses. Although we would have preferred to maintain ecoregional contiguity, the spatial nature of USDA Forest Service lands and the applicability of the *Endangered Species Act* required strict adherence to political boundaries.

Ricketts et al. (1999) classified the biological importance of each ecoregion based on species distribution, i.e., richness and endemism, rare ecological or evolutionary phenomena such as large-scale migrations or extraordinary adaptive radiations, and global rarity of habitat type, e.g., Mediterranean-climate scrub habitats. They used species distribution data for seven taxonomic groups: birds, mammals, butterflies, amphibians, reptiles, land snails, and vascular plants (Ricketts et al. 1999). Each category was divided into four rankings: globally outstanding, high, medium, and low. The rankings for each of the four categories were combined to assign an overall biological ranking to each ecoregion. Ecoregions whose biodiversity features were equaled or surpassed in only a few areas around the world were termed "globally outstanding." To earn this ranking, an ecoregion had to be designated "globally outstanding" for at least one category. The second-highest category, or continentally important ecoregions, were termed "regionally outstanding," followed by "bioregionally outstanding" and "nationally important" (Ricketts et al. 1999). Although our analyses focused on those ecoregions characterized as globally and regionally outstanding, even the lowest category, nationally important, contains important biodiversity in a local context.

Threatened, endangered, and imperiled species

Currently, public land managers are required to monitor populations of threatened and endangered (TE) species and, where appropriate, develop management plans to conserve these populations and their habitat requirements (U.S. Fish and Wildlife Service 1973). Previous studies have analyzed the distribution of TE species based on counties, or boroughs in Alaska, and identified high-concentration areas of TE species and associated habitats (Dobson et al. 1997, Flather et al. 1998, Stein et al. 2000). Despite their valuable findings, these previous studies were limited by the coarse level of spatial resolution and the use of political units of disparate sizes. To avoid similar limitations with our analysis, we use data of a finer resolution to identify levels of concordance between the locations of IRAs and TE species.

The NatureServe central database (Table 1) provided the finer-resolution data for the identification of the locations of TE species. Data for this database are developed by state natural heritage programs and managed by NatureServe. Natural heritage programs have documented and tracked the occurrence of threatened, endangered, and imperiled species for nearly 30 yr (Jenkins 1985, 1988, 1996). The system assigns global conservation status ranks known as "element global ranks" or "G-RANKS" to species and communities that are intended to estimate the extent of their imperilment or vulnerability. Conservation status ranks are assigned based on an assessment of rarity, the extent of recent decline of populations, threats, biological fragility, and other factors (Stein et al 2000). The most imperiled species and communities are ranked G1, and the most stable ones are ranked G5.

The NatureServe central database includes fields for federal ESA listing status and for global conservation status. We selected records of species that are federally listed as threatened or endangered (TE) according to the U.S. Fish and Wildlife Service or the National Marine and Fisheries Service and those that are ranked by NatureServe as critically imperiled (G1) or imperiled (G2). The output file was a vector file of 109,125 occurrences of species with G1 or G2 rankings or federal ESA listings. These occurrences were collated into 7.5-min quadrangles from the U.S. Geological Survey. The largest quadrangles, in the southern part of the United States, are 179 km^2. We used two data products for our analyses. The first contains only TE species (Fig. 2), and the second contains TE, G1, and G2 species (Fig. 3). The spatial resolution of the locational data varied according to the equipment and methodologies that natural heritage programs used in collecting the data. However, the maximum uncertainty for the data set was less than the area of a quadrangle grid cell.

Fig. 2. Threatened and endangered (TE) species distributions by the 7.5-min quadrangles of the U.S. Geological Survey.

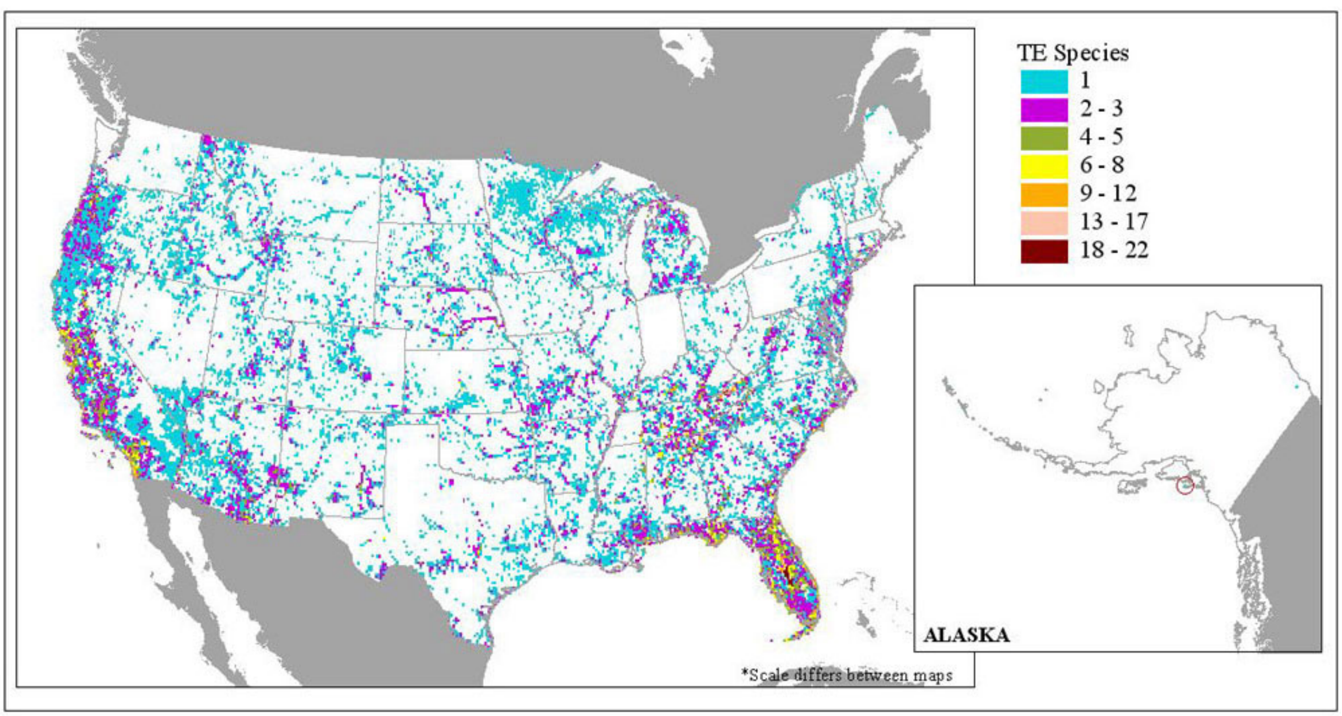
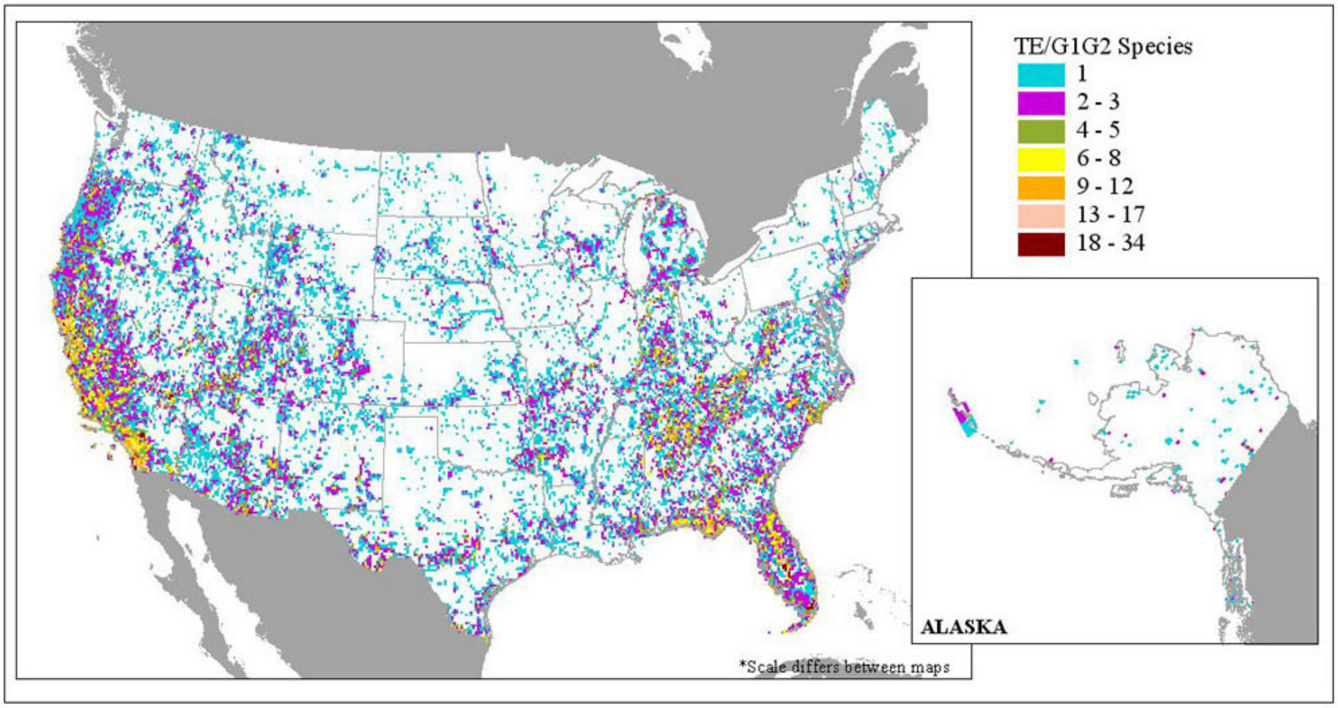


Fig. 3. Threatened and endangered (TE) species and critically imperiled (G1) and imperiled (G2) species distributions by the 7.5-min quadrangles of the U.S. Geological Survey.



The TE, G1, and G2 data sets demonstrate only a moderate degree of overlap. These discrepancies occur partly because the NatureServe system evaluates only biological factors, whereas species are assigned to federal listings for both scientific and political reasons. There are 75,000 occurrences of TE species, but only 27,000 are ranked G1 or G2 by the NatureServe system. Of the 1409 ESA-listed TE species in the NatureServe database, 1109 are ranked G1 or G2. Conversely, there are 5997 species ranked G1 or G2 that are not classified as TE species. Of the 61,000 occurrences of G1 and G2 species recorded in the NatureServe database, more than 33,000 occurrences lack a TE species designation. One of the reasons for the disparity between the high concordance of species but the low concordance of occurrences is the fact that certain species are wide-ranging. For example, the grizzly bear, which is a threatened species but not a G1 or G2 species, is recorded often across its wide range, so that it accounts for far more records than a narrow endemic species that is both TE and listed as G1 or G2.

The NatureServe database contains information gaps (Table 2). However, although the missing data for Idaho, Montana, and Washington are critical for the conservation of individual species, the lack of them served only to make our analysis a more conservative estimate of the potential contributions of IRAs to species conservation. There are no IRAs in Massachusetts and only one in Maine, with a total area of 24 km^2.

We overlaid both the TE species and TE/G1–G2 species databases with the uniquely named IRAs to identify the percentage of IRAs that contain known occurrences of TE or G1–G2 populations. In instances where multiple quadrangles containing species occurred within a single IRA unit, we erred on the conservative side and used only the quadrangle that contained the most species, i.e., we assumed that multiple quadrangles would contain the same species.

We also analyzed the relative increase in conservation reserves that IRAs would confer to TE and TE/G1–G2 species. We overlaid the TE and TE/G1–G2 databases with a conservation area database compiled by the Conservation Biology Institute and World Wildlife Fund (Table 1). This database includes all federal, state, county, and municipal public lands and some private lands. The private lands have not been systematically surveyed and do not include conservation easements. We used only lands that are classified for strict biodiversity conservation, which we define as those designated as categories I–III by the IUCN. Category I is for Strict Nature Reserves/Wilderness Areas, category II covers National Parks, and category III includes National Monuments (The World Conservation Union 1978, The World Conservation Union 1994). Hereafter we refer to the areas that meet these criteria as "conservation reserves." We did not include protected-area categories IV–VI, which allow road building, timber harvesting, and other extractive activities in our analysis. Of 78 x 10^6 ha of National Forest land, 14 x 10^6 ha are designated as National Wilderness Areas, and an additional 2.5 x 10^6 ha are classified as Special Designated Areas that are IUCN category I reserves. The remaining 61.5 x 10^6 ha of National Forest land, which are not classified as conservation reserves, are governed by periodic management plans that may allow or restrict resource uses and extraction.

Table 2. Gaps in data available for this study.

State	Missing data
Idaho	Fish data
Maine	Animal data
Massachusetts	All data
Montana	Canada lynx, bull trout, gray wolf data
Washington	Most animal data

Grizzly bear case study

Finally, because national analyses can obscure important details of individual species, we also analyzed the potential contribution of IRAs to grizzly bears (*Ursos arctos horribilis*), specifically in relation to the regions designated as grizzly bear recovery areas by the U.S. Fish and Wildlife Service (Table 1). We overlaid these grizzly bear recovery zones with the IRAs to assess the concordance of these areas. We chose grizzly bears because they are a federally listed threatened species in the conterminous United States and require large and contiguous habitat areas to survive.

All spatial databases were in vector format and put into a common projection prior to the overlap analysis. All spatial estimates derived from our analyses were obtained by summarizing the area of overlap of the respective GIS databases. One caveat of our methodology is that the combination of multiple GIS layers may lead to the propagation of spatial errors and increased uncertainty (Flather et al. 1998, Heuvelink 1998). This concern is a generalized methodological one. Our errors are no greater or smaller than those of any similar analysis that uses multiple spatial data from multiple sources. The TE species databases, protected areas database, and IRA coverages represent a vast collection of data from many sources. It is likely that errors are associated with each of these layers. However, most of our analyses were conducted at a sufficiently broad scale that we believe the error rate is not large enough to affect our ultimate conclusions.

RESULTS

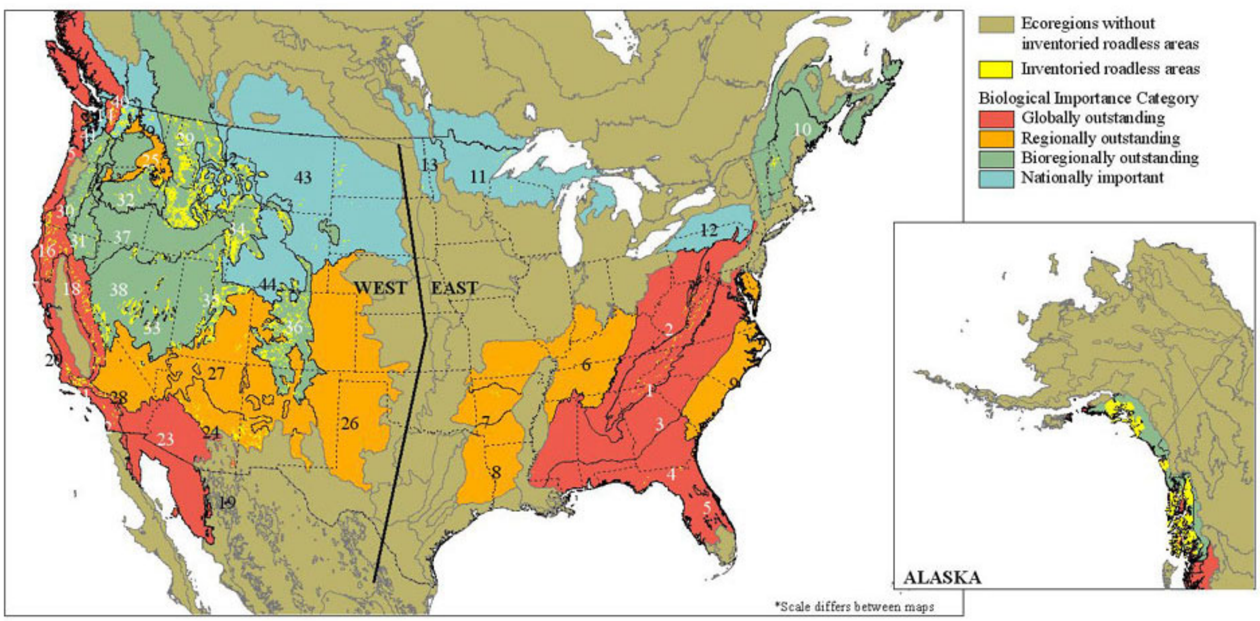
Ecoregions

Across the United States, we found that more than 20% of inventoried roadless areas (IRAs) were located within ecoregions that have been classified as globally outstanding (Table 3, Fig. 4). In the eastern region, approximately 70% of the IRAs are found in globally or regionally outstanding ecoregions (Table 3, Fig. 4). More than 50% of these forests occur in two Appalachian ecoregions, the Appalachian-Blue Ridge forests and the Appalachian mixed mesophytic forests. Both are considered globally outstanding for their diverse endemic species, which range across many taxa (Stephenson et al. 1993, Ricketts et al. 1999). The vast majority of the IRAs in eastern forests are less than 10.1 km^2 in size, and few are adjacent to existing wilderness areas (DeVelice and Martin 2001).

Table 3. Distribution of inventoried roadless areas (IRAs) by category of ecoregion biodiversity as per Ricketts et al. (1999). The percentage is the percentage of IRAs that fall into that particular category.

Biodiversity category	km^2	Percentage
Globally outstanding	50,221	21.2
Regionally outstanding	12,648	5.4
Bioregionally outstanding	164,600	69.5
Nationally important	9268	3.9

Fig. 4. Overlap of USDA Forest Service inventoried roadless areas and ecoregions classified by biological importance (see Ricketts et al. 1999).



1. Appalachian-Blue Ridge Forests
2. Appalachian Mixed Mesophytic Forests
3. Southeastern Mixed Forests
4. Southeastern Conifer Forests
5. Florida Sand Pine Scrub
6. Central US Hardwood Forests
7. Ozark Mountain Forests
8. Piney Woods Forests
9. Middle Atlantic Coastal Forests
10. New England-Acadian Forests
11. Western Great Lakes Forests
12. Allegheny Highlands Forests
13. Northern Tall Grasslands
14. British Columbia Mainland Coastal Forests
15. Central Pacific Coastal Forests
16. Klamath-Siskiyou Forests
17. Northern California Coastal Forests
18. Sierra Nevada Forests
19. Madrean Sky Islands Montane Forests
20. California Interior Chaparral & Woodlands
21. California Montane Chaparral & Woodlands
22. California Coastal Sage & Chaparral
23. Sonoran Desert
24. Arizona Mountains Forests
25. Palouse Grasslands
26. Western Short Grasslands
27. Colorado Plateau Shrublands
28. Mojave Desert
29. North Central Rockies Forests
30. Central and Southern Cascades Forests
31. Eastern Cascades Forests
32. Blue Mountains Forests
33. Great Basin Montane Forests
34. South Central Rockies Forests
35. Wasatch and Uinta Montane Forests
36. Colorado Rockies Forests
37. Snake-Columbia Shrub Steppe
38. Great Basin Shrub Steppe
39. Okanogan Forests
40. Cascade Mountains Leeward Forests
41. Puget Lowland Forests
42. Montana Valley and Foothill Grasslands
43. Northwestern Mixed Grasslands
44. Wyoming Basin Shrub Steppe
45. Northern Pacific Coastal Forests
46. Pacific Coastal Mountain Tundra & Ice Fields

In the western region, IRAs are found predominantly in bioregionally outstanding ecoregions, with only 18% in globally or regionally outstanding ecoregions (Table 3, Fig. 4). Although globally and regionally outstanding IRAs are found mainly in the states of California, Oregon, Washington, and Arizona, the intermountain west contains most of the nation's bioregionally and nationally important IRAs. Western IRAs are on average larger than eastern IRAs, and the vast majority are adjacent to existing wilderness areas. If the IRAs were combined with the wilderness areas, the western forests would contain 34 of the 45 largest contiguous areas of strictly protected forests in the United States (DeVelice and Martin 2001).

Table 4. Comparison of the degree of overlap between inventoried roadless areas (IRAs) and quadranges containing threatened or endangered (TE) species or quadranges containing TE species that are also ranked as highly imperiled (G1–G2) by the IUCN. The mean number of TE or TE/G1–G2 species present in each IRA is given.

Region	Total no. of IRA units†	No. of IRA units with TE species quadrangles (% of total)	Mean no. of species‡	No. of IRA units with TE/G1–G2 species quadrangles (% of total)	Mean no. of species‡
Eastern United States	286	201 (70.3)	2.1	228 (79.7)	4
Western United States	2159	1317 (61.0)	1.6	1692 (78.3)	2.9
Alaska	150	2 (1.3)	1	88 (58.6)	1.3

†Units are defined by each named inventoried roadless area.
‡Where multiple quadrangles occurred in a single IRA unit, we used only the quadrangle with the greatest number of species.

Threatened, endangered, and imperiled species

Of the 2595 IRA units, approximately 58% of them overlap with TE species quadrangles (Table 4). When separated into geographic regions, the IRAs in the eastern and western United States demonstrate overlaps of 70.3 and 61.0%, respectively. Of the IRAs that contain TE species, the mean number of TE species found in IRAs is highest in the east (2.1 species) and lowest in Alaska (1.0 species).

When G1–G2 species are included in the analysis, both the number of IRAs that contain TE/G1–G2 species and the mean number of species of concern found in each IRA increase (Table 4). In sum, approximately 77% of the IRAs overlap with quadrangles that contain species at risk. The Alaska region contains the largest increase in IRAs when G1–G2 species are included, increasing to 58.6 from 1.3%. The west increases to 78.3%, and the east increases to 79.7%. However, the east shows the largest increase in mean number of TE/G1–G2 species found in IRAs, increasing from 2.1 to 4.0 species (Table 4).

The IRAs could also contribute a significant amount of land area to existing conservation reserves for both TE and TE/G1–G2 species in all geographic regions (Table 5). The largest increase in area and the greatest percent increase in conservation reserves are found in the western United States, with the exception of the 100% increase from the single quadrangle in Alaska. IRAs would contribute to a 96% increase in available habitat in conservation reserves for TE species, whereas the inclusion of G1–G2 species expands that increase to 210%. Although the eastern region would see similar but more modest gains, habitat in conservation reserves in the Alaska region would increase 113% for TE/G1–G2 species (Table 5). Overall, IRAs would increase the conservation reserve network containing TE, G1, or G2 species by 156%.

Table 5. The concordance of occurrences of threatened or endangered (TE) species or of TE species that are also classified as highly imperiled (G1–G2) by the IUCN with the existing conservation reserve network (IUCN I–III) and inventoried roadless areas (IRAs).

Region	No. of TE species quadrangles in IUCN I–III conservation reserves	No. of TE species quadrangles in IRAs	Percent increase	No. of TE/G1–G2 species quadrangles in IRAs	No. of TE/G1–G2 species quadrangles in IUCN I–III conservation reserves	Percent increase
Eastern United States	995	217	22	1027	431	42
Western United States	1752	1679	96	2200	4627	210
Alaska	0	1	100	38	43	113

Grizzly bear case study

As seen in Fig. 5, the inclusion of IRAs in the existing system of conservation reserves in Washington, Idaho, Montana, and Wyoming shows a strong concordance with the grizzly bear recovery zones of the U.S. Fish and Wildlife Service, as well as bear habitat range (Martin et al. 2000, USDA Forest Service 2000). In total, the six grizzly bear recovery zones include approximately 15,300 km^2 of IRAs. Approximately 24,750 km^2 of almost contiguous IRAs surround the Salmon-Selway (Bitterroot) Recovery Zone (SSRZ), which has already been designated a wilderness area and assigned to IUCN category I.

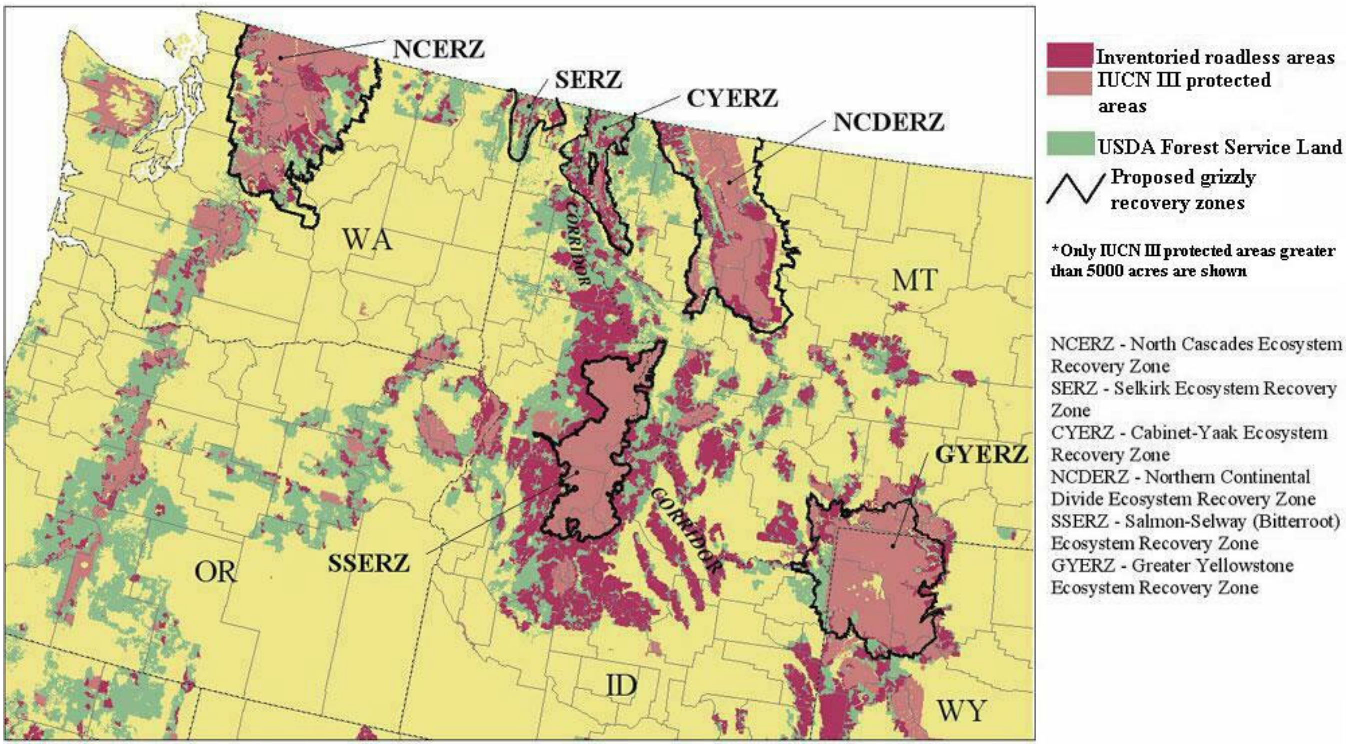
DISCUSSION

Our analyses found that one-quarter of the inventoried roadless ares (IRAs) are found in globally or regionally outstanding ecoregions, and that they have the potential to provide important habitat for numerous species, including threatened, endangered, and imperiled species. This conclusion is further illustrated by an investigation of the potential benefit of IRAs to grizzly bear conservation.

Based on these findings, the assignment of IRAs to IUCN category III or higher could increase the area of conservation reserves in the United States from 4.8 to 8.5%. This broad national conclusion has different implications depending on geographic region. For example, whereas fewer than 3% of the IRAs are found in the eastern United States, the vast majority of eastern IRAs are found in the ecoregions with the greatest amount of biodiversity and the least amount of existing protection. In addition, despite the fact that western forests currently have some of the highest existing protection levels in the United States, Scott et al. (2001) found that many existing reserves in the United States are concentrated in areas of high elevation and low soil productivity. Therefore, despite the current levels of perceived protection, the nation's biological diversity may be under-represented in the current system, particularly in the mountainous west (Scott et al. 2001). DeVelice and Martin (2001) have shown that approximately 40% (about 91,300 km^2) of the IRAs are at an elevation below 1500 m and that 35% of the total IRAs are adjacent to designated wilderness areas. The combination of increased protection of forest habitat and the potential increase in size of conservation reserves would have a positive effect on the conservation of large mammals in the western United States.

The purpose of the *Endangered Species Act* is to " ... provide a means whereby the ecosystems upon which endangered and threatened species depend may be conserved ... " (U.S. Fish and Wildlife Service 1973). The act further directs that " ... all Federal departments shall seek to conserve endangered species and threatened species." In this regard, many IRAs function as biological refugia for terrestrial and aquatic species, including numerous threatened, endangered, and imperiled species. The maintenance of natural values in IRAs could contribute to their long-term viability (Brown and Archuleta 2000). IRAs contain more than 220 TE species, i.e., approximately 25% of listed or proposed animal species and 13% of listed plant species (USDA Forest Service 2000).

Fig. 5. Overlap of USDA Forest Service inventoried roadless areas and grizzly bear recovery zones.



Among TE species, 88% are imperiled by habitat destruction and degradation (Wilcove et al. 1998). Dobson et al. (1997) found that, if the habitats of TE species were more extensively protected, a large number of them would be efficiently conserved. Our analysis showed that the vast majority of IRAs hosted TE or G1–G2 imperiled species and that, by adding the IRAs to the existing conservation reserve system, the conservation of species at risk and their habitat could be better realized. Although we recognize that not all threatened, endangered, or imperiled species require lands free of active land management to survive, limiting the human footprint by placing IRAs off limits to road construction and maintenance, resource extraction, and other development activities could provide a counterpoint to the multiple-use activities taking place elsewhere within the National Forest System.

Furthermore, although there may be duplicate species populations within IRAs or existing conservation reserves, the high level of endangerment of these species should predicate that we conserve as many populations as possible. Therefore, the potential issues of complementarity or duplication of species across IRAs should not diminish the contribution that IRAs could make to conserving species at risk. Our analyses have shown that, despite the small size and extent of IRAs in the eastern United States, they contain a greater number of endangered or imperiled species across more IRAs than do the west and Alaska. However, many of the western IRAs are missing data or have not been surveyed. This error of omission serves only to emphasize that our findings are a conservative estimate of potential species endangerment particularly in IRAs in Alaska and the western United States.

Top carnivore species, such as the grizzly bear, often have the largest species-level area requirements in an ecosystem and maintain ecological structures and resilience by top-down trophic interactions. They need large, contiguous habitat blocks to persist, and there must be landscape connectivity among core areas to ensure sufficient habitat for viable populations (Soulé and Noss 1998, Carroll et al. 2001). As a result of these requirements, large reserves are necessary to maintain populations of these wide-ranging species. Woodroffe and Ginsberg (1998) recently estimated that habitats of 20,000 km^2 are needed to provide a 90% chance for the long-term survival of the grizzly bear in the wild. Indeed, only those wilderness areas that were 20,000 km^2 or larger in 1920 still support grizzly bears today (Mattson and Merrill 2002). The 40,000 km^2 of IRAs in and near designated grizzly recovery zones in the northern Rockies will help improve the long-term habitat viability for grizzly bears in the region (Martin et al. 2000, USDA Forest Service 2000).

Carroll et al. (2001) proposed the need for a comprehensive conservation strategy for carnivores in the Rocky Mountains that considers the requirements of several species, including grizzly bear, wolverine, fisher, and lynx. The regions where these four species overlap show a strong concordance with grizzly bear recovery zones. IRAs may benefit all of these species by providing expanded and buffered habitat and, in turn, secure the ecological integrity of those ecosystems (Terborgh and Soulé 1999, Conner et al. 2000, Martin et al. 2000). If grizzly bear populations remain limited by the size and configuration of current conservation reserves, their long-term survival in the conterminous United States cannot be assured (Mattson and Merrill 2002).

Bruner et al. (2001) found a clear relationship between the existence of a viable and well-connected system of conservation reserves and biodiversity conservation. Because of the stable long-term ownership tenure associated with USDA Forest Service lands, as opposed to privately held forests, many of these forested areas contain a wealth of biological diversity. Historically, land within the Forest Service has been managed under a multiple-use strategy, with timber extraction being a main component of many of these plans. However, multiple-use management may not ensure the protection of the full range of biodiversity, because anthropogenic habitat degradation and destruction are the primary causes of biodiversity loss (Ehrlich 1988, Myers 1988, Wilcove et al. 1996, Haila 1999, Wood 2000).

Setting aside IRAs for stricter protection from extractive or economically driven activities may indeed meet many biological objectives, e.g., integration of fish and wildlife values and watershed and forest health, consistent with the agency's multipurpose agenda. In addition, IRAs may also contribute invaluable benchmarks to gauge ecological changes on managed U.S. Forest Service lands. A representative system of natural habitats, set aside from active management, would allow natural ecological processes, including a full suite of existing native species, to survive free of human activities. Without strict conservation areas that represent all forest habitat types, it will be difficult to make objective assessments on the sustainability of forest management (Noss and Cooperrider 1994, Norton 1999, Noss et al. 1999). Based upon our analyses, we conclude that IRAs support many at-risk species and thereby greatly contribute to the conservation of biodiversity throughout the United States. For some species with only a few remaining populations, the strict and permanent protection of IRAs may represent the final, critical refuge.

Responses to this article can be read online at:
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Fire history and fire–climate relationships along a fire regime gradient in the Santa Fe Municipal Watershed, NM, USA

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ABSTRACT

The Santa Fe municipal watershed provides up to 40% of the city's water and is at high risk of a stand-replacing fire that could threaten the water resource and cause severe ecological damage. Restoration and crown fire hazard reduction in the ponderosa pine (PP) forest is in progress, but the historic role of crown fire in the mixed-conifer/aspen (MC) and spruce-dominated forests is unknown but necessary to guide management here and in similar forests throughout the southwestern United States. The objective of our study was to use dendroecological techniques to reconstruct fire history and fire–climate relationships along an elevation, forest type, and fire regime gradient in the Santa Fe River watershed and provide historical ecological data to guide management. We combined systematic (gridded) sampling of forest age structure with targeted sampling of fire scars, tree-ring growth changes/injuries, and death dates to reconstruct fire occurrence and severity in the 7016 ha study area (elevation 2330–3650 m). Fire scars from 141 trees (at 41 plots) and age structure of 438 trees (from 26 transects) were used to reconstruct 110 unique fire years (1296–2008). The majority (79.0%) of fires burned during the late spring/early summer. Widespread fires that scarred more than 25% of the recording trees were more frequent in PP (mean fire interval $(MFI)_{25\%}$ = 20.8 years) compared to the MC forest (31.6 years). Only 24% of the fires in PP were recorded in the MC forest, but these accounted for a large percent of all MC fires (69%). Fire occurrence was associated with anomalously wet (and usually El Niño) years preceding anomalously dry (and usually La Niña) years both in PP and in the MC forest. Fire in the MC occurred during more severe drought (mean summer Palmer Drought Severity Index; PDSI = −2.59), compared to the adjacent PP forest (PDSI = −1.03). The last fire in the spruce forest (1685) was largely stand replacing (1200 ha, 93% of sampled area), recorded as fire scars at 68% of plots throughout the MC and PP forests, and burned during a severe, regional drought (PDSI = −6.92). The drought–fire relationship reconstructed in all forest types suggests that if droughts become more frequent and severe, as predicted, the probability of large, severe fire occurrence will increase.

1. Introduction

Large areas of forest throughout the southwestern United States (Arizona, New Mexico and adjacent areas) are unnaturally dense due a century of fire exclusion, and are consequently at high risk of historically unprecedented large crown fires (Covington and Moore, 1994; Allen et al., 2002). Given limited resources for treatment, a triage approach must be adopted to identify areas with high resource value or that are located strategically within the larger landscape. Historical ecological data describing the range of variability of disturbance regimes and their climatic controls are vital to guide forest restoration (Swetnam et al., 1999), particularly when facing the additional challenge of a changing climate (Millar et al., 2007).

The upper Santa Fe River watershed, New Mexico is arguably the most at risk, high-profile municipal watershed in the southwestern U.S. Santa Fe is the oldest state capital, founded on the Santa Fe River in the early 17th century (Debuys, 1985). Sitting at 2137 m elevation on the alluvial plane of a steep, forested, montane watershed, Santa Fe is inextricably linked to the ecosystem services (e.g., drinking water) and natural hazards (e.g., fire and floods) associated with the wildland urban interface. Surface water that originates high in the spruce-fir forests of the Pecos Wilderness Area is regulated through a system of reservoirs that provides up to 40% of the city's water supply (Grant, 2002). The population in Santa Fe County has tripled in recent decades (1970–2007; USCB, 2009), overtaxing the already limited water

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supply. Like much of the West, there has not been a widespread fire in the ponderosa pine (PP) and mixed-conifer (MC) forests of the watershed for 130 years, increasing the area at high risk of crown fire beyond the spruce-fir forests, where they naturally occur.

Recent, large, crown fires in near by watersheds have produced runoff and erosion events two orders of magnitude greater than pre-fire events (Veenhuis, 2002). This type of event in the Santa Fe watershed could destroy the water supply infrastructure and flood the historic heart of the city. The threat of catastrophic fire sparked years of contentious public debates, which ultimately led to U.S. Congressional earmarks of seven million dollars to fund planning and implementation of crown fire hazard reduction and forest restoration in the lower elevation PP forests (USDA, 2001). However, the ecological role of fire and the consequences of fire exclusion in the upper elevation mesic MC and spruce-fir forest types remain largely unknown, and it is these forests that cover the majority of the area that supplies the main reservoir.

1.1. Gradients: elevation, forest types and fire regimes

Gradients (e.g., elevation and vegetation) are common in terrestrial ecosystems and are a valuable way to study how ecological processes vary across a range of conditions (Whittaker and Niering, 1965; Whittaker, 1967). In the southwestern U.S. fire is a keystone ecological process that has affected vegetation across ecosystem gradients for hundreds to thousands of years (Swetnam and Baisan, 1996; Allen, 2002; Anderson et al., 2008). The size, frequency and severity of fire over time define the fire regime (Agee, 1993). Fire regimes are commonly classified by the extremes of the fire severity gradient (low severity or high severity). Recently, the term *mixed-severity fire regime* has been described as including a range of fire severities across a spatially complex mix of forest patches, including unburned, low, moderate, and high severity fire (Agee, 2005).

At landscape scales (1000–100,000 ha; watersheds to mountain ranges), fire can move across gradients of elevation, forest types and likely, between fire regimes. The PP forest type in the southwestern U.S. is a classic low severity, high frequency fire regime (Swetnam and Baisan, 1996). Subalpine spruce-fir forests in the southern Rocky Mountains exemplify the other extreme: a high severity, low frequency fire regime (Romme and Knight, 1981; Sibold et al., 2006). The steep topography of the southwestern U.S. juxtaposes these two forest types (representing the extremes of the fire severity gradient) in close proximity (<10 km separation) along a continuous elevation gradient with continuous fuels. MC forests are intermediately located between PP and spruce-fir (Dick-Peddie, 1993). Lower elevation, xeric, MC forests historically burned with low severity, but less frequently than PP (Dieterich, 1983; Brown et al., 2001). Some upper elevation, mesic MC forests have evidence of high severity fire (Fule et al., 2003; Margolis et al., 2007; Margolis, 2007). Historically, drought synchronized fire occurrence within and between low and high severity fire regimes regionally (Swetnam and Baisan, 1996; Margolis et al., 2007), but there is limited research examining connectivity between low and high severity fire regimes along a continuous forest gradient in a single, continuous landscape (Fule et al., 2003).

The implications of low and high severity fire regime connectivity are important given the well-documented changes associated with fire exclusion in ecosystems of the southwestern U.S. Over a century of fire exclusion in PP forests of the region has dramatically increased forest density and the risk of large crown fires (Covington et al., 1997; Allen et al., 2002). While there is historical evidence of high severity fire patches in some MC forests (Fule et al., 2003), increased forest density in other MC forests due to fire exclusion has increased the size of forest patches at risk of crown fire (Fule et al., 2003; Cocke et al., 2005; Heinlein et al., 2005; Margolis et al., 2007).

There is comparatively less information about the effects of fire exclusion on forest density, composition, and crown fire risk in the upper elevation spruce-fir forests of the region (Fule et al., 2003; Cocke et al., 2005). It is generally thought that a century of fire exclusion has not had dramatic impacts in these naturally dense forest types (Sibold et al., 2006), because high elevation, high severity forest fire regimes burn at long (centennial-scale) return intervals (Turner and Romme, 1994). Evidence of decreased fire frequency during the fire suppression period, compared to previous centuries has been observed in some subalpine forests of the Southern Rockies (Kipfmueller and Baker, 2000), but not others (Sibold et al., 2006). If forest ecosystems along steep elevation gradients are connected by fire spread across vegetation and fire regime gradients, then a century of fire exclusion in the lower elevation pine-dominated and MC forests is likely to have affected the high elevation, high severity forest fire regimes as well.

The semi-arid climate of the southwestern U.S. is highly variable, with frequent (2–7 years) wet/dry oscillations that are partially driven by multiple ocean-atmosphere oscillations, particularly the El Niño Southern Oscillation (ENSO; Diaz and Markgraf, 2000). Fire–climate analyses indicate that moisture variability largely explains patterns of fire occurrence in tree-ring reconstructed and contemporary records in low- and mid-elevation forests of the southwestern U.S. (Swetnam and Betancourt, 1990; Crimmins and Comrie, 2004). Warmer temperatures in recent decades have increased the length of the fire season, resulting in more large fires throughout the western U.S. (Westerling et al., 2006). The established link between climate variability and fire, coupled with predicted warmer global temperatures (IPCC, 2007) and drier conditions in the southwestern U.S. (Seager et al., 2007) has led to predictions of more large fires in the future (Westerling et al., 2006). Better understanding of the link between climate variability and fire occurrence along the elevation gradient of forest types and fire regimes is necessary to proactively manage our forests with a science-based approach, in the face of climate change.

The goal of the research is to provide essential historical ecological data across a gradient of forest types and fire regimes to guide management in the upper Santa Fe watershed and similar upper montane forest types in the region. Our first objective was to reconstruct fire history (frequency, severity, and size) along an elevation, vegetation and fire regime gradient in the upper Santa Fe Watershed. Our second objective was to reconstruct and compare historic fire–climate relationships between forest types. Our third objective was to test for evidence of direct connectivity of fire regimes along the fire severity gradient from low, to mixed, to high severity.

1.2. Study area

The study area encompasses the upper Santa Fe River watershed (7016 ha), which includes the headwaters located within the U.S. Forest Service Pecos Wilderness Area (Fig. 1). The watershed is located on the west slope of the Sangre de Cristo Mountains, northeast of the city of Santa Fe, NM, near the southern terminus of the Southern Rocky Mountains. The upper watershed has been closed to the public since 1932 to protect the water supply for the city of Santa Fe (USDA, 2001). Elevation ranged from 2237 m at the lowest point in the stream channel to 3847 m on the peaks that rise above tree-line and define the headwaters of the basin. Tree-ring samples were collected from 2328 m to 3650 m.

The climate is semi-arid and continental. Precipitation peaks during summer monsoon convective storms (July–August), and winter snowpack is common except during extreme drought years

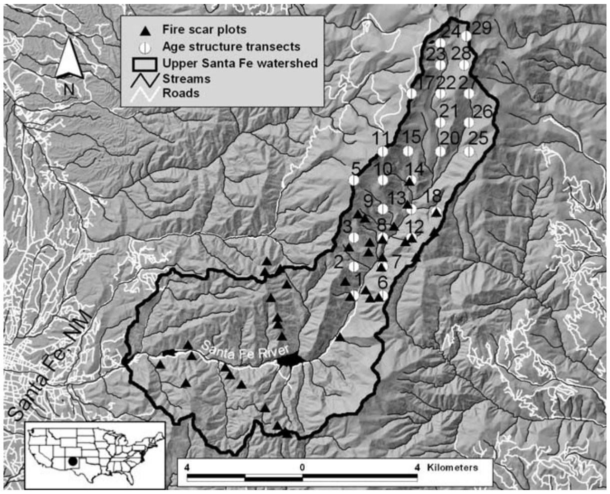

Fig. 1. Location of gridded age structure transects (numbered) and fire scar plots used to reconstruct fire history in the upper Santa Fe watershed, NM. The upper watershed, containing only age structure transects, is spruce-dominated forest. The lower watershed, containing only fire scar plots, is PP. The middle-elevation forest area where both age structure and fire scars overlap is MC.

(e.g., 2002). Temperature (1972–2005) at Santa Fe, NM (2060 m) ranged from an annual average minimum of 2.3 °C to an annual average maximum of 18.2 °C. Total annual average precipitation was 34.8 cm and total annual average snowfall was 44.2 cm (Western Regional Climate Center, www.wrcc.dri.edu). Fire occurrence records were available for 222 fires (1970–2003) from the ranger district containing the watershed and the adjacent district to the east (Española and Pecos/Las Vegas Ranger Districts). The majority (93%) of fires occurred between May and August, peaking in July, but monthly area burned peaks in May and June during the dry foresummer. Eighty percent of all fires were started by lightning (USFS, unpublished data).

Along the elevation gradient, forest types transitioned from PP dominated forests in the lower part of the study area, to MC in the middle elevations, to spruce-fir in the upper elevations. The spruce-fir type was composed of Engelmann spruce (*Picea engelmannii* Parry) and corkbark fir (*Abies lasiocarpa* [Hook] Nutt. var. *arizonica* [Merriam] Lemmon), but Engelmann spruce was dominant in all locations sampled and often present in pure stands. This general upper elevation forest type is hereafter referred to as "spruce-dominated." The MC forest was relatively diverse and species composition varied largely by aspect. The following species were present in various combinations in this forest type, listed in order of abundance of the dominant tree: Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco.), ponderosa pine (*Pinus ponderosa* Lawson), southwestern white pine (*Pinus strobiformis* Engelm), quaking aspen (*Populus tremuloides* Michx.), white fir (*Abies concolor* [Gord. and Glend.] Lindl. Ex Hildebr.), and Engelmann spruce. There were no conspicuous, large (>50 ha) stands dominated by quaking aspen that might indicate recent stand-replacing fire patches. The lower section of the study area was dominated by ponderosa pine, with associated species including Colorado pinyon pine (*Pinus edulis* Engelm.), Rocky Mountain juniper (*Juniperus scopulorum* Sarg.), Gambel oak (*Quercus gambelii* Nutt.), Douglas-fir, white fir, and southwestern white pine.

2. Methods

2.1. Tree-ring fire history methods

A combination of tree-ring methods was necessary to reconstruct fire history along the elevation, vegetation and fire regime gradient. Fire scar-based methods (Dieterich and Swetnam, 1984; Swetnam and Baisan, 1996) were used to reconstruct surface fire frequency, seasonality, and extent for the PP and dry MC portions of the watershed where fire scars were present. However, in the upper elevation spruce-dominated and mesic MC forests, fire-scarred trees were rare or non-existent because: (1) high severity, high intensity, stand-replacing crown fires destroy (kill and burn) direct tree-ring evidence of past fires, (2) the thin bark of spruce and fir species is more susceptible to being fatally girdled, even by low-intensity fire, thus leaving no evidence of the most recent fire (e.g., fire scars), and/or (3) long fire return intervals may allow the rare fire-scarred trees to heal over so that open fire scar wounds are not visible.

In forest types where fire scars are not abundant, age structure-based fire history methods are commonly applied (Heinselman, 1973; Agee, 1993; Johnson and Gutsell, 1994). These methods largely rely upon the establishment dates of tree cohorts that regenerate following stand-replacing fire events to date the fire and determine aerial extent of the stand-replacing patches. Thus, in the spruce-dominated forest type we used age structure sampling methods to reconstruct the age of the dominant, presumably oldest trees, thereby estimating the time since the last fire (Kipfmueller and Baker, 1998). Satellite imagery, aerial photography, and field observations were used to identify any potential post-fire forest patches.

Age structure data alone may not be sufficient to determine if the forest patch was a post-stand-replacing fire cohort and ultimately date the fire. Unlike lodgepole pine (*Pinus contorta*) or quaking aspen, spruce and fir trees may take years to decades to regenerate following stand-replacing fire (Antos and Parish, 2002). This is likely due to a combination of variability in seed sources, dispersal, and post-fire weather and climate. The precision of fire dates derived strictly from forest age will not be annual because of lagged regeneration. Decadal precision of fire dates can be sufficient when calculating area-based estimates of fire frequency (e.g., natural fire rotation; Heinselman, 1973), due to the long return intervals (100 years to >400 years) of forest crown fire regimes (Turner and Romme, 1994). Annually precise stand-replacing fire dates may be reconstructed if fire scars, fire-killed trees or injured trees are present in adjacent forest stands, or on the perimeter of unburned patches (Johnson and Gutsell, 1994; Margolis et al., 2007). Annually resolved fire dates are necessary for inter-annual fire–climate analyses, which can provide specific climate information associated with the relatively rare, but important, stand-replacing fire events.

In forest types such as MC, where a combination of low severity and high severity fire occurs (i.e., mixed-severity fire), it is necessary to use a combination of fire scar and age structure-based methods to reconstruct the fire history (Agee, 2005). For example, some stands within the MC zone had no fire scars or potentially fire-killed trees, and stand age was the best evidence of past fire. Alternatively, other stands had abundant fire scars (i.e., evidence of repeated, low severity surface fire) and no clear post stand-replacing fire cohorts.

2.2. Sampling design

In the PP forests of the lower portion of the study area we used a targeted approach to locate and collect fire scars. Targeted fire scar sampling in Southwestern ponderosa pine provides similar estimates of fire frequency compared to systematic sampling, particularly for widespread fires, with the added benefit of providing a longer record (Van Horne and Fule, 2006). Samples were primarily collected at 50 m-radius plots along two transects; a north-facing transect, and a south-facing transect. These transects were located in the middle of the PP zone and followed a series of ridges that extended from the river up to the respective watershed boundaries. From the ridgetop location we searched both slopes that descended from the ridge and adjacent slopes that could be seen from the ridge. Additional plots were located within the PP zone to provide broader spatial coverage of the PP forest fire history. One group of plots was located west (downstream) of the transects in the area surrounding the lower reservoir. An additional plot was located east (upstream) of the transects, above the second reservoir. The resulting spatial patterning of the plots was determined by a combination of our search effort, topography and the presence of fire-scarred material.

In the high elevation spruce-dominated forests, a systematic, gridded age structure sampling approach provided the best evidence of fire history (i.e., "time since last fire"; Johnson and Gutsell, 1994). We generated a 1 km grid beginning with a random location in the study area (Fig. 1). The grid was oriented along cardinal directions to facilitate navigation in the field. Two grid points (24 and 28) initially fell within unforested vegetation types and were relocated 50 m inside the nearest forested area.

In the middle-elevation, MC forest evidence of fire was present as both fire scars and post-fire tree regeneration cohorts. We extended the 1 km-spaced age structure grid into the MC zone, and because fire scars were only present at 5 of 12 MC age structure transects we used a targeted approach to locate and sample fire scars in this forest type. In the MC forest, fire-scarred trees were most abundant on the relatively flat ridges, apparently because of lower fire severity that allowed trees to survive fires that were otherwise stand-replacing on the adjacent steep slopes. We searched and sampled ridges with the goal of obtaining a relatively even spatial distribution of fire scar plots and to maximize the length of the fire history record. The final spatial distribution of the fire scar sample plots was ultimately determined by the location of fire-scarred trees, in part determined by topography and chance, and therefore is not evenly distributed in space.

In the topographically complex mountains of the semi-arid southwestern U.S., elevation and aspect can be important variables mediating vegetation type (e.g., Whittaker and Niering (1965)) and fire regimes (e.g., Iniguez et al. (2008)). To ensure that the distribution of aspect class (N, S, E, W) at our gridded, age structure transects was proportional to the relative abundance of aspect classes in the study area we stratified the sampling grid by aspect class. The percent of sample points in the four primary aspect classes was distributed similar to the percent of land area in each aspect (Table 1), with a slight over-sampling of east-facing slopes and under-sampling of the south-facing slopes.

2.3. Field sampling

Where multiple fire-scarred trees were present we used a plot-based field sampling approach. A plot was sampled where two or more fire-scarred trees were located less than 15 m apart. The plot center was located between the samples. Samples from multiple fire-scarred trees were collected within a 50 m search radius that defined the plot. Collecting multiple trees within a plot increased the probability of recording all fires that actually occurred in that area. This is necessary because trees are imperfect recorders of fire and individual trees may not record all fires (as fire scars) that burned around the tree (Dieterich and Swetnam, 1984). Wedges and cross-sections were collected from fire-scarred logs, stumps and rarely from live trees with a cross-cut saw in the MC forest (within the Pecos Wilderness Area) and with a chainsaw in the PP forests using standard procedures (e.g., Arno and Sneck (1977)).

To determine stand age at the gridded age structure transects in the spruce-dominated and MC zones we sampled the 20 largest (diameter at breast height (dbh)) trees along a 100 m by 20 m belt transect. The transect was centered on the grid point and the long axis was oriented parallel to the contour of the slope (i.e., sideslope). To determine tree age, increment cores were collected as close to the base of the tree as possible (<0.3 m). We angled the borer down to intersect the estimated location of the root crown in an attempt to sample all the years of tree growth. We re-sampled trees until we extracted a core containing rings estimated to be within 10 years of the pith.

2.4. Lab methods

All tree-ring samples were sanded with progressively finer sandpaper until the ring structure was visible and then crossdated using standard dendrochronological procedures (Stokes and Smiley, 1968). For fire scar samples, we determined the calendar year of the scar and the season of fire occurrence by analyzing the relative position of each scar within the annual growth ring: dormant season, early earlywood, middle earlywood, late early-wood, latewood, or unknown (Baisan and Swetnam, 1990). Predominant occurrence of spring or early summer fires in northern New Mexico and the southwestern U.S. is widely supported by fire seasonality data from observed 20th century fires in the region (Barrows, 1978), locally, and from hundreds of tree-ring reconstructed fires (Swetnam and Baisan, 1996). Based on our observations and conventional season of montane fire occurrence in the region, all fire years with fire scars recorded only in the dormant season were assigned to the spring/summer of the next year (ring).

For age structure samples, we estimated the date of the first year of growth (pith) for increment cores that did not contain the pith ring, using a concentric circle pith estimator (Applequist, 1958). Cores that were estimated to be greater than 30 years from the pith ring or that had no curvature in the inner rings were not included in the age structure data. Because cores were collected at a downward angle to intercept the root crown, the error associated with the age to core height was assumed to be within the resolution of the age class bins (10 years) and was not estimated.

A qualitative description of the initial tree-ring growth of cored trees (open, average, or suppressed) was recorded to provide information regarding the growth environment when trees established (Romme and Knight, 1981). Spruce and fir species are shade tolerant and are able to survive in low light conditions under canopies, but the growth rates in these conditions can be very slow (i.e., "suppressed"). Growing conditions for trees germinating in an open forest, such as following a stand-replacing fire, would be more favorable and should be indicated as relatively wide initial ring widths (i.e., open). This information was combined

Table 1
Aspect class of land area and age structure transect grid in the MC and spruce-dominated forests.

Aspect class	Area (ha)	Area (% of total)	Age structure transect (#)	Age structure transect (% of total)
Flat (0% slope)	0.22	0.01	0	0
N (315–45°)	221.11	8.29	3	11.54
E (45–135°)	773.59	28.99	9	34.62
S (135–225°)	852.73	31.95	6	23.08
W (225–315°)	821.03	30.77	8	30.77

with tree ages and fire scar dates to determine if trees were likely part of post stand-replacing fire cohorts.

2.5. Data analysis

The fire scar data were entered into a database and analyzed using FHX2 software (Grissino-Mayer, 2001). Because fire scar return intervals are rarely normally distributed and more commonly fit a Weibull distribution (Grissino-Mayer, 1999), we tested for the fit of the Weibull model (Kolmogorov–Smirnov (K-S) test) and estimated the Weibull Median Probability Interval (WMPI). Central tendency parameters (mean, median and WMPI) of fire frequency were calculated for five ''filtered'' subsets of the composite fire history data for (1) the PP forest and (2) the MC forest. The following filtered subsets of reconstructed fires were used for the analysis: (1) all fires, (2) fires recorded by a minimum of 2 trees, (3) a minimum of 2 trees and >10% of recording trees, 10% scarred, (4) a minimum of 2 trees and >20% of recording trees, 20% scarred and (5) fires recorded by a minimum of 2 trees and >25% of recording trees, 25% scarred. ''Recording trees'' refers to previously fire-scarred samples that have intact wood (i.e., not burned away or missing pieces) and an open wound (not covered by bark) during the time period in question. Many montane conifers have thick bark that protects trees from damage to the cambium by fire. These full-bark trees may not record fires as fire scars, while the same fire is recorded on adjacent trees with pre-existing open ''cat face'' fire scar wounds.

Filtering the fire scar data by the percent of recording trees scarred is used to infer relatively large, spreading fires, as compared to less widespread fires that only scar a relatively small number (percent) of trees (see discussion in Swetnam and Baisan, 2003). Widespread fires are thought to be more ecologically important because of the extent of the effects. Too few fire-scarred trees were present on the landscape and/or collected to confidently allow plot-based fire interval analysis (e.g., Iniguez et al., 2008). In addition, high severity fire in parts of the MC forest killed and burned evidence of prior fires at individual plots, so fire dates from all plots were combined to make a site composite for each forest type (Dieterich, 1980). We also chose not to analyze fire intervals for individual trees (point intervals), because our attempt to extend the record back in time by targeting remnant wood resulted in many samples having an incomplete record due to being burned and/or eroded. This was particularly the case in the MC zone of the wilderness area, where a majority of samples were remnant wood. Given these limitations of a relatively long record, we still are confident that the percent of trees recording fire is a good indicator of widespread vs. localized fires and that the widespread fires that we focus on are the most robust to variability in sampling (Van Horne and Fule, 2006). Because fire intervals vary over time with changes in fuels and climate (e.g., Swetnam (1993)), central tendency statistics (e.g., MFI) oversimplify historic fire regimes. We report additional statistics (e.g., minimum and maximum fire intervals) and interpret these data in terms of fire management to provide a better understanding of the historic range of variability of the fire regime.

To test for differences in historical fire frequency between the PP and MC forests we used the Student's *t*-test to compare MFI, the Folded-*f* test to analyze differences in variance, and the K-S test to analyze differences in distributions (FHX2, Grissino-Mayer, 2001). Because these tests assume that the data are normally distributed, the data are transformed to the standard normal distribution (i.e., mean of zero and a standard deviation of one) before the comparisons (Grissino-Mayer, 2001). To quantify synchrony of burning (i.e., fire spread) between the PP and the MC forests we counted the number of coincident fire years between the two forest types, and calculated the percent of all fire years in each forest type that were synchronous between forest types. As a more robust test of synchrony we used Chi-squared analysis to test for independence between MC fire years and PP fire years (1550–1880) for all filtered subsets of fire years.

We used superposed epoch analysis (SEA; Baisan and Swetnam, 1990) to test for inter-annual relationships between variability in four tree-ring reconstructed measures of climate and fire occurrence in (a) the PP forest and (b) the MC forest. The tree-ring reconstructed climate variables included (1) Palmer Drought Severity Index (PDSI), (2) annual precipitation from El Malpais, NM, (3) an index of El Niño/Southern Oscillation (ENSO), and (4) an index of the Pacific Decadal Oscillation (PDO).

PDSI is a commonly used measure of available moisture (Palmer, 1965). Summer (June–August) PDSI is a good indicator of moisture conditions prior to and during the southwestern U.S. fire season and is highly correlated with variability in historical fire occurrence (Swetnam and Baisan, 2003) and 20th century fire occurrence (Crimmins and Comrie, 2004). A 2.5° gridded tree-ring reconstruction of summer PDSI exists for much of North America and in the southwestern U.S. it extends hundreds of years prior to the 20th century instrumental climate data (Cook et al., 2004). PDSI gridpoint 133 is nearest to our study site and is used in the SEA analysis. A tree-ring based precipitation reconstruction from El Malpais National Monument, in west-central NM (Grissino-Mayer, 1996), was also used as a sub-regional climate variable.

Indices of Pacific Ocean-atmosphere oscillations (e.g., ENSO and PDO) that have been shown to affect climate variability in the southwestern U.S. (Diaz and Markgraf, 2000; Brown and Comrie, 2002) were also used as variables in the SEA analysis. As a proxy index for ENSO we used the tree-ring reconstructed Niño3 index (Cook, 2000) of winter (December–February) sea surface temperature (SST) from the eastern equatorial Pacific Ocean (5°N–5°S, 90°–150°W). Positive (negative) Niño3 index values represent warm (cool) SST's - El Niño (La Niña).

We used the (D'Arrigo et al., 2001) annual PDO index reconstruction derived from temperature sensitive tree-ring sites from coastal Alaska (5) and Oregon (1), and two tree-ring reconstructed PDSI grid points in northern Mexico. Positive (negative) index values of PDO correspond with warm (cold) phases of the primary mode of variability in Pacific Ocean SST's polewards of 20°N (Mantua et al., 1997).

To test whether drier conditions were associated with fire in the MC forest than in PP, we compared mean PDSI during widespread (25% scarred) and ''all fire'' years with a *t*-test. To test whether widespread fires occurred on drier years than ''all fire'' years we compared mean PDSI between fire years for each vegetation type with a *t*-test.

3. Results

3.1. Fire scars—PP

In the PP zone (1600 ha search area) we crossdated a total of 442 fire scars from 76 trees at 20 locations, for a total of 99 unique fire years (Tables 2 and 3). The PP fire scar record covers 709 years (1296–2004), with fire scars recorded from 1331 to 1966 (Fig. 2, Table 2). The period 1550–1880 was chosen for fire interval analysis as the best compromise between record length and sample depth.

The season of fire occurrence was determined for 331 (75%) of the fire scars (Table 4). The remaining fire scars were in poor condition or were in rings too narrow to accurately determine the season. When fire scar season could be determined, the most frequent occurrence (69%) was in the dormant (D) season (i.e., between ring boundaries). All but 3% of the remaining fire scars were recorded in the earlywood (E) portion of the ring and the

Table 2
Fire scar record statistics.

Forest type	Search area (ha)	Plots (#)	Fire-scarred trees (#)	Fire scars (#)	Unique fire years (#)	Full record (years)	Fire scar record (years)	Fire interval analysis (years)
PP	1600	20	76	442	99	1296–2004	1331–1985	1550–1880
MC	1200	21	65	139	35	1337–2008	1339–1879	1495–1880
Spruce-dominated	1200	26	0	–	–	–	–	–

Table 3
Upper Santa Fe watershed fire scar dates (all fires). Fire years recorded in both forest types indicated in bold.

Century	<1500	1500s	1600s	1700s	1800s	1900s
PP (1296–2004)	1331, 1398, 1415, 1434, **1445**, 1479, **1495**	1503, **1516**, **1522**, 1532, **1542**, 1551, 1558, **1562**, 1573, 1580, **1587**, 1591	1600, 1604, 1605, 1606, **1608**, 1612, 1616, 1617, **1619**, **1622**, 1623, **1624**, 1628, 1631, 1633, 1636, 1638, 1642, 1644, 1646, 1648, **1654**, 1656, 1659, 1661, 1664, 1672, 1676, 1683, **1685**, 1687, 1696,	**1700**, 1705, **1715**, 1719, 1724, 1725, **1729**, **1737**, 1739, 1742, **1748**, 1763, **1773**, 1778, 1779, 1784, 1786, 1788, 1794, **1795**	1803, 1805, 1808, 1809, 1810, 1814, **1819**, 1823, 1825, 1826, 1831, 1835, **1842**, 1858, **1860**, 1864, 1867, 1877, **1879**, 1883, 1885, 1886, 1893	1902, 1904, 1911, 1931, 1946, 1966
MC (1337–2008)	1399, 1444, **1445**, **1495**	1500, **1516**, **1522**, **1542**, 1546, **1562**, 1579, **1587**, 1599	**1608**, 1614, **1619**, **1622**, **1624**, **1654**, **1685**	**1700**, **1715**, 1716, **1729**, 1730, **1737**, **1748**, **1773**, **1795**	**1819**, 1820, **1842**, 1857, **1860**, **1879**	

Table 4
Fire scar seasonality reconstructed from the relative position of the fire scar in the tree-ring. Period of record: PP, 1296–2006 and MC, 1337–2008.

Scar position	Number of fire scars (PP/MC)	Percent of scars with season determined (PP/MC)
Dormant	229/39	69.2/42.4
Early earlywood	40/27	12.1/29.3
Middle earlywood	35/15	10.6/16.3
Late earlywood	18/11	5.4/12.0
Latewood	9/0	2.7/0.0

majority of those were in the first third of the earlywood (early earlywood, EE). The remaining 3% of the fires were recorded in the latewood (A) portion of the tree-ring. Overall, 81% of the fires in the PP zone were burning in the beginning of the growing season (May or June; D or EE).

The fire frequency of the reconstructed PP fire regime was highly variable through time (Fig. 2), and cannot adequately be described by one metric (e.g., MFI). The fire interval data (1550–1880) were not normally distributed (K-S d-statistic = 0.438, $p < 0.001$) and were fit with the Weibull model (K-S d-statistic = 0.132, $p = 0.144$). Increasingly exclusive filters increased the fire interval central tendency statistics by eliminating the (small) fires recorded by only a few trees, such that the WMPI increased from 3.8 years (all fires) to 18.8 years (25% scarred; Table 5). MFI was similar and ranged from 4.3 years (all fires) to 20.8 years (25% scarred). Thus, somewhere within the 1600 ha PP search area there was a fire recorded by at least one tree approximately every four years, on average, and relatively widespread fires scarring more than 25% of the trees occurred approximately every 18–21 years, on average. The minimum fire interval ranged from 1 year (all fires) to 7 years (25% scarred). The maximum fire interval ranged from 16 years (all fires) to a fire-free period of 63 years (1779–1842, 25% scarred). No widespread fires (25% scarred) occurred in the 20th century.

3.2. Fire scars—MC

In the mixed–conifer/aspen forests (1200 ha search area) we crossdated a total of 139 fire scars from 65 trees at 21 locations, for a total of 35 unique fire years (Tables 2 and 3). The MC fire scar record covers 672 years (1337–2008) with fire scars recorded between 1399 and 1879 (Fig. 2, Table 2). The period from 1495 to 1880 was chosen for fire interval analysis.

The season of fire occurrence was determined for 92 (66%) of the fire scars (Table 4). Based on the observed dominance of earlywood fires and the absence of latewood fires we used the same convention as in the ponderosa zone to assign fires only recorded in the dormant season to the spring/summer of the next year ($n = 7$). The majority (72%) of the fire scars dated to the season in the MC zone were burning in the spring or early summer (May or June; D or EE).

The MC fire interval data (1495–1880) were fit with the Weibull model (K-S d-statistic = 0.103, $p = 0.897$). The WMPI ranged from 10.3 years (all fires) to 27.8 years (25% scarred, Table 5). MFI was similar and ranged from 12.4 years (all fires) to 31.6 years (25% scarred). Minimum fire intervals ranged from 1 year for all fires, to 6 years for widespread (25% scarred) fires. Maximum fire intervals ranged from 31 years for all fires, to 94 years for widespread fires. No widespread fires (25% scarred) occurred in the 20th century. Further comparisons of fire intervals among the 5 filtered datasets and between vegetation types are discussed later in the paper.

Table 5
Fire interval analysis statistics for the PP (1550–1880) and the MC forests (1495–1880) for five filtered subsets of fire years (e.g., 20% = fires recorded by >20% of the recording trees).

Filter	Intervals (#) PP/MC	Mean fire interval (years) PP/MC	Median fire interval (years) PP/MC	Weibull median probability interval (years) PP/MC	Minimum interval (years) PP/MC	Maximum interval (years) PP/MC
All fires	76/31	$4.3^{+}/12.4^{+}$	4.0/12.0	3.8/10.3	1/1	16/31
>2 Trees	48/18	$6.8^{+}/21.3^{+}$	5.0/16.5	5.8/18.9	1/6	20/71
10%	34/18	$9.1^{+}/21.3^{+}$	7.0/16.5	8.0/18.9	1/6	25/71
20%	17/14	$17.1^{+}/27.4^{+}$	15.0/22.5	15.0/24.4	7/6	63/94
25%	14/11	20.8/31.6	15.5/25.0	18.8/27.8	7/6	63/94

$^{+}$ Indicates significantly different ($p < 0.05$) MFI between PP and MC (Student's t-test).

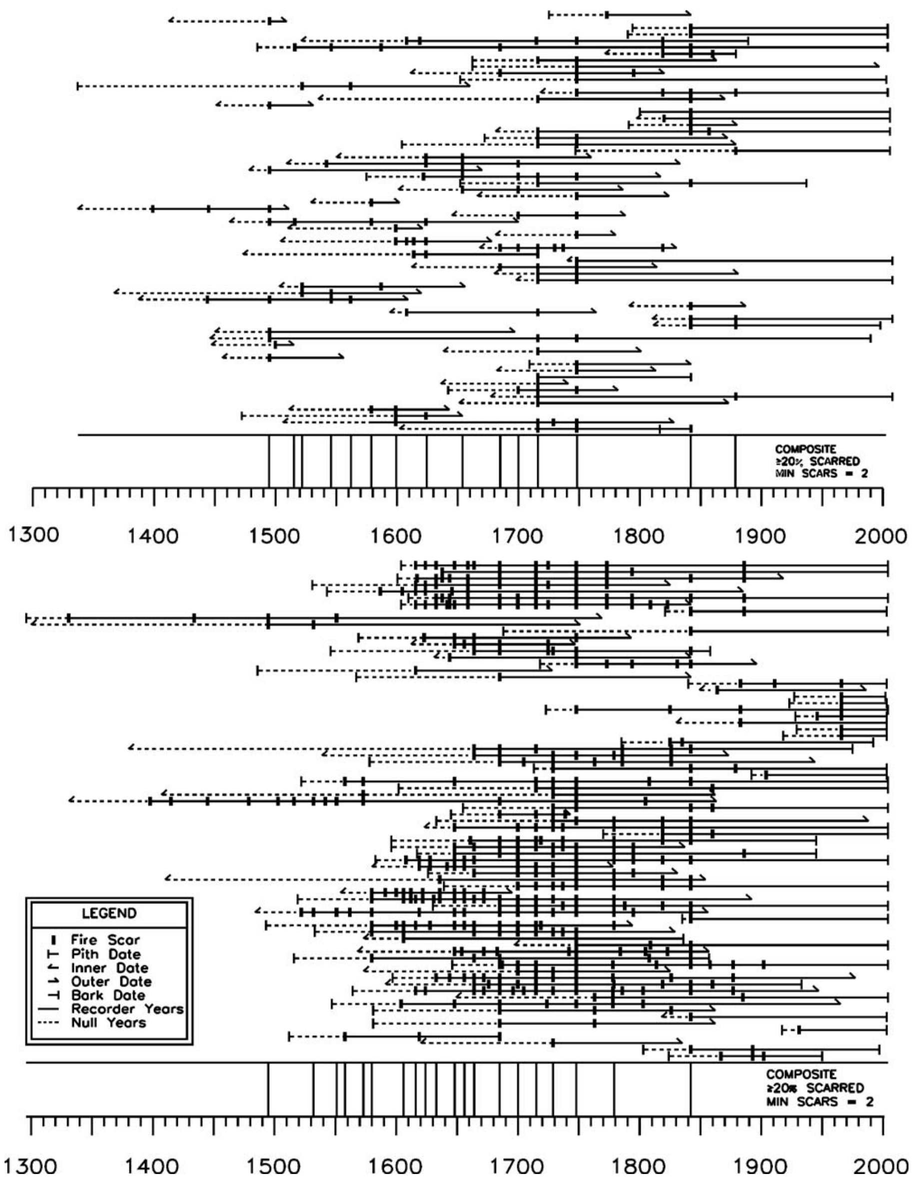

Fig. 2. Historical fire occurrence recorded by fire scars (1296–2004) in the PP forest (bottom) and MC forest (top) of the Santa Fe watershed. Each horizontal line is a tree and each vertical line is a dendrochronologically crossdated fire scar. The fire occurrence composite (bottom of each fire chart) indicates "widespread" fires recorded by a minimum of 2 trees and at least 20% of the trees recording fire.

3.3. Fire scars—spruce-dominated forest

No fire scars or any other direct evidence of fire (e.g., charred wood) were encountered at or between the age structure transects in the spruce-fir zone (1200 ha search area). Fire history in this vegetation type is presented in the age structure section.

3.4. PP vs. MC

The number of fire scars and individual fire years in the PP zone was approximately three times greater than that in the MC forest (Table 2). Historic fire intervals (1550–1880) in the PP zone were significantly shorter than in the MC forest for four of the five filtered subsets of fire years (all fires, ≥2 trees scarred, 10% scarred, and 20% scarred, Table 5). Although the MFI in the MC zone for the 25% scarred class (31.6 years) was approximately 10 years longer than in the ponderosa zone (20.8 years), the values were not statistically different (Student's *t*-test with equal variance, *t*-statistic = -1.780, $p = 0.092$).

Twenty-four fire years were synchronous between the two forest types (Table 3). Multiple synchronous fire years occurred every century from the 1400s to the 1800s. The number of synchronous fire years was greater than that would be expected by chance for all fire years ($\chi^2 = 39.22$, $p < 0.005$) and widespread (25% scarred) fire years ($\chi^2 = 29.15$, $p < 0.005$, with Yates correction for continuity). Sixty nine percent of all fires in the MC forest were also recorded in the PP zone. Only 24% of all fires in the PP forest were recorded in the MC zone.

3.5. Age structure

All of the age structure transects were located in the MC and spruce-dominated forest. Age structure transects were classified as spruce-dominated ($n = 14$) if the plurality of dominant trees was

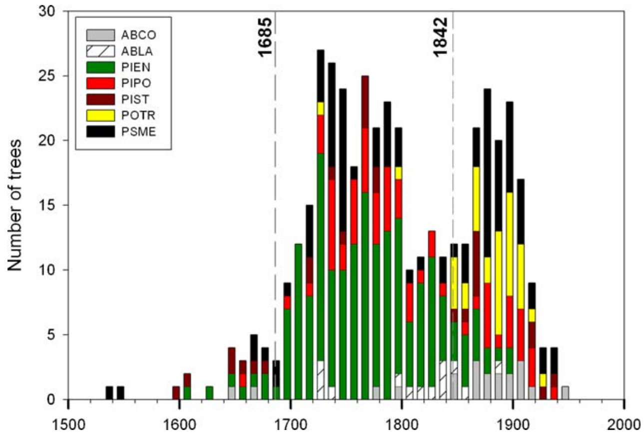

Fig. 3. Age structure and species composition of the dominant trees from the MC and the spruce-dominated forests. Data are from all age structure transects, in 10-year classes (plotted on the last year of the decade), and presented as estimated pith dates. Quaking aspen (POTR) was sub-dominant, but was sampled as a potential indicator of high severity fire. The last widespread fires with a stand-replacing component in the spruce-dominated (1685) and the MC forests (1842) are indicated as dashed lines.

Engelmann spruce. The remaining transects were classified as MC ($n = 12$). We collected 594 cores from 488 trees at 26 age structure transects (Fig. 1). We were not able to collect cores from all 20 dominant trees at 5 transects due to decomposed wood near the tree center and inclement weather. We were able to estimate pith dates for 438 (90%) of the sampled trees. Cores from the remaining 10% of the trees had no curvature in the inner rings or were estimated to be greater than 30 years from the pith so the number of rings to pith could not be estimated. The major cause for inadequate cores for pith estimation was decomposed wood near pith.

The collective age structure of dominant trees at all 26 transects in the MC and spruce-dominant forest has two recruitment peaks (i.e., a bi-modal distribution, Fig. 3). Less than 3% of the dominant trees established prior to 1650. A change in recruitment occurred in the late 1600s, increasing from a local minima of three trees (1681–1690) to the mode of 27 trees only 40 years later (1721–1730). This recruitment peak is dominated by Engelmann spruce. A second major tree recruitment pulse occurred in the mid-1800s. This younger recruitment peak is dominated by MC species. The recruitment peaks follow the last widespread fires in the MC (1842) and the spruce-dominated forests (1685) and there are relatively few trees dating to the decades prior to these two widespread fires.

The age structure at the individual transects illustrates both commonality and variability within and between the MC and the spruce-dominated forests (Figs. 4 and 5). The youngest MC stands all established after 1850 (1, 2, 6, 7) and were located nearest to the PP zone. The two oldest MC stands established circa 1600 (14, 18) and were located on rocky, relatively fire-protected sites near the upper MC/spruce ecotone. The youngest spruce-dominated stand began regenerating in the 1760s and the oldest trees date to the 1530s. The average age of the dominant trees in the spruce-dominated stands (mean [median] estimated pith date = 1769 [1763]) was approximately 60–100 years greater than in the MC forest (1829 [1861]).

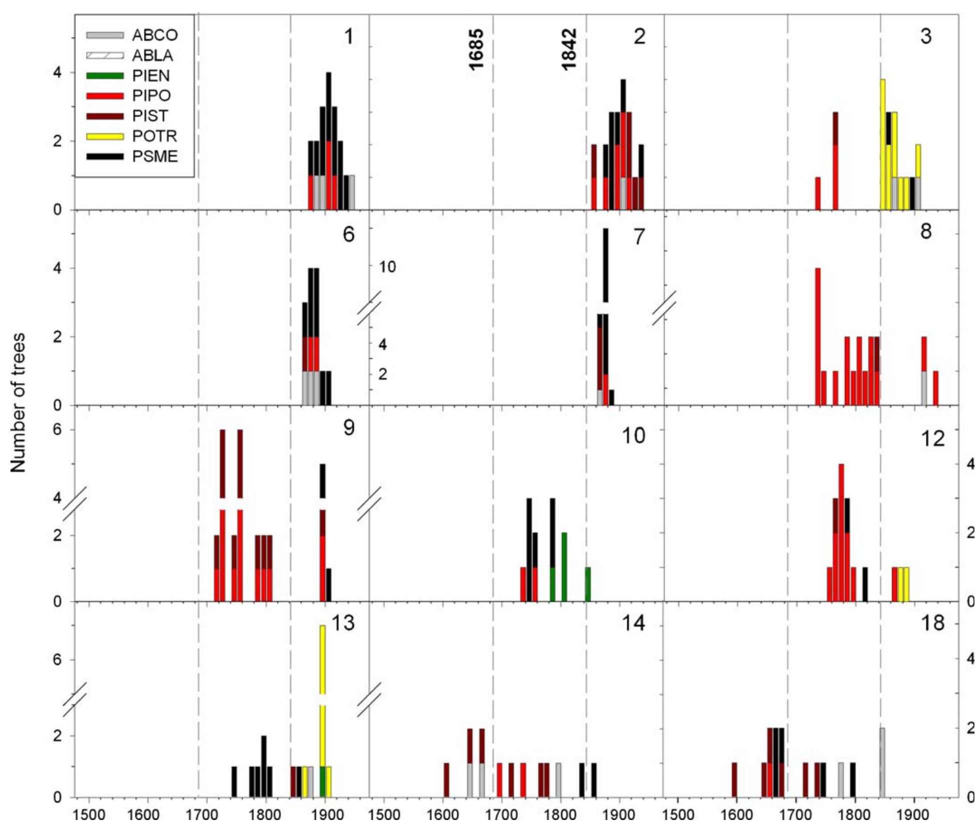

Fig. 4. Age structure and species composition of the dominant trees at individual age structure transects (e.g., 1) from the MC forest. Tree age data (estimated pith dates) are in 10-year classes. Note different scale for transects 7, 9, and 13.

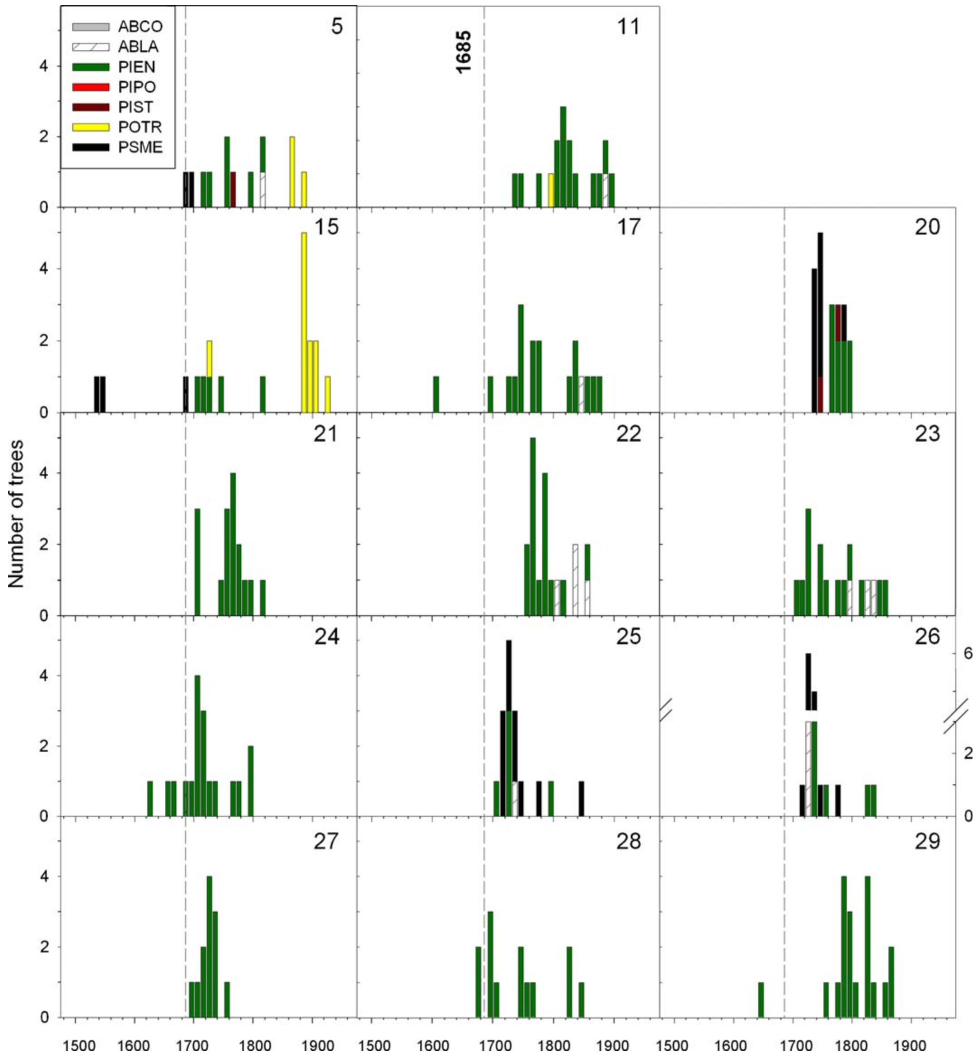

Fig. 5. Age structure and species composition of the dominant trees at individual age structure transects (e.g., 11) from the spruce (co-) dominated forest. Tree age data (estimated pith dates) are in 10-year classes. The lack of trees pre-dating the 1685 fire at 9 of the 14 transects suggests that this fire was largely stand-replacing in the upper, spruce-dominated portion of the watershed. Four of the five transects with trees surviving the fire (15, 17, 24, 28) had growth changes or injuries (i.e., traumatic resin ducts) in the tree-rings in 1685 (see Fig. 6).

3.6. *Evidence of stand-replacing fire*

The 1685 fire was recorded as fire scars by 57% ($n = 35$) of the recording fire-scarred trees at 68% ($n = 19$) of the recording fire scar plots throughout the MC and PP zones. Nine of the 14 spruce-dominated age structure transects and 10 of the 12 MC transects had no living trees that pre-date 1685. Four out of the five spruce-dominated transects that pre-date 1685 (15, 17, 24, and 28) had trees with growth changes or injuries/resin ducts in the tree-rings in 1685 (e.g., Fig. 6). The combination of age structure, growth changes/injuries, and widespread fire scar evidence indicates that the 1685 fire was relatively large and stand-replacing in the upper elevation forest.

The interpolated area of the 1685 fire within the upper Santa Fe watershed based on the spatial distribution of tree-ring evidence was 4730 ha. Approximately 25% of the reconstructed fire area was stand-replacing (1200 ha), all within the spruce-dominated zone (Fig. 7). It is likely that some of the younger forest stands below the spruce-dominated zone also burned with stand-replacing severity in 1685, but subsequent fires killed and burned any evidence of prior post-fire cohorts. We were conservative when reconstructing fire area and included these younger age structure transects as "not recording." The gaps between polygons in the reconstructed 1685 fire area are likely due to this lost record of fire.

Other fires that were widespread throughout the watershed (i.e., recorded by >50% of recording fire scar plots in the MC and PP forests, 1748, 1842; Fig. 2) were not recorded in the spruce-dominated forest. Age structure transects with many trees that

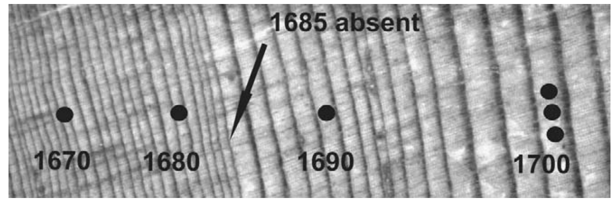

Fig. 6. Tree-ring growth release in a Douglass-fir core inferred to be a result of reduced competition due to tree mortality following the 1685 high severity fire.

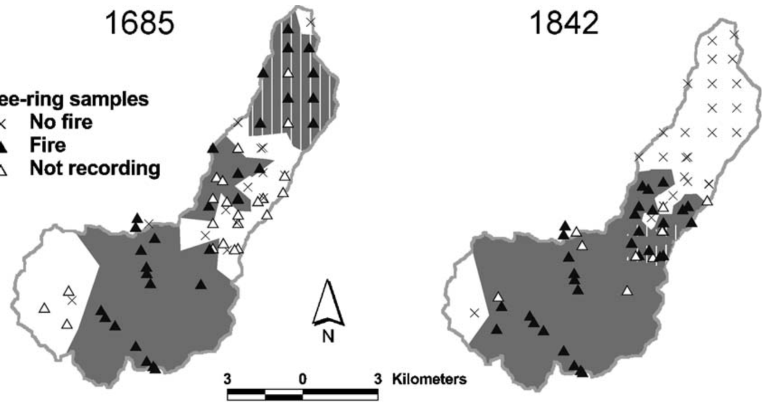

Fig. 7. Reconstructed fire area (gray) derived from thiesen polygon interpolation of tree-ring fire history data (fire scars, death dates, growth changes/injuries, and forest age structure). Areas with vertical white lines indicate stand-replacing fire patches.

pre-date these fires suggest that although these fires were widespread in the mid-elevation MC and lower pine forests, climate and/or fuel conditions were not suitable for fire spread into the mesic upper elevation spruce-dominated forest. It is possible however, that some widespread fires (e.g., 1716) may have burned with localized stand-replacing severity in the lower spruce-dominated forest and may explain the lack of trees in the early 1700s at some transects (e.g., 20).

3.7. Mixed-severity fire

There was evidence of mixed-severity fire in the MC zone. We use the term "mixed-severity" to indicate that some forest stands experienced high severity, stand-replacing fire (recorded as a tree recruitment pulse with no surviving trees) and other, adjacent stands experienced low-severity surface fire (recorded as fire scars). The landscape structure in the lower MC zone is such that north- and south-facing slopes are located on opposite sides of ridges. The youngest stands in the watershed (transects 1, 2, 6 and 7; Figs. 1 and 4) were on the more productive north- and east-facing slopes in this zone, near the ecotone with PP. These stands established in the mid-to-late 19th century and had the highest percentages of trees with "open" inner-ring growth (85–95%).

The 1842 fire was recorded as fire scars by 82% ($n = 24$) of the recording plots and 57% ($n = 42$) of the recording fire-scarred trees in the MC and PP forests. In addition to the four transects with no trees surviving the 1842 fire, three transects (3, 9, and 12) had growth changes or injuries/resin ducts in the tree-rings in 1842. Transect three had an aspen recruitment pulse beginning immediately following 1842 and the dominant PP trees that survived the fire had multi-year growth suppressions in the tree-rings beginning in 1843. The fire scar plot located less than 200 m southwest of age structure transect six had no samples post-dating 1842 and one of the fire-scarred trees had a bark-ring date of 1841.

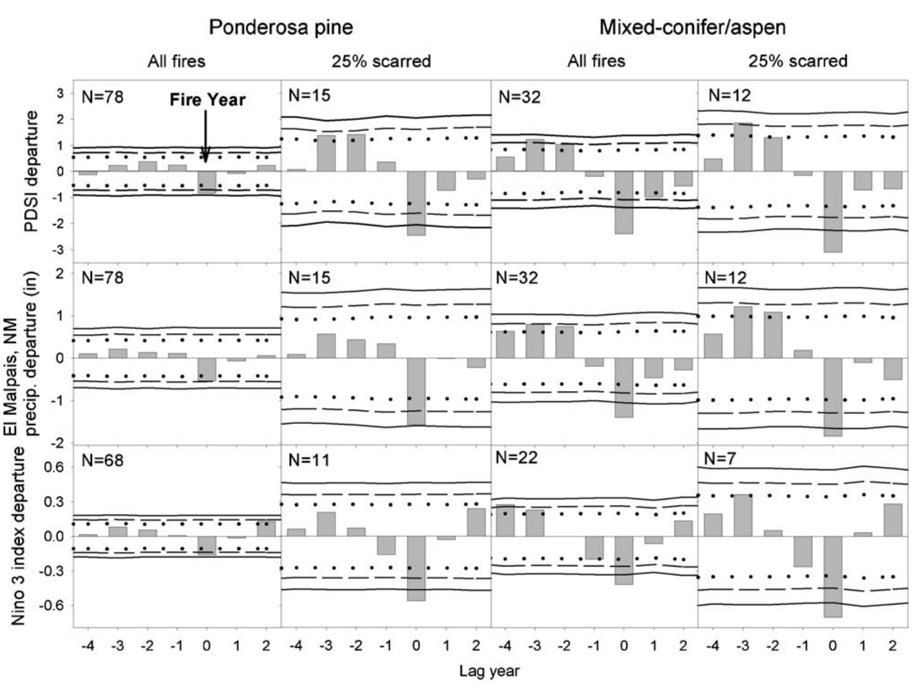

Fig. 8. Superposed epoch analysis for the PP and MC forests illustrating departure from the mean of reconstructed climate indices (PDSI, El Malpais, NM precipitation, and Niño3 index) for all fires and widespread (25% scarred) fires. Dotted, dashed and solid lines represent 95, 99, and 99.9% confidence intervals derived from 1000 Monte Carlo simulations; *n*, number of fire years.

The proximity of high severity (age structure) and low-severity (fire scar and tree-ring growth change) evidence within the lower MC zone indicates that the 1842 fire burned with mixed-severity within this forest type. Based on the multiple lines of fire evidence presented above, the reconstructed 1842 fire area within the upper Santa Fe watershed was 4642 ha (Fig. 7). The reconstructed stand-replacing fire area was 182 ha, consisting of multiple patches ranging from 34 ha to 110 ha.

3.8. Fire–climate

The results of the SEA indicate that all four filtered subsets of fire scar data (all fires, 10% scarred [not shown], 20% scarred [not shown], and 25% scarred) from both the PP and MC forests were significantly associated with negative (dry) departures during the fire year from mean summer PDSI and El Malpais, NM precipitation (Fig. 8). Fire occurrence in both forest types was also associated with positive (wet) departures from mean summer PDSI two to three years prior to the fire year. Fire occurrence was also associated with positive (wet) departures from mean annual precipitation at El Malpais, NM two to four years prior to the fire year in the MC forest. All sets of fire scar data in both forest types were associated with negative (cool ocean phase—La Niña) SST departures from the mean Niño3 index during the fire year. All fires and widespread (25% scarred) fires in the MC forest and 10% scarred fires in the PP forest (not shown) were associated with positive (warm ocean phase—El Niño) SST departures three to four years prior to the fire year. Fire occurrence in both forest types was not associated with inter-annual variations in PDO (results not shown). The period of analysis was the same used in the fire interval analyses, except when the reconstructed climate series was limiting (i.e., earliest date for reconstructed Niño3 index, 1600 and PDO index, 1700).

Although the results of the SEA indicate surprisingly similar inter-annual fire–climate relationships between the MC and the PP forest types, there were some differences. Mean summer PDSI associated with all fires in the MC forest (-2.59) was significantly drier than in the PP (-1.03; $t = 3.428$, $p < 0.001$, t-test with equal variance; SPSS 16.0). Widespread fire years (25% scarred) in the mixed/conifer aspen forests occurred on drier years (mean PDSI = -3.22) than in the PP forest (-2.57), but the difference was not statistically significant ($t = 0.798$, $p = 0.432$). Widespread fires occurred during drier years on average compared to all fires in the PP forest ($t = 2.498$, $p = 0.014$). The same was true in the MC forest, but the difference was not significant ($t = 0.896$, $p = 0.375$). The PDSI during the one reconstructed stand-replacing fire (1685) in the spruce-fir zone was -6.92.

4. Discussion

Fire in the upper Santa Fe River watershed historically spread between forest types and fire regimes. Low severity fires burned frequently in the PP forests. During sufficiently dry conditions fire spread up the watershed into the MC forests and burned with mixed-severity. During an extreme drought (1685), fire continued to spread into the highest elevation spruce-dominated forests and burned primarily with high severity. The connectivity of forests through fire, the removal of this important process, and historical evidence of large (100–1200 ha) stand-replacing fire patches in MC and spruce-dominated forests have important implications for both fire and water management in the upper Santa Fe watershed and similar forests throughout the region.

4.1. Human influence on the fire regime

Santa Fe was settled by the Spanish earlier than other locations in the southwestern U.S. (1600s), making this site unique. The most striking feature of the Santa Fe watershed fire scar record is the lack of widespread fire since the mid-to-late 19th century (Fig. 2). Fires stopped earlier (i.e., last widespread fire in the PP and MC, 1842) compared to the general pattern of circa 1900 fire exclusion in the southwestern U.S. (Swetnam and Baisan, 1996, 2003). The start of fire exclusion at a particular site has been linked to the timing of intensive land use practices (e.g., grazing and fuel wood collecting) by the Spanish and Anglo-American settlers (Savage and Swetnam, 1990; Baisan and Swetnam, 1997). Sheep herding in the vicinity of Santa Fe began in the 1600s, became a stable industry regionally by the mid-1700s, and peak numbers in the pre-American Civil War era were recorded in the 1820s and 1830s (Baxter, 1987). This early, intensive land use may have created a pattern of anomalously early fire exclusion (e.g., early 1700s, Sandia Mountains, NM; Baisan and Swetnam, 1997) on the east side of the Rio Grande valley along the Camino Real Spanish travel and settlement route. A long gap between widespread fires in the PP and MC forest in the Santa Fe watershed beginning in the 1700s may indicate initial effects of early grazing, but may also have a climatic explanation.

In specific locations in the southwestern U.S. the fire scar record has revealed periods of anomalously high fire frequency (e.g., repeated 1-year fire intervals) or a change in the seasonality of fire occurrence, indicating possible human ignitions (e.g., Chiricahua Mountains, Arizona; Seklecki et al. (1996)). Very few (<2%) latewood fires were recorded in the Santa Fe watershed and there was not evidence of anomalously high fire frequency, despite the long record of settlement. The high percentage of lightning-caused fires (80%, $n = 178$, 1970–2003) in the local area supports the general premise that sufficient lightning ignitions occur in the southwestern U.S. to account for the reconstructed frequency of fire occurrence (Allen, 2002).

4.2. Spruce-fir fire history

Very little fire history and/or age structure data exist for old-growth spruce-fir forests of Arizona and New Mexico. Fule et al. (2003) reconstructed a mixed-severity fire regime with surprisingly frequent small fires ($MFI_{all\ fires} = 2.6$ years) and less frequent widespread fires ($MFI_{25\%} = 31.0$ years) in a relatively low elevation (<2800 m) spruce-fir forest that contained a mix of species (including PP) on the north rim of the Grand Canyon, AZ. A higher elevation spruce-fir forest (average elevation 3200 m) in the San Francisco Peaks, AZ, has not burned catastrophically for over 200 years based on the age of the oldest trees (Cocke et al., 2005). Other high elevation (>3000 m) pure spruce-fir forests in the southern sky island region (Pinaleño Mountains, AZ, and Mogollon Mountains in the Gila Wilderness, NM) had not experienced significant stand-replacing disturbance for at least 300 years prior to the recent crown fires beginning in the late 1990s (Grissino-Mayer et al., 1995; Margolis, 2007). Multiple lines of tree-ring evidence suggest that the Pinaleño spruce-fir stand regenerated after a stand-replacing fire in 1685 (Grissino-Mayer et al., 1995; Margolis, 2007; Swetnam et al., 2009), the same year as the upper Santa Fe watershed. Drought conditions in 1685 were remarkably severe and widespread throughout the southwestern U.S. (Cook et al., 2004). This climate event synchronized these rare stand-replacing fire events, and potentially others, hundreds of kilometers apart.

4.3. Comparing the PP and MC fire regimes

Historical MFI was significantly shorter in PP compared to the higher elevation MC forest in four of the five filtered subsets of fire years (Table 5). Widespread fires in the MC forest occurred on average at intervals that were 10 years (50%) longer than in PP

(Table 5; PP $MFI_{25\%} = 20.8$ years, MC $MFI_{25\%} = 31.6$ years). The difference in fire frequency might be partially explained by a larger area in the PP zone (PP, 1600 ha vs. MC, 1200 ha), different sampling intensity (PP, 76 trees: MC, 65 trees) or the spatial distribution of samples. However, these sampling differences are relatively small and with sufficient sample numbers, 25% scarred MFI is robust to differences in sampling (Van Horne and Fule, 2006) and thus likely does not account for the magnitude of observed fire frequency differences. Regionally, MC forests burned less frequently than pine-dominant forests based on comparisons from dozens of Southwestern fire history studies (Swetnam and Baisan, 1996; Heinlein et al., 2005). $MFI_{25\%}$ of widespread fires at six other MC sites in New Mexico ranged from 16.0 years to 26.4 years (Swetnam and Baisan, 1996), which is shorter than the Santa Fe watershed (MC $MFI_{25\%} = 31.6$ years). The relatively long MFI could be a result of settlement and land-use (e.g., grazing) by the Spanish beginning in the 1600s (Debuys, 1985), which could have reduced fine fuels and consequently fire occurrence in the watershed earlier than in other locations (e.g., Savage and Swetnam, 1990; Baisan and Swetnam, 1997).

An inverse relationship between fire frequency and elevation exists broadly across the montane forests of the western U.S. (Martin, 1982) and at individual sites (Caprio and Swetnam, 1995; Brown et al., 2001)), but site-specific topographic factors may weaken the relationship in some locations (Brown et al., 2001). A hypothesized mechanism for this pattern relates to increased moisture in the higher elevation forests and consequently less frequent occurrence of drought conditions severe enough to dry fuels sufficiently to sustain fire spread. Our results indicate that, on average, fires in the MC forest occurred during drier conditions compared to the adjoining lower elevation PP, providing quantitative support for this hypothesis (Fig. 8). Specifically, the grassy understory of the drier, relatively open PP forest was more likely to carry fire, even if fuels in the mesic mixed-conifer zone were not primed by drought for widespread fire.

4.4. Fire–climate relationships

The relationship between fire occurrence in MC and PP forests and drought during the fire year is intuitive and commonly observed in fire history reconstructions across fuel types in the southwestern U.S. (Fig. 8; Swetnam and Baisan, 1996). The relationship between fire occurrence and wet conditions in prior years is less intuitive, but also well replicated in pine-dominant forests of the southwestern U.S. from fire history studies (Baisan and Swetnam, 1990; Swetnam and Baisan, 1996) and the instrumental record (Crimmins and Comrie, 2004; Baisan and Swetnam, 1990) hypothesize that wet years increase fine fuels (e.g., grass and pine needles) that carry fire, which are burned during subsequent dry years.

This antecedent wet-year relationship is not present in high elevation sub-alpine forests and upper montane seral MC forests of the Southern Rockies (e.g., Sibold et al., 2006; Margolis et al., 2007). A similar drought-only fire–climate relationship exists at multiple MC fire history sites in the region (Swetnam and Baisan, 1996; Touchan et al., 1996). These more mesic, higher elevation forest types are generally not fuel-limited, but require more severe drought for fire occurrence than lower elevation forests.

Based on this prior research, the relationship between fire occurrence and antecedent wet years in the MC forests of the Santa Fe watershed was somewhat surprising (Fig. 8). This result suggests that variability in fine fuels may have been important for fire occurrence (i.e., the system was fuel-limited). But how can a fire regime with a 20- to 30-year mean return interval for widespread fires be fuel-limited? Twenty years in a MC forest should be sufficient to produce enough fuel to sustain fire spread, even in the semi-arid southwestern U.S. It is possible that due to the topographic heterogeneity of the landscape (opposing north and south-facing slopes), wet conditions followed by drought were needed to produce sufficient fuel on the drier south aspects to connect the more productive forest patches and allow fire to burn across aspect and forest types. Grazing could amplify the aspect-driven fuel discontinuity by further reducing fuels on the drier, grassy, south-facing slopes.

A second factor that may explain the wet lags in the MC SEA results is the connectivity of the MC forest to the adjacent, large, frequent burning PP forest. The PP forest in the Santa Fe watershed, similar to others throughout the southwestern U.S., had an herbaceous understory that fueled the frequent fires. As expected, historical fire occurrence in the Santa Fe watershed PP forest was associated with prior wet years that replenished this herbaceous fuel layer (Fig. 8). Prevailing wind direction and the tendency for fire to move upslope would push fires from the PP into the MC forest. Based on our analysis of fire synchrony, 24% of the PP fires spread to the MC forests, but these accounted for a large proportion (69%) of all fires in the MC forest (Table 3). Thus, if fires in PP were in part fueled by prior wet years, and it was sufficiently dry during the fire year, fires would continue to spread up the "fireshed" into the MC forest. The connectivity between forest types would indirectly link fire occurrence in the MC zone to antecedent wet years.

4.5. Landscape scale connectivity of fire regimes

By reconstructing fire history along an elevation, vegetation and fire regime gradient we were able to reconstruct evidence of the transition of fire regimes (and individual fires) from surface fire, to mixed-severity fire (e.g., 1842 fire), to widespread stand-replacing fire (e.g., 1685 fire) in a single watershed. We present multiple lines of evidence of connectivity between forest types and fire regimes through fire as a continuous process that moves across artificially drawn fire regime and vegetation boundaries (Caprio and Swetnam, 1995; Fule et al., 2003). An important implication of this connectivity is that by altering the fire regime in one location (forest type) there may be effects in other forest types. The disruption of the surface fire regime in the mid-elevation, pine-dominated forest throughout the southwestern U.S. (Swetnam and Baisan, 1996, 2003) may not only have serious consequences for that vegetation type (Allen et al., 2002), but is also likely to have effects all along vegetation/elevational gradients. In the Santa Fe watershed, early fire exclusion in PP (i.e., last widespread fire, 1842) from grazing followed by active fire suppression removed an important source of fires for the MC and the spruce-dominated forests. As a result, fire frequency was dramatically reduced in the upper elevation MC forest (Fig. 2).

4.6. Mixed-conifer/aspen forest change due to fire exclusion

Over 120 years of fire exclusion in the MC forest has contributed to changes in structure and composition similar to what occurred regionally and locally in PP and MC forests (Fig. 9). We present age structure data from two fire sensitive species (white fir and quaking aspen) as examples of changes in species composition in the MC forest that occurred coincidently with fire exclusion. Seventy-five percent of the dominant white fir in the MC zone recruited since the last widespread fire (1842; Figs. 3–5). Young white fir has thin bark, making them particularly sensitive to even low-intensity surface fire. In the absence of fire these trees survived to occupy a dominant canopy position, and because they are shade tolerant, they have continued to recruit in the understory, creating ladder fuels, and increasing crown fire hazard. This pattern has been documented in PP dominated systems (Allen

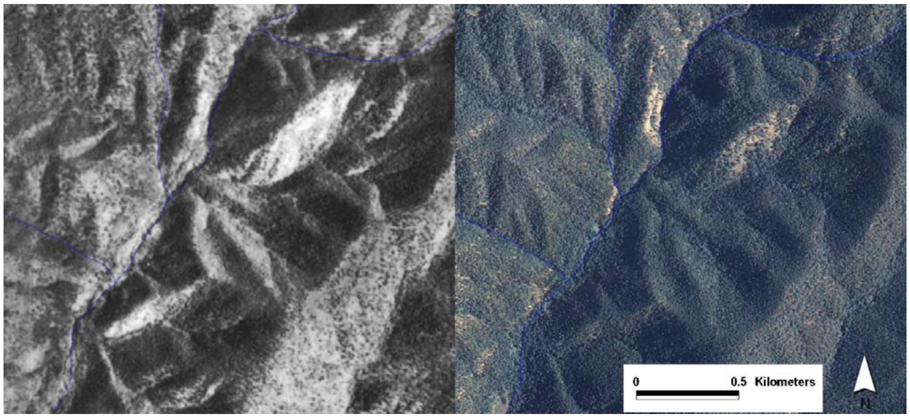

Fig. 9. Comparison of aerial photos (1935 on the left, 2005 on the right) from the MC forest of the Santa Fe watershed indicating a dramatic increase in forest cover on south- and southeast-facing slopes. Images encompass age structure plots 8, 12, and 7. Photos from the U.S.F.S. Santa Fe National Forest, courtesy of Julie Luetzelschwab.

et al., 2002) and other southwestern U.S. MC forests (Mast and Wolf, 2006).

Fire was historically an important determinant of quaking aspen mortality and natality in many upper elevation forests across the western U.S. (Kulakowski et al., 2006; Margolis, 2007; Margolis et al., 2007) and the cessation of fire has been identified as one cause of widespread stand-deterioration throughout its range (Bartos and Campbell, 1998; Kashian et al., 2007). In the Santa Fe watershed, only one (2.5%) quaking aspen stem pre-dated the last widespread fire (1842, Figs. 3–5). Quaking aspen recruitment pulses occurred at three transects following the last fire (3, 13, and 15). Conifers survived the fire at these locations, indicating mixed-severity fire effects by species (i.e., quaking aspen were top-killed and re-sprouted while the overstory conifers survived). This evidence of fire killing and regenerating quaking aspen stems at multiple locations throughout the MC forest illustrates the substantial effect of fire (occurrence and exclusion) on quaking aspen age structure.

4.7. Spruce-fir forest: potential for fire exclusion effects

In the high elevation spruce-fir forests of the region, limited research has assessed the potential for changes related to fire exclusion (Fule et al., 2003; Cocke et al., 2005). Cocke et al. (2005) recorded increased density in spruce-fir since 1876, but this is consistent with natural succession in this forest type. Because *Picea* and *Abies* species are shade tolerant and fires are infrequent, these forests naturally increase in density through time. Different approaches (e.g., examining effects of fire interval length on successional pathways) may be necessary to evaluate potential effects of fire exclusion in this forest type.

Changes in the length of fire-free intervals, even if they were naturally long, may affect successional pathways and forest composition (Romme and Knight, 1981; Kipfmueller and Kupfer, 2005). For example, Romme and Knight (1981) found that sites with naturally longer fire-free intervals and more rapid succession were dominated by spruce-fir forests, compared to sites with more frequent fire and slower succession, which were dominated by lodgepole pine. In the southwestern U.S., quaking aspen is the upper elevation tree species most likely to be sensitive to changes in the length of fire intervals. Seral quaking aspen in the upper montane forests of the region depend on stand-replacing fire for widespread regeneration and long-term perpetuation of the stand (Margolis et al., 2007). Following fire in these seral stands, the aspen-conifer successional pathway proceeds and shade tolerant conifer species regenerate under the canopy, eventually overtopping and shading out the aspen stems in the absence of fire (Dick-Peddie, 1993). Lengthening fire-free intervals in seral aspen stands beyond the life of the above-ground stems and the below-ground clonal root resources could potentially remove aspen from the site, affecting the long-term forest composition.

We hypothesize that although fire intervals were naturally long in high elevation forests of the southern Rocky Mountains, because fire historically spread between forest types, fire exclusion in the lower elevation forests has likely affected some high elevation forests. Future research should be designed to test for changes (e.g., altered successional pathways) resulting from fire exclusion. Upper elevation spruce-fir forests are naturally dense, so although forest density has been an indicator of change in PP and MC forests, it is not likely the best variable to test for change in the spruce-fir zone.

4.8. Will Santa Fe flood?

Large patches of high severity fire (>100 ha) historically occurred on some north-facing slopes in the MC forests of the Santa Fe watershed. The dramatic increase in forest density and canopy cover in these forests, evident from repeat photos (Fig. 9), has very likely increased the size of forest patches at risk of high severity fire. Areas that historically burned with mixed-severity (i.e., 100 ha patches of high severity fire adjacent to equally large low-severity patches) now are likely to burn as larger, contiguous high severity patches. This increased area of forest at risk of stand-replacing fire could subsequently result in a larger, historically unprecedented post-fire hydrologic response in this vital municipal watershed (e.g., Veenhuis, 2002).

One approach to evaluating post-fire flood risk would be to use a combination of our historical fire reconstructions and a hydrological model. The 1685 fire was the worst-case scenario in the spruce-dominated forest; 93% of the sampled spruce forest burned with stand-replacing severity (~1200 ha). The reconstructed spatial extent and location of low and high severity fire patches from this fire and others (e.g., 1842) could be used to populate a GIS-based hydrologic model such as The Automated Geospatial Watershed Assessment Tool (Goodrich et al., 2006). Alternatively, fire behavior and fire spread models (e.g., FARSITE) could be used to estimate the range of high severity patch sizes under current forest conditions for comparison with reconstructed patch size. The different fire scenarios (modeled and reconstructed) could then be used to populate the hydrologic model. Modeled post-fire runoff and erosion output would provide the best possible answer to the big question in the Santa Fe watershed: what will happen to the water supply when the forest burns?

5. Conclusions

Historical fire in the upper Santa Fe River watershed burned across gradients of elevation, forest types and fire severity. Widespread fires that burned up to 80% of the MC forest area occurred on average at intervals 10 years longer ($MFI_{25\%}$ = 31.6 years) than in the adjacent, lower elevation PP forest ($MFI_{25\%}$ = 20.8 years). The historical MC fire regime is best described as mixed-severity, where patches of stand-replacing fire greater than 100 ha were located adjacent to stands with evidence of repeated surface fire. The upper elevation spruce-dominated forest last burned in 1685 in a climate-driven stand-replacing fire that affected greater than 93% (1200 ha) of the sampled spruce forest and at least 68% of the MC and PP forests (total fire area, 4730 ha). This history of fire that includes natural stand-replacing patches in the upper elevation forests presents challenges for fire management in the watershed. Restoring the aspect-driven heterogeneity of fuels in the MC forest is both ecologically sound and would reduce the area at risk of crown that could threaten the water supply. Given the natural occurrence of large (>1000 ha) stand-replacing fire patches in the spruce-fir zone of the Pecos Wilderness Area, where fire hazard reduction treatment options are limited and would be ecologically unsound, hydrologic models should be used to develop a contingency plan for a large, high severity fire.

Climate variability has strongly influenced fire regimes for centuries in the montane forests of the southwestern U.S. (Swetnam and Betancourt, 1990; Swetnam and Baisan, 1996) and more broadly across western North America (Kitzberger et al., 2007). Fire synchrony between the MC and the PP forest during 24 individual fire years (69% of all MC fires) indicates both top-down control of fire occurrence by climate and connectivity between forest types and fire regimes. More severe drought was required on average for the higher elevation MC forest to burn (sometimes with mixed-severity), compared to the lower PP forest. The worst single-year drought in over 700 years (1685) was associated with the last major fire in the upper elevation spruce-dominated forests of the Santa Fe watershed and synchronized high severity fire in the upper elevations of multiple, distant mountain ranges. This evidence of a direct relationship between drought severity, fire occurrence, and fire severity in MC and spruce-dominated forests suggests that if temperatures continue to increase (IPCC, 2007) and droughts become more frequent and severe as predicted (Seager et al., 2007), the probability of large and severe fire occurrence will increase (Westerling et al., 2006). This emphasizes the urgency for creative and science-based fire and watershed planning and management in this and other fire prone, vitally important watersheds across the West.

Acknowledgements

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Drought, multi-seasonal climate, and wildfire in northern New Mexico

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Abstract Wildfire is increasingly a concern in the USA, where 10 million acres burned in 2015. Climate is a primary driver of wildfire, and understanding fire-climate relationships is crucial for informing fire management and modeling the effects of climate change on fire. In the southwestern USA, fire-climate relationships have been informed by tree-ring data that extend centuries prior to the onset of fire exclusion in the late 1800s. Variability in cool-season precipitation has been linked to fire occurrence, but the effects of the summer North American monsoon on fire are less understood, as are the effects of climate on fire seasonality. We use a new set of reconstructions for cool-season (October–April) and monsoon-season (July–August) moisture conditions along with a large new fire scar dataset to examine relationships between multi-seasonal climate variability, fire extent, and fire seasonality in the Jemez Mountains, New Mexico (1599–1899 CE). Results suggest that large fires burning in all seasons are strongly influenced by the current year cool-season moisture, but fires burning mid-summer to fall are also influenced by monsoon moisture. Wet conditions several years prior to the fire year during the cool season, and to a lesser extent during the monsoon season, are also important for spring through late-summer fires. Persistent cool-season drought longer than 3 years may inhibit fires due to the lack of moisture to replenish surface fuels. This suggests that fuels may become increasingly limiting for fire occurrence in semi-arid regions that are projected to become drier with climate change.

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1 Introduction

Over the past decade, wildfires have made headlines due to their increasing size, severity, and cost. Warming temperatures, drought, and earlier snowmelt—all consistent with projected future climate in the southwestern USA—have been linked to an increasing number of large fires (Dennison et al. 2014; Westerling 2016). The legacy of late nineteenth and twentieth century land use and forest management has also played an important role by increasing fuels that have led to recent megafires (10,000 ha to >100,000 ha), particularly in dry conifer forests (Stephens et al. 2014). Natural climate variability, in addition to human influences, has long been a primary driver of variability in wildfire occurrence, severity, and seasonality (Littell et al. 2016; Swetnam et al. 2016). Climate change will likely alter fire regimes globally, but the mechanisms and the directions of the effects are complex and will vary geographically (Moritz et al. 2012).

Understanding the relationships between climate variability and wildfire by analyzing instrumental and paleoecological data (e.g., tree rings or sediment charcoal) is increasingly valuable for fire management and modeling future fire regimes. Robust relationships have been established in North America between variability in instrumental period (twentieth and twenty-first century) and paleo (pre-twentieth century) fire records and a suite of climate variables, climate patterns, and ocean-atmosphere oscillations (Swetnam and Betancourt 1990; Westerling et al. 2006; Kitzberger et al. 2007; Marlon et al. 2012; Williams et al. 2015). However, fire-climate relationships are spatially and temporally complex, with significant variability within and among regions in fire and moisture seasonality, and lagging relationships that drive fire (Swetnam and Betancourt 1998; Littell et al. 2009; Keeley and Syphard 2016). To date, there is limited understanding of the impacts of the seasonality of moisture and persistent drought on wildfire size and seasonality.

The relationships between cool-season moisture and fire over past centuries (circa 1600–1900 CE) have long been established in the southwestern USA using data from fire-scarred trees (for background on fire scars, see Text S1). A pattern of one or two wet cool seasons followed by cool-season drought is consistently associated with fire occurrence in dry conifer forests of the region (Swetnam and Betancourt 1990; Swetnam and Betancourt 1998). In contrast, the role of summer moisture, delivered through the North American monsoon (NAM) and accounting for up to 50% of the annual precipitation in the southwestern USA, has not been well investigated. Limited research indicates a potential influence of the NAM on fire through increased fine fuels from prior wet monsoons (Crimmins and Comrie 2004; Text S2), or the possibility of monsoon drought leading to more monsoon-season fires (Grissino-Mayer and Swetnam 2000). Until recently, there have been no tree-ring proxies of summer moisture, but a large new network of partial ring-width chronologies now enables the reconstruction of both cool- and monsoon-season moisture in the southwestern USA (Griffin et al. 2013; Text S3).

In this study, we compile the largest known collection of fire scar data for a single mountain range and develop new reconstructions of cool- and monsoon-season moisture to investigate relationships between historical fire regimes and multi-seasonal climate in northern New Mexico. Our main research questions are (1) How do monsoon- and cool-season moisture variability affect fire occurrence, extent, and seasonality? and (2) What is the relationship between fire and prolonged drought? Our goal is to improve the understanding of fire-climate relationships in the past to help inform how climate change may impact fire regimes in the future.

2 Study area and data

The Jemez Mountains are located in northern New Mexico within NAM region 3 (Gochis et al. 2009; Fig. 1). Approximately 44% of the annual precipitation falls in the cool season (October–April) and 43% in the monsoon season (July–September). The warmest and driest months of the year are May and June, when the largest fires occur. Multiple large fires have burned in the Jemez Mountains in recent years, including the 2011 Las Conchas fire (63,400 ha). Vegetation in the Jemez Mountains ranges from grasslands at the lower forest border (~2000 m a.s.l.), to ponderosa pine and mixed conifer forests, to montane meadows and spruce forests at the highest elevations (~3000 m a.s.l.). The majority of the landscape was historically dry conifer forest that included ponderosa pine. The region has extensive networks of fire-scarred and climatically sensitive trees, making it an ideal location for tree-ring fire-climate analyses (Swetnam et al. 2016).

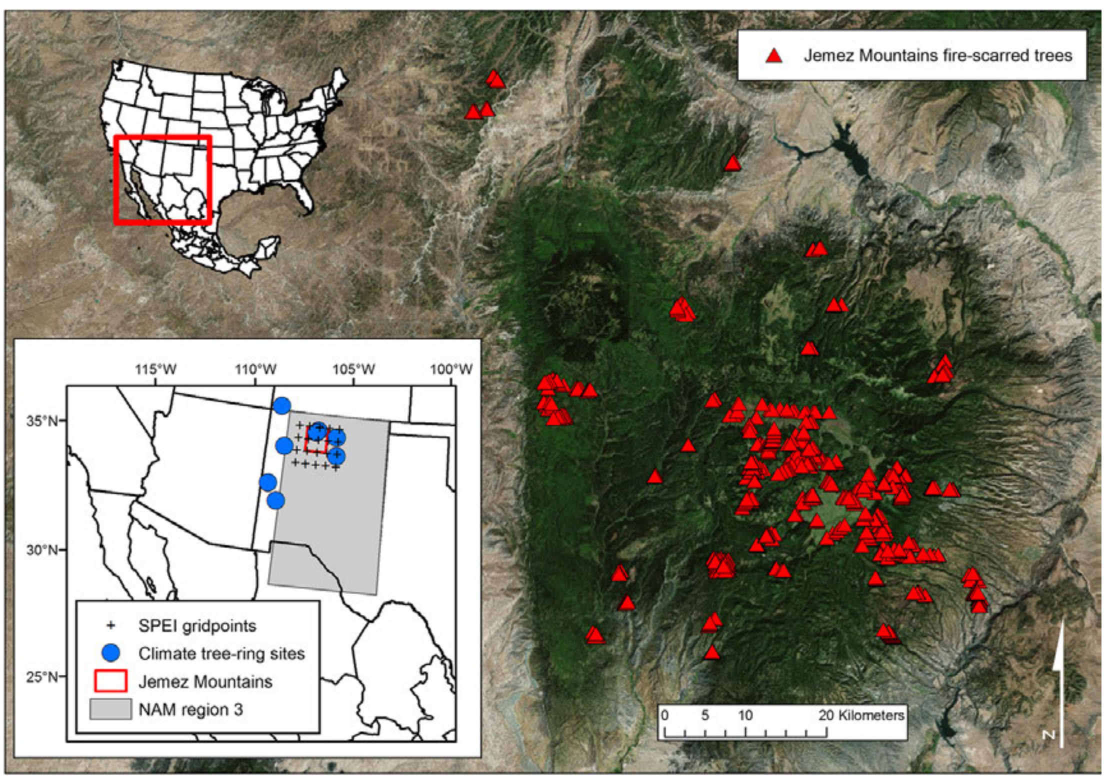

Fig. 1 Study area in southwestern North America focused on the North American monsoon (NAM) region 3. Inset map indicates the location of the climate-sensitive tree-ring sites, the standardized precipitation-evapotranspiration index (SPEI) gridpoints used in the climate reconstructions, the Jemez Mountains, and NAM region 3. The aerial photo is of the Jemez Mountains in New Mexico, which contain a network of 1343 fire-scarred trees

2.1 Tree-ring climate reconstructions

To reconstruct cool- and monsoon-season moisture, we used existing earlywood and adjusted latewood chronologies from 23 sites in New Mexico and southern Colorado located within and adjacent to the Jemez Mountains and NAM region 3 (Text S3). Adjusted latewood chronologies have the dependence of latewood growth on earlywood removed statistically (Griffin et al. 2011). We reconstructed the standardized precipitation-evapotranspiration index (SPEI), because fire is influenced by the combined effects of temperature and moisture that are integrated into SPEI (Williams et al. 2015). SPEI data were obtained from the Global SPEI Database, which uses monthly precipitation and potential evapotranspiration at a 0.5 degree spatial resolution. A regional time series was generated based on the average of 20 grid points centered on the Jemez study area (Text S3, Fig. 1). Monthly SPEI was averaged for the cool (October–April) and monsoon (July–August) seasons.

Reconstruction models were developed by calibrating earlywood chronologies with October–April SPEI and adjusted latewood chronologies with July–August SPEI separately, using stepwise regression (1896–2007). Models explained 67 and 52% of the total variance for October–April and July–August SPEI, respectively. Models met the assumptions of linear regression, and cross-validation statistics indicate reasonable skill. Details of regression results are in supplemental materials (Text S3, Table S1, Fig. S1). The October–April SPEI reconstruction extends 1594–2007 and July–August SPEI, 1599–2008. The relationship between the seasonal SPEI variables is preserved, for the most part, in the reconstructions. There is no relationship between the instrumental cool- and monsoon-season SPEI ($r = -0.09$, $p > 0.05$), but there is a weak correlation between the reconstructed cool- and monsoon-season SPEI in the instrumental period ($r = 0.23$, $p < 0.05$). Over the full common reconstruction period, 1599–2007, the cool and monsoon-season SPEI are uncorrelated ($r = 0.09$, $p > 0.05$).

2.2 Tree-ring fire history reconstructions

The tree-ring fire scar data were compiled from existing collections in the Jemez Mountains. The data cover approximately 300,000 ha of historically dry conifer forests that used to burn predominantly with low-severity fire. This network, the largest in North America for a single mountain range, is a compilation of 19 studies conducted over 40 years (Text S4). A total of 8588 fire scars from 1295 trees were dated to the year (1599—1899). Fire seasonality was determined for 77% of the scars ($n = 6581$) from the position of the scar within the annual ring. Categories for scar positions and their seasonal timing include: dormant (D—early spring); early, mid, and late earlywood (E, M, L—late spring through mid-summer); and latewood (A—late summer and fall). Most fire years historically had scars in multiple fire seasons (Fig. 2 and S2). Details of the fire scar seasonality methods are described in the supplementary materials (Text S1).

Fire scar data were compiled and analyzed with the “burnr” fire history package in R (Malevich et al. 2015; R Core Team 2015). Percent of recording trees scarred was used as a proxy for relative fire size (e.g., Farris et al. 2010). Fires recorded by a single tree were not included in the analysis. After 1899, the number of fires in the Jemez Mountains declines precipitously due to increased human land use, so the common period for the fire and climate data is 1599–1899. Native Americans influenced fire regimes through the mid-1600s in the southwest Jemez Mountains (Swetnam et al. 2016), which could affect fire-climate relationships in the early part of the record.

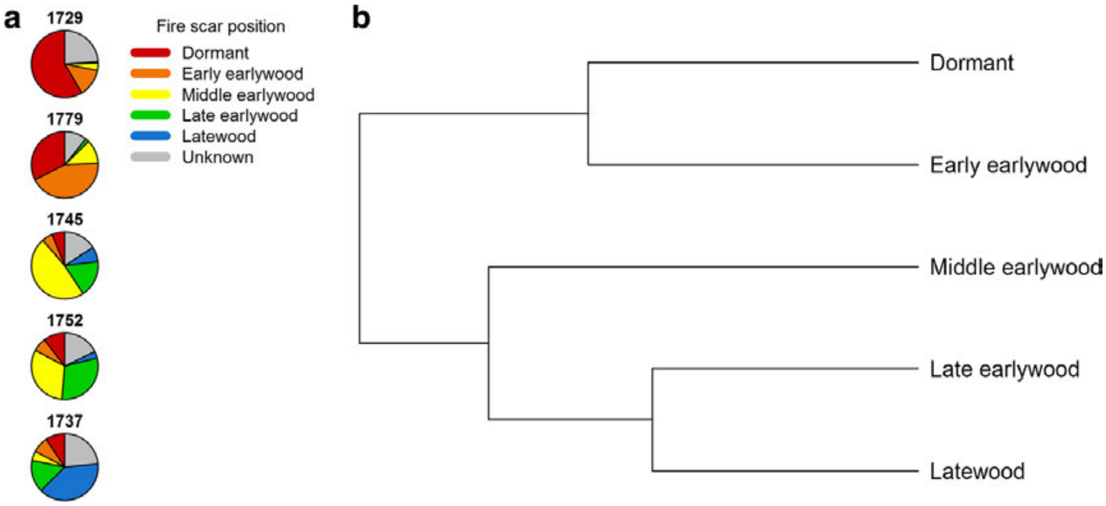

Fig. 2 **a** The proportion of trees scarred by fire in the Jemez Mountains in each fire scar position, or season, for five large fire years. The selected years have the largest number of fire scars in the spring dormant (1729) through late summer/fall latewood (1737) fire seasons in the 1700s. Note the inter- and intra-annual variability in the distribution of fire seasonality. **b** Cluster dendrogram of large, 95th percentile fire years by fire-scar position (n = 16 years for each scar position). Note the grouping of dormant and early earlywood (DE) fire years and middle earlywood, late earlywood and latewood fire years (MLA)

Analyses focused on the extreme fire years. Extreme large and small fire years were determined by the 95th and 5th percentile rank of the percent of recording trees scarred in a year (Table S2). Extreme fire years were first determined for all fires (combining all fire seasons, including unknown seasonality), and then for each of the five individual fire scar seasonalities (spring through fall). A total of 16 fire years fell within the 95th percentile (Fig. S2). The 5th percentile years were all years when no fires occurred.

3 Analysis methods

3.1 Multi-seasonal fire-climate analysis

We used superposed epoch analysis (SEA) to test whether fire occurrence and fire seasonality were associated with cool- and monsoon-season SPEI anomalies (Swetnam 1993). SEA is a compositing approach that uses block re-sampling and bootstrap simulations to evaluate the significance of the concurrence between fire event years and wet or dry conditions in the event year or lagged years. We examined 7-year blocks of cool- or monsoon-season SPEI spanning 4 years before, and 2 years after the fire year (year zero). We first used SEA to test whether cool- and monsoon-season SPEI anomalies were associated with all extreme large fire years and no fire years, and then for SPEI associations with the separate individual fire seasons.

To determine associations among the different fire scar positions, as well as relationships between fire-scar positions and seasonal climate, we used hierarchical cluster analysis of extreme large fire years for all five individual fire scar positions (hclust; R Core Team 2015). The analysis includes all possible combinations, not just adjacent scar positions. The groups that resulted from the cluster analysis were used as a framework for combining multiple fire scar positions for analyzing the relationships between sequences of cool- and monsoon-season moisture and the related fire scar positions, as well as the drought-fire analysis.

3.2 Drought-fire analysis

Reconstructed cool- and monsoon-season SPEI series were first analyzed to investigate characteristics of seasonal drought. This included the number and length of droughts (single and consecutive years with negative SPEI values) and comparisons of these metrics between cool- and monsoon-season droughts. The relationships between droughts and large fire years in the early (D and E) and mid-to-late (M, L, and A) fire seasons—as grouped by the cluster analysis—were then examined to determine (1) the length of droughts in which the large fires occurred and (2) the year in the drought that large fires occurred. On the basis of the SEA results, early-season fires were evaluated with cool-season droughts, and mid- to late-season fires were evaluated with both cool- and monsoon-season droughts.

We also assessed whether the driest decades of the cool- and monsoon-season SPEI reconstructions were associated with increased fire. Here, we relax the threshold for fires to include those with at least 2.5% of trees scarred (74th percentile, $n = 79$ fire years for early-season fires and 85th percentile, $n = 46$ fire years for mid- to late-season fires). Decadal dry periods were identified as the five driest non-overlapping decades for each climate season. Decades with the highest fire activity for early and mid- to late-season fires were defined as the five non-overlapping decades with the largest sum of the percent of recording trees scarred. These decadal measures of climate and fire were compared visually to assess the correspondence between the most active fire periods and the driest periods.

4 Results

4.1 Cool-season climate associated with large fire years

The SEA analysis for the largest fire years, regardless of fire season, highlights the importance of cool-season drought during the fire year (Fig. 3a, top row). The largest fire years were also associated with wet cool seasons 2 and 3 years prior to the fire year. No significant associations were found between all large fires and monsoon-season moisture, although a similar pattern of dry conditions during the fire year preceded by wet years is suggested (Fig. 3b, top row). Years without fire were associated with wet cool seasons in the fire year, but not with monsoon moisture.

When the largest fire years for each fire season are analyzed, several different fire-climate relationships are revealed. The SEA results indicate that early season (D and E) fires are most strongly associated with cool-season drought during the fire year, with the strength of the association decreasing by mid-summer through fall (Fig. 3a). Similarly, the importance of prior wet cool seasons associated with large fire occurrence decreases through the fire season; spring (D) fires are associated with two prior wet cool seasons 2 and 3 years before the fire year; early- to mid-summer fires (E, M, and L) are associated with one wet cool season 2 or 3 years prior to the fire year; and late-summer and fall (A) fires have no significant relationship with prior wet cool seasons.

4.2 Monsoon-season climate associated with large fire years

Monsoon-season drought during the fire year is significantly associated with large late season (L and A) fire occurrence (Fig. 3b). There is a suggestion of a similar relationship with the

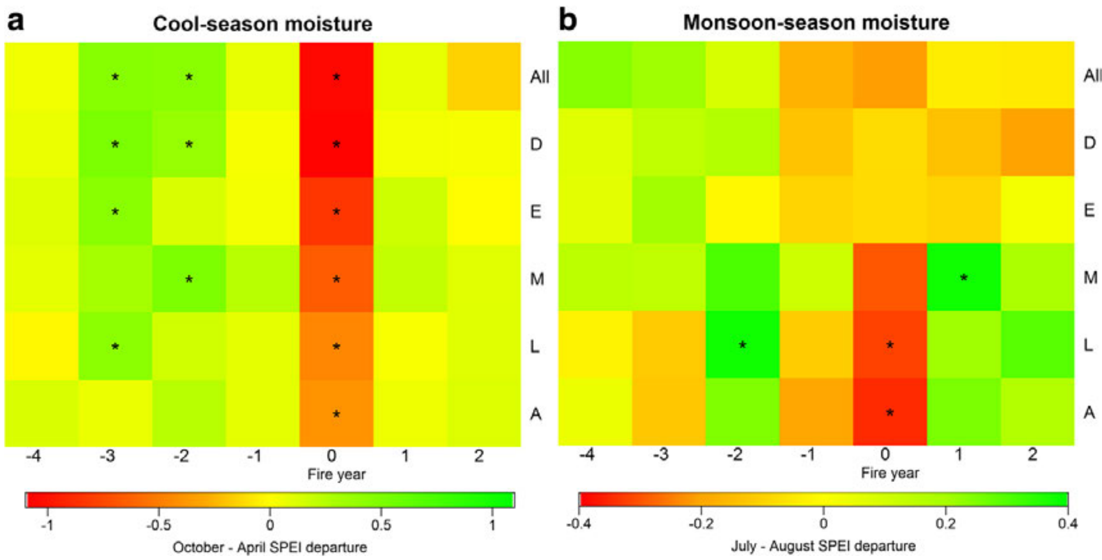

Fig. 3 Superposed epoch analysis of **a** cool-season moisture and **b** monsoon-season moisture by fire-scar position for large, 95th percentile fire years in the Jemez Mountains (n = 16 fire years for each seasonality, 1599–1899). All = all fire scar positions, D = dormant, E = early earlywood, M = middle earlywood, L = late earlywood, and A = latewood. SPEI = standardized precipitation-evapotranspiration index. *Asterisks* in cells denote significant departures from mean SPEI based on bootstrap simulations ($p < 0.05$)

monsoon and mid-season (M) fires. Since late-season fires are also associated with cool-season drought during the fire year, joint drought in the cool and monsoon seasons appears important for widespread late-season fires. Wet conditions in the cool and monsoon seasons 2 or 3 years prior to the fire year also appear to be important in this sequence favoring late-season fires. There is no significant association between monsoon moisture and D or E fires.

4.3 Fire seasonality patterns

The cluster analysis of the fire-scar seasonality of large fire years supports results from the SEA. There were two main groups of fire scar positions: (1) dormant and early earlywood (DE) and (2) middle earlywood, late earlywood, and latewood (MLA) (Fig. 2b). The patterns of fire-climate relationships from the SEA suggest a similar grouping of M, L, and A fires, particularly in association with monsoon moisture (Fig. 3b). This implies different climatic controls on spring and early summer (DE) fires compared with the mid-summer to fall (MLA) fires. Large early-season fires, by virtue of their timing, are strongly linked to cool-season drought, and they rarely continue to burn throughout the summer (Fig. 2 and S2). Whereas the largest MLA fires burn through the summer under dry monsoon conditions and consequently are associated with drought in both the cool and monsoon seasons. Mid-summer (M) fires are most likely to continue burning through the late summer and fall (L or A scars), but only during dry monsoons (e.g., differing fire-scar position distributions of the 1729 dormant fire year compared to the 1745 middle earlywood fire year, Fig. 2 and S2).

4.4 Relationships between persistent drought and fire

Results from the SEA and cluster analysis suggest that a sequence of both wet and dry years in the cool and monsoon seasons lead to large fires. Thus, a short-term drought might be most favorable for fire, while persistent (multi-year) droughts that do not include intervening wet

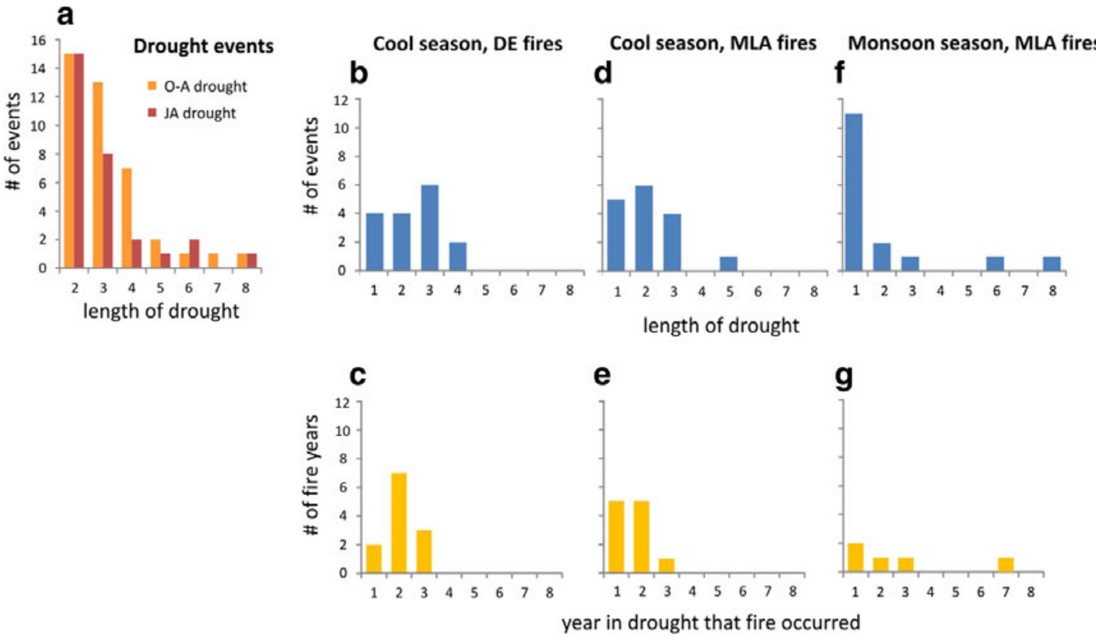

Fig. 4 **a** Numbers and length of droughts (consecutive years of negative SPEI) for October–April (*orange*) and July–August (*brown*) SPEI, 1599–1899. **b**, **d**, and **f** Lengths of the droughts in which the largest fire years occurred, and numbers of fire years corresponding to each drought length for cool season DE fires, cool season MLA fires, and monsoon season MLA fires. **c**, **e**, and **g** The year within the drought in which the fire occurred

conditions could inhibit large fires. The SPEI drought analysis revealed differences in the distributions of drought lengths between the cool and monsoon seasons (Fig. 4a). Single dry years are more common in the monsoon and multi-year droughts occur more frequently in the cool season. There were 25 cool-season droughts of 3 years or more, compared to 14 for the monsoon (Table S3). The longest cool-season droughts lasted up to 8 years. In order to explore the relationship between fire occurrence and drought length, we examined when large fires occurred relative to multi-year cool- and monsoon-season droughts.

Although there are cool-season droughts lasting 5 to 8 years, none of the largest early-season (DE) fires occur during these persistent droughts (Fig. 4). Of the 16 largest early season fire years, ten occurred within a 2- to 3-year cool-season drought, and two within a 4-year cool-season drought (Fig. 4b). The remaining four large early season fire years occurred during single-year cool-season droughts. Within a multi-year cool-season drought, fires only occurred in the first 3 years and primarily in the second year of the drought (Fig. 4c). This result generally supports the SEA, which indicates that conditions most strongly linked to large early-season fires include a wet year 2 or 3 years prior to the fire year, but not the year prior to the fire year.

Relationships between persistent cool-season droughts and the largest mid- to late-season fires (MLA) are similar to the early-season fires. Most of the mid- to late-season fires occur during droughts of 2 or 3 years (Fig. 4d). Almost all large mid- to late-season fires occur in the first 2 years of a cool-season drought (Fig. 4e).

Similarly, persistent monsoon-season drought was not related to large fire occurrence. Only 31% (5 of 16) of the large mid- to late-season fire years occurred during persistent monsoon droughts (Fig. 4f). These monsoon droughts lasted 2 to 8 years. All but one of these large fires occurred within the first 3 years of the persistent drought (Fig. 4g). Results for the monsoon droughts may reflect the fact that, compared to the cool season, monsoon droughts are more likely to occur as single years (Fig. 4a).

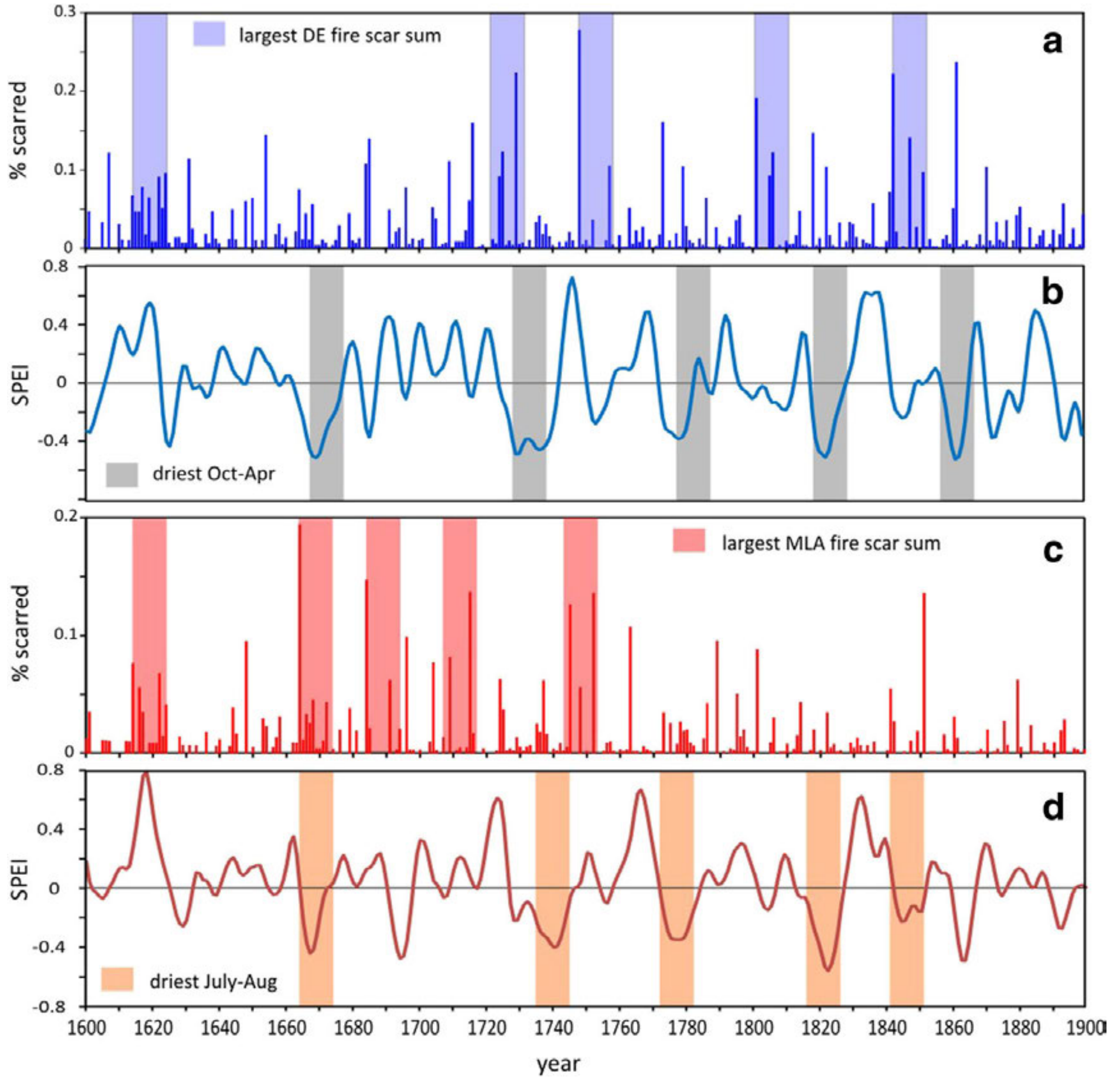

Fig. 5 **a** Percent of recording trees scarred by early-season (DE) fires in *dark blue bars*. *Light blue vertical bars* are the five non-overlapping decades with the largest sums of percent DE scarred trees. **b** October–April SPEI smoothed with a 10-year spline; *vertical bars* are the five non-overlapping decades with the lowest SPEI values. **c** Percent of recording trees scarred by mid- to late-season (MLA) fires in *dark red bars*. *Light red vertical bars* are the five non-overlapping decades with the largest sums of percent MLA scarred trees. **d** July–August SPEI smoothed with a 10-year spline; *vertical bars* are the five non-overlapping decades with the lowest SPEI values

When the driest decades of both SPEI seasons were assessed, they were not consistently related to decades of high fire occurrence. For early season (DE) fires, the five decades with the largest sum of percent trees scarred—periods of widespread fire—had little correspondence with the driest decades of cool-season moisture (Fig. 5a, b). Since both cool- and monsoon-season drought appear to influence mid- to late-season fires (Fig. 3), we compared dry decades for both seasons with high MLA fire decades. These dry periods are distributed across three centuries (Fig. 5b, d), whereas the decades with the largest MLA fire scar sums are concentrated in the first half of the record (Fig. 5c). As with DE fires, there is little correspondence between the decades with the most widespread MLA fires and the driest decades of cool-season SPEI. The one exception is the mid-1660s to mid-1670s (Fig. 5b, c). However, when looking at monsoon moisture, two of the five driest decades do overlap with high mid- to late-season fire scar sums. The mid-1660s is unique, with widespread mid- to late-season fires coinciding with some of the driest decades in both seasons. This period includes the year with the highest percent of trees scarred in the mid-to-late fire season, 1664.

5 Discussion

5.1 Multi-seasonal climate associated with large fire years

The largest early-season fires tend to occur when wet cool seasons are followed by cool-season drought. Years without fires occur after wet cool seasons, with no influence from climate in prior years. These results emphasize the historical importance of cool-season moisture for promoting conditions conducive to large fires in the dry conifer forests of the Jemez (Touchan et al. 1996), the southwestern USA (Swetnam and Betancourt 1998), and the western USA (Swetnam et al. 2016). Modern studies confirm the importance of cool-season wet-dry oscillations in the cool season for fire occurrence across the western USA, but highlight regional differences. Cool-season drought is an important predictor of twentieth century area burned in northern or mountainous ecoprovinces across the western USA, whereas wet cool seasons in prior years are also important in drier ecoprovinces (Westerling et al. 2003; Littell et al. 2009).

Our results are the first documented effects of the NAM on fire occurrence prior to the twentieth century. Monsoon moisture has the greatest effect on mid- to late-season (M, L, and A) fires. The monsoon must be dry for these mid- to late-summer and fall fires to be widespread, as hypothesized by Grissino-Mayer and Swetnam (2000). Large late-season fires may also depend on cool-season conditions, such that dual-season drought preceded by dual-season wet conditions are important for large late-season fire occurrence. Modern studies indicate that prior-year NAM moisture was associated with fires in Arizona and the Great Basin (Westerling et al. 2003; Crimmins and Comrie 2004; Littell et al. 2009). In these studies, wet summers 1 and 2 years prior to the fire likely increased fine fuels, such as grasses, that were important for fire spread.

The intra-annual distribution of fire seasonality derived from tree-ring fire scars provides additional insights into the effects of the monsoon on fire seasonality. The largest early-season fires appear to burn until the onset of the monsoon (Fig. 2 and S2). This is consistent with modern fires in the region, many of which are extinguished by monsoon moisture. Historically, many of the largest late summer and fall fires appear to have occurred when dry monsoons allowed relatively small early-season fires to continue to burn into the summer and fall. This is indicated by all of the largest late summer and fall fires having some proportion of trees scarred in the early (DE) fire seasons (see distribution of fire scar positions for large latewood fires in Fig. 2 and S2). It is also possible that some large late-season fires may have ignited during a dry monsoon season. Multiple ignitions over the fire season could confound these interpretations.

5.2 Persistent drought and fire

Analysis of cool- and monsoon-season droughts and fire occurrence indicates that, overall, fires most often occur during the first or second year of multi-year droughts. Long droughts do not appear to promote large fires in the later years of the drought. The occurrence of all but one large fire in the first 3 years of a drought is not surprising (Fig. 4c, e, and g), but reinforces the importance of short droughts for fires in the region. This is further supported by the sequence of climate conditions leading to fires, which include a wet cool season several years prior to the fire. Because of the key role of wet cool seasons 2 and 3 years prior to a large fire year, and to a lesser degree in the monsoon season, prolonged drought may actually limit the occurrence of large fires in dry conifer forests. Once these dry forests burn, they need moisture to replenish surface fuels before the area can burn again.

The decades with the driest cool seasons were not consistently related to periods of high fire occurrence. These dry decades do not provide the necessary periodic wet conditions that precede the biggest fire years. Fitch and Meyer (2016) also found that extended dry periods in the Jemez Mountains, going back multiple millennia, did not necessarily correspond with increased fire activity, likely due to fuel limitations. While extremely dry winters are a necessary component for the most widespread fires, regardless of fire seasonality, if dry conditions persist beyond several years, the chances of widespread fire likely diminish. This result of persistent drought reducing fire occurrence in a fuel-limited ecosystem supports observations of the importance of biomass variability for modeling fire regimes globally and their response to climate change (Krawchuk et al. 2009).

Overall, these results suggest that the strongest climatic controls over fire regimes in the Jemez Mountains were seasonal and inter-annual to sub-decadal in scale. Decadal fire-climate relations were generally weak. This suggests that fine fuel biomass production (grasses, tree needles, and cones), which can respond to these short time-scale variations in climate, was likely the most important mechanism of climatic influence. It is probably not coincidental that the El Niño-Southern Oscillation (the key synoptic climate control over wet-dry oscillations in the southwestern USA), the phenological cycle of ponderosa pine (*Pinus ponderosa*) needle and cone production, and the frequency of surface fires, all typically occur over time scales of about 2 to 7 years (Maguire 1956; Swetnam and Betancourt 1990). That is, natural wet-dry oscillations might readily entrain inherent (and evolved) vegetative and reproductive cycles of flammable fuel production, which in turn promote synchronized, extensive surface fires.

5.3 Insights from the past for future fire regimes

Projecting fire response to climate change in semi-arid, biomass-limited regions is challenging, and future fire regimes will likely vary temporally in accordance with biomass availability. Climate-driven changes in vegetation will further confound forecasts of future fire regimes. Williams et al. (2015) suggest that future increased drought and moisture stress will increase fire occurrence in the southwestern USA, until fuel becomes limiting. Our results suggest that in the semi-arid southwestern USA, fuel was historically limiting in dry conifer forests and that persistent cool-season drought actually reduced fire occurrence. This differs from wetter, more productive mixed-conifer, aspen, and spruce-fir forests that are not fuel limited and where prior wet years are not associated with fire occurrence, only severe drought during the fire year (Swetnam and Betancourt 1998; Margolis and Swetnam 2013). Fine and heavy fuel loads in dry conifer forests have increased significantly over the last century due to fire exclusion (Fulé et al. 1997), although mega-fires in recent decades are beginning to reduce these overabundant fuels in portions of the landscape (Stephens et al. 2014). As warming continues to increase drought stress and increase large fire occurrence, some of the drier ecosystems in the region may move back toward being fuel-limited, with consequences for forecasting future fire regimes.

A major uncertainty for future fire regimes in fuel-limited systems is future moisture variability. Forecasting future precipitation is particularly complex in the southwestern USA, because of the two seasons of moisture. Projected extended drying in the region, due to reduced cool-season moisture (e.g., Seager and Vecchi 2010) would likely continue to increase fire occurrence in coming decades. However, as biomass becomes limiting, fire occurrence could ultimately decrease in dry forests and woodlands where fine-fuels are important for fire spread. A transition to a shortened or a weak NAM (e.g., Cook and Seager 2013) could extend the fire season in the southwestern USA through the summer and into the fall, which is currently rare, but consistent with the tree-ring record. Failed monsoons could represent the scenario with the greatest fire occurrence in the near term, before moisture stress from increased temperature supersedes any potential increases in precipitation (e.g., Williams et al. 2013), and biomass becomes increasingly limiting to fire occurrence.

6 Conclusions

We present the first in-depth, landscape-scale analysis of historical multi-seasonal climatic controls of fire size and seasonality using tree rings. Our findings suggest different seasonal climate controls on early season and mid- to late-season fires, but in both cases, sequences of wet and dry conditions are critical for preconditioning forests to burn. Dry conditions in the year of the fire—dry in the cool season for early-season fires, and dry in the monsoon season for late-season fires—are critical. Equally important are wet conditions, particularly in the cool season, 2 to 3 years preceding the fire year. The importance of this sequence of wet and dry years has key implications for relationships between fire activity and drought, and our results indicate persistent drought is not associated with the largest fires or periods of high fire activity in this region.

Our results suggest that as moisture stress increases in the southwestern USA due to warming (Seager et al. 2007; Williams et al. 2013), large fire occurrence may decrease in some fuel-limited ecosystems. Many model projections of global fire response to climate change use multi-decadal climate "normals" and lack inter-annual or intra-annual climate variability (e.g., Krawchuk et al. 2009; Moritz et al. 2012). We demonstrate that inter- and intra-annual climate variability is an important control for large fire occurrence and fire seasonality in a semi-arid, monsoon-affected region of southwestern North America. Accurate projections of inter- and intra-annual moisture variability will likely be important to accurately model future fire in the southwestern USA, particularly due to the bimodal precipitation regime and a likely future increase in biomass limitations on fire occurrence (i.e., requiring wet conditions to produce fuels to burn). In the future, in semi-arid regions such as the southwestern USA, prolonged droughts driven by warming could decrease fire activity due to biomass limitations.

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Research Article

Historical Stand-Replacing Fire in Upper Montane Forests of the Madrean Sky Islands and Mogollon Plateau, Southwestern USA

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ABSTRACT

The recent occurrence of large fires with a substantial stand-replacing component in the southwestern United States (e.g., Cerro Grande, 2000; Rodeo-Chedeski, 2002; Aspen, 2003; Horseshoe 2, Las Conchas, and Wallow, 2011) has raised questions about the historical role of stand-replacing fire in the region. We reconstructed fire dates and stand-replacing fire patch sizes using four lines of tree-ring evidence at four upper montane forest sites (>2600 m) in the Madrean Sky Islands and Mogollon Plateau of Arizona and New Mexico, USA. The four lines of tree-ring evidence include: (1) quaking aspen (*Populus tremuloides*) and spruce-fir age structure, (2) conifer death dates, (3) traumatic resin ducts and ring-width changes, and (4) conifer fire scars. Pre-1905 fire regimes in the upper montane forest sites were variable, with drier, south-facing portions of some sites recording frequent, low-severity fire (mean fire interval of all fires ranging from 5 yr to 11 yr among sites), others burning with stand-replacing severity, and others with no evidence of fire for >300 yr. Reconstructed fires at three of the four sites (Pinaleño Mountains, San Francisco Peaks, and Gila Wilderness) had stand-replacing fire patches >200 ha, with maximum patch sizes ranging from 286 ha in mixed conifer-aspen forests to 521 ha in spruce-fir forests. These data suggest that recent stand-replacing fire patches as large as 200 ha to 500 ha burning in upper elevation (>2600 m) mixed conifer-aspen and spruce-fir forests may be within the historical range of variability.

Keywords: fire history, mixed conifer, quaking aspen, spruce-fir, tree ring

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INTRODUCTION

The number and duration of large fires in the western United States has increased in recent decades due in part to increasing temperatures (Westerling *et al.* 2006). In the southwestern US (Arizona, New Mexico, and proximate areas), many of the recent large fires included large (100 ha to >1000 ha) high-severity fire patches, which raises questions about the historical role of stand-replacing fire in the region. Many of the recent stand-replacing fire patches in the southwestern US have occurred in the overstocked, mid-elevation ponderosa pine (*Pinus ponderosa* C. Lawson) and dry mixed conifer forests, where extensive stand-replacing fires are unreported in the documentary records prior to *circa* 1950 (Cooper 1960, Allen *et al.* 2002). However, in the upper elevation (>2600 m) mixed conifer-aspen and spruce-fir forests, historical photographs and tree-ring data from seral quaking aspen (*Populus tremuloides* Michx.) stands provide direct evidence that fires with large (100 ha to >1000 ha) stand-replacing patches occurred in parts of the region as recently as the early twentieth century (Abolt 1997, Romme *et al.* 2001, Margolis *et al.* 2007).

Relatively little is known about pre-EuroAmerican settlement fire regimes (size, severity, frequency, and seasonality) of upper elevation forests in the southwestern US (Grissino-Mayer *et al.* 1995, Fulé *et al.* 2003, Margolis *et al.* 2007, Margolis and Balmat 2009). Extensive fire histories from upper montane and subalpine forests of southern Wyoming, Colorado, and northern New Mexico indicate that infrequent (>100 yr intervals) stand-replacing fire is a dominant disturbance in upper elevation forests of the southern Rocky Mountains (Kipfmueller and Baker 2000, Sibold *et al.* 2006, Margolis *et al.* 2007). Thus, it is logical to hypothesize that upper elevation mixed conifer-aspen and spruce-fir forests of the southwestern US outside of the southern Rocky Mountains potentially had a historical fire regime that included infrequent, relatively large (>100 ha) patches of stand-replacing fire.

Reconstructing Stand-Replacing Fire

Age-structure-based methods for reconstructing fire history were developed in coniferous subalpine and boreal forests of North America where stand-replacing fire regimes are dominant (Clements 1910, Heinselman 1973, Agee 1993, Johnson and Gutsell 1994). By definition, stand-replacing fires leave few or no surviving trees to record direct evidence of those fires within the highest burn severity patches (but note that fire-scarred survivors can sometimes be found on the edges of such patches; e.g., Margolis *et al.* 2007). Post-fire tree cohorts, assumed to have established soon after the fire, are the most common type of evidence used to date and map stand-replacing burns. In the Rocky Mountains, the assumption that there is typically rapid recruitment of a post-fire cohort (i.e., <5 yr) within stand-replacing burn patches is well supported in the case of quaking aspen, because it has evolved mechanisms for rapid regeneration, and has been commonly observed to do so following fires (Clements 1910, Patton and Avant 1970). Post-fire cohort evidence (dates and mapped perimeters) can be combined with the relatively rare direct conifer evidence of fire (e.g., fire scars, tree death dates, ring-width changes or traumatic resin ducts) to reconstruct annually resolved stand-replacing fire dates (Johnson and Gutsell 1994, Margolis *et al.* 2007).

In the current study, we separate the upper elevation forest into mixed conifer-aspen (2600 m to 3100 m) and spruce-fir (>3100 m) because of differing fire ecology, and potentially different fire regimes and use of differing fire history methods. Age-structure-based fire history methods in mixed conifer-aspen forests have been applied in a few studies in the southwestern US, primarily focusing on quaking aspen regeneration dates as a proxy for stand-replacing fire (Abolt 1997, Romme *et al.* 2001, Margolis *et al.* 2007). Romme *et al.* (2001) reconstructed a 140-year stand-replacing fire rotation period from aspen stand age in the La Plata Mountains of southwestern Colorado. They noted that the lack of fire-scarred trees in aspen stands was a limitation to dating past fires. Abolt (1997) used coincident aspen pith dates and conifer fire scars from lower elevations to date stand-replacing fire patches in mixed conifer forests of the Mogollon Mountains of southwestern New Mexico. Margolis *et al.* (2007) combined four lines of tree-ring evidence (aspen age structure, conifer fire scars, conifer death dates, and conifer injury dates) to reconstruct synchronous, drought-related stand-replacing fire dates and patch sizes from aspen stands embedded in upper montane mixed conifer and spruce-fir forests at a network of twelve sites in the upper Rio Grande Basin (New Mexico and Colorado). These studies indicate that, because of the unique fire ecology of quaking aspen (i.e., high sensitivity to being killed by fire and ability to re-sprout), the age structure from seral aspen stands is a potential indicator of historical stand-replacing fire in upper elevation forests in the southwestern US.

Fewer studies have evaluated the effectiveness of age-structure-based fire history methods in southwestern US spruce-fir forests. In the Pinaleño Mountains, Arizona, Grissino-Mayer *et al.* (1995) used intensive, but spatially limited, age structure sampling in spruce-fir forests, combined with lower elevation fire scars, to hypothesize that the spruce-fir zone regenerated following a stand-replacing fire. Due to limited spatial coverage of the sampling, stand-replacing fire area was not estimated. Fulé *et al.* (2003) used fire scars, tree age and species, and spatial patterns of forest stands to reconstruct fire-initiated tree groups at the plot scale (20 m × 50 m), which likely originated after severe eighteenth century fires in high-elevation forests (including aspen and spruce-fir) on the north rim of the Grand Canyon, Arizona. They were not able to identify distinct fire-created stands in the study area from aerial photos or satellite data, which differs from the stand-replacing fire history methods used in the Rocky Mountains. In the Santa Fe Watershed, New Mexico, Margolis and Balmat (2009) combined a systematic spatial grid sampling of spruce-fir age structure with conifer ring-width growth changes and conifer fire scars to conclude that approximately 90 % of the spruce-fir zone (1200 ha) regenerated following stand-replacing fire. These studies provide evidence of past stand-replacing fires in spruce-fir forests in the southwestern US, but leave questions about patch sizes, variability between sites, and the ability to apply fire history methods from other regions and forest types.

Fire Patch Size and Severity

Fire patch size and severity have strong influences on the ecological effects of fire on terrestrial and aquatic systems. Stand-replacing fire patch size is a key determinant of post-fire vegetation composition and structure (Agee 1993, Turner *et al.* 1994, Turner and Romme 1994). Following the extensive (>250 000 ha) fires in Yellowstone National Park, Wyoming, in 1988, the size and severity of burn patches were shown to affect overall plant cover, tree seedling recruitment, and herbaceous recruitment (Turner *et al.* 1994). High-severity fires remove overstory vegetation and ground cover that dramatically affects watersheds and water resources by altering the important processes of evapotranspiration, interception, surface flow, and subsurface flow (Swanson 1981). The size of high-severity fire patches is important in determining the probability of fire-induced flooding or debris flows (Pearthree and Wohl 1991, Cannon and Reneau 2000). Recent, large stand-replacing fires in the southwestern US have produced runoff and erosion events as much as two orders of magnitude greater than pre-fire conditions (Veenhuis 2002).

High-severity (stand-replacing) fire patches are usually part of a "mosaic" of burn severities, within fire perimeters that include moderate- and low-severity surface fire patches, as well as unburned patches (Turner and Romme 1994). For example, less than half of the 1988 Yellowstone fires burned with high severity (Turner *et al.* 1994). Reconstructing the complex spatial patterns and wide range of burn severities of pre-twentieth century fires at high resolution (i.e., less than a few hectares) is not possible. However, the largest stand-replacing fire patches often leave a persistent and identifiable legacy in the form of tree ages and, less commonly, as conifer death dates, conifer fire scars, and tree-ring growth patterns in conifers injured by the fire. From these legacies, stand-replacing fire patch sizes and dates can be reconstructed and compared with recent fires even if overall size (extent) of the entire fire is unknown.

Research Objectives

Our primary objective was to use dendroecological methods to expand the upper elevation stand-replacing fire history network of Margolis *et al.* (2007) to four new sites in mixed conifer-aspen forests (2600 m to 3100 m elevation) in the Mogollon Plateau and Madrean Sky Island regions of the southwestern US, focusing on quaking aspen as a potential indicator of the dating and patch size of past stand-replacing fires. The secondary objective was to test the utility of using spruce-fir forest age structure to expand the reconstruction of stand-replacing fires above the local elevation range of quaking aspen (>3100 m) at two test sites. We did not attempt to reconstruct a complete inventory of all historical stand-replacing fire patches at these four sites; rather, we mapped and dated the largest and potentially most ecologically significant patches.

METHODS

Study Area

To expand the existing southwestern US network of upper elevation stand-replacing fire history sites of Margolis *et al.* (2007) beyond the upper Rio Grande Basin, we selected two sites on the Mogollon Plateau and two sites from the Madrean Sky Islands (Figure 1, Table 1). The sites were selected based on the presence of the largest seral aspen stands, which potentially represented historical stand-replacing fire patches. We used the regional gap analysis program vegetation map, USDA National Forest vegetation maps, black and white and color infrared digital ortho-rectified quarter-quadrangle photographs (DOQQs) and field surveys to map and verify the largest aspen patches on the Mogollon Plateau and Madrean Sky Islands on US Forest Service land. We set the minimum aspen patch size threshold at 5 ha to eliminate smaller patches. We targeted seral aspen stands embedded within conifers to eliminate self-replacing aspen and aspen within high-elevation parklands that likely experienced frequent surface fires (Jones and DeByle 1985).

The largest potential post-stand-replacing fire aspen patches on the Mogollon Plateau were in the San Francisco Peaks (SFP) and the Mogollon Mountains (Gila Wilderness, GIL; Table 1 and Figure 2). On the Mogollon Plateau, we chose GIL as our test site for age-structure-based fire history methods in spruce-fir (>3100 m) because the patches were smaller than at SFP and required less sampling. In the Sky Islands, the Chiricahua Mountains (CHI) and the Pinaleño Mountains (PIN) had the largest potential historical post-stand-replacing fire aspen patches (Figure 2). At PIN, aspen was not present in homogeneous patches; rather, aspen stems were scattered throughout the mixed conifer forest, potentially representing older stand-replacing fire patches that had infilled with conifers. The PIN contains the only spruce-fir forest in the Sky Islands, which we used as the second test site for spruce-fir fire history methods.

Mean elevation of the study sites was 2982 m and tree-ring samples were collected between 2694 m and 3257 m (Table 1). All sites are managed as US Forest Service wilderness areas except PIN, which is closed to the public to protect the endangered Mount Graham red squirrel (*Tamiasciurus hudsonicus grahamensis*). We did not see evidence of logging (e.g.,

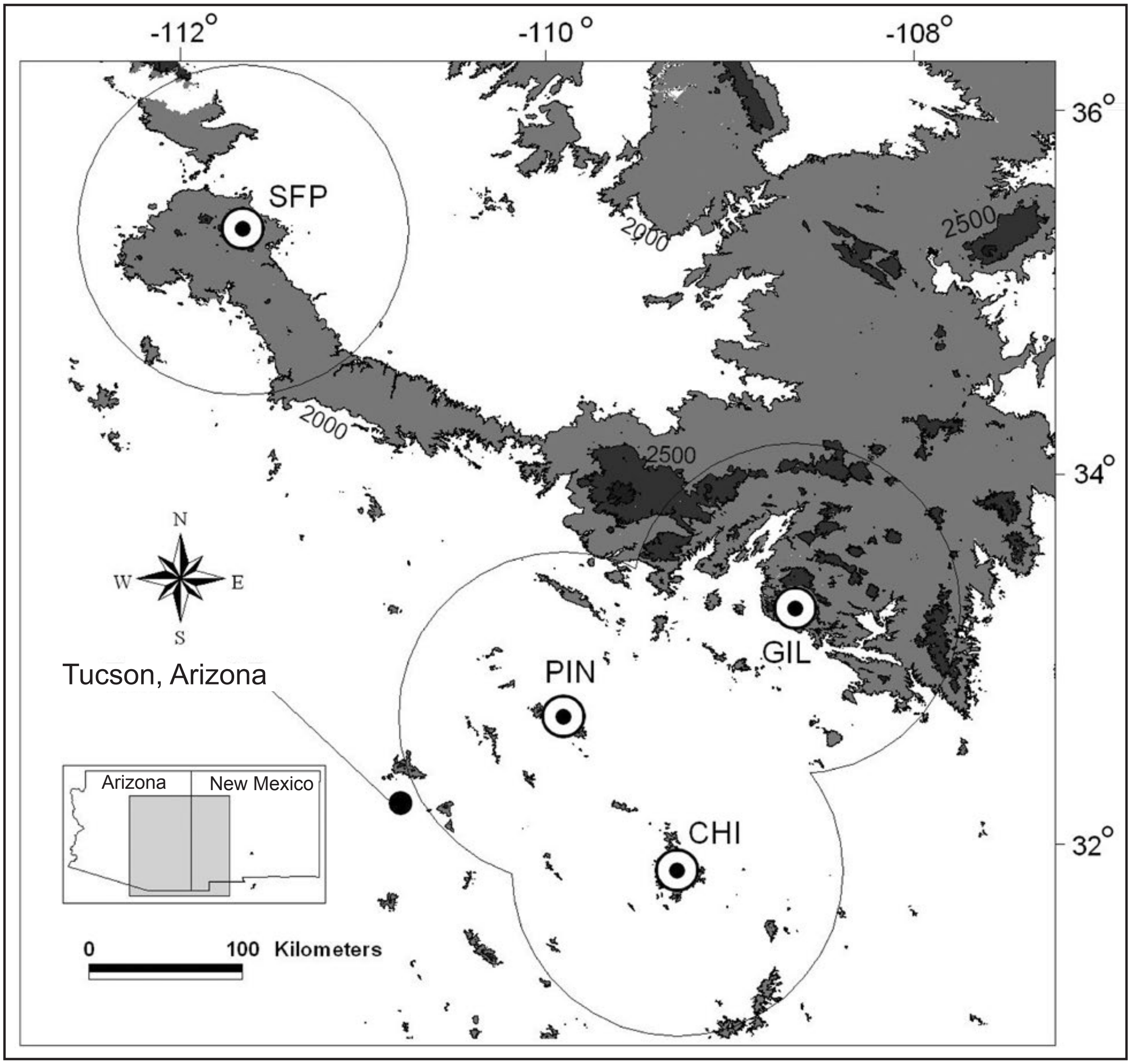

Figure 1. Map of site locations (e.g., SFP) in the Mogollon Plateau and the Madrean Sky Islands of Arizona and New Mexico, USA. Shading indicates major topographic features >2000 m in elevation at 500 m intervals. Large circles indicate the 100 km search radius around the fire history sites used to select recent fires (1984 to 2008) to quantify the size of recent stand-replacing fire patches.

Table 1. Site information for four upper elevation fire history sites from the Mogollon Plateau and Madrean Sky Islands, USA.

Site ID	Site name	Vegetation type[a]	Sampled aspen area (ha)	Sampled spruce-fir area (ha)[b]	Number of plots	Mean sample elevation (m)
CHI	Chiricahua Mountains	MC/S	139	--	26	2856
GIL	Mogollon Mountains	MC/SF	744	1639	32	3060
PIN	Pinaleño Mountains	MC/SF	0*	521	33	3057
SFP	San Francisco Peaks	MC/SF	990	--	25	2954

[a] MC = mixed conifer-aspen, SF = spruce-fir, S = spruce
[b] Spruce-fir was only mapped and sampled at two test sites (GIL and PIN).
* Distinctive seral aspen patches greater than 5 ha were not present.

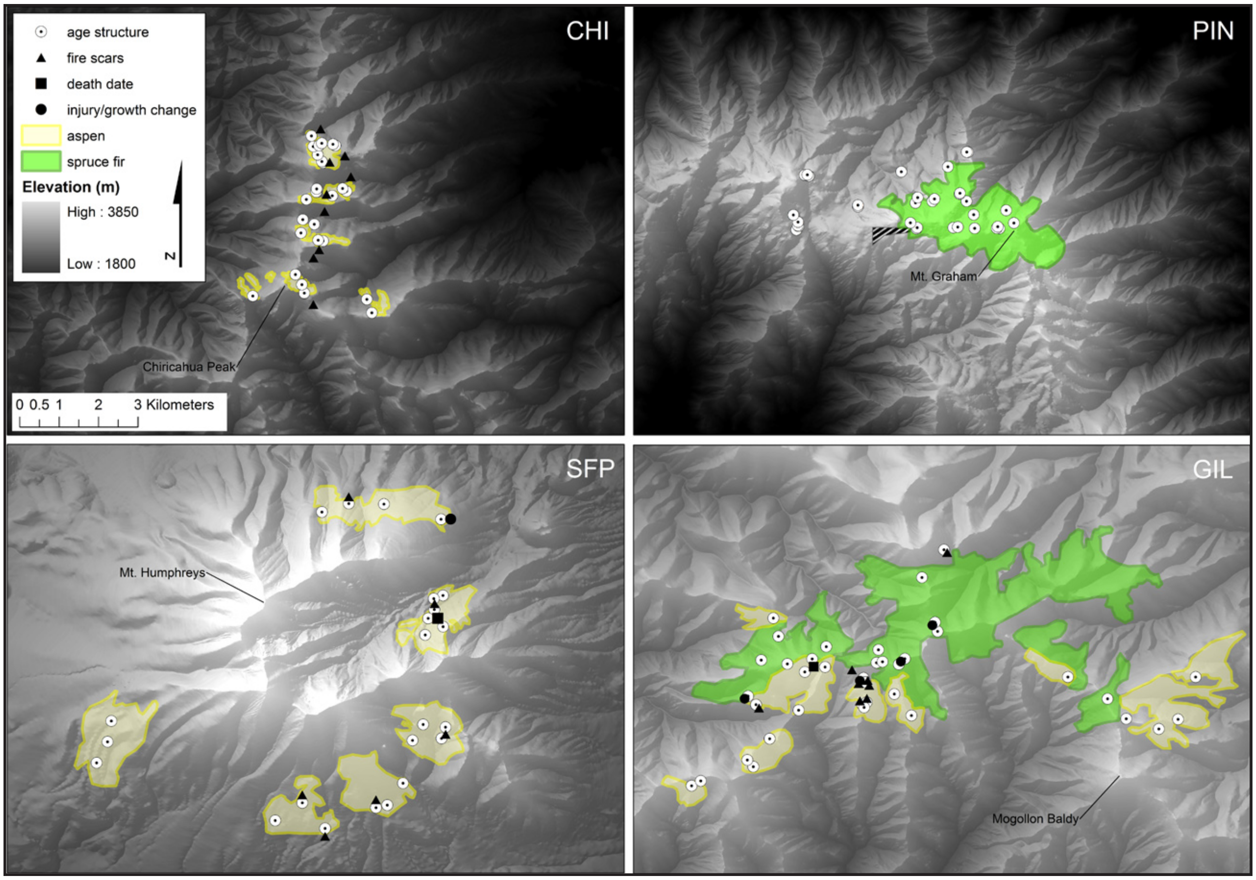

Figure 2. Tree-ring sample locations and analyzed aspen and spruce-fir stands at the study sites in the Chiricahua Mountains (CHI), Pinaleño Mountains (PIN), San Francisco Peaks (SFP), and Gila Wilderness (GIL) of the Mogollon Mountains. Hatched polygon at PIN indicates fire scar sample area from Grissino-Mayer *et al.* (1995).

stumps or skid trails) within the sampled stands. Fire exclusion resulting from late nineteenth century grazing followed by twentieth century fire suppression occurred at all sites, similar to most montane forests in the southwestern US (Dieterich 1980, Bahre 1985, Swetnam and Baisan 1996, Allen *et al.* 2002).

The general climate of the study area is continental with a bimodal precipitation regime. All sites receive an average of 40% to 50% of annual precipitation from summer (July to September) monsoon convective thunderstorms (1910 to 2009; http://www.prism.oregonstate.edu/). Average annual precipitation was similar amongst sites, ranging from 800 mm to 950 mm. Average annual maximum temperature ranged from 12.5 °C to 17 °C and minimum temperature ranged from 0° C to –4.5° C (1910 to 2009; http://www.prism.oregonstate.edu/). All sites receive winter snow, but snowpack varies widely from year to year depending on the winter storm track. The majority of area that burns in the study area occurs during a consistently dry and warm pre-monsoon period that begins in April or May and lasts through June (Barrows 1978). The potential severity and length of the fire season in the high-elevation forests of the region is largely a function of the snowpack and residual moisture that persists into the early summer pre-monsoon period.

The sampled seral quaking aspen stands at all four sites were located adjacent to mixed conifer or spruce-fir forests. The following conifer tree species were observed within and adjacent to the aspen stands, listed in descending order of occurrence: Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), southwestern white pine (*Pinus strobiformis* Engelm.), white fir (*Abies concolor* [Gord. & Glend.] Lindl. Ex Hildebr.), subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.), ponderosa pine, and Rocky Mountain bristlecone pine (*Pinus aristata* Engelm.).

Although all sites contained quaking aspen, there were differences between and within sites. Aspen patches in the two Sky Island sites were smaller than on the Mogollon Plateau (Table 1, Figure 2). This pattern can be partially explained by less land area in the aspen zone (2600 m to 3100 m) at the Sky Island sites (2927 ha in CHI, and 5945 ha in PIN) compared to Mogollon sites (7088 ha in SFP, and 7645 ha in GIL). Within-site differences in vegetation that could affect fire regimes were driven by aspect, with south-facing slopes containing drier, more open forests, and north-facing slopes generally supporting more mesic, denser forests.

Stand-Replacing Fire History Methods—Mixed Conifer-Aspen Forest

Our general sampling methods follow Margolis *et al.* (2007), in which large quaking aspen patches embedded in mesic mixed conifer and spruce-fir forests of the upper Rio Grande Basin were mapped and tree-ring dated with multiple lines of evidence to reconstruct stand-replacing fire patch sizes and dates. The four lines of evidence included 1) quaking aspen age structure, 2) conifer death dates, 3) conifer traumatic resin ducts or ring-width changes, and 4) conifer fire scars. All conifer death dates were bark-ring dates. Bark-ring dates indicate that either bark or other evidence of an intact outer ring (e.g., insect galleries) was present on the samples—this ensures that the outer ring dates are actual tree death dates.

Age structure plots were randomly located within each mapped aspen patch at a minimum density of three to four plots per 100 ha (e.g., SFP in Figure 2). Aspen patches were visually surveyed in the field to ensure plot locations were representative of the stand. Additional plots were added in the field at locations with conifer evidence of fire to verify stand boundaries, or to age potentially older trees (fire survivors) indicated by anomalously large diameter.

Aspen age structure plots had a 10 m fixed radius. Within the plots, we cored the two aspen stems with the greatest diameter at breast height (dbh). Trees were cored at <0.3 m core height until the pith was present in one sample at the plot. In a post-stand-replacing fire aspen stand, sampling two stems per plots at multiple plots within a patch has been shown to be sufficient to determine stand age (Margolis *et al.* 2007). This is because of the immediate asexual regeneration response of aspen following aboveground stem mortality, which creates a distinct recruitment pulse and a single-tiered, even-aged stand (Barnes 1966, Patton and Avant 1970). In upper montane seral aspen stands, subsequent regeneration is relatively rare and the dominant post-fire cohort is easily identified as the stems with largest dbh (Margolis *et al.* 2007). A more intensive sampling design would be necessary to fully describe a multi-cohort age structure, but this was not our goal. Post-fire quaking aspen regeneration can grow up to 1 m in the first year of growth (Jones 1975); thus, <0.3 m core height seems adequate to capture the first year of the aspen regeneration pulse (Margolis *et al.* 2007).

We searched within aspen patches and along the patch boundaries for conifers with potential direct evidence of fire (e.g., fire scars, conifer death dates, and ring-width changes and injuries). Cross sections and partial cross sections were collected with handsaws from remnant conifer logs, living trees, and standing dead snags with intact outer rings. Increment cores were collected from potentially injured live conifers without basal scars. Potential evidence of fire injury included char, scars on the undersides of branches, elevated crown base height, and unilateral loss of branches. Fire scars were not collected at PIN due to the existing fire scar collection located within our study site (Grissino-Mayer *et al.* 1995).

All tree-ring samples were prepared and crossdated according to standard dendrochronological procedure (Stokes and Smiley 1968). To estimate the date of the first year of growth (pith) for age structure increment cores that did not contain the pith ring, we used a concentric circle pith estimator (Applequist 1958). Dates from the four lines of tree-ring evidence were plotted together to determine fire dates (from conifer fire scars, death dates, and tree-ring growth changes and injuries) and stand-replacing fire patches (from age structure of aspen patches).

A mapped aspen patch was determined to represent the minimum extent of a previous stand-replacing burn patch if: 1) the oldest aspen estimated pith dates were associated with (≤5 years following) a fire event recorded by conifer death dates from within the patch or fire scars on surviving trees along the periphery of the patch, and 2) estimated aspen pith dates were part of a site-level (i.e., multi-patch) aspen recruitment pulse. The rarity and poor spatial coverage of fire-scarred trees at some sites (e.g., $n = 6$) prevented the use of percent-scarred filters to categorize and compare relatively widespread versus local fires between sites (Swetnam and Baisan 2003). Instead, we categorized fires recorded by ≥5 conifer samples at a site (e.g., conifer death dates, growth changes or traumatic resin ducts, and fire scars) as likely being more widespread than fires recorded by fewer trees.

Testing Fire History Methods in Spruce-Fir Forest

Within our study area, potential post-stand-replacing fire quaking aspen patches were generally found between 2600 m and 3100 m, and forests above 3100 m were generally dominated by spruce and fir. Pure spruce-fir forests with no living aspen stems would not be expected to contain quaking aspen regeneration following fire, so above 3100 m in this region, past stand-replacing fire patch size and dates cannot be estimated using post-fire aspen patches. We tested the utility of age structure fire history methods in spruce-fir forests at two sites, PIN and GIL. These sites were chosen because the relatively small size of the spruce-fir patches was more manageable for testing the efficacy of the methods. Therefore, the extensive spruce-fir stands at SFP were not sampled.

Aerial photographs and field observations were used to map spruce-fir patches and identify differences in texture, density, color, or differences in tree height, potentially representing fire boundaries (Johnson and Larsen 1991, Agee 1993, Johnson and Gutsell 1994). We were not able to identify any evidence of potential fire boundaries (e.g., discrete changes in canopy height) within the spruce-fir stands at PIN or GIL. Therefore, we treated each spruce-fir stand as a single potential stand-replacing fire patch.

In contrast to the predominance of asexual reproduction in aspen, spruce and fir trees recruit from seed, so the initial post-fire cohort can lag behind the fire date and may be distributed over decades (e.g., Antos and Parish 2002). Subsequent cohorts of these shade-tolerant conifers are able to regenerate under the canopy of the initial post-fire cohort. This multiple-aged structure makes the initial post-fire cohort in spruce-fir more difficult to identify with age or size structure data. We collected age structure samples using similar methods for dating aspen patches (see above), but with two differences to account for the differing fire ecology. First, we doubled the number of trees cored at each plot to include the four trees of largest dbh in order to account for the potentially complex age structure. Conifers were cored as low on the bole as possible and angled down to intersect the root crown and capture the earliest years of growth.

The second difference from the aspen age structure methodology was in the criterion to qualify as a stand-replacing fire patch. A spruce-fir patch was determined to be a post-stand-replacing fire patch if the oldest estimated conifer pith dates were ≤10 years following a fire recorded by conifer death dates from within the patch or by fire scars on the periphery of the patch. We increased the cut-off criteria to 10 yr (compared to 5 yr for aspen) to account for potentially lagged seedling recruitment (compared to immediate asexual regeneration in aspen). A 10-year lag window is likely conservative, given reports of greater than 50-year lags for subalpine forest regeneration following stand-replacing fire (Stahelin 1943). Because of relatively high fire frequency recorded by fire scars in some of the mixed conifer forests immediately below the spruce-fir stands, we determined that a 10-year lag would help to avoid spurious matches between fire scar dates and age structure that could be interpreted as stand-replacing fire dates. All tree-ring samples were collected in 2003 and 2004.

Historical Stand-Replacing Fire Patch Size

The aspen and spruce-fir patches that were dated to historical stand-replacing fires were used to derive minimum estimates of historical stand-replacing fire patch sizes. Patch area was calculated with a geographic information system (GIS). This data set provides the first estimate of historical stand-replacing burn patch sizes within two elevation and vegetation ranges at our study sites, including: 1) the aspen zone (2600 m to 3100 m) and 2) the spruce-fir zone (>3100 m).

RESULTS

Tree-ring dates from 178 aspen stems and 139 conifers were used to reconstruct upper montane fire history, including stand-replacing fire patch dates and sizes (Tables 2 and 3, Figures 3-6). Annually dated, direct conifer evidence of fire (e.g., fire scars and tree death dates) was used to reconstruct 77 new fires in addition to the existing fire dates (from Grissino-Mayer *et al.* 1995) for PIN. Across the four sites, 100 fires occurring on 87 unique fire dates were analyzed (1623 to 1904; Table 3). Twenty five percent of the fires (n = 25) were recorded by ≥5 conifers (including fire scars) at a site. An average of 59% of all sampled aspen regenerated within five years after fire, ranging from 27% to 89% among sites. Three fires (1685 in PIN, 1879 in SFP, and 1904 in GIL) met our criteria for stand-replacing fire within mapped aspen or spruce-fir patches (Figures 4-6). Evidence of stand-replacing fire included aspen and conifer recruitment pulses, coincident conifer death and fire scar dates, and a lack of trees that survived (pre-date) the fire.

Table 2. Number of trees with crossdated tree-ring samples used to reconstruct fire history in the Mogollon Plateau and Madrean Sky Islands, USA.

Site ID	Aspen age structure	Conifer age structure	Conifer fire scar	Conifer death date	Conifer growth change or injury	Total
CHI	44	0	26	0	6	76
GIL	58	44	10	1	6	119
PIN	31	25	12*	0	0	68
SFP	45	0	6	1	2	54
Total	178	69	54	2	14	317

* Data from Grissino-Mayer *et al.* 1995 (PIN).

Table 3. Stand-replacing fires, fires recorded by ≥5 trees, and all additional fires reconstructed from multiple lines of tree-ring evidence.

Site	Stand-replacing fire dates	Fires recorded by ≥5 trees	All additional fires
CHI		1685, 1711, 1725, 1748, 1763, 1773, 1785, 1817, 1826, 1841, 1851, 1868, 1877, 1886	1654, 1661, 1688, 1697, 1698, 1700, 1701, 1703, 1709, 1716, 1721, 1723, 1727, 1733, 1737, 1739, 1749, 1752, 1760, 1765, 1775, 1779, 1787, 1789, 1794, 1798, 1800, 1805, 1806, 1807, 1818, 1822, 1835, 1838, 1840, 1848, 1849, 1859, 1863, 1875, 1883, 1894, 1903, 1904
GIL	1904	1904, 1748, 1773	1716, 1765
PIN*	1685	1685, 1773, 1785, 1819, 1842, 1858, 1871	1623, 1648, 1668, 1670, 1674, 1687, 1691, 1696, 1709, 1719, 1733, 1745, 1748, 1752, 1760, 1847
SFP	1879	1879	1752, 1773, 1809, 1818, 1836, 1840, 1847, 1851, 1855, 1857, 1860, 1863, 1876

* Fire scar data from Grissino-Mayer *et al.* 1995 (PIN).

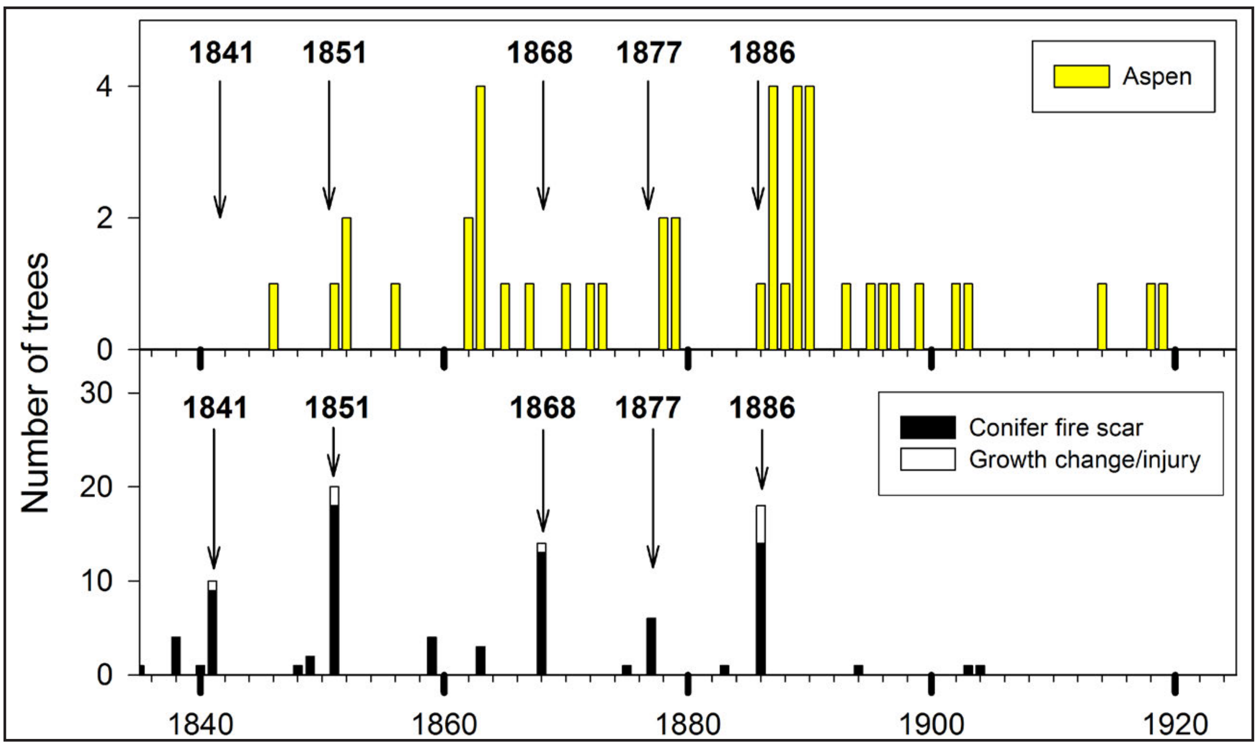

Figure 3. Chiricahua Mountains (CHI) estimated aspen pith dates (top) and direct conifer evidence of fire (bottom) in 1-year classes used to reconstruct fire history in the upper elevation forests. Years (e.g., 1886) indicate annually dated fire events recorded by ≥5 conifer trees, including fire scars.

Chiricahua Mountains

Eight small quaking aspen patches were mapped at CHI, totaling 139 ha (Table 1, Figure 2). No single post-stand-replacing fire quaking aspen cohort was present at CHI, but 89% of the aspen stems regenerated within five years after a fire (Figure 3). Surface fires recorded by conifers on south-facing slopes adjacent to the aspen stands were relatively

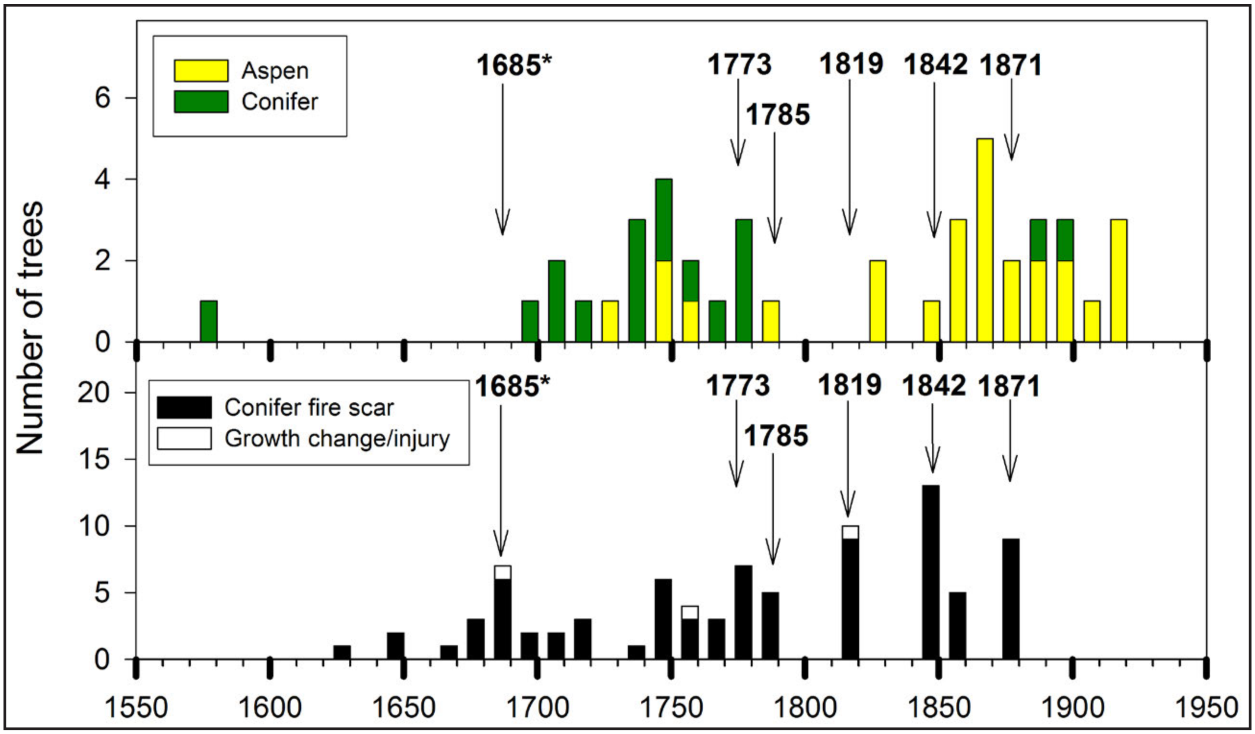

Figure 4. Pinaleño Mountains (PIN) estimated pith dates (top) and direct conifer evidence of fire (bottom) in 10-year classes used to reconstruct fire history in the upper elevation forests. Years (e.g., 1685) indicate annually dated fire events recorded by ≥5 conifer trees, including fire scars. * Indicates stand-replacing fire date. Fire scar data from Grissino-Mayer *et al.* (1995).

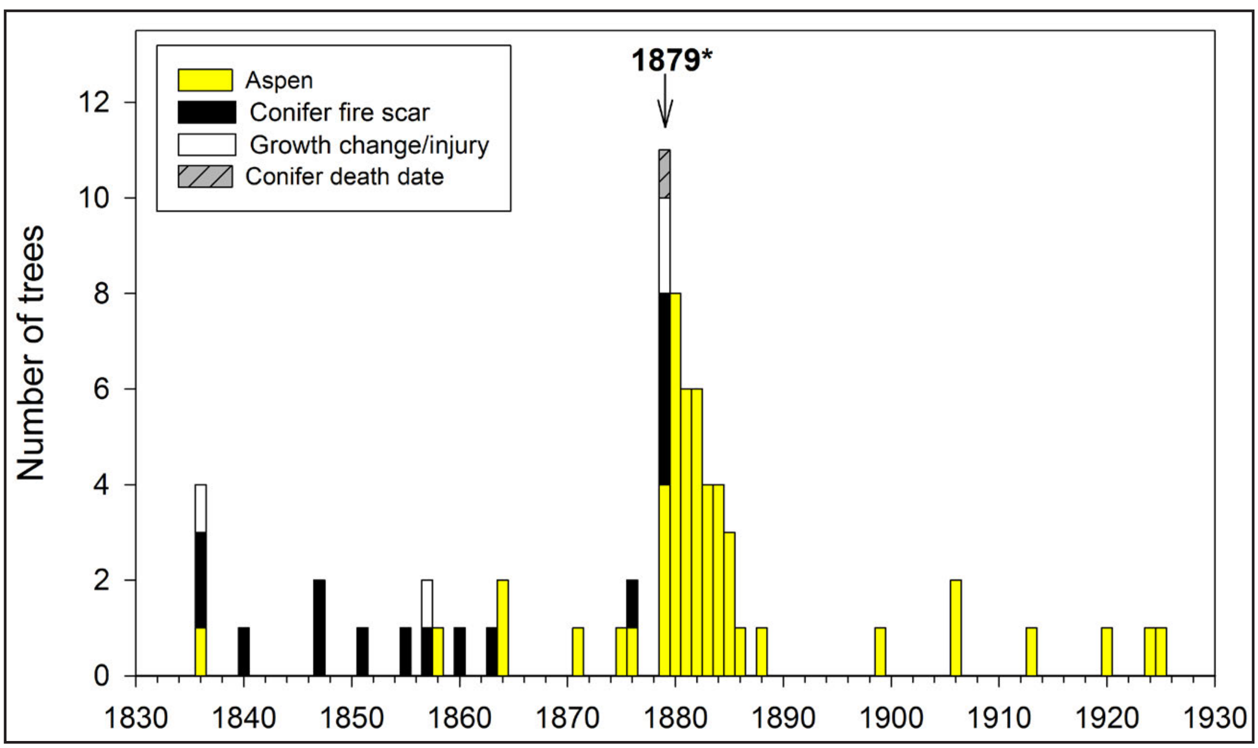

Figure 5. San Francisco Peaks (SFP) estimated aspen pith dates and direct conifer evidence of fire in 1-year classes used to reconstruct fire history in the upper montane forests. Years (e.g., 1879) indicate annually dated fire events recorded by ≥5 conifer trees, including fire scars. * Indicates stand-replacing fire date.

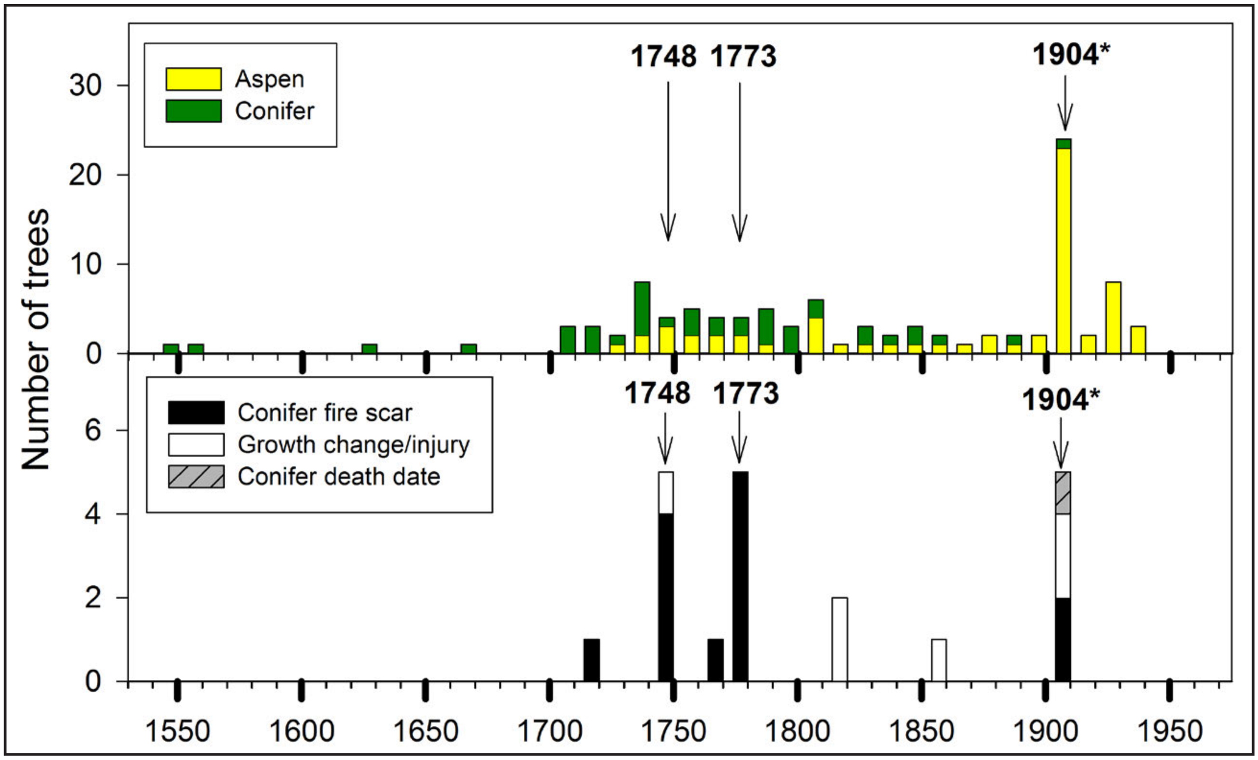

Figure 6. Mogollon Mountains (GIL) estimated pith dates (top) and direct conifer evidence of fire (bottom) in 10-year classes used to reconstruct fire history in the upper montane forests. Years (e.g., 1904) indicate annually dated fire events recorded by ≥5 trees, including fire scars. * Indicates stand-replacing fire dates.

frequent prior to *circa* 1900 (mean fire interval from 1654 to 1904 for all fires was 4.5 yr; Table 3).

The mapped aspen patches at CHI were not post-stand-replacing fire patches based on our criteria. The age structure of the dominant aspen within each patch was multi-aged, with some trees surviving (pre-dating) multiple fire events. For example, aspen from the 1886 post-fire cohort were scattered throughout multiple patches, but were often located adjacent to older aspen stems (e.g., 1851 post-fire regeneration) that survived the 1886 fire.

Pinaleño Mountains

The combined age structure of the multiple, small aspen groups (5 to 10 stems) scattered throughout the mixed conifer forest showed no evidence of a single, widespread, post-fire cohort (Figure 4). Only 27% of the dominant aspen at PIN regenerated within 5 years after a fire. Many aspen pre-dated (survived) fires recorded by multiple conifers as fire scars (e.g., 1871 fire), with the oldest living aspen dating to 1724 (estimated pith date).

Without post-fire aspen cohorts or large contiguous patches of seral, post-fire quaking aspen at PIN, the spruce-fir stand was the best potential evidence of past stand-replacing fire. The oldest tree (Engelmann spruce; 1692 estimated pith date) in the spruce-fir stand regenerated within 10 years after the 1685 fire that scarred all recording trees in the adjacent mixed conifer-aspen zone (Figure 4). The one tree that pre-dated the 1685 fire was a Douglas-fir located on the edge of the spruce-fir zone. These data met our criteria for stand-replacing fire in the spruce-fir zone at PIN in 1685.

San Francisco Peaks

Seventy-one percent of the dominant aspen at SFP regenerated within five years after a fire. Multiple lines of tree-ring evidence indicate that the 1879 fire was stand-replacing in some of the mapped patches (Figure 5). A distinct and immediate aspen recruitment pulse began in 1879, accounting for 63% of the sampled aspen. This site-level aspen age structure, dominated by a single post-fire aspen cohort, was different from the two Sky Island sites that had no dominant aspen cohort (CHI and PIN, Figures 3 and 4). The few aspen at SFP that pre-date 1879 were from the southeastern part of the site where there was no fire-scar evidence of the 1879 fire (Figures 2 and 5). In total, tree-ring evidence of the 1879 fire was present in all but one aspen patch.

The seral, post-stand-replacing fire aspen patches at SFP were located on the north-facing slopes and had the largest mean reconstructed stand-replacing fire patch size of all of the sites (145 ha). The drier, south-facing slopes contained conifers with multiple fire scars within the aspen stands. The ten fires recorded between 1836 and 1879 were all recorded by fire-scarred conifers on the south slope (Figures 2 and 5). This frequent fire regime ($MFI_{All\ fires}$ = 4.8 yr) that scarred, but did not kill, conifers within the south-facing aspen stands differed from the post-stand-replacing fire aspen patches, with no surviving conifers, on the north-facing slopes at SFP.

Mogollon Mountains (Gila Wilderness)

The aspen age structure at GIL was dominated by a post-1904 fire recruitment pulse (Figure 6). No sampled trees from within the mapped aspen patches survived the 1904 fire. These homogenous, even-aged aspen patches contained fire-killed Douglas-fir that died in 1904. Based on this evidence, all mapped aspen patches at GIL (totaling 744 ha) were determined to be stand-replacing fire patches from the 1904 fire. The aspen stems that predated the 1904 fire were located in the spruce-fir stands as scattered, co-dominant stems, some of which were >250 years old. A synchronous recruitment pulse was not evident from these old aspen. Overall, 42% of the sampled aspen at GIL regenerated within five years after a fire.

No direct evidence of fire (e.g., charred wood or fire scars) was observed within the spruce-fir stands at GIL. Relatively continuous conifer regeneration was recorded in the decades from 1700 to 1910, and the oldest individual (Engelmann spruce) in the spruce-fir patches dated to 1707. Multiple spruce trees were older than the oldest crossdated fire scar (1716) recorded adjacent to the spruce-fir patches (Figure 6). Therefore, the sampled age structure did not meet our criteria to be a post-fire recruitment cohort. There was no evidence that the 1904 fire, which our results suggest burned with stand-replacing severity in adjacent mixed conifer-aspen forests, burned into the spruce-fir zone.

Historical Stand-Replacing Fire Patch Size

We derived historical stand-replacing fire patch size estimates from the 10 tree-ring dated post-stand-replacing fire aspen patches (1879 to 1904) and the one post-stand-replacing fire spruce-fir patch (1685 fire; Table 4). Fires at three of the four sites (GIL, PIN, and SFP) had stand-replacing fire patches >200 ha. The maximum reconstructed historical stand-replacing fire patch size was 286 ha in the mixed conifer-aspen zone (2600 m to 3100 m) and 521 ha in the spruce-fir zone (>3100 m; Table 4).

DISCUSSION

Historical Stand-Replacing Fire

We found evidence of historical stand-replacing fire in upper elevation forests

Table 4. Historical and recent stand-replacing fire patch area statistics. Historical burn patch areas derived from combined tree-ring reconstructed aspen and spruce-fir stand-replacing fire patches. Recent burn patch area derived from fire severity maps (1984 to 2008, n = 352 fires). The conservative estimate of recent stand-replacing fire patch size includes only high-severity patches (H), and a more inclusive estimate includes high- and moderate-severity patches (H+M). Recent data only include patches >30 ha, equal to the smallest reconstructed historical stand-replacing fire patch.

	Historical burn patches		Recent burn patches					
	2600 m to 3100 m	>3100 m	2600 m to 3100 m		>3100 m		All elevations	
	Aspen	Spruce-fir	H	H+M	H	H+M	H	H+M
Count	10	1	64	85	1	2	204	675
Mean (ha)	110	521	129	206	33	110	136	233
Median (ha)	63	521	80	86	33	110	65	74
Standard deviation (ha)	89	--	134	300	--	70	204	500
Minimum (ha)	30	521	32	31	33	60	32	31
Maximum (ha)	286	521	637	1540	33	159	1929	5136
Sum (ha)	1104	521	8251	17507	33	219	27810	157482

(>2600 m) at three of the four sites. Fires with multiple large (>100 ha) stand-replacing fire patches were tree-ring dated at the two Mogollon Plateau sites using quaking aspen age structure and associated direct conifer evidence of fire (1904 in GIL and 1879 in SFP). Aldo Leopold (1922), while on a fire assignment in the Gila Wilderness, referenced a 1904 fire in the Mogollon Mountains. Abolt (1997) identified a widespread fire with a stand-replacing component in 1904 in the Mogollon Mountains from tree-rings and historical documents. In the San Francisco Peaks, Heinlein *et al.* (2005) recorded a fire in 1879 in lower elevation ponderosa pine-mixed conifer forests using fire scars, but does not report evidence of stand replacement. Historical photographs taken in 1910 at SFP show standing dead and downed trees in the spruce-fir and mixed conifer-aspen zones that likely resulted from a fire with large (estimated >500 ha) stand-replacing patches in the late nineteenth century (http://www.rmrs.nau.edu/imagedb/viewrec.shtml?id=22141&colid=fv).

We were able to associate spruce-fir age structure with direct conifer evidence of fire (i.e., fire scars) at one of the spruce-fir fire history test sites (PIN). The age structure data we collected, and prior sampling of more than 290 trees by Grissino-Mayer *et al.* (1995) from the large (521 ha) spruce-fir stand at PIN, support the hypothesis of a stand-replacing fire in 1685 (Swetnam *et al.* 2009, but see Stromberg and Patten 1991). Margolis and Balmat (2009) reconstructed a 1200 ha stand-replacing fire patch in the spruce-fir forests of the Santa Fe Watershed, New Mexico, also in 1685. This year was extremely dry (−5.0 reconstructed Palmer Drought Severity Index: Cook *et al.* 2004) and a common fire year throughout the southwestern US (Swetnam and Baisan 2003). Thus, it is plausible that the typically mesic spruce-fir zone at PIN could have been dry enough in 1685 to burn.

Frequent Fire and Quaking Aspen

We did not find evidence of past stand-replacing fire in the sampled aspen stands at CHI. Although 89 % of the aspen stems at this site regenerated within five years after reconstructed fires, the mapped aspen patches were multi-aged. This indicates that some aspen stems survived multiple fires, while other aspen in the same patch were top-killed by the same fires and then regenerated by sprouting.

We found direct evidence of repeated low-severity fire (e.g., conifers with multiple fire scars) adjacent to aspen stands at CHI and within the mixed conifer-aspen forests of PIN. Frequent fire occurred at these two upper montane Sky Island sites prior to *circa* 1900: MFI-$PIN_{All\ fires}$ = 10.9 yr (1685 to 1871), approximately 150 ha sample area (Grissino-Mayer *et al.* 1995), and MFI-$CHI_{All\ fires}$ = 4.4 yr (1654 to 1904), approximately 250 ha sample area. This history of frequent fire may have prevented sufficient fuel accumulation to sustain stand-replacing fire. This suggests that the cessation of fire for over 120 years due to late nineteenth century grazing and twentieth century fire suppression may be a cause of fuel structure changes and buildup that contributed to the recent occurrence of stand-replacing fires in the mixed conifer-aspen forests at these Sky Island sites (Swetnam *et al.* 2009).

Similar evidence of frequent low-severity fire (i.e., logs and living conifers with multiple scars) was present within and adjacent to the aspen stands on the south slope of SFP (Figures 2 and 5). The lower borders of these aspen stands are connected with ponderosa pine-mixed conifer forests that historically burned with frequent low-severity fire (e.g., Heinlein *et al.* 2005). Based on this evidence of repeated surface fire in aspen on south aspects at SFP, it is likely that the present stand structure, dominated by >20 m tall, mature aspen stems (>120 years old) may be in part an artifact of fire exclusion. These fire-sensitive aspen stems would have been historically exposed to frequent fire, thus the same stands likely looked very different in the nineteenth century. One hypothesis is that they were smaller diameter aspen "thickets" that were top-killed and regenerated after each fire (Maini 1960, Allen 1989). Alternatively, some larger diameter stems at the center of the stand may have been protected from being girdled by fire, creating a multi-cohort age and stand structure. Binkley *et al.* (2006) proposed a similar hypothesis of altered quaking aspen stand-structure in response to twentieth century fire exclusion on the Kaibab Plateau in north-central Arizona. The following hypothesis should be tested with future research: the age and stand structures of quaking aspen that historically experienced frequent fire have shifted from young or multi-aged, dense stands, to the current open structure dominated by a single mature cohort, largely due to >120 years of fire exclusion.

Spruce-Fir Fire History Challenges

The lack of burn boundaries within the spruce-fir stands at our two test sites (PIN and GIL) differs from higher latitude, Rocky Mountain landscapes where old stand-replacing fire patch boundaries are visible as obvious stand-height and structural differences that are used to map and date historical crown fires (e.g., Kipfmueller and Baker 2000, Sibold *et al.* 2006). Fulé *et al.* (2003) reported a similar lack of fire-related patch boundaries identifiable with remote sensing data in mixed conifer, aspen, and spruce-fir forests of the north rim of the Grand Canyon, Arizona. The lack of old fire boundaries within the spruce-fir zone of the current study, and on the north rim of the Grand Canyon may suggest that, in these spruce-fir forests, large crown fire patches were not as common within recent centuries as they were in the Rocky Mountains.

The inconclusive evidence of stand-replacing fire in the spruce-fir zone at GIL was possibly due to an insufficient number of tree ages to determine the complex and relatively old (>300 yr) age structure, and the relative scarcity of old (pre-1700 AD) fire scar material in this high-elevation forest type (Figure 6). Age-structure transects with a higher density of samples may be necessary to determine patch age in old (>300 years old) southwestern US spruce-fir forests. Repeated, sample-intensive age structure transects distributed throughout the mapped stands may be the best method to confidently evaluate the age structure of old spruce-fir forests in this region (e.g., Margolis and Balmat 2009). The number of trees sampled for age structure could be adjusted based on the estimated age of the stand (e.g., <150 yr old, > 250 yr old) so that only the oldest stands would require intensive sampling to overcome these challenges.

Multiple mapping and age-structure sampling methods should be tested on known and potential post-fire spruce-fir stands. The subalpine forests of the upper Rio Grande Basin or at SFP could be used to select test sites because there are large spruce-fir stands adjacent to large, post-fire aspen patches from historically documented nineteenth century fires (e.g., Santa Fe Ski Basin, New Mexico). Dating and mapping these sub-alpine conifer stands is the best available method to improve the accuracy of estimates of historical stand-replacing fire area in the highest elevations (>3100 m) in the southwestern US. These data are necessary to estimate fire frequency statistics (e.g., fire cycle or natural fire rotation) of the stand-replacing fire regimes in the upper montane mesic mixed conifer-aspen and spruce-fir forests of the region.

Historical Stand-Replacing Burn Patch Size

The occurrence of historical stand-replacing fire patches >200 ha at three of the four upper elevation sites suggest that recent large (200 ha to 500 ha) stand-replacing patches are within the historical range of variability in upper elevation forests (>2600 m) of the southwestern US outside of the southern Rocky Mountains. Based on our reconstructions, stand-replacing fire patches as large as 286 ha historically occurred in the mixed conifer-aspen zone, and patches as large as 521 ha historically occurred in the spruce-fir zone. Within these upper elevation forests, it is possible that older, larger stand-replacing fire patches were burned over by the late nineteenth century fires, or that such patches were re-colonized by mixed conifer species instead of aspen. We did not observe obvious even-aged mixed conifer stands with abundant fire-killed, remnant conifer logs or snags at our study sites that might indicate evidence of past stand-replacing fire. However, extensive (>500 ha) mixed conifer and spruce-fir patches exist in the region and could be systematically sampled to determine whether they regenerated following stand-replacing fire.

The largest historical stand-replacing fire patch we reconstructed was in the spruce-fir zone at PIN (521 ha). Historical photographs at SFP, discovered after our sampling was completed, illustrate large late-nineteenth century stand-replacing fire patches (estimated >500 ha) in the spruce-fir zone (http://www.rmrs.nau.edu/imagedb/viewrec.shtml?id=22141&colid=fv). In the southern Rocky Mountains of New Mexico, Margolis and Balmat (2009) reconstructed a 1200 ha stand-replacing fire patch in spruce-fir forest. Thus, documentary and tree-ring evidence at multiple sites in the southwestern US indicates the potential for large (500 ha to >1000 ha) stand-replacing fire patches in spruce-fir forest.

Recent Stand-Replacing Burn Patch Size

All four of our study sites have recently burned with high-severity patches. As an ancillary investigation to summarize the recent (1984 to 2008) fires, we quantified patch sizes of 352 fires >404 ha with high- or moderate-severity patches within 100 km of the four fire history study sites (Figure 1, http://www.mtbs.gov/index.html). We stratified the recent burn severity patch size data by elevation and vegetation type and fire severity to produce six subsets: 1) high severity with no elevation limit, 2) high plus moderate severity with no elevation limit, 3) high severity 2600 m to 3100 m, 4) high plus moderate severity 2600 m to 3100 m, 5) high severity >3100 m, and 6) high plus moderate severity >3100 m. The elevation ranges are the same used to categorize the upper elevation fire reconstructions. The subset with no elevation limit includes lower elevation, pine-dominant oak or shrub vegetation. Recent patch sizes were limited to ≥30 ha, equal to the minimum reconstructed stand-replacing fire patch size. Data from all sites were pooled. We were most interested in the largest patches since they arguably have the greatest ecological effects.

Significant direct and delayed mortality from crown scorch and insect attack has been documented in moderate-severity burn patches in recent fires (McHugh and Kolb 2003). Based on an assumption that a substantial percentage of the trees in moderate-severity burn patches die, high- and moderate-severity patches were combined in one subset of the data. We posit that the actual area of fire-related tree mortality (i.e., stand replacement) was probably somewhere between the "high severity" and "high plus moderate severity" patch size estimates.

The largest recent stand-replacing fire patch size with no elevation limit was 1929 ha (high severity) and 5136 ha (high plus moderate severity), with 37 patches >1000 ha (2 high severity and 35 high plus moderate severity; Table 4). In the mixed conifer-aspen zone (2600 m to 3100 m), the largest recent high-severity patch was 637 ha, and the largest high-plus moderate-severity patch was 1540 ha. Above 3100 m, in the spruce-fir zone, the largest recent high-severity patch was 33 ha, and the largest high- plus moderate-severity patch was 159 ha.

Direct comparison between recent and historical stand-replacing fire patch sizes are challenging. Due to reasons discussed above, our historical estimates are likely conservative estimates of stand-replacing patch size. Thus, we cannot confidently test whether the largest recent high- or moderate-severity patches are larger than have occurred in past fires. However, given these limitations, the data suggest that recent high- (or moderate-) severity patches that are smaller than the historical estimates (maximum reconstructed patch size, 286 ha in mixed conifer-aspen forest and 521 ha in spruce-fir forest) are likely within the historic range of variability.

In summary, historical fire regimes at multiple upper elevation (>2600 m) mixed conifer-aspen and spruce-fir sites on the Mogollon Plateau and Madrean Sky Islands included large (>200 ha) stand-replacing fire patches. Aspen recruitment was historically associated with fire, with an average of 59% of the dominant aspen stems regenerating within five years after fire (ranging from 27% to 89% among sites). In the drier portions of the mixed conifer-aspen sites, the cessation of historically frequent fires for the last 130 years has likely altered the current aspen age and stand structures. Tree-ring and photographic evidence of historical stand-replacing fire in the spruce-fir zone indicates that recent fires that burned with high severity in this forest type at the study sites (e.g., 2004 Nuttall Fire at PIN) are rare events, but not unprecedented. Based on the reconstructed estimate, recent stand-replacing fire patches as large as 286 ha in the mixed conifer-aspen zone and 521 ha in the spruce-fir zone may be within the historic range of variability and should be expected in future fires, particularly when considering predictions of a warmer and drier climate in the southwestern US (e.g., Seager *et al.* 2007).

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EVERYTHING YOU WANTED TO KNOW ABOUT WILDLAND FIRES IN FORESTS BUT WERE AFRAID TO ASK: LESSONS LEARNED, WAYS FORWARD

Salmon August fire in the Marble Mountains, California (L. Ruediger)

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March 30, 2018

EXECUTIVE SUMMARY

Wildfires are a fact of life for westerners. They mark the beginning of the spring season and have been a keystone architect of biodiverse ecosystems for millennia. While wildfires are not eco-catastrophes, they are a health concern, evoke public fear-of-fire exploited by decision makers seeking to push through anti-environmental policies, and generate conflicts over the best ways to coexist with this force of Nature that is not going away (nor should it), no matter how hard we try. This white paper summarizes some of the latest science around top-line wildfire issues, including areas of scientific agreement, disagreement, and ways to coexist with wildfire. It is a synopsis of current literature written for a lay audience and focused on six major fire topics:

1. Are wildfires ecological catastrophes?
2. Are acres burning increasing in forested areas?
3. Is high severity fire within large fire complexes (so called "mega-fires") increasing?
4. What's driving the recent increase in burned acres?
5. Does "active management" reduce wildfire occurrence or intensity?
6. Will more wildfire suppression spending make us safer?

Key findings

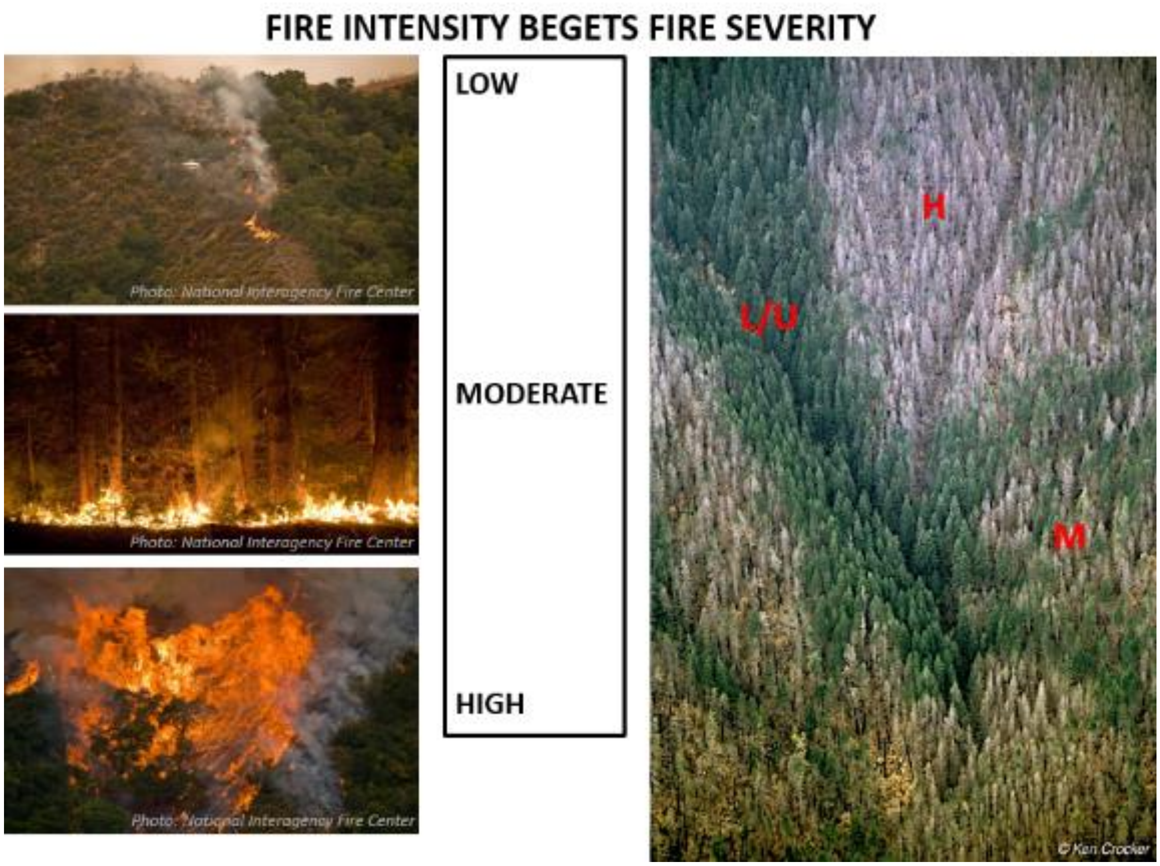
- Large wildland fire complexes, including patches of high severity fire, generate critical ecological pulses of dead trees (biological legacies) that are associated with extraordinary levels of biodiversity under-appreciated by most.
- Using long historical timelines, wildfire acres are currently at historical lows, but have been increasing in recent decades due mainly to three factors: (1) climate change; (2) human-caused fire ignitions (including suppression firing operations such as burnout and backfires); and (3) conversion of fire-resilient native forests to flammable plantations that experience relatively more high fire severity fire.
- Throwing more money at fire suppression will not abate fire concerns as more and more homes are built in indefensible places and are not designed or built with fire-resistant materials.
- Post-fire logging and associated activities (including roads) are unequivocally damaging to fire-rejuvenated forests and related aquatic ecosystems.
- Thinning small trees and prescribed burning can lower fire intensity at the stand level if done properly but this has significant limitations and ecological consequences given the scale of the perceived need and a changing climate.
- The most effective pathway to fire coexistence is to: (1) limit ex-urban sprawl through land-use zoning; (2) lower existing home ignition factors by working from the home-out with vegetation management and home retrofitting (defensible space), instead of the wildlands-in

(logging); (3) thin small trees and prescribe burn in ecologically appropriate settings (e.g., flammable plantations) while prioritizing wildland fire use in most forests away from homes; (4) store more carbon in ecosystems by protecting public forests and incentivizing carbon stewardship on non-federal lands; and (5) shift to a low-carbon economy as quickly as possible. Anything else will not achieve desired results to scale.

Issue 1: Are Wildfires "Catastrophic" or "Disastrous" Events?

Background

Large landscape wildfires are most often referred to as catastrophic "mass fires" or "megafires." Demonizing wildfires has placed this natural process in the same conversation as hurricanes and floods. Such disaster-speak and presumed logging remedies are now inculcated in the "Wildfire **Disaster** Funding Act" (emphasis added) recently passed by Congress as part of federal omnibus appropriations that also included rollbacks to forest protections. But what really goes on after a wildfire may be surprising in terms of the high biodiversity and rejuvenation capacity of forests after large fires, including severe ones.



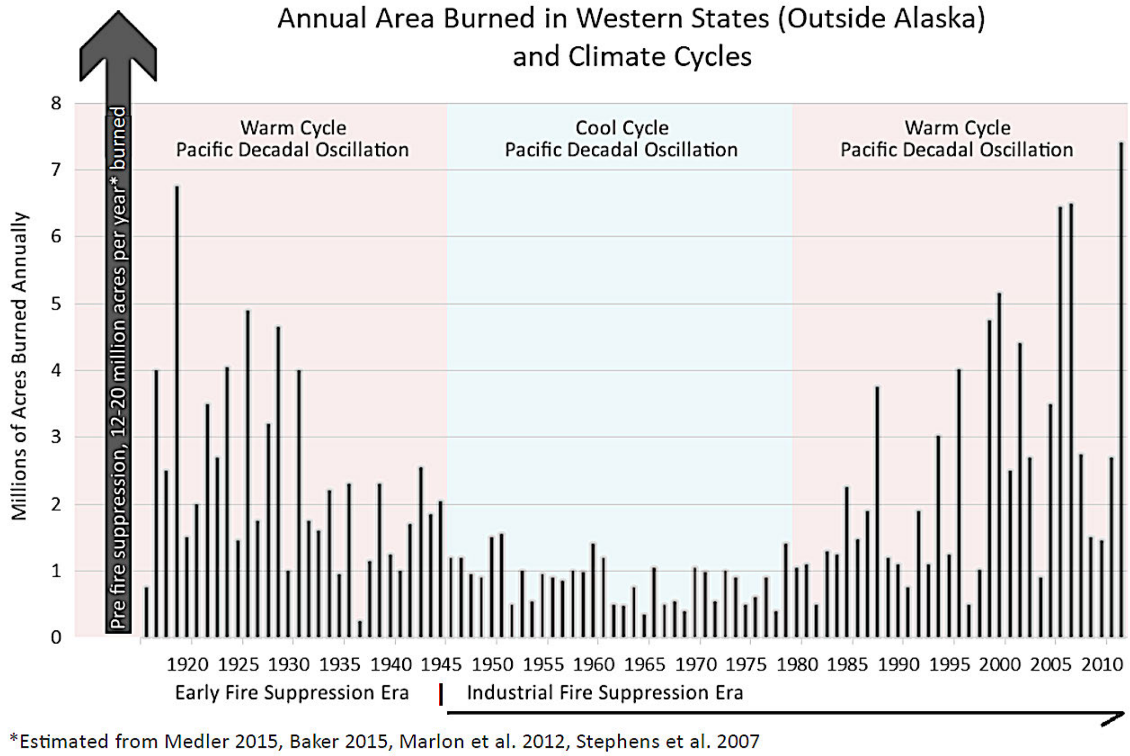
In general, fire effects are the result of heat energy released during a fire (fire intensity – left photos) and resulting effects on ecosystems (fire severity, right). Most large fires (right) produce a mosaic of burn severity effects on vegetation (H-high severity, M-moderate, U-unburned, L-low). Fire-mediated landscape heterogeneity is habitat for a diverse assortment of species distributed across the successional gradient (new to old forests) and has been referred to as "pyrodiversity begets biodiversity" (see below)[1]. Note – in some cases a fast-moving high-intensity "running" surface fire can produce low severity effects, while a slow-moving low intensity "creeping" fire can produce high severity effects (e.g., smoldering piles of slash or logs).

Issue 2: Are Total Wildfire Acres Burning Increasing (independent of severity)?

Background

Nearly every fire season, the news media and politicians announce another "unprecedented" wildfire season. Such proclamations are incorrectly based on comparisons of contemporary wildfire acres to a recent historical timeline. This has been widely criticized in the scientific literature as the "shifting baseline perspective" (i.e., when a baseline is shifted to a more recent historical time period)[2]. Importantly, in the early part of the 20th century during a warm climatic cycle (Pacific Decadal Oscillation - PDO), wildfire acres were at least five times more abundant than today. A mid-century cool down accompanied by industrial fire suppression resulted in a substantial decline in acres burning[3]. The current warm period is associated with a recent increase in both acres burning and fire suppression (see below). In other words, wildfire activity tracks broad-scale climatic phenomenon (top-down drivers) that also influence fire suppression efficacy.

[1] DellaSala, D.A., and C.T. Hanson. 2015. The ecological importance of mixed-severity fires: nature's phoenix. Elsevier: Boston.
[2] See Jackson, B.C., et al. 2011. Shifting baselines. Island Press: DC.
[3] For an excellent historical resource read NY Times Best Seller, Timothy Egan's "The Big Burn." Mariner Books: NY.



*Estimated from Medler 2015, Baker 2015, Marlon et al. 2012, Stephens et al. 2007

Figure interpretation caveats: prior to 1984, standardized datasets are difficult to obtain. Contemporary wildfires also have a strong back-burning influence not prevalent in historical times–i.e., errors in estimation exist on both ends of the wildfire acreage continuum. However, historical accounts (including General Land Office records and pollen-sediment core analyses) confirm very active fire seasons in the early part of the 20th century and before[4] (Figure compliments of John Muir Project).

Areas of Agreement

Fewer wildfire acres burning in forests today compared to the early 20th century has resulted in what many are calling a wildland fire deficit[5], which may seem as a surprise given fire hyperbole. The main exception to this deficit is southern California chaparral and shrub-steppe communities (too much human-caused fire is leading to ecosystem type conversions).

Areas of Disagreement

Current science debate is focused mainly on what is the best way for putting fire (i.e., “the right fire” “good fire”) back on the landscape in order to restore wildland fire-forest relationships.

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[4] Whitlock, C., et al. 2008. Long-term relations among fire, fuel, and climate in the north-western US based on lake-sediment studies. Int. J. Wildland Fire 17:72-83. Baker, W.L., and M.A. Williams. 2018. Land surveys show regional variability of historical fire regimes and dry forest structure of the western United States. Ecol. Applic. 28:284-290.
[5] Parks, S.A. et al. 2015. Wildland fire deficit and surplus in the western United States, 1984-2012. Ecosphere 6:275. 13 pp.

Many claim that this cannot be done safely without massive thinning to reduce "fuels"[6], others state that we need to get to coexistence with wildland fire as the amount of thinning needed is prohibitively costly[7], and has significant consequences to ecosystems (see below). Still others want more of the "right kind" of fire in the "right places"– meaning less high severity fire, despite ecological importance of this type in low to mid elevation pine and mixed conifer forests (i.e., even predominately low severity ponderosa pine systems have a component of high severity) throughout the West.

Issue 3: Is High Severity Fire Within Wildland Fire Complexes Increasing?

Background

High severity fires that kill most of the trees in older forests are associated with extraordinary levels of biodiversity not present in low severity burns due mainly to the abundance of biological legacies (e.g., snags and down logs, shrubs)[8]. This fact is now widely accepted by the scientific community; however, the amount and spatial distribution of high severity fire patches within wildland fire complexes remains in question as to whether ecosystem thresholds are being crossed in large fires.

[6] Hessburg, P.F., et al. 2015. Restoring fire-prone Inland Pacific landscapes: seven core principles. Landscape Ecol. 30:1805-1835.

[7] Moritz, M.A., et a. 2014. Learning to coexist with wildfire. Nature 515:58-66.

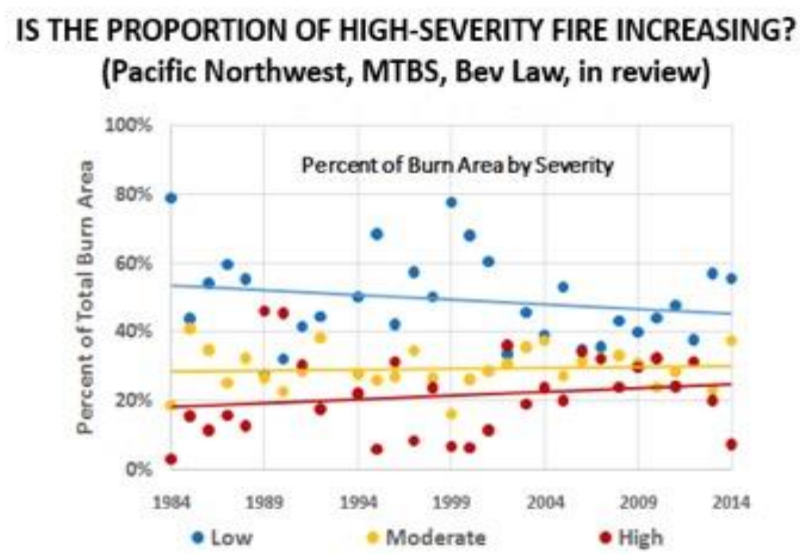
[8] Donato, D.C., J.L. Campbell, and J.F. Franklin. 2012. Multiple successional pathways and precocity in forest development: can some forests be born complex? J. Vegetation Science 23:576-585. DellaSala, D.A. and C.T. Hanson. 2015. The ecological importance of mixed-severity fires: nature's phoenix. Elsevier: Boston.

High Severity Fire Patches Become Biodiverse Snag Forests

Complex early seral forest after 12 years of natural conifer regeneration, native shrub patches, and deciduous trees (C. Hansen, Eldorado Starr Fire, Sierra).

Areas of Agreement

Nearly all studies have detected no statistically significant trend in high severity acres or proportion of high severity fire within large fire complexes (Colorado is an exception and there is debate in the Sierra)[9].



This figure shows no discernable increase in percent of various fire severities in the Pacific Northwest over a three-decade period (compliments of Bev Law, Oregon State University). Data prior to 1984 are not available for fire severity comparisons.

[9] Keyser, A., and A. LeRoy Westerling. 2017. Climate drives inter-annual variability in probability of high severity fire occurrence in the western United States.

Areas of Disagreement

Concern has now shifted to whether the size of high severity patches is increasing, believed to be a product of 21st century "mega-fires," and whether this is leading to type conversions (forests to shrubs)[10]. High severity patch data obtained from hundreds of forest fires across the West show no statistical increase in patch sizes in recent decades (DellaSala et al. in peer review). This is important as the patch size debate is used to make claims about "mega-fires" and to justify large-scale thinning, post-fire logging, and tree planting based on perceptions of inadequate tree recruitment or lack of forest resilience to fires. However, most high severity patches have high levels of internal heterogeneity that include small patches of live trees or nearby low-moderate burn areas as seed sources (in review).

Issue 4: What's Driving Recent Increases in Wildfire Acres Burning?

Background and Areas of Agreement

Recent increases in wildfire acres burning (see above PDO figure) can be traced to three main factors acting in concert: (1) a warming PDO from climate change; (2) increases in human-caused fire starts (accidental, arson, back burns); and (3) conversion of native forests to flammable tree plantations[11].

Over half of recent increases in wildfire acres burning has been attributed to climate change[12] (see top figure below as generalization) with 84% of all fire ignitions nationwide in recent decades caused by people (bottom figure below)[13]. Human-caused wildfire ignitions vary regionally based on population densities and remoteness.

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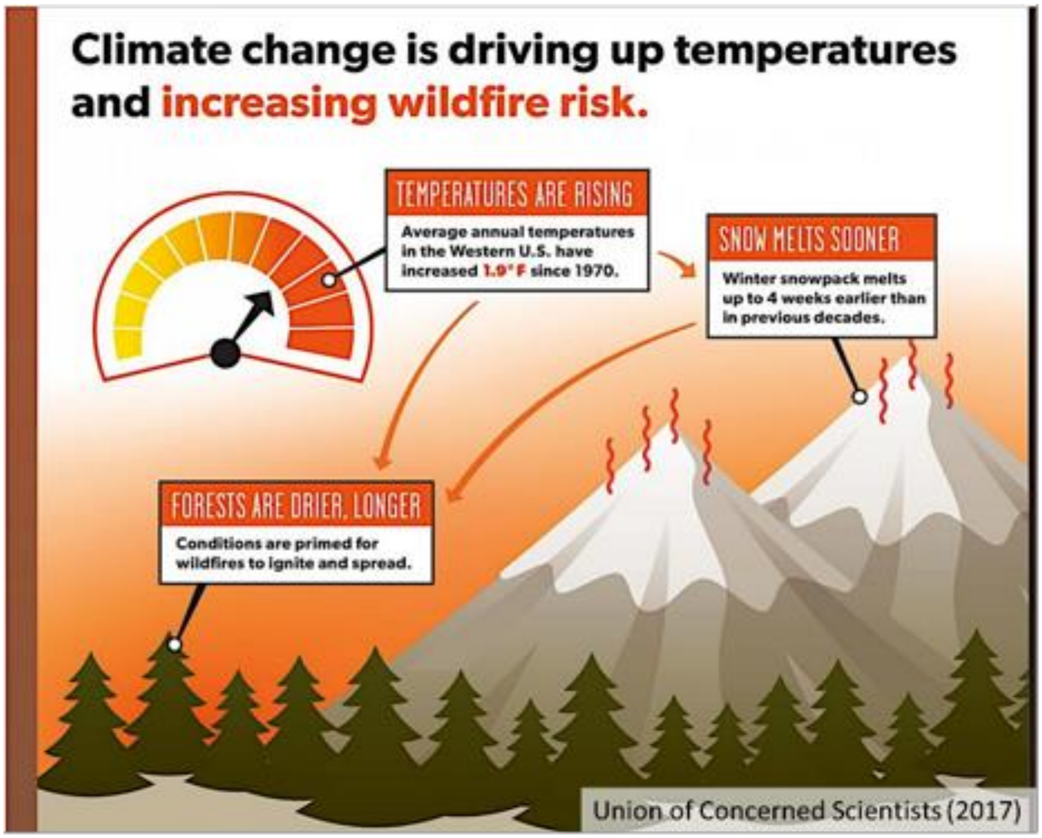
[10] Hessburg P.F. et al. 2015. Restoring fire-prone inland Pacific landscapes: Seven core principles. Landscape Ecology 30, 1805–1835.

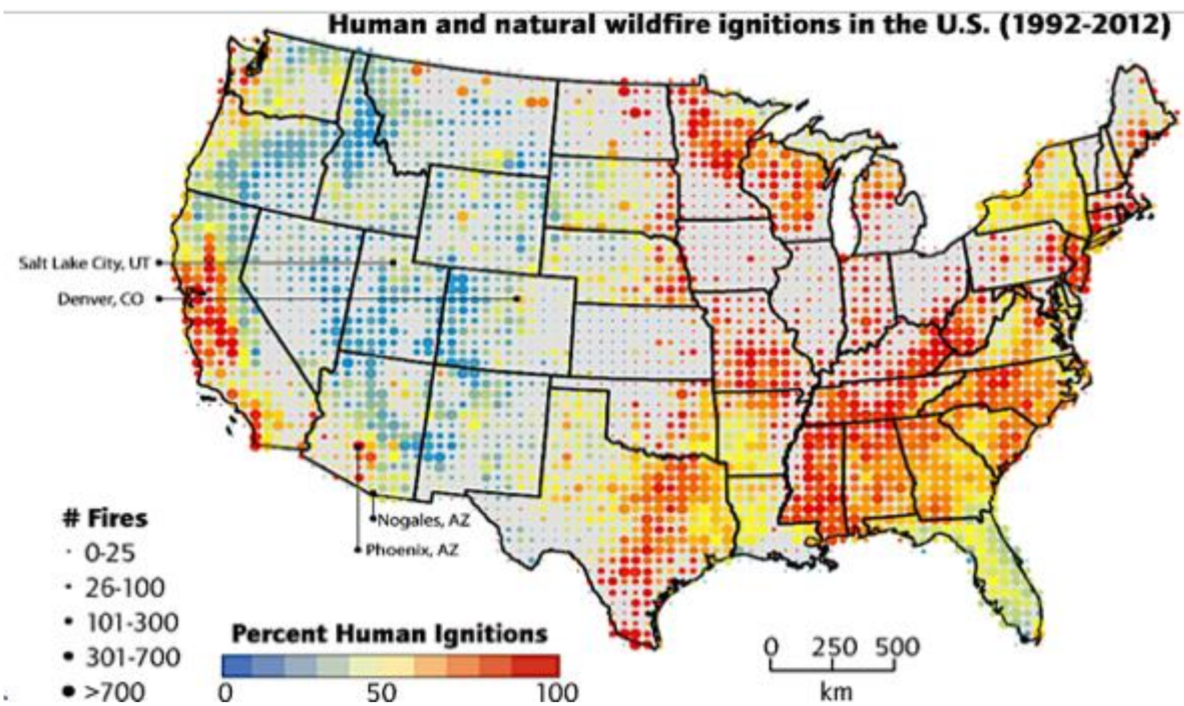
[11] Bradley, C., C.T. Hanson, and D.A. DellaSala. 2016. Does increased forest protection correspond to higher fire severity in frequent-fire forests of the western United States? Ecosphere 7:1-13. Odion, D.C., et al. 2004. Fire severity patterns and forest management in the Klamath National Forest, northwest California, USA. Conservation Biology 18:927-936

[12] Abatzoglou J.T., and A.P. Williams. 2016. Does Impact of anthropogenic climate change on wildfire across western US forests. PNAS 113:11770-11775

[13] Balch et al. 2017. Human-started wildfire expand the fire niche across the United States. PNAS 114:2946-2951.



Climate change is driving up temperatures and increasing wildfire risk.
TEMPERATURES ARE RISING
Average annual temperatures in the Western U.S. have increased 1.9° F since 1970.
SNOW MELTS SOONER
Winter snowpack melts up to 4 weeks earlier than in previous decades.
FORESTS ARE DRIER, LONGER
Conditions are primed for wildfires to ignite and spread.
Union of Concerned Scientists (2017)



Human and natural wildfire ignitions in the U.S. (1992-2012)
Salt Lake City, UT
Denver, CO
Nogales, AZ
Phoenix, AZ
Fires
0-25
26-100
101-300
301-700
>700
Percent Human Ignitions
0
50
100
0 250 500
km

Areas of Disagreement

While most land managers and decision makers are preoccupied with "fuels," two of the main drivers of fire behavior (climate change, human-caused ignitions) are largely ignored (except when used to justify logging for forest resilience). Additionally, roads (a principal source of human-caused fire ignitions) are almost never addressed in fire risk reduction measures. Uncertainty exists regarding whether large-scale thinning will work in a changing climate where fire behavior will be increasingly governed by extreme fire weather (high temperatures, low soil moisture, high winds, see below)[14]. Storing more carbon in ecosystems will help mitigate climate effects, although land managers often prioritize generating revenue from commercial sales over carbon storage[15].

Issue 5: Does "Active Management" Reduce Wildfire Intensity and Lower Fire Risks?

Background

Active management encompasses a wide spectrum of actions and opinions mostly focused on pre- (thinning) and post-fire ("salvage" logging) logging widely debated by scientists, conservation groups, and decision makers. This is arguably the number one area of fire-related conflicts on public lands with the underlying assumption that forests are overstocked, they need active management to reduce fire risks, and environmental safeguards are preventing management of forests that otherwise will burn out of control.

Areas of Agreement

Post-fire logging is unequivocally damaging to the pyrodiverse landscapes and complex early seral forests. In general, the larger the fire, the bigger the logging project[16]. Post-fire logging involves clearcutting both live and mostly dead trees, kills naturally regenerating conifers, and often is followed by herbicides to reduce competing yet beneficial vegetation and allow for subsequent planting of artificially grown trees (from nursery stock) in dense rows. As artificial plantations increasingly replace native forests, plantations act as kindling for intense fires (i.e.,

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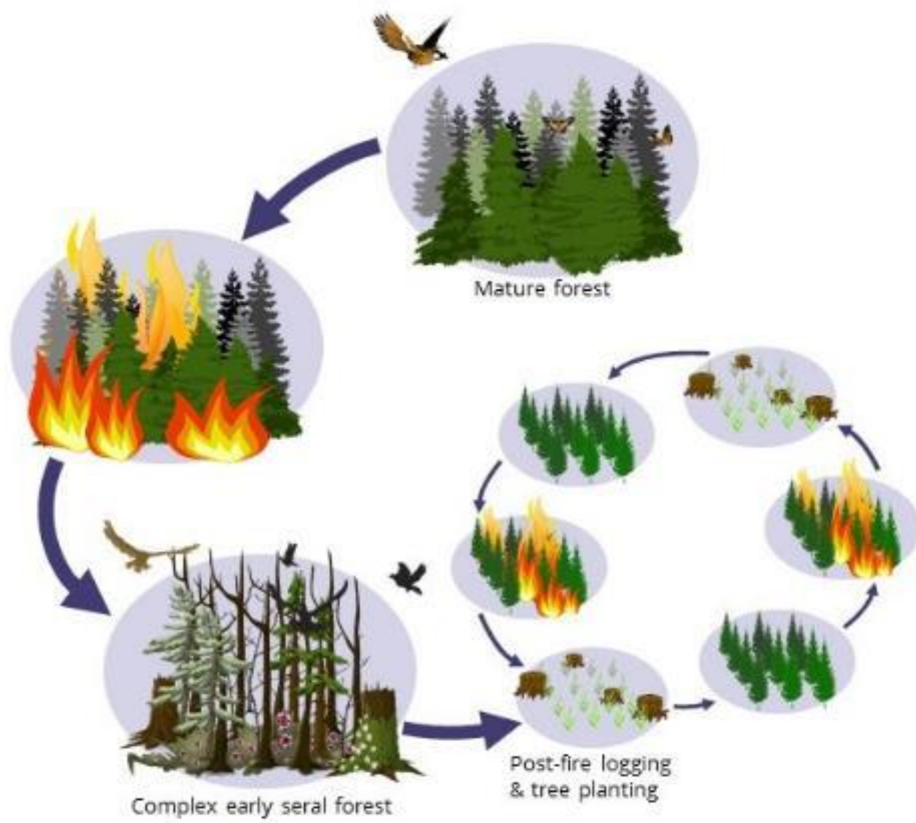
[14] Cary, G.J. et al. 2016. Importance of fuel treatment for limiting moderate-to-high intensity fire: findings from comparative fire modeling. Landscape Ecol. 32:1473–1483. Kalies, E.L., and L.L.Y. Kent. 2016. Tamm review: are fuel treatments effective at achieving ecological and social objectives? A systematic review. Forest Ecology and Management 375:84-95.

[15] Moritz, M.A. et al. 2014 (ibid). Law, B.E et al. 2018. Land use strategies to mitigate climate change in carbon dense temperate forests. PNAS http://www.pnas.org/cgi/doi/10.1073/pnas.1720064115

[16] DellaSala, D.A., et al. 2015. In the aftermath of fire: logging and related actions degrade mixed- and high-severity burn areas. Pp. 313-347, *In* DellaSala, D.A., and C.T. Hanson (eds), The ecological importance of mixed-severity fires: nature's phoenix. Elsevier, United Kingdom

"fire's gasoline")[17]. Post-fire logging creates a catastrophic feedback loop where fires in older forests create ecologically beneficial snag forests, those forests are then clearcut and replanted with small trees in dense rows lacking structural complexity, only to burn in higher intensities and so on (see figure below)[17,18]. Legacy trees removed by logging operations anchor soils, provide shade for developing seedlings, "nurse logs" for new growth and soil moisture retention for amphibians and invertebrates, habitat for aquatic species when snags fall into streams, and they store vast amounts of carbon as they slowly (decades to centuries) decompose. The scientific community is generally at consensus with regard to post-fire logging as damaging to ecosystems[19], particularly to spotted owl habitat[20].



Fire in a mature forest produces complex early seral (snag) forest that connects the stages of forest development through time. This cycle is interrupted by post-fire logging and tree planting leading to type conversions (native forest to flammable plantation) and unnatural fire severity.

[17] Odion, D.C., et al. 2004. Ibid. Thompson, J.R., et al. 2007. Reburn severity in managed and unmanaged vegetation in a large wildfire. PNAS 104:10743-10748.
[18] Bradley, C.M., et al. 2016. Does increased forest protection correspond to higher fire severity in frequent-fire forests of the western United States? Ecosphere 7:1-13.
[19] Lindenmayer, D.B., P.J. Burton, and J.F. Franklin. 2008. Salvage logging and its ecological consequences. Island Press: Washington, D.C.
[20] C.T. Hanson, M.L. Bond, and D.E. Lee. 2018. Effects of post-fire logging on California spotted owl occupancy. Nature Conservation 24:93-105.

Areas of Disagreement

In contrast to post-fire logging, thinning involves partial logging of trees for various purposes, including reducing competition among nearby trees, increasing tree vigor, and accelerating tree growth (e.g., in wet forests it is commonly used to accelerate development of older forest conditions as specified under the Northwest Forest Plan). Thinning also is commonly used to reduce "fuels" in dry forests and has support in the scientific community and with NGOs. When done properly, thinning of small trees followed by prescribed burning[14], or prescribed burning alone in some cases[21], can reduce fire intensity. However, it remains controversial, has significant ecological consequences (short and long-term), and substantial limitations given high costs and the massive scale believed needed to influence fire behavior especially in a changing climate (Box 1).

Large trees (dbh inches marked on trees) marked for removal on a BLM "fuels" project, southwest Oregon (L. Ruediger).

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[21] Zachmann, L.J., D.W.H. Shaw, and B.G. Dickson. 2018. Prescribed fire and natural recovery produce similar long-term patterns of change in forest structure in the Lake Tahoe basin, California. Forest Ecol. & Manage. 409:276-287

Box 1. General limitations of thinning (and collateral ecosystem damages)

(1) Thinning reduces habitat for canopy dependent species, including spotted owls[22], requires an expansive road network damaging to aquatics, can spread invasive and flammable weeds, and, when implemented over large landscapes, releases more carbon emissions than fires, even severe ones[23].

(2) There is a very low probability (3-8%) that a thinned forest will encounter a fire during the narrow period (10-20 years depending on site factors) of reduced "fuels" [24], resulting in large-scale thinning proposals that alter forest conditions over large areas[6].

(3) Excessive thinning (e.g., reducing bulk crown density below 60%) can increase wind speeds and solar radiation to the ground causing increased flammable vegetation growth and fire spread.

(4) Thinning needs to be followed by prescribed fire to reduce flammable slash but this can cause soil damage especially if burning is concentrated in piles (intensifies heat effects.

(5) Thinning is seldom cost effective without public subsidies or removing large fire-resistant trees.

(6) In some regions (Sierra, Klamath-Siskiyou), time since fire is not associated with increasing fire risks (i.e., as forests mature, they become less flammable[25]).

(7) Thinning efficacy is limited under extreme fire weather (principal factor governing large fires).

(8) At regional scales, active management (unspecified forms of logging) have been associated with uncharacteristic levels of high severity fires (see figure below)[26].

[22] Odion, D.C., et al. 2014. Effects of fire and commercial thinning on future habitat of the northern spotted owl. Open Ecology Journal 7:37-51.

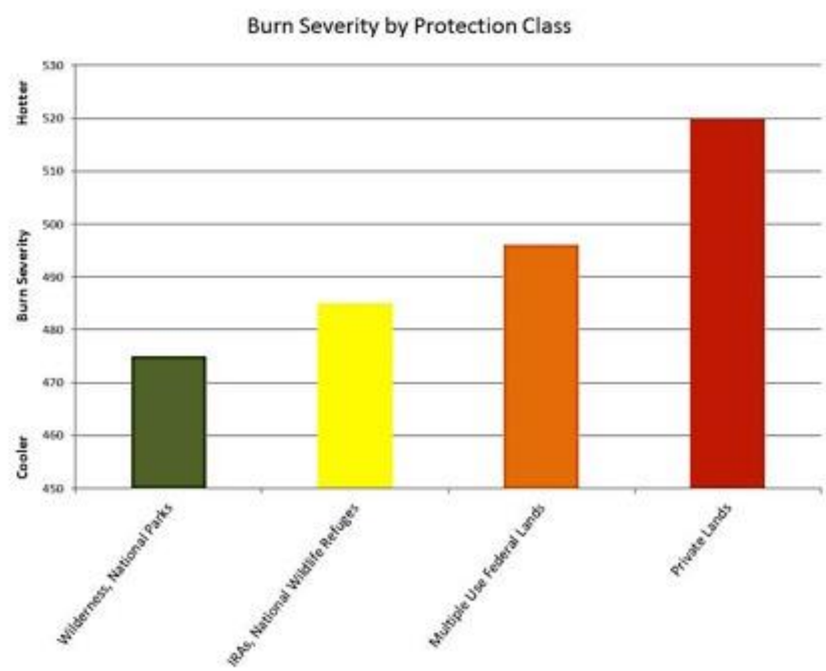
[23] Campbell, J.L., M.E. Harmon, and S.R. Mitchell. 2012. Can fuel-reduction treatments really increase forest carbon storage in western US by reducing future fire emissions? Frontiers in Ecol. & Environ. doi:10.1890/110057

[24] Rhodes, J.J., and W.L. Baker. 2008. Fire probability, fuel treatment effectiveness and ecological tradeoffs. The Open Forest Science Journal, 2008, 1, 1-7

[25] Odion, D.C., et al. 2004. Fire severity patterns and forest management in the Klamath National Forest, northwest California, USA. Conservation Biology 18:927-936. Zachmann, L.J., et al. 2018. Ibid.

[26] Bradley, C.M., C.T. Hanson, and D.A. DellaSala. 2016. Does increased forest protection correspond to higher fire severity in frequent-fire forests of the western United States? Ecosphere 7: Ecosphere 7:1-13.

Thinning on the Deschutes National Forest, Oregon (G. Wuerthner).



Burn severity as a function of protection levels from lower burn severity in Wilderness and National Parks (green) to greater high severity amounts in actively managed areas (red)[26]. Figure prepared by C. Bradley, CBD.

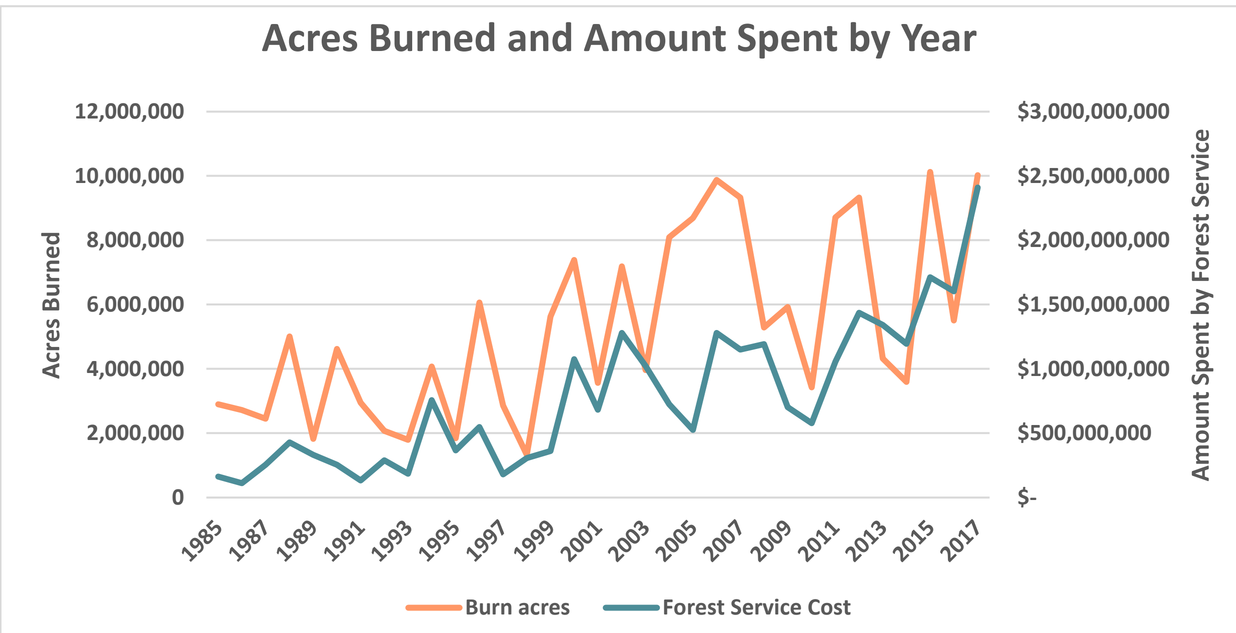
Issue 6: Will More Suppression Spending Make Us Safer?

Background

On March 21, 2018, Congress passed an omnibus spending package that established a dedicated wildfire "disaster" fund of > $2 billion per year that would increase steadily over a 10-year period. Spending measures include expanding the use of controversial categorical exclusions for logging projects up to 3,000 acres each that can conceivably be located adjacent to one another with no regard for cumulative impacts.

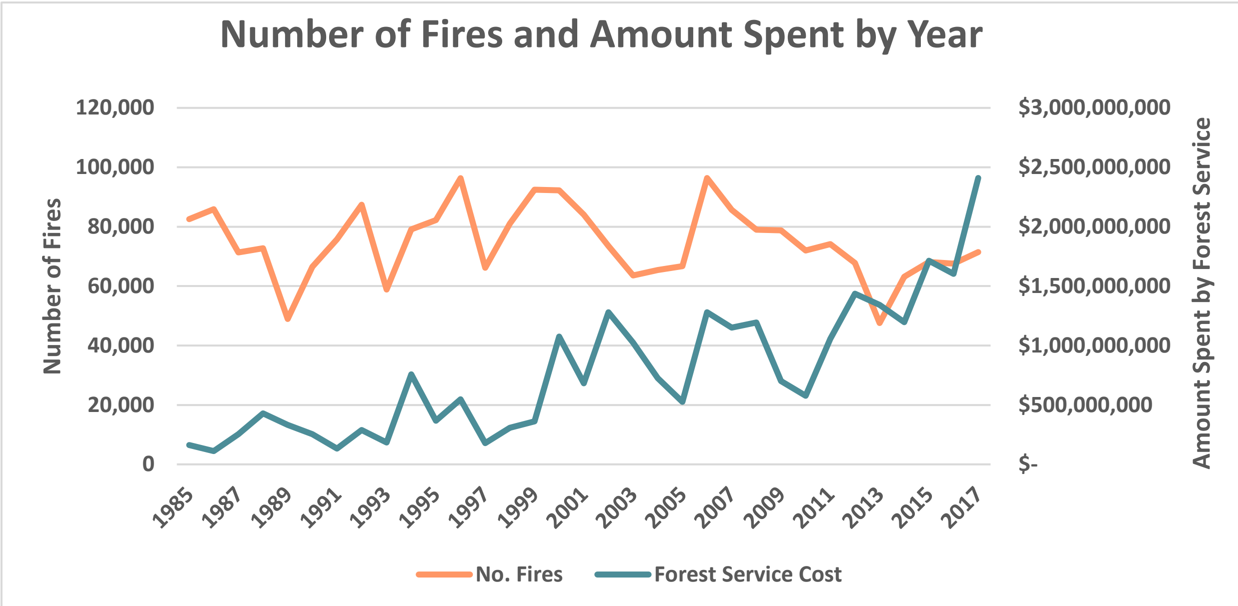
Areas of Agreement and Disagreement (combined)

While conservation groups pushed for a rider-free wildfire spending fix, throwing more money at fire suppression while expecting fewer fires is highly uncertain. In many ways, the two figures below illustrate the common definition of crazy – doing the same thing over and over again but expecting a different outcome. In sum, both acres burning and wildfire suppression costs of the Forest Service have risen dramatically over the past three decades (top figure) calling into question whether more money will achieve fewer fires or less acres burning. Interestingly, in some years (e.g., 2006-2012, bottom figure) total wildfire ignitions steadily dropped while costs generally rose presumably from fighting more fires in remote areas and few controls on spending[27] (figures prepared by J. Leonard, Geos Institute using fire data from National Interagency Fire Center[28]).



[27]Ingalsbee, T., and U. Raja. 2015. The rising cost of wildfire suppression and the case for ecological fire use. Pp. 348-317 In: D.A. DellaSala, C.T. Hanson (eds.). The ecological importance of mixed-severity fires: nature's phoenix. Elsevier: Boston.

[28] https://www.nifc.gov/fireInfo/fireInfo_statistics.html



As an example of unmitigated suppression spending, the 132,127-acre Soberanes fire in California (started by an illegal campfire) cost ~$236 million (nearly $1800 per acre) and deployed thousands of fire fighters and numerous air-tankers, making it the most expensive wildfire to fight in US history. Although the fire destroyed 57 homes (and took the life of a bulldozer operator), suppression forces were used on the fire as it burned safely in the back country far removed from homes. The fire was eventually extinguished by fall rains.

Conclusion: Moving Forward in the New Climate Wildfire Era

When it comes to fire, we each see what we want: land managers view the world as ready-to-burn ecosystems just lacking an ignition source and needing "fuels" reduction; ecologists see habitat restored by wildfires as part of the circle of life and death in a forest; the public fears fire and understandably has concerns about smoke emissions; the media portrays death and destruction during fires; conservation groups are either for or against large-scale thinning; and politicians race to sensationalize fire to justify increased commercial logging on public lands. This is no doubt the most difficult public lands issue we have ever faced as its wrapped in emotion, human health, self-interests, avarice, hyperbole, point-counter point arguments, and nearly everyone wants to do something – even if doing something is worse than the perceived problem. Moving beyond this will require communicating about fire with empathy and clear intent especially while recognizing genuine fear and health issues. It will involve a combination of science publications, public support for managing wildfires for ecosystem benefits (once safety has been addressed), tolerance for temporary smoke levels, and our own limitations in being able to influence ecological processes increasingly governed by top-down drivers (climate) rather than bottom up forest management. Based on climate change models, extreme fire conditions are predicted to be more common this century and thus the extensive thinning involved to theoretically reduce fire intensity (e.g., wide spacing among trees, open-park like conditions) would create novel or greatly engineered forest systems that impact biodiversity and ecosystem services (carbon stores, clean water) in undesirable ways.

Importantly, we need to solve for human safety with the most significant challenges coming from ex-urban sprawl (enabled by scant land-use zoning and building in the wrong places), a rapidly changing climate, an expanding logging footprint focused increasingly on extracting the "new coal" ("feed stock") for biomass burning. Rational fire approaches and communication strategies that do not sacrifice native forests for perceived fire safety are an area of much needed research and financial resources.

We know a lot more about wildfire today than in the last decade; however, much of the science is still in debate, it almost always lags behind or is ignored by decision makers, land managers, and even some scientists and conservation groups with entrenched views about fire (Box 2).

Box 2. What we know and do not know about wildfires.

- Complex early seral forests are as biodiverse as old growth, containing comparable levels of species richness (although species composition varies across seral stages).
- Wildfire effects on vegetation are highly variable (mixed)[29], calling into question fuel reduction projects (especially those that use a shifting baseline) based on restoring forests to an "historical" open park-like condition when there was a lot more variability and the climate is changing.
- It will be impossible to mechanically treat the substantial acres alleged to need fuel reduction to reduce fire intensity[7] (58 million acres according to the Forest Service), and, even if possible, this would have severe consequences to ecosystems, especially aquatics, and come with substantial taxpayer funded costs.
- Thinning under extreme fire weather ("the new norm") is highly uncertain in a changing climate.
- Additional increases in homes built within the Wildland Urban Interface (WUI) (now totaling 43.4 million)[30] will result in more human-caused fire ignitions and out of control suppression spending regardless of where the money comes from. Wildfire problems will not abate if this growth along with climate change accelerates.

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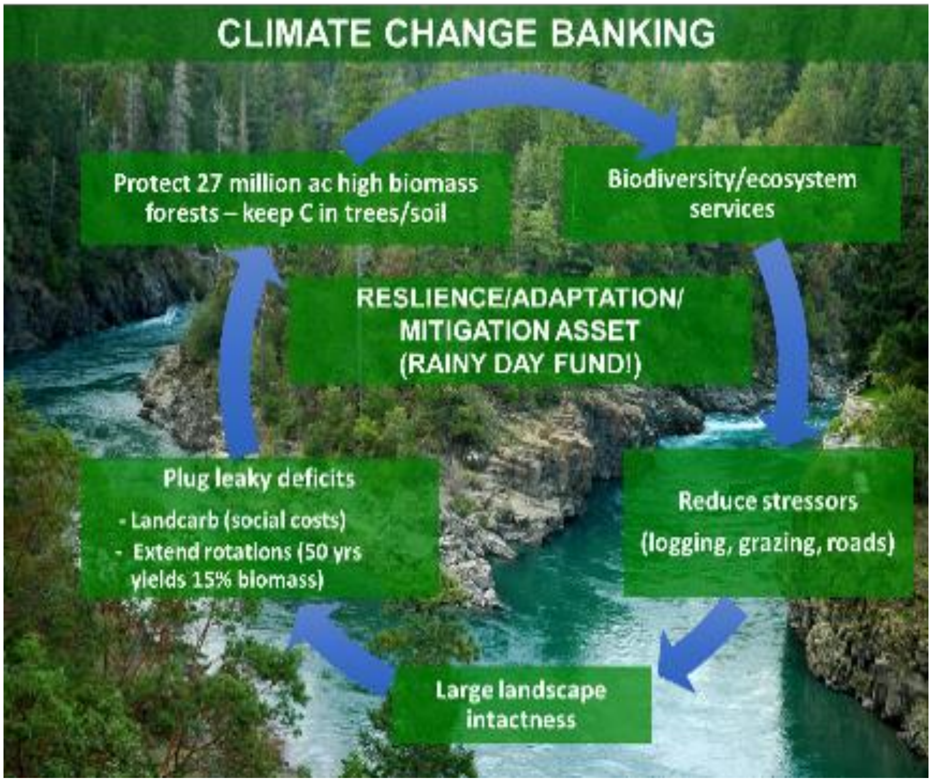
[29]Odion, D.C., et al. 2016. Areas of agreement and disagreement regarding ponderosa pine and mixed conifer forest fire regimes: a dialogue with Stevens et al. PLoSOne DOI:10.1371/journal.pone.0154579 May 19, 2016
[30]Radeloff, V., et al. 2018. Rapid growth of the US wildland -urban interface raises wildfire risk. PNAS http://www.pnas.org/cgi/doi/10.1073/pnas.1718850115

There is no “right” or “wrong” or “good” or “bad” fire. Fire is a predatory force of Nature resulting in ecological winners and losers (at least temporarily). We in the environmental community do not speak of “good” wolves or “bad” mega-wolves (that eat sheep) yet the fire debate embraces this terminology. In sum, we do not have a fire problem per se but rather a people management problem – homes built in the wrong places and with the wrong materials, fire-fighters dropped into unsafe areas, hyped-up thinning projects that may or may not work, and a rapidly changing climate that will produce surprises.

There are plenty of management options that are compatible with western forest resilience and fire-mediated biodiversity in a changing climate, including:

- Removing land-use stressors (e.g., mining, livestock, Off Highway Vehicle impacts that accumulate in space and time) so that ecosystems can adapt to climate change;
- Maintaining viable populations of imperiled species and habitats, including climate sanctuaries such as older forests, forests on north-facing slopes, and riparian areas[31];
- Curtailing the spread of invasive species;
- Managing wildfires for ecosystem benefits and prescribed fire in appropriate types;
- Thinning and girdling (killing) small trees in young plantations (along with prescribed fire) to increase structural complexity and reduce fire intensity (but see limitations discussion);
- Replacing ineffective culverts (especially important in areas where climate change will trigger more floods); restoring floodplains so they can naturally store more water (e.g., reintroducing beavers) and attenuate floods; and removing damaging roads by re-contouring the road prism to natural features (e.g., to reduce sediments to streams and improve hydrological functions);
- Managing for connectivity (up-down elevation, latitudinal-longitudinal gradients); and
- Storing more carbon in forest ecosystems (see climate robust strategies).

[31]Olson, D.M., et al. 2012. Climate change refugia for biodiversity in the Klamath-Siskiyou ecoregion. Natural Areas Journal 32:65-74.



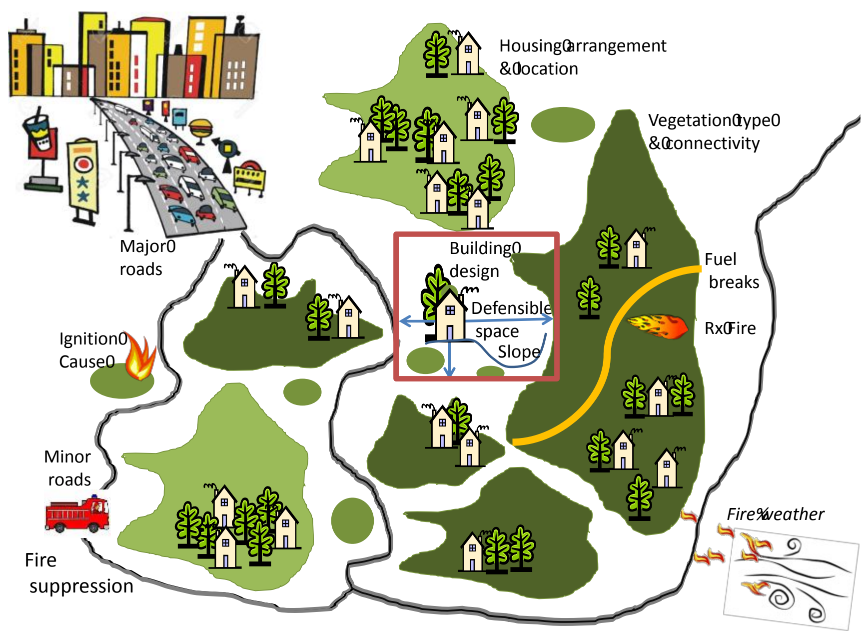
Climate robust conservation means protecting carbon dense forests nationwide as a foundation for biodiversity and ecosystem services, reducing land-use stressors, connecting landscapes for wildlife migrations and reducing carbon emissions from logging. Fire safety measures discussed herein are compatible with this overall strategy and represent a comprehensive path forward.

Importantly, managing wildfire for ecosystem benefits is not the same as “let burn.” Instead, this involves monitoring wildfire behavior initially, targeting suppression at fires likely to spread near towns, “loose-herding” and directing fire in the back-country under safe conditions, cutting fire lines nearest homes, and keeping fire fighters out of harm’s way. The same fire can be compartmentalized for different treatments. The Forest Service already has existing authorities that allow them to use such approaches in deciding when to attempt to use suppression vs. managing wildfire for ecosystem benefits[32]. Implementing this policy would help keep spiraling wildfire suppression costs in check[27].

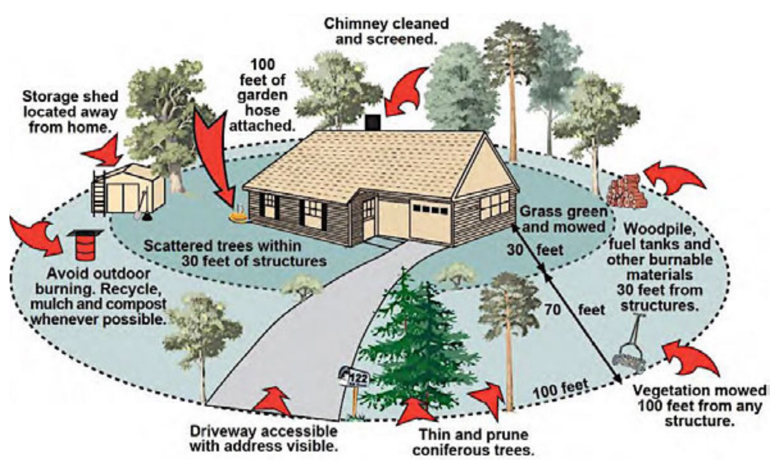
In addition, local governments need to start embracing smart growth measures to limit sprawl within the WUI. Fire safety for existing homes is about reducing risks from the home-out (defensible space), rather than from the wildlands-in (logging)[33]. Defensible space has to become as routine as changing the batteries in a home’s smoke detectors and building with metal roofs the norm in home construction.

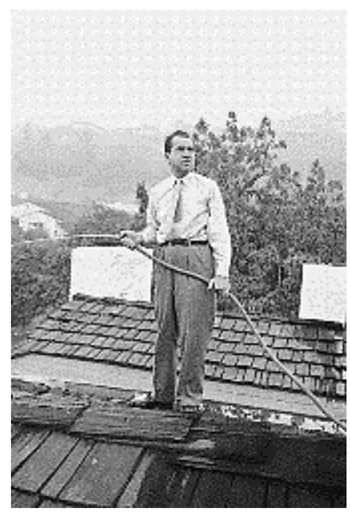
[32] https://www.nifc.gov/policies/policies_documents/GIFWFMP.pdf

[33] Syphard, A.D., T.J. Brennan, and J.E. Keeley. 2014. The role of defensible space for residential structure protection during wildfires. Int. J. Wildland Fire. http://www.publish.csiro.au/wf/WF13158



Fire prevention begins with land-use planning that limits growth in unsafe areas and includes defensible space management (figure prepared by A. Syphard, CBI; historical Nixon photo courtesy of San Francisco Chronicle[34]; lower figure – Homeowner fire safe guide for Montana).





Potential synergies and framing messages around forest issues cut across public lands campaigns that could benefit from working together, including the “keep it [carbon] in the ground,” “350.org,” and a much needed “keep it [carbon] in the forest” campaign. For instance, researchers at Oregon State University recently showed that the best way to increase carbon stores in Northwest forests is to reduce federal lands logging by at least 50%, increase the length of timber harvest rotations on private lands to 80 years, afforestation, and reforestation[35]. Notably, wildfires are currently not a significant contributor to greenhouse gas

[34] http://www.sfgate.com/news/article/Skirball-Fire-recalls-1961-Bel-Air-inferno-that-12410921.php
[35]Law, B.E., et al. 2018. Land use strategies to mitigate climate change in carbon dense temperate forests. PNAS www.pnas.org/cgi/doi/10.1073/pnas.1720064115

emissions, contrary to many assertions[36]. Importantly, the Northwest Forest Plan resulted in ancillary climate benefits by shifting federal forest management from a substantial source of logging emissions in the 1980s to a current "sink" (warehouse) for carbon storage due to reduced (by 80%) timber harvest on federal lands[37]. As this forested warehouse continues to accumulate carbon, it is critical to protect carbon-dense older forests on public lands and incentivize forest carbon stewardship on non-federal lands. Making the link between climate mitigation and intact forest conservation currently lacks the recognition needed to offset fossil fuel emissions and keep the planet from heating above 2° C, which cannot be accomplished without forests in the mix[38].

The long-range prognosis for public lands forests is generally favorable. On the one hand, conservation groups with significant support of the donor community have held the line on decades of hard-fought victories centered on the Northwest Forest Plan and wilderness/ roadless protections. On the other hand, the pressure to develop forests is unprecedented globally and regionally with an urgent need to solve for increasingly complex social, economic, and engrained perceptions about forest management. Conservation science continues to be a leading voice for public lands by supporting effective communications, grass-roots organizing and campaigning, and responding to maladaptive climate policies by proposing climate robust conservation strategies. When it comes to fire science, however, we have as many questions as answers, more debate than consensus, but there have been important strides forward.

In closing, we have much work to do to change public attitudes about forest fires but optimism begins when we open our hearts and minds to the intricate dance between green and burned forest orchestrated by the natural disturbance processes that have been at play since the age of dinosaurs and will continue in largely unpredictable ways in the emerging novel climate. Preparing for these changes must be comprehensive, science-based, and solve for top-down drivers of change while we hold the line and then expand on a robust conservation vision.

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[38]https://primaryforest.org/

RESEARCH ARTICLE

Negative Feedbacks on Bark Beetle Outbreaks: Widespread and Severe Spruce Beetle Infestation Restricts Subsequent Infestation

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Abstract

Understanding disturbance interactions and their ecological consequences remains a major challenge for research on the response of forests to a changing climate. When, where, and how one disturbance may alter the severity, extent, or occurrence probability of a subsequent disturbance is encapsulated by the concept of *linked disturbances*. Here, we evaluated 1) how climate and forest habitat variables, including disturbance history, interact to drive 2000s spruce beetle (*Dendroctonus rufipennis*) infestation of Engelmann spruce (*Picea engelmannii*) across the Southern Rocky Mountains; and 2) how previous spruce beetle infestation affects subsequent infestation across the Flat Tops Wilderness in northwestern Colorado, which experienced a severe landscape-scale spruce beetle infestation in the 1940s. We hypothesized that drought and warm temperatures would promote infestation, whereas small diameter and non-host trees, which may reflect past disturbance by spruce beetles, would inhibit infestation. Across the Southern Rocky Mountains, we found that climate and forest structure interacted to drive the 2000s infestation. Within the Flat Tops study area we found that stands infested in the 1940s were composed of higher proportions of small diameter and non-host trees ca. 60 years later. In this area, the 2000s infestation was constrained by a paucity of large diameter host trees (> 23 cm at diameter breast height), not climate. This suggests that there has not been sufficient time for trees to grow large enough to become susceptible to infestation. Concordantly, we found no overlap between areas affected by the 1940s infestation and the current infestation. These results show a severe spruce beetle infestation, which results in the depletion of susceptible hosts, can create a landscape template reducing the potential for future infestations.

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Data Availability Statement: All climate data are available from the PRISM database (http://www.prism.oregonstate.edu). All Aerial detection survey data are available from the USFS (http://www.fs.usda.gov/detail/r2/forest-grasslandhealth/?cid = fsbdev3_041629). All ecoregion data area available from the EPA (http://www.epa.gov/wed/pages/ecoregions/level_iii_iv.htm). All vegetation data for Rocky Mountain National Park area available from the USGS (http://www.usgs.gov/core_science_systems/csas/vip/parks/romo.html). All vegetation data for USFS Region 2 are available from the USFS (http://

www.fs.usda.gov/detail/r2/landmanagement/gis/?cid¼stelprdb519523). Vegetation data for the state of Colorado are available from the Colorado Gap Project (http://gapanalysis.usgs.gov). Additional vegetation data for the state of Colorado are available from Landfire (http://www.landfire.gov/vegetation.php).

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Introduction

In the context of a changing climate and increases in forest disturbances such as bark beetle infestations and wildfires, disturbance interactions are receiving increased attention in ecological research [1,2]. In particular, there is a need to better understand when, where and how one disturbance event may alter the severity, extent, or probability of occurrence of a subsequent disturbance, a concept known as *linked disturbances* [3]. A prior disturbance may amplify the second by increasing its likelihood or severity through positive feedbacks (e.g. blowdowns may increase the amount breeding material thereby increasing insect populations and likelihood of outbreak [4]). Or, alternatively the first disturbance may dampen the probability of occurrence or severity of the second (e.g. stand-replacing fire may decrease the probability of subsequent fire [5]).

During the late 20th and early 21st century, warm and dry conditions and suitable hosts have promoted landscape-scale (sensu [6]) and severe bark beetle outbreaks, resulting in tree mortality across 8.4 ± 2.5 Mha in the western North America (1997–2010; [7]). Given this extensive tree mortality, there is an increased need for understanding how bark beetle infestations alter subsequent disturbance dynamics. Considerable research has emphasized the potential effects of bark beetle outbreak on the fire behavior [3,8–14], occurrence [15–19], and severity [15,20–22]. Recent research has also emphasized the compound effects of bark beetle outbreak and fire on ecosystem recovery [20–23]. Far less is understood about how one bark beetle outbreak affects a subsequent outbreak.

Bark beetles of the *Dendroctonus* genus inhabit the inner bark and feed on the tree's phloem tissues. Heavy colonization and reproduction within the inner bark interrupts the flow of water and nutrients throughout the tree and usually causes tree death. When and where bark beetle outbreaks occur is constrained by both weather and forest structure conditions [6,24,25]. Warm temperatures promote the rapid growth of beetle populations by increasing the proportion of beetles that develop within one year and decreasing overwintering mortality [26–28]. Drought may stress host trees, increasing the susceptibility of trees to infestation [29–32]. Forest structure also affects the occurrence of bark beetle infestations. Bark beetles prefer large diameter trees, growing in dense stands composed predominantly of the host tree species [6,33].

In the Southern Rocky Mountains, outbreaks of spruce beetles (*Dendroctonus rufipennis*) are among the most important broad-scale disturbances in subalpine forests. Spruce beetles are found in Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) forests, where they most frequently colonize large diameter (> 23 cm diameter at breast height; DBH) spruce trees. However when beetle population levels are high and host trees are severely-drought stressed, spruce beetles may attack trees less than 10 cm DBH [34]. Like other bark beetles, heavy colonization and reproduction within the inner bark usually kills the host tree. In northwestern Colorado, severe spruce beetle infestations tend to occur at median intervals of c. 70 years for the same stand [30,35]. The return interval of spruce beetle infestations to the same stand or relatively homogeneous landscape is hypothesized to be in part a function of a negative linkage between infestations. Thus, for forest stands (100s of hectares) and forest landscapes (1000s to tens of 1000s) that are characterized by similar forest compositions and tree population age structures, forest attributes are likely to affect the probability of occurrence and severity of an outbreak [32]. For example, a severe spruce beetle outbreak, which may result in the mortality of 90% of the mature host trees (Engelmann spruce), has been hypothesized to decrease the likelihood of subsequent infestation [33]. This decrease in susceptibility to infestation is hypothesized to persist until host trees reach a suitable size for infestation. While there are studies documenting the collapse of an outbreak evidently due to host depletion [29], there is no published empirical evidence for a bark beetle infestation negatively influencing the occurrence of a subsequent bark beetle infestation.

A widespread spruce beetle outbreak affected a large part of the spruce-fir forests of western Colorado in the 1940s. This outbreak was most severe in the Flat Tops Wilderness area of White River National Forest in northwestern Colorado where 99% of the overstory spruce were killed over an area of 2,700 km^2 [33,36]. The second most severely affected area in the 1940s outbreak was Grand Mesa National Forest to the southwest of White River National Forest where mortality was estimated at over 50% [32]. There are no other known 20th century spruce beetle outbreaks in Colorado of a comparable magnitude to the 1940s outbreak that was centered on the Flat Tops area of White River National Forest. Thus, in the context of the recent spruce beetle outbreak of 1997–2012, the concentration of high tree mortality during the 1940s outbreak in one large contiguous area created the opportunity to quantitatively evaluate the potential for a landscape-scale bark beetle infestation to negatively affect the probability of a subsequent infestation ca. 60 years later. Mapping of the recent spruce beetle outbreak from Aerial Detection Surveys [37] indicate a low spruce beetle infestation in the Flat Tops area in comparison with spruce-fir forests throughout Colorado (Fig 1). Thus, the primary aim of this study is to determine if the relative lack of recent spruce beetle infestation in the Flat Tops area is due to a negative feedback from host depletion attributable to the 1940s outbreak. Because spruce beetle infestation depends on both climate and forest conditions [6], we first assess the suitability of climate and forest attributes for spruce beetle infestation during 1997–2012 in the Flat Tops study area in comparison with the entire Southern Rocky Mountain Ecoregion of the U.S. Second, we examine forest attributes across the Flat Tops study area in relation to the mapped extent of the 1940s spruce beetle infestation and compare current forest structure in field sampled stands infested and not infested in the 1940s. Thus, by documenting the climatic suitability of the Flat Tops study area for the recent infestation, we are able to associate the relative absence of infestation with host depletion from the 1940s outbreak.

Materials and Methods

Study area

The study region (Fig 1) is the spruce-fir forest type of the Southern Rocky Mountain Ecoregion. The study region is characterized by high elevations (3215 ± 205 m), cold, wet winters (mean minimum January temperature -14°C and mean total January-March precipitation 241 mm; 1981–2010) and warm, dry summers (mean maximum July temperature 20.6°C and mean total June-August precipitation 169 mm; 1981–2010) [38]. Engelmann spruce and subalpine fir co-dominate the spruce-fir forest type.

We examine the potential for spruce beetle infestation to affect the area of forest structure suitable for subsequent spruce beetle infestation within a subset of the study region comprised of the Flat Tops Wilderness and adjacent areas of White River National Forest of northwestern Colorado, USA (Fig 1). The Flat Tops study area was chosen because of the unique availability of maps of both the 1940s spruce beetle infestation derived from air photo interpretation [5] and the current (1997–2012) spruce beetle infestation produced from Aerial Detection Surveys (ADS; [37]). Historical reports document widespread spruce beetle infestation in the 1940s, when about 25% of the merchantable volume of Colorado's spruce was killed. The Flat Tops study area experienced particularly abundant mortality, characterized by more than 90% canopy mortality [33].

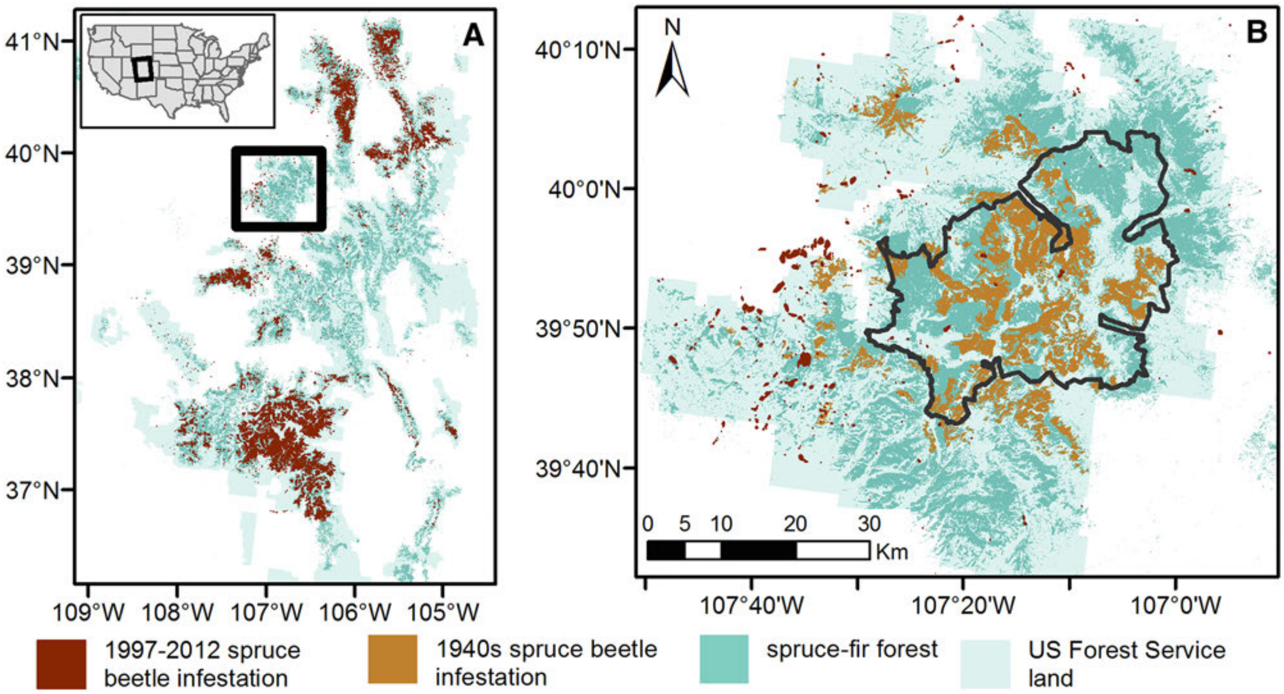

Fig 1. The larger study region and study area. (A) Map of the Southern Rocky Mountain study region displaying spruce-fir forests infested by spruce beetles during the 1997–2012 period. The upper left inset displays the location of the study region in relation to the entire United States. The black box indicates the study area displayed in B. (B) Map of the Flat Tops study area comprised of the Flat Tops Wilderness (black line) and adjacent areas of White River National Forest and areas infested by spruce beetles during the 1940s and 1997–2012 periods. Sources are given in Table 1.

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Data processing

We first obtained data on the occurrence of spruce-fir forest across the Southern Rocky Mountain study region (Table 1). Most vegetation cover-type datasets express only moderate (40–60%) overall agreement between field plot data and forest cover-type at 30 x 30 m spatial scale [39,40], thus we combined three datasets depicting the occurrence of spruce-fir forest [41]. For each vegetation dataset, we listed the presence of a spruce beetle host within a 990 x 990 m pixel, which approximates a stand scale [41]. We adopted a conservative criterion for mapping spruce-fir forest based on requiring its presence in all three datasets.

Next we obtained spatially explicit data on the presence of spruce beetle infestation over the time period from 1998–2013 from the United States Forest Service Region 2 ADS database [37]. Aerial Detection Surveys have been conducted annually in the Southern Rocky Mountains since 1994. To our knowledge robust accuracy assessments of ADS maps of spruce beetle infestation do not exist. However, accuracy assessments between ADS and ground reference data listing the presence/absence of bark beetle infestation in lodgepole pine show moderate-high agreement at coarse (500 m) spatial grains [39,40]. Thus we assumed ADS maps of spruce beetle infestation are most appropriate for assessing coarse-grain trends in presence/absence of infestation. To account for the ca. 1-year lag between initial infestation and ADS detection, we shifted the year of detection back one year to obtain year of attack [7]. Annual spatial polygon data listing the year of spruce beetle attack (1997–2012) were then converted to a 990 x 990 m grid listing the presence of spruce beetle infestation. Annual grids were then summed to obtain the cumulative area infested (1997–2012) and multiplied by a raster of spruce-fir presence to obtain a cross-validated grid of spruce beetle infestation [24].

Table 1. The GIS data layers and attributes used to examine linked spruce beetle disturbance.

Variable	Description	Data	Type	Resolution	Year
Damage casual agent	Name of forest pest or pathogen causing damage	Aerial Detection Survey Database [37]	Polygon	Compiled at 1:100,000 scale	1998–2013
1940s infestation	Presence /absence of 1940s spruce beetle infestation	Bebi et al. 2003 [5]	Polygon	Interpreted at 1:10,000 scale	Based on 1971 color & 1984 IR aerial imagery
R2VEG Cover type	Dominant life forms, based on Society of American Foresters classification	R2VEG [42]	Polygon	Interpreted at 1:24,000 scale	Based on 2002 aerial imagery
LANDFIRE EVT	Existing vegetation type, based on Nature Serve's ecological systems classification	LANDFIRE [43]	Raster	30 x 30 m	Based on 2001–2010 Landsat imagery
GAP Analysis Project Cover type	Primary cover type	GAP	Polygon	Interpreted at 1:100,000 scale	Based on 1989–1998 Landsat imagery
R2VEG Diameter at breast height	Tree DBH binned (cm): 1) <2.5, 2) 2.5–12.4, 3) 12.5–22.9, 4) 23–40.4, 5) $\geq$40.5	R2VEG [42]	Polygon	Interpreted at 1:24,000 scale	Based on 2002 aerial imagery
Southern Rocky Mountain Ecoregion	Level III Ecoregions	North America Ecoregions [44]	Polygon	Compiled at 1:250,000 scale	2013
August maximum temp	average monthly maximum temperature (°C)	PRISM [38]	Raster	4 x 4 km	1997–2012
Annual precipitation	average annual precipitation (mm)	PRISM [38]	Raster	4 x 4 km	1997–2012
March minimum temperature	average monthly minimum temperature (°C)	PRISM [38]	Raster	4 x 4 km	1997–2012
October minimum temperature	average monthly minimum temperature (°C)	PRISM [38]	Raster	4 x 4 km	1997–2012

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We also obtained a map of the presence of the 1940s infestation within the Flat Tops study area [5]. To our knowledge no other maps of the 1940s outbreak exist for the Southern Rocky Mountains. Maps of the 1940s infestation were developed from visual stereoscopic examination of 1971 color and 1984 IR aerial imagery (minimum mapping unit 5 ha). Stands mapped as infested by spruce beetles during the 1940s were defined as stands in which >30% of canopy trees were dead [5]. Spatial polygon data on the occurrence of the 1940s infestation was then converted to a 990 x 990 m grid listing the presence of spruce beetle infestation and multiplied by the raster of spruce-fir presence to obtain a cross-validated grid of spruce beetle infestation [24].

Finally we obtained spatial data on climate and forest structure variables, which were hypothesized to be important in predicting the occurrence of spruce beetle infestation. We obtained gridded monthly precipitation and temperature data from the Parameter-elevation Regressions on Independent Slopes Model (PRISM; [38]) (Table 1). To determine if warm and dry weather was associated with infestation, we calculated the 1997–2012 means of maximum August temperature and total annual precipitation, which previous research has shown to predict occurrence of spruce beetle infestations [25,29,45]. To determine if anomalously cold weather during the late autumn to early spring was associated with the presence/absence of infestation we calculated the 1997–2012 means of minimum October and March temperature, which are understood to inhibit infestation [25,45]. Next, we obtained vegetation layers depicting the mean diameter at breast height (DBH) size classes for the dominant canopy species, which were created from manual aerial photo interpretation of 1-m resolution color aerial photographs in 2002 (Table 1).

Determining the biophysical drivers of spruce beetle infestation

We used two methods to assess the biophysical variables driving the spatial variability in the occurrence of 1997–2012 spruce beetle infestation in the Southern Rocky Mountain study region. First, we used a spatial overlay approach [46,47], where spatial data on spruce beetle infestation were compared with spatially explicit climate and forest structure data (Table 1). We used spatial overlays to calculate the conditional probability of the presence/absence of spruce beetle infestation given each value of the independent variable. Conditional probability is a measure of the probability of the dependent variable (presence or absence of spruce beetle infestation) occurring given each value of the independent variable (biophysical variables). Continuous climate variables were first binned into four equal-interval classes [48]. Then we tabulated the number of 990 x 990 m pixels of all values of each independent variable that occurred in uninfested and infested areas and calculated the conditional probability of infestation. The null hypothesis is that spruce beetle infestation is independent of all values of each independent variable and thus observed conditional probabilities of infestation should equal conditional probabilities of uninfested stands. Our spatial overlays assessed entire populations and not samples. Thus all deviations between conditional probabilities are viewed as real differences between the datasets and statistical tests are not necessary. However, given that our spatial datasets exhibit classification error, we conservatively assumed that only differences greater than 10% are meaningful (e.g. [46]).

Second, to complement our conditional probability analysis of univariate relationships between biophysical predictors and the presence/absence of spruce beetle infestation, we used a Conditional Inference Framework (CIF; [49]) to assess multivariate relationships. CIF is similar to Random Forests [50] in that many classification trees are constructed by dividing the data into increasingly homogenous groups based on splits in the independent variables [49,51,52]. Classification trees are useful for detecting nonlinear relationships and interactions between variables [51]. In contrast to Random Forests where variable selection is based on the maximization of an information criterion (e.g. Gini coefficient), CIF uses conditional permutation-based significance tests to select variables [49]. This decreases selection bias in cases where independent variables have substantially different numbers of potential splits (e.g. categorical vs. continuous independent variables) [53], or where independent variables are correlated [54]. To evaluate the variables most important for predicting the presence/absence of spruce beetle infestation, we calculated conditional variable importance scores, a measure of each independent variable's contribution to overall model fit [54]. Because the calculation of conditional variable importance is computationally intensive, we randomly selected 2000 cases, stratified by spruce beetle infestation (1000 infested; 1000 uninfested). Model accuracy was asssed using overall accuracy and model sensitivity and specificity.

Effects of the 1940s spruce beetle infestation on the 1997–2012 infestation

To determine if the effects of the 1940s spruce beetle infestation on forest structure may affect the susceptibility of a stand to subsequent infestation in 1997–2012, we first used our model of the presence/absence of spruce beetle infestation to determine the relative importance of forest structure versus climate variables in constraining infestation within the Flat Tops study area. We tabulated the number of pixels within each model node and evaluated the relative importance of splits in climate vs. forest structure variables in predisposing the Flat Tops study area to infestation in 1997–2012. To this end, we calculated the percent of pixels in each model node for the entire Southern Rocky Mountain Study region and just the Flat Tops study area (Southern Rocky Mountain Study Region % | Flat Tops study area %). If the percent of pixels that met the condition were greatly different (>10%) for the Flat Tops study area than for entire Southern Rocky Mountain Study region, then that condition was interpret to be disproportionately important in constraining/promoting infestation within the Flat Tops study area.

Next, we coupled fine-scale field data with stand-level spatial data to determine if forest structure was altered by previous spruce beetle infestation. First, to test if large trees were depleted in areas of the 1940s infestation, we tabulated the number of 990 x 990 m pixels of all values of tree size that occurred in areas with and without 1940s infestation [55]. Then we calculated the conditional probability of the dominant tree size class (2.5–12.4, 12.5–22.9, 23–40.4, or $\geq$40.5 cm DBH) given the presence/absence of 1940s infestation.

Because the available GIS dataset depicting tree size is not species specific, we collected stand-scale (0.01 ha) field data to determine the delayed effects of a severe spruce beetle infestation on species composition. Field data were collected in the summer of 2013 at 7 sites (4 sites without evidence of 1940s infestation and 3 sites with evidence of severe spruce beetle infestation in the 1940s) across the Flat Tops study area. Plots were located using maps of the presence/absence of the 1940s infestation [5]. We field verified that our sites were located in areas affected by the 1940s infestation by locating large, dead, standing snags with spruce beetle galleries. At each site we collected data from a cluster of 10 randomly-located 100 m^2 plots. For each tree in the plot, we recorded the species, the diameter at breast height (DBH), and tree status (live, dead, or fallen). We then aggregated data for stands that experienced and did not experience severe spruce beetle infestation in the 1940s and calculated the 2000s density of live spruce and fir. We then compared 2000s stand structure and composition in stands uninfested and infested during the 1940s.

Finally, we used spatial data to assess if these structural differences between stands uninfested and infested during the 1940s affected the distribution of 1997–2012 infestation within the Flat Tops. We overlaid a 990 x 990 m grid of 1997–2012 spruce beetle infestation presence/absence with a 990 x 990 m grid of 1940s infestation presence/absence and calculated the area of overlap.

Results

Biophysical drivers of the 1997–2012 spruce beetle infestation

Across the Southern Rocky Mountain study region, spruce beetles infested approximately 15% of the spruce-fir zone over the period from 1997–2012 (areas mapped as infested in ADS surveys 1998–2013; Fig 1A). Over this time period, the Flat Tops study area has experienced very little infestation (2% of the spruce fir-zone recorded presence of infestation; Fig 1). While the annual area infested by spruce beetles across the Southern Rocky Mountain study region has been growing since 1998 [55], ADS data indicate that most spruce beetle activity in the Flat Tops study area occurred prior to 2005 (S1 Fig).

Across the Southern Rocky Mountain study region, spatial overlay analysis revealed meaningful differences between the conditional probabilities of uninfested and infested spruce-fir forest given climate and forest structure variables (Fig 2). Contrary to expectations, spruce beetle infestation was less likely in areas with high maximum August temperatures ($\geq$19.5°C; Fig 2B). However this difference was only meaningful in areas where the average maximum temperature was greater than $\geq$20.5°C. Areas with cooler maximum August temperatures ($<$18.5°C) were more likely to be infested. Also contrary to expectation, areas with high annual precipitation ($\geq$ 1050 mm/year) were more likely to experience spruce beetle infestation, while areas with moderately low annual precipitation (650–849 mm/year) were less likely to experience infestation (Fig 2A). There were no meaningful differences between the probabilities of uninfested and infested forest given any of the four classes of minimum March temperature or minimum October temperature (Fig 2C and 2D).

We also found that forest structure differed between forests uninfested and infested by spruce beetles in 1997–2012 (Fig 2E). Spruce beetle infestation was more likely to occur in areas with

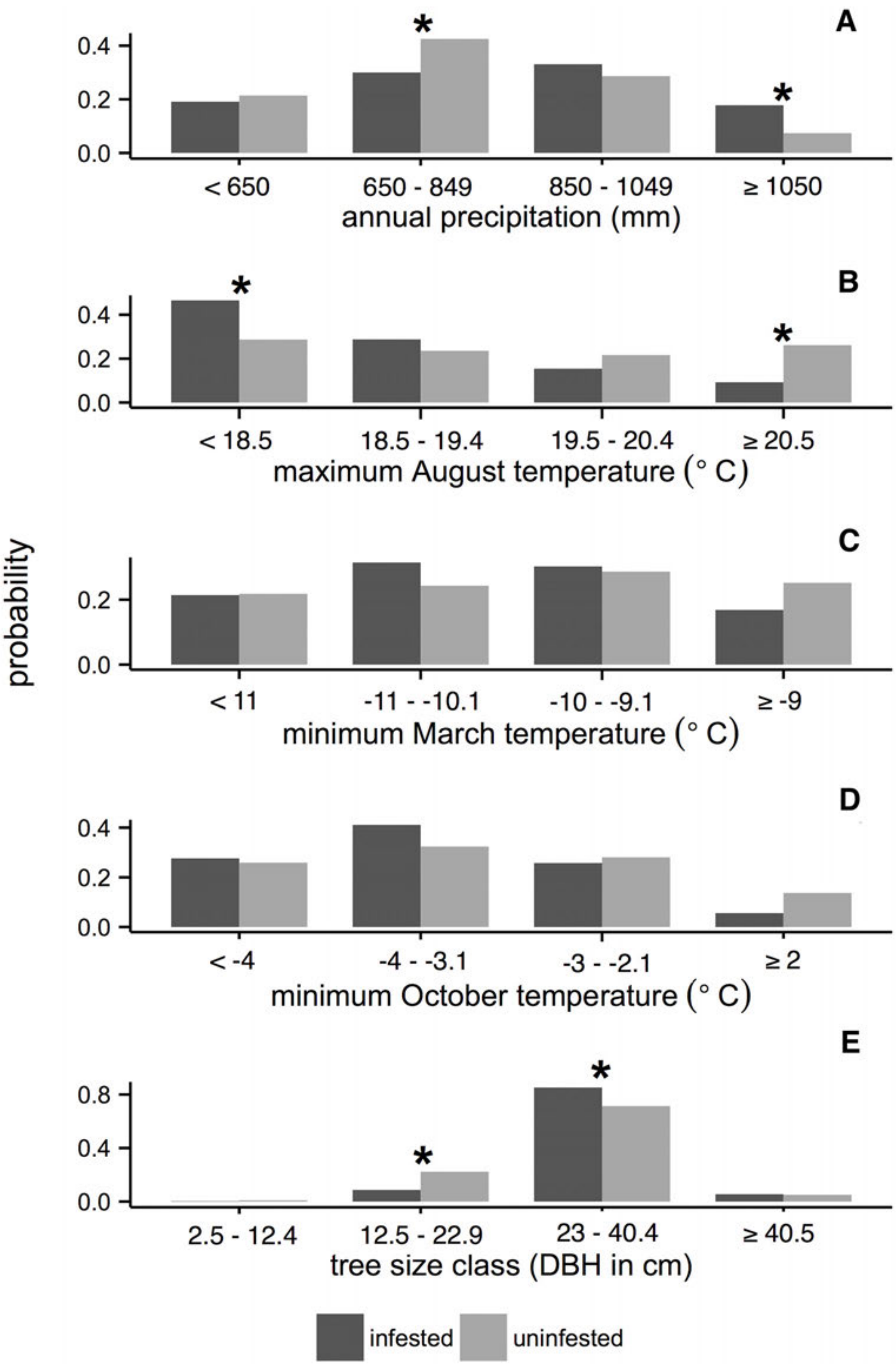

Fig 2. Conditional probabilities of the presence/absence of spruce beetle infestation (1997–2012) given selected bioclimatic variables in the Southern Rocky Mountains study region. (A) annual precipitation, (B) maximum August temperature, (C) minimum March temperature, (D) minimum October temperature, and (E) tree size class for uninfested and infested stands. Dark gray bars indicate conditional probability of spruce beetle infestation given that value of a bioclimate variable across the Southern Rocky Mountain study region. Light gray bars indicate the conditional probability of uninfested forest. The asterisk symbol (*) above a pair of bars indicates a meaningful difference between conditional probability of uninfested and infested forest (i.e. difference > 10%, see *Methods* for more description). Note y-axes extend over different ranges.

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large diameter trees (≥ 23 cm DBH; Fig 2E). For stands with smaller diameter trees (< 23 cm DBH), the probability of infestation was < 0.22 (Fig 2E).

The multivariate model of 2000s spruce beetle uninfested and infested forest performed reasonably well. The CIF model correctly predicted 809 of the 1000 pixels with spruce beetle infestation (i.e. sensitivity = 0.81), and correctly predicted 819 of the 1000 pixels without spruce beetle infestation (i.e., specificity = 0.82). Variables important in predicting 2000s spruce beetle infestation included maximum August temperature, annual precipitation, and tree size class (Fig 3A). Spruce beetle infestation was unlikely to occur in areas with maximum August temperatures above 20.3°C (probability of infestation = 0.276). Infestation was particularly unlikely when temperatures exceed 21.6°C (probability of infestation = 0.164; Fig 3B). However, more than 75% of the study area was characterized by 1997–2012 mean maximum August temperatures cooler than 20.3°C (S2 Fig). In these areas, spruce beetle infestation was particularly like to occur in areas with large trees (≥ 23 cm DBH; Fig 3A) and high precipitation (>1063 mm/year) (Fig 3B and S2 Fig).

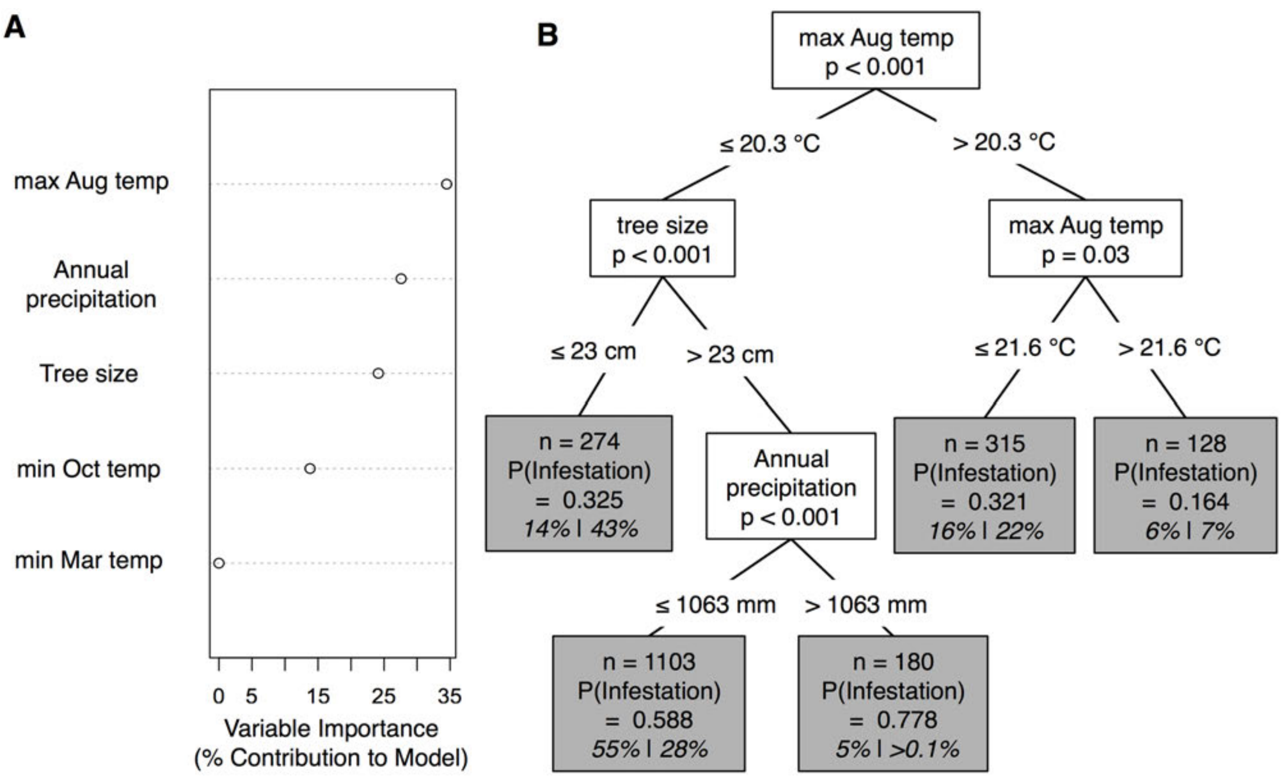

Fig 3. Results from conditional inference forest analysis of the presence/absence of spruce beetle infestation with climate and forest structure data in the Southern Rocky Mountain study region. (A) Conditional variable importance for the five biophysical variables used to model the occurrence of spruce beetle infestation across the Southern Rocky Mountain study region. Conditional variable importance scores were calculated following the Random Forest principle of mean decrease in accuracy and then transformed to express the contribution of each variable to the overall model. Higher values indicate variables are more important to the classification. Conditional variable importance scores represent 1000 model runs. All trees were built using a random sample of 2000 cases, stratified by the presence/absence of spruce beetle infestation (1000 infested and 1000 uninfested). Overall prediction accuracy is 81%. (B) A classification tree for determining the presence of spruce beetle infestation from uninfested spruce-fir stands across the Southern Rocky Mountains study region. On the tree, if condition is satisfied, proceed to the left of the tree. Tree nodes (gray boxes) describe the number of pixels across the entire Southern Rocky Mountain study region that meet the condition and the probability of spruce beetle infestation. The gray boxes also list the percent of pixels that meet the conditions for the entire Southern Rocky Mountain Study region and just the Flat Tops study area (Southern Rocky Mountain Study Region % | Flat Tops study area %). If the percent of pixels that meet the condition are greatly different (>10%) for the Flat Tops study area than for entire Southern Rocky Mountain Study region, then that condition is disproportionately important in constraining/promoting infestation within the Flat Tops study area.

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Effects of the 1940s spruce beetle infestation on the 1997–2012 infestation

Applying the decision tree to the pixels within the Flat Tops study area provided insight into the biophysical predictors important in constraining infestation in the Flat Tops study area. Across the Flat Top study area about 29% of pixels were characterized by 1997–2012 mean maximum August temperatures unsuitable for infestation (<20.3°C; Fig 3B and S2 Fig). An additional 43% of the pixels within the Flat Tops study area were characterized by small diameter trees (<23 cm DBH), which inhibit infestation (Fig 3B and S2 Fig). In comparison to the entire Southern Rocky Mountain study region, the percent of pixels with small diameter trees (<23 cm DBH) in the Flat Tops study area was three times greater (43% vs. 14%, for the Flat Tops study area and entire Southern Rocky Mountain study region, respectively; Fig 3B and S2 Fig). As a result, the percentage of pixels that were split based on annual precipitation was far lower for the Flat Tops study than the Southern Rocky Mountain study region.

Within the Flat Tops study area, comparison of forest structure of 990 x 990 m in areas uninfested and infested by the 1940s infestation indicates that infested stands are characterized by smaller tree sizes (12.5–22.9 cm DBH) 60 years following infestation (Fig 4). This coarse-scale finding based on mapping from aerial photographs (Table 1) is supported by stand-level field measurements. During the 1997–2012 period of spruce beetle infestation, field data revealed that in comparison with stands infested during the 1940s, stands not infested in the 1940s had consistently higher densities of spruce in all size classes including the largest class (i.e. $\geq$ 40.5 cm DBH; Fig 5). In contrast, 60 years following the 1940s spruce beetle outbreak

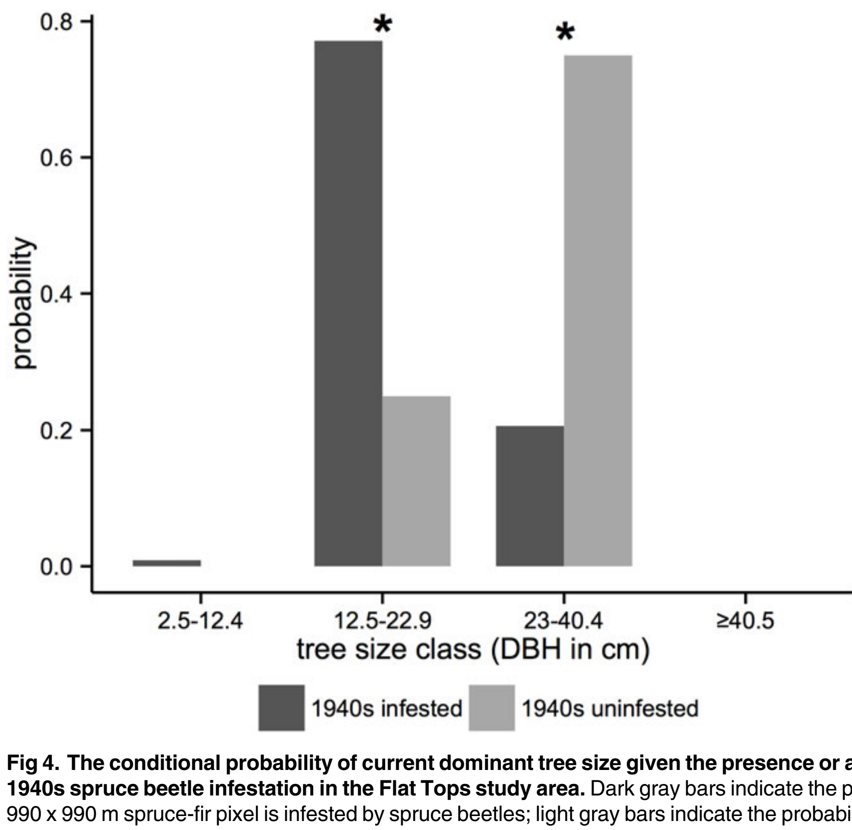

Fig 4. The conditional probability of current dominant tree size given the presence or absence of the 1940s spruce beetle infestation in the Flat Tops study area. Dark gray bars indicate the probability that a 990 x 990 m spruce-fir pixel is infested by spruce beetles; light gray bars indicate the probability a pixel is uninfested. The asterisk symbol (*) above a pair of bars indicates a meaningful difference between conditional probability of uninfested and infested forest.

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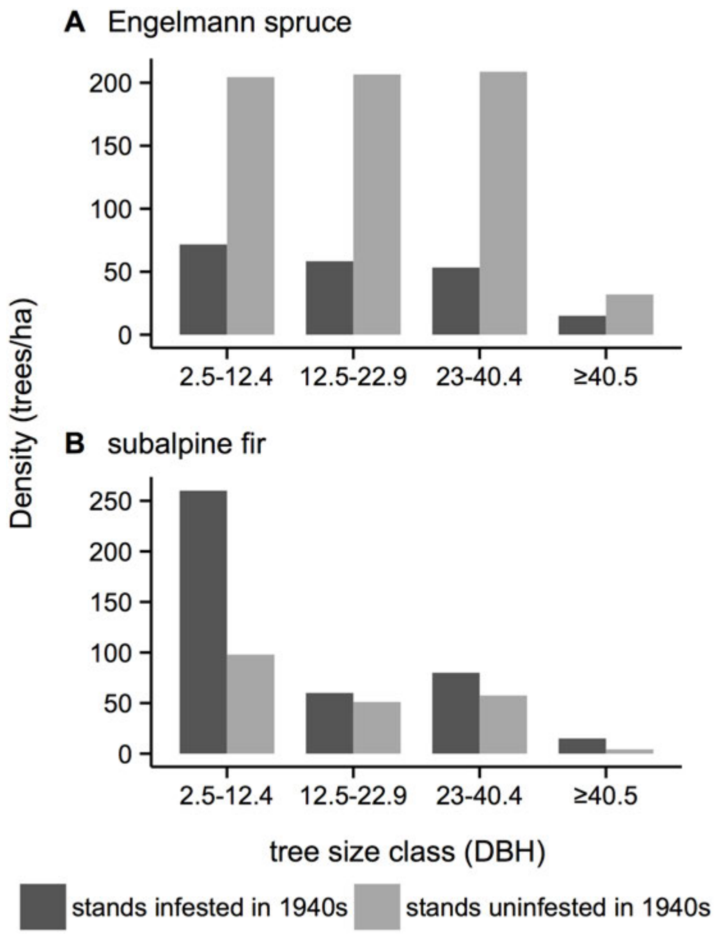

Fig 5. Current (2000s) tree size class distributions in stands uninfested and infested during the 1940s infestation within the Flat Tops and adjacent areas of White River National Forest. Data represent the aggregate of all plots (stands uninfested during the 1940s outbreak, n = 4 sites each with 10 ca. 100 m^2 plots; stands infested during 1940s outbreak, n = 3 sites each with 10 ca. 100 m^2 plots).

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subalpine fir was more abundant in all size classes in stands infested during the 1940s outbreak compared to uninfested stands. Concordantly, we found no overlap between areas infested during the 1940s and the 2000s infestation (Fig 1A). Within the Flat Tops region only three 990 x 990 m pixels were infested in 1997–2012, but none of those overlapped with the 254 pixels infested in the 1940s. Instead, all pixels infested in 1997–2012 were located in areas with large diameter trees ($\geq$ 23 cm DBH).

Discussion

Across the Southern Rocky Mountain study area, spruce beetle infestation was more likely to occur in areas with cool to moderately warm mean maximum August temperatures and higher amounts of annual precipitation. Although these results at first glance seem counter-intuitive given the importance of drought in triggering spruce beetle outbreaks [28–30,56], our results are spatial associations of infestation with mean conditions rather than temporal associations with drought events measured as departures from longer-term average conditions. While bark beetles preferentially attack drought-stressed trees [28, 29], our results describe habitat suitability for spruce beetle, which clearly is greater at the cooler and wetter sites where spruce is more common. In contrast, warmer sites are likely to be characterized by greater proportions of non-host species (e.g. lodgepole pine) and provide less potential for spruce beetle outbreak. Overall, we interpret the association of spruce beetle infestation with cooler and wetter sites as being explained primarily by the greater presence of host species at those sites.

Across the Southern Rocky Mountain study area, spruce beetle infestations have occurred overwhelmingly in spruce-fir stands dominated by large trees ($\geq$ 23 cm DBH). This corresponds with empirical results from Grand Mesa National Forest in western Colorado, which showed early 2000s spruce beetle infestation was significantly more likely in spruce larger than 24 cm DBH [57]. Spruce beetles prefer large trees, which provide both higher amounts of phloem for beetles to feed upon and thicker bark that increases overwinter survival rates [32].

While both climate and forest structure interacted to drive the occurrence of spruce beetle infestation across the Southern Rocky Mountains, our data suggest that the current infestation in the Flat Tops was severely constrained by a low proportion of large trees ($\geq$ 23 cm DBH). Our model suggests that climate variables were conducive to bark beetle infestation across most of the Flat Tops study area. Relative to the entire Southern Rocky Mountains (inclusive of the Flat Tops), infestation in the Flat Tops was severely constrained by forest structure. The paucity of large diameter trees within the Flat Tops study area was a result of a severe spruce beetle outbreak that occurred 60 years ago. Stands infested during the 1940s in Flat Tops were notably depleted of large spruce relative to uninfested stands. Given the preference of spruce beetles for large diameter spruce and relative absence of large diameter spruce in areas affected by the 1940s infestation, it is not surprising that we found no overlap between areas of current infestation and areas affected by the 1940s infestation. These results support the hypothesis that stands affected by severe spruce beetle infestation are less susceptible to infestation c. 60 years later due to a decrease in large diameter spruce.

The 1940s spruce beetle infestation in northwestern Colorado was most severe in the Flat Tops area, where three-quarters of the 1940s spruce beetle-induced tree mortality occurred [32]. Nearby spruce-fir forests in Grand Mesa National Forest also experienced 1940s spruce beetle infestation, however it was significantly less severe [32,58]. For instance, the basal area of beetle-killed spruce was ca. 4—7.5x greater in the Flat Tops than in Grand Mesa [58]. In contrast, the 1997–2012 spruce beetle infestation has affected 2% of the spruce-fir forest in Flat Tops and 19% of Grand Mesa's spruce-fir forest [34]. This suggests that the 1940s infestation in Grand Mesa was not severe enough to cause significant host depletion and thus Grand Mesa forests were much more susceptible to the 1997–2012 infestation.

Our study is notably limited by the availability of spatial datasets of both the 2000 and 1940s spruce beetle infestation. In particular we note that comparisons between these two datasets may be limited by the different methods used to map spruce beetle infestation (interpretation of aerial photography vs. aerial sketch mapping). However, our ability to accurately model the 1997–2012 infestation from a few ecologically meaningful biophysical predictors and the agreement between field data and maps of the 1940s outbreak suggest these datasets were appropriate for coarse assessment of the linkage between spruce beetle outbreaks. Subsequent analyses with datasets depicting severity of infestation at a fine spatial resolution would serve to advance our understanding of this linkage, however to our knowledge no such datasets exist for the Southern Rocky Mountains.

The findings of the current study indicate that at a broad spatial scale, severe spruce beetle outbreaks are linked disturbances (sensu [3]) at least over the 60-year period considered in our study. We suggest that the host depletion feedback not only may cause infestation collapse (sensu [32]), but may enhance ecological resistance (sensu [59]) of beetle-affected systems to spruce beetle infestation through long lasting effects of host depletion. Given that predictions of future beetle disturbance from climate-driven beetle population models do not incorporate process dynamics of disturbance-caused tree mortality and forest recovery [60], our results underscore the need for additional research on forecasting future forest dynamics, which may affect host availability for bark beetle infestations. In particular, the dampening effect of the 1940s spruce beetle infestation on the spread of the early 2000s infestation in the Southern Rocky Mountains implies that future infestations in the 21st century may be similarly restricted by disturbance-caused depletion of susceptible hosts.

Most previous studies of linked disturbances in the coniferous forests of the Rocky Mountain region have addressed how previous fire affects subsequent bark beetle outbreaks [61,62] or how previous bark beetle outbreaks alters the probability, extent or severity of subsequent fire [5,15,20,36]. To our knowledge this is the first broad-scale analysis of how prior bark beetle outbreak affects susceptibility to subsequent bark beetle outbreak. Our findings of a dampening effect of the 1940s spruce beetle outbreak on susceptibility to spruce beetle infestation 60 years later highlights the need for incorporating the process dynamics of tree growth and mortality in predictive modeling of the likelihood of bark beetle outbreaks under future climate scenarios. Simulation modeling of the probability of future insect outbreaks based on climate suitability for the growth of the insect populations has been important in identifying likely trends over relatively short time periods. However, our results show that even at a time scale of 60 years, failure to incorporate negative feedbacks into prediction of future bark beetle outbreaks is likely to over-predict the extent or severity of future outbreaks and by implication under-estimate forest resistance to altered disturbance regimes under climate change.

Supporting Information

S1 Fig. Time series displaying the percentage of spruce-fir forest infested by spruce beetles. Data is shown for the Southern Rocky Mountain Ecoregion (inclusive of the Flat Tops) and only in the Flat Tops. For each region, the percent area was calculated by the determining the number of 990 x 990 m pixels within the spruce-fir zone identified as infested by the United States Forest Service in annual Aerial Detection Surveys (ADS) and dividing it by the total number of spruce-fir pixels.
(TIFF)

S2 Fig. The importance of biophysical predictors in promoting spruce beetle infestation the Southern Rocky Mountain study region. Maps of (A) mean maximum August temperature (1997–2012), (B) mean annual precipitation (1997–2012), and (C) tree size class. The probability of infestation (derived from the classification tree in Fig 3B) is indicated by pixel color. Dark green indicates a probability of infestation <0.3, light green indicates a probability of infestation of 0.3–0.49, dark yellow indicates a probability of infestation 0.5–0.69, and dark brown indicates a probability of infestation >0.7. Sources are given in Table 1.
(TIFF)

Acknowledgments

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Author Contributions

Conceived and designed the experiments: SJH TTV NM DK. Analyzed the data: SJH. Wrote the paper: SJH TTV NM DK. Collected field data: NM.

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Conservation Planning for US National Forests: Conducting Comprehensive Biodiversity Assessments

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The US Forest Service has proposed new regulations under the National Forest Management Act that would replace a long-standing requirement that the agency manage its lands "to maintain viable populations of existing native and desired non-native vertebrate species." In its place, the Forest Service would be obligated merely to assess ecosystem and species diversity. A landscape assessment process would rely on ecosystem-level surrogate measures, such as maps of vegetation communities and soils, to estimate species diversity. Reliance on such "coarse-filter" assessment techniques is problematic because there tends to be poor concordance between species distributions predicted by vegetation models and observations from species surveys. The proposed changes would increase the likelihood of continued declines in biodiversity and fail to address the original intent of the act. We contend that responsible stewardship requires a comprehensive strategy that includes not only coarse-filter, ecosystem-level assessment but also fine-filter, species-level assessments and viability assessments for at-risk species.

Keywords: forestry, forests, management, policy, conservation

The US National Forest Management Act (NFMA) is an essential statute for maintaining biotic diversity on 192 million acres of national forests and national grasslands. It was enacted in 1976 as reform legislation in response to environmental impacts from timber harvest, grazing, and mining on national forest lands, which the public and Congress found increasingly unacceptable (Wilkinson and Anderson 1987). Among many provisions for resource protection, a primary emphasis was the protection of individual species. The statutory language of NFMA requires management of the national forests and grasslands to "provide for diversity of plant and animal communities based on the suitability and capability of the specific land area in order to meet overall multiple-use objectives" (16 US Code 1604[g][3][B]). Since 1982, the regulations governing implementation of NFMA have addressed this diversity provision by requiring that "fish and wildlife habitat shall be managed to maintain viable populations of existing native and desired non-native vertebrate species in the planning area" (36 Code of Federal Regulations, sec. 219.19, app. 13). Revisions to NFMA regulations adopted in 2000 retained the requirement for viable populations and expanded it to include all plant and animal species (Federal Register 65 [218]: 67514–67581).

Although NFMA has remained essentially unchanged since its enactment, the US Forest Service has now proposed regulations that eliminate an explicit population viability requirement and that restrict management responsibility to vertebrates and vascular plants (Federal Register 67 [235]: 72770–72816). The proposed regulations require only a "hierarchical, sequential approach to consider and assess both ecosystem diversity and species diversity" and that the Forest Service "identify species for which substantive evidence exists that continued persistence in the planning or assessment area is at risk, specific risks or threats to these species, and measures required for their conservation or restoration" (Federal Register 67 [235]: 72801). No specific language to compel species-level analyses of viability has been proposed. Moreover, the proposed regulations would subsume the existing species conservation requirement into a landscape assessment process that would use a variety of unproven

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ecosystem-level surrogates to estimate species diversity without necessarily examining the condition or status of individual species. Although not explicitly stated, the substance of these proposed regulations hinges on two underlying assumptions: (1) Land-use planning that relies solely on such "coarse-filter" (Hunter et al. 1988) approaches to assess the distributions and status of ecological communities is adequate to assess how well the needs of all their constituent species will be met, and (2) the uncertainty that accompanies indirect assessments of species status provided by coarse-filter tools is acceptable because species-level assessments are too difficult or too expensive to implement. These assumptions are not only counter to current understanding of the role and dynamics of specific species in sustaining ecosystem processes (e.g., Kinzig et al. 2002), they also negate the nature and appropriate role of population viability analyses in land-use planning.

Inadequacies of assessments employing only a coarse-filter approach

To understand the functioning of any complex system, it is necessary to identify and attempt to elucidate the parts that it comprises. For ecological systems, the most fundamental "parts" are species. Sir Arthur Tansley originally defined ecosystems as biotic communities or assemblages of species and their physical environment in specific places (Tansley 1935). Directly contradicting this view of ecosystems as collections of interacting species, the proposed regulations focus resource assessments almost entirely on vegetation types and successional stages, geology, landforms, and soils. The logic behind this coarse-filter approach is that the majority of species can be protected by conserving examples of natural vegetation communities, obviating the need to evaluate the status of each species individually (Noss 1987, Noss and Cooperrider 1994).

The original intent of coarse-filter approaches to landscape planning was to provide distribution maps of land cover that could be used to inform the conservation of entire species assemblages, including communities of interacting or potentially interacting species (Jennings 2000, Groves et al. 2002). Broadscale applications of coarse-filter methods have relied on ecoregional classifications determined by a variety of measures of climate, substrate, and plant composition. However, they commonly and often exclusively default to dominant vegetation, because vegetation types can be assessed by remote-sensing technologies and have been linked, using general habitat models, to the distributions of many vertebrate species (Scott et al. 1993). For example, recent planning efforts by the Forest Service for 4.4 million hectares of public forests and grasslands in the Sierra Nevada of California assessed the effects of various management alternatives on vertebrate species using wildlife–habitat relationship models (Mayer and Laudenslayer 1988) to classify habitats based on three attributes—dominant vegetation type, successional stage, and canopy closure. When these models were coupled with a vegetation growth and yield model (Davis and Johnson 1987), they allowed a comparison of how competing forest management scenarios would be likely to affect future wildlife populations (Forest Service 2001).

Coarse-filter approaches to assess the viability of species for land-use planning purposes can provide cost-efficient, indirect methods of assessing species distributions, but to assess the viability of species, at least three assumptions must hold true: (1) Attributes that define the coarse filter (i.e., dominant vegetation types) are sufficient and reliable surrogates for habitat and can effectively predict the occurrence of a given species; (2) managing coarse-filter attributes will address the factor(s) currently limiting abundance, density, and persistence of each species; and (3) the spatial resolution of the coarse filter matches the scale at which given species respond to environmental heterogeneity. Although these assumptions may be valid for some species in many circumstances, especially species that are small-bodied, abundant, and tightly linked to a particular vegetation community, the likelihood that the assumptions are met for all, or even most, species in an assemblage is low. For that reason, landscape planning employs "fine-filter" assessments, which are based on direct measures of the status and trends of individual species or on models of population viability to evaluate the needs of species at risk of decline.

The utility of the coarse-filter approach has been tested for many individual species with equivocal success (see Scott et al. [2002]). In general, there has been poor concordance between predicted and observed distributions. Commission errors (false positives, or predictions that a species is present when it is absent) have been shown to be more common than omission errors (false negatives, or predictions that a species is absent when it is present) at spatial scales appropriate to regional conservation planning—for example, vertebrates in the state of Maine and in national parks in Utah and breeding birds in California (Edwards et al. 1996, Boone and Krohn 1999, 2000, Garrison et al. 2000, Garrison and Lupo 2002, Robertson et al. 2002). Thus, coarse-filter assessments often overestimate the presence and, presumably, the viability of species on the planning landscape.

Only by increasing the resolution of the coarse filter (which reduces the area predicted to be suitable habitat for the species), as well as the number of land-cover types (usually by stratifying the vegetation communities more finely), can commission and omission errors be simultaneously reduced (Karl et al. 2000). Prediction errors are also related to ecological attributes of a species: Species that are rare, colonial, or habitat specialists, or that have small home ranges, are most likely to be misclassified (Karl et al. 2000, Scott et al. 2002). The misclassified groups of species usually include those most likely to be at risk of population declines or extirpation—that is, those that should be targets of conservation planning efforts (McKinney 1997). In sum, these prediction errors suggest that employing a coarse-filter approach alone is inadequate to meet NFMA requirements to provide for the diversity and viability of plant and animal communities.

Integrating the fine filter with population viability analysis

Coarse- and fine-filter approaches to conservation planning differ in both the extent and resolution of measurement employed and the targeted level of biological organization. In general, mapped coarse-filter attributes reflect higher-level processes and patterns that arise, for example, from disturbance processes that operate across entire landscapes. For pragmatic reasons, coarse-filter attributes considered during the planning process are often those that can be measured inexpensively using remote imagery. Coarse filters rarely will accurately reflect the complex and dynamic habitat requirements of any individual species. In contrast, a fine filter makes measurements directly at the species level for the subset of species whose habitat requirements were not captured by the attributes that define the coarse filter.

Neither coarse- nor fine-filter assessments alone can prescribe the extent or area of habitat necessary to maintain viable populations of plant and animal species on the landscape. Many rare and declining species are limited primarily by the availability of suitable habitat (Wilcove et al. 1998), and the viability of such species depends to a great extent on how much of their habitat is conserved. Population viability analysis (PVA) is an in-depth method of fine-filter assessment used to evaluate habitat loss or similar risk factors for specific species (Boyce 2002, Shaffer et al. 2002).

An assessment approach that includes both coarse and fine filters and PVA was recommended by the Committee of Scientists to the US Forest Service and incorporated into the 2000 NFMA regulations (COS 1999). In addition to rare and at-risk species, the committee recommended that two groups of species be evaluated using fine filters—those that provide comprehensive information on the state of a given ecosystem (indicator species) and those that play significant functional roles in ecosystems (focal species). The latter category includes species that contribute disproportionately to the transfer of matter and energy (e.g., keystone species), structure the environment and create opportunities for additional species (e.g., ecological engineers), or exercise control over competitive dominants, thereby promoting increased biotic diversity (e.g., strong interactors). Thus, fine-filter assessments might be needed for 10 to 50 of the 200 to 1100 species typically evaluated in regional planning efforts carried out by the Forest Service and may need to include select invertebrates as well as vertebrates and plants.

Formal PVAs are needed only for species in decline or at high risk or for species with such functional significance that their loss might have unacceptable ecological effects. Many methods of viability assessment exist to accommodate diverse sources and amounts of data (Beissinger and Westphal 1998, Andelman et al. 2001). All methods explicitly or implicitly require some sort of model that relates population dynamics to environmental variables, including variables affected by management. The range of available methods offers a tradeoff between complexity of analysis and generality of results.

Population viability analysis is neither inherently difficult nor expensive, but it does require thoughtful model choice and construction and good judgment in the implementation of analyses. Perhaps the most demanding aspect of building realistic PVA models for assessment of alternative management scenarios is acquisition of sufficient data to yield accurate and precise parameter estimates (Beissinger and Westphal 1998). These models then permit reliable assessments of alternative management scenarios (Noon and McKelvey 1996). The choice of models and data collection methods depends in part on the life history characteristics of the species to be assessed, the quality and quantity of existing data, the time and money available for additional data acquisition, and the resolution and extent of analysis (Beissinger and Westphal 1998, Andelman et al. 2001). A method that uses a formal mathematical model of analysis is often preferable to less quantitative methods for analyzing viability when there is sufficient knowledge of demography, dispersal, habitat use, and threats.

Currently, population viability analyses are required to address the viability requirements of NFMA. In the context of the act, viable populations consist of "self-sustaining and interacting populations that are well distributed through the species' range. Self-sustaining populations are those that are sufficiently abundant and have sufficient diversity to display the array of life history strategies and forms to provide for their long-term persistence and adaptability over time" (Federal Register 65 [218]: 67580–67581). Many population attributes included in this definition can be evaluated using population viability analyses, but they cannot be addressed solely through the application of coarse-filter analyses.

A scientifically credible approach to national forest planning

An expert panel convened by the National Center for Ecological Analysis and Synthesis, at the request of the Forest Service, concluded that "viability assessment is an essential component of ongoing forest management and forest planning processes. A variety of methods can and should be incorporated into viability assessments" (Andelman et al. 2001, p. 136). A scientifically credible approach to management of a diversity of plant and animal communities in US national forests and national grasslands combines coarse-filter and fine-filter approaches to identify conservation targets, including the judicious use of PVA for focal species and species at risk. Scientifically valid and pragmatic management does not require that the status of all species be directly assessed. But failure to detect declining species and to address the putative threats to their persistence leaves only the prohibitive provisions of the Endangered Species Act to serve as a safety net.

Although coarse-filter, fine-filter, and PVA assessment tools are imperfect, their weaknesses are sufficiently understood that the information they provide is, on balance, useful, and the Forest Service's failure to require their use is irresponsible. Insights provided by the use of these tools will inform managers about the condition of the ecosystems they are charged with protecting and the likely consequences of the management decisions they are empowered to make. Acting on these insights to change management practices when needed will aid biodiversity conservation and enable the Forest Service to meet its stewardship responsibilities.

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ECOSYSTEMS

Fire Severity in Conifer Forests of the Sierra Nevada, California

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Abstract

Natural disturbances are an important source of environmental heterogeneity that have been linked to species diversity in ecosystems. However, spatial and temporal patterns of disturbances are often evaluated separately. Consequently, rates and scales of existing disturbance processes and their effects on biodiversity are often uncertain. We have studied both spatial and temporal patterns of contemporary fires in the Sierra Nevada Mountains, California, USA. Patterns of fire severity were analyzed for conifer forests in the three largest fires since 1999. These fires account for most cumulative area that has burned in recent years. They burned relatively remote areas where there was little timber management. To better characterize high-severity fire, we analyzed its effect on the survival of pines. We evaluated temporal patterns of fire since 1950 in the larger landscapes in which the three fires occurred. Finally, we evaluated the utility of a metric for the effects of fire suppression. Known as Condition Class it is now being used throughout the United States to predict where fire will be uncharacteristically severe. Contrary to the assumptions of fire management, we found that high-severity fire was uncommon. Moreover, pines were remarkably tolerant of it. The wildfires helped to restore landscape structure and heterogeneity, as well as producing fire effects associated with natural diversity. However, even with large recent fires, rates of burning are relatively low due to modern fire management. Condition Class was not able to predict patterns of high-severity fire. Our findings underscore the need to conduct more comprehensive assessments of existing disturbance regimes and to determine whether natural disturbances are occurring at rates and scales compatible with the maintenance of biodiversity.

Key words: Condition Class; ecological restoration; Jeffrey and ponderosa pine; fire rotation interval; fire severity; fire spread; mixed conifer forests; spatial heterogeneity.

Introduction

The diversity of species in ecosystems is linked to natural disturbances and the environmental heterogeneity they create (Connell 1978; Huston 1979). However, managing the rates and scales of disturbance processes to allow for natural levels of environmental heterogeneity has its inherent risks and difficulties. This is particularly true for large disturbances that have profound influences on ecosystem structure, function, and composition (Turner and Dale 1998). Thus, although natural disturbances are vital to ecosystem integrity, maintaining their full range of variability is often at odds with management (Holling and Meffe 1996). How can disturbance-mediated environmental heterogeneity be most effectively maintained or restored where it has been suppressed over large areas? How can we recognize the levels and types of disturbance and heterogeneity that are appropriate for maintaining biodiversity? Here we explore these questions by focusing on the management of fire. Enormous resources are expended

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worldwide in efforts to manage this important disturbance or restore its effects.

To date there has been little direct assessment of how fire-mediated spatial heterogeneity might be restored or managed for in many fire-prone systems, such as the conifer forests of western North America (Rocca 2004). In many of these areas management policy is focused on the use of mechanical treatments to modify forest structure as a means of counteracting the effects of fire suppression. These efforts are controversial and are often not based on a sound understanding of the ecological role of fire as a disturbance process and the methods needed to restore its effects (Johnson 2003; DellaSala and others 2004). Perhaps nowhere in western North America has the appropriateness of structure-versus process-based forest management approaches been more controversial than in the conifer forests of the Sierra Nevada Mountains of California, USA (Stephenson 1999; Miller and Urban 2000).

Since the 1850s, grazing and fire suppression have reduced fire frequencies in the forests of the Sierra Nevada (Stephenson 1999; Miller and Urban 2000). The prevailing management view is that, because of fire exclusion, forest fires in the Sierra, which once varied considerably in severity, are now almost exclusively large, high-severity, stand-replacing events (Skinner and Chang 1996). As a consequence, an extensive program for the management of national forest lands was initiated in 2004. Its goal is to modify the structure of 283,000 ha of vegetation per decade, mainly in the dominant mixed conifer forests (USDA 2004). However, the actual severity of contemporary fire on these lands has yet to be analyzed to determine how well the prevailing view of dramatically increased fire severity and decreased heterogeneity is supported by empirical evidence.

Under the provisions of the National Forest Management Act of 1976, the national forests in the Sierra Nevada and throughout the United States are directed to ''provide for diversity of plant and animal communities.'' Natural variation and the maintenance of biodiversity in ecosystems can be assessed based on the concept of ecological integrity. ''Ecological integrity'' refers to ecosystem wholeness, including the occurrence of ecological processes such as natural disturbances at appropriate rates and scales to maintain natural levels of biodiversity (Karr 1991; Angermeier and Karr 1994). To determine the appropriateness of process-based versus structure-based management approaches for the maintenance biodiversity, we need to understand how ecological integrity is affected by contemporary fires. Thus, one of our primary objectives is to evaluate the rates and scales of contemporary fire as a disturbance process and assess their appropriateness in the context of ecological integrity.

To pursue this objective, we analyzed fire-severity data from the three largest fires that have occurred in the Sierra Nevada since 1999, accounting for most of the area burned over this time. These fires occurred in landscapes where timber harvest and silvicultural activities have been uncommon. After these burns, fire severity was classified by multi–US agency Burned Area Emergency Rehabilitation (BAER) teams. The BAER fire-severity data are derived from pre- and post-burn satellite and photo images and are used to map the effects of the fire on overstory vegetation canopy. We supplement these data with measures of ponderosa and Jeffrey pine mortality taken on the ground in areas of high-severity as defined by BAER. These pines have been harvested in many areas, and there is considerable interest in restoring their natural abundance (SNEP 1996). To gain further insight into the rates and scales of disturbance by fire under current management, we also evaluated temporal patterns of burning since 1950 in the broader landscapes in which the three fires occurred. Fire suppression has been mechanized in its current form since about 1950.

Another of our objectives was to evaluate the effectiveness of a national approach for the assessment of fire regimes and to discover how they have changed. The current basis for this approach, now used throughout the United States, is Fire Regime Condition Class (hereafter Condition Class), (Hann and Bunnell 2001); see also http://www.frcc.gov). It is an index that Estimates departure from reference conditions in vegetation, fuels, and disturbance regimes. In the national forests of the Sierra Nevada, Condition Class has been based on the number of fires estimated to have been exclude in the landscape due to fire suppression. Considerable research has revealed that historically Sierran forests were burned mostly by surface fire, but that this regime has decreased dramatically due to fire suppression (Caprio and Swetnam 1995; Skinner and Chang 1996). Condition Class predicts that these circumstances will lead to a dramatic increase in fire severity and place forest ecosystems at high risk losing key components due to fire (Hann and Strohm 2003).

A new approach to mapping departure from reference conditions, LANDFIRE, is currently under development (http: www.landfire.gov). In addition to Condition Class, it relies on the rapid assessment and mapping of wildland fuels to identify potential conditions that promote fire. The use of approaches that map departure from historic reference conditions in management is advancing rapidly. In the United States, 25 million ha have been identified for fuel treatments based on Condition Class (Brown and others 2004). Thus, it is especially timely now to evaluate the efficacy of approaches that map departure from historic reference conditions as a means of predicting fire severity.

Methods

Study Areas

The Sierra Nevada Mountains of California are a high-elevation (3000–4000 + m tall), 8-million-ha, north/south-trending mountain range (Figure 1, inset). They are forested primarily by conifer vegetation. We evaluated fire severity in the three largest burns in the Sierra since 1999—the McNally, Manter, and Storrie fires. Older fires lacked comparable fire-severity data in digital form. Smaller burns since 1999 in the main part of the Sierra occurred in areas that have been altered by past or recent timber harvesting and silvicultural activities. These effects were rare in the three burns we studied. The 2002 McNally and 2000 Manter fires occurred in close proximity in the southern Sierra (Figure 1), whereas the 2000 Storrie fire occurred in the northern Sierra near the southern Cascades (Figure 2). Together, these fires encompassed most of the area of Sierran conifer forest that has burned in the last 5 years, for a total of 49,917 ha. The McNally fire burned within the Sequoia National Forest from 22 July until 27 August 2002. The Manter and Storrie fires burned in 2000, the former from 7 July until 10 August and the latter from 17 August until 17 September. Weather initially conducive to fire spread, combined with rugged topography, enabled these fires to escape control and subsequently burn for 4–5 weeks under variable weather conditions. All three of the burns occurred in landscapes where most forests were not located within known, historic fire perimeters. In the McNally fire area, shrub ages indicate that fires had occurred there 125–150 years earlier in locations where there was no mapped record of fire (Keeley and others 2005).

Conifer forests typical of midelevations of the western Sierra (for a more detailed description of Sierran forests, see Rundel and others 1977) were abundant in the landscape that burned in the fires, particularly mixed or individually dominated for-

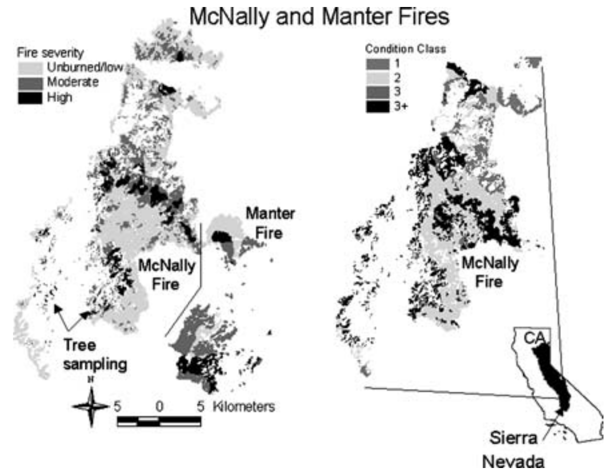

Figure 1. Patterns of burn severity in conifer-forested portions of the 2002 McNally and 2000 Manter fires in the southern Sierra Nevada, California. Preburn Condition Class is shown for the McNally fire area, not including the northernmost portion of the burn in the Inyo National Forest.

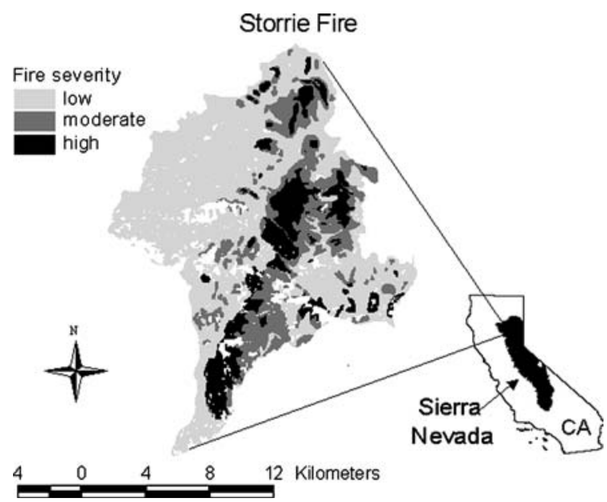

Figure 2. Patterns of burn severity in conifer-forested portions of the 2000 Storrie fire in the northern Sierra Nevada, California.

ests of red and white fir (*Abies magnifica, A. concolor*); Jeffrey, ponderosa, and sugar pine (*Pinus jeffreyi, P. ponderosa, P. lambertiana*); and incense cedar (*Calocedrus decurrens*). These species are often mixed with a deciduous and an evergreen oak (*Quercus kellogii, Q. chrysolepis*). Trees in these forests are generally tall, with many overstory trees exceeding 40–50 m. Canopies are usually closed but can be open as a result of rocky substrata and other edaphic factors, particularly on granitic ridges. Open forests are mostly dominated by Jeffrey pine, often with shrubs in the understory. These forests are common in the Manter fire area and a portion of the McNally fire area. Closed mixed conifer forests predominated in the Storrie and McNally burn areas. One conifer, Douglas-fir (*Pseudotsuga menzesii*), is common in the Storrie fire area but absent from the southern Sierra.

Spatial Patterns of Fire Severity

BAER severity Mapping is designed to identify areas with high potential for soil erosion, which is generally based on the extent to which the fire affects the vegetation overstory canopy. The ability of remotely sensed data to identify patterns of fire severity based on the spectral response of tree canopies has been demonstrated in the Sierra (van Wagtendonk and others 2004). BAER severity in the McNally fire was mapped with Landsat 7 and SPOT multispectral satellite imagery (30-m pixel resolution) obtained immediately before and after the fire (Parsons 2002). A band ratio of mid-infrared and near-infrared reflectance was calculated from pre- and postburn image data. The band ratio data were classified and interpreted by staff at the USDA Forest Service Remote Sensing Applications Center in Salt Lake City Utah. BAER severity for the Manter and Storrie fires was mapped using aerial reconnaissance, infrared aerial photographs, and ground surveys (USDA 2000, 2002). General guidelines for severity classes are from the Forest Service Handbook (USDA 1995).

The BAER mapping identified three to four classes of fire severity based on the level of canopy effects detected. *Unburned* included areas where 0–10% canopy change was detected; this classification was distinguished only in the McNally fire. *Low severity* included areas where fire-caused crown scorch (heat-induced mortality of canopy foliage) affected less than 40% of overstory canopy foliage. The unburned and low-severity classes killed primarily conifer seedlings and saplings. *Moderate severity* included areas where fire scorched 40–89% of the forest canopy in the McNally fire and 40–80% in the other two fires. This level of severity was lethal to most conifer seedlings, saplings, and many small trees, but most overstory trees survived. *High severity* included areas where 90% or more of the canopy was scorched or affected by varying levels of incineration (direct consumption of crown foliage) in the McNally fire, whereas an excess of 80% of canopy showing these effects was considered high-severity in the Manter and Storrie fires. High-severity fire generally resulted in complete understory mortality. Overstory mortality ranged from complete to mixed depending on degree of canopy scorch and consumption (incineration), forest composition, and whether the threshold was 80% or 90% canopy mortality. Depending on imagery and other factors, different thresholds may be used for these severity levels in BAER mapping.

To characterize the spatial scales of the effects of high-severity fire in conifer forests, we describe the size of high-severity patches in each fire. To better characterize the effects, we evaluated the mortality of ponderosa and Jeffrey pine in areas of high-severity burn. Mortality assessments were restricted to a section of roadway in the McNally fire along which initial crown scorch had been assessed before there was any flushing of foliage. We identified five patches along this roadway that were dominated by trees that had no green foliage after the fire. These patches had fire effects ranging from 100% crown scorch (needles killed but not consumed) to needles consumed by crown fire. Within the patches, we chose to monitor all pines showing this range of high-severity effects that had a diameter at breast height (dbh) of more than 25 cm. These trees were generally within 50 m of the road. Our survival data are from 2 years postfire, following Stephens and Finney (2002). We did not observe any further indirect mortality caused by bark beetles over this period. Some trees were considered dead and were harvested over the course of the monitoring. We classified them as having been fire-killed, thus providing a maximum estimate of direct fire-induced mortality in the five sites.

Spatial and Temporal Patterns of Fire

To help assess the landscape-level influence of fire over time under modern fire suppression management, we calculated fire rotation intervals (amount of time needed for an area the size of the area of interest to burn one time) using fire perimeter data obtained from the US Forest Service and the California Department of Forest and Fire Protection. We used the total area of fire that has occurred from 1950 to 2005. Fire perimeters are complete and accurate over this period, and modern fire suppression was a consistent factor. Only conifer-forested areas were analyzed. The landscape we used to calculate fire rotation intervals in the McNally and Manter fire region was the southern portion of the Sequoia National Forest (210,932 ha of conifer forest), along with a smaller amount of the adjacent Inyo National Forest (10,000 ha of conifer forest), including and just beyond the northern boundary of the McNally fire (Figure 1). The landscape used to calculate fire rotation intervals in the Storrie fire region was the largest area within the Lassen and Plumas National Forests; that had the same forest vegetation types found within the Storrie fire region, which was in the center of this landscape. This landscape was more strongly dominated by conifer vegetation, which totaled 488,337 ha, than the landscape where the other two burns had occured. An estimate of rotation intervals for different severity classes in the two landscapes was calculated by assuming that all the conifer forest landscape that burned from 1950 to 2005 had the same severity proportions for the respective landscapes as either the McNally and Manter fires combined or the Storrie fire. This estimate integrates frequency and severity to help illustrate the influence of fire in the two landscapes under current management.

Fire Patterns and Condition Class

We evaluated fire patterns as a function of Condition Class in detail for the McNally fire, where preburn Condition Class data were available. These Condition Class data were mapped to the same vegetation units used in the Cal-Veg map (see Data Analysis). The Condition Class data were based on preburn Fire Return Interval Departure (FRID) and have been applied in planning efforts across the Sierra (USDA 2003, 2004). In other regions of the United States, the Condition Class approach is not necessarily based only on the estimation of FRID (http://www.frcc.gov). We obtained FRID data from the Southern Sierra Geographic Information System Cooperative, which helped to prepare them and still had a version that had not been updated after the McNally fire.

The Fire Return Interval Departure is the number of fires that, on average, may have been excluded. It is based on the time when fire last occurred in an area and the estimated historical fire frequency for the type of vegetation in that area. FRID was thus calculated as:

$$\text{FRID} = (Tsf - Fri)/Fri \qquad (1)$$

where Tsf equals time since the last fire in the landscape and Fri is the estimated fire interval for a vegetation type in the landscape. Estimated historical fire intervals for forests were developed from fire scar studies undertaken in the Sierra Nevada, southern Cascades, and the mountains of northwest and southern California, as reported by Skinner and Chang (1996). Table 1 shows estimated historic fire intervals for each forest type that burned in the McNally fire.

The FRID data we obtained identify the following categories of the number of fires that, on average, may have been excluded: *Extreme* denotes more than five (Condition Class 3 in the national three-level system), *High* is between two and five (Condition Class 3), *Moderate* is between one and two (Condition Class 2), and *Low* is less than one, or not outside the estimated historic fire return interval for a forest type (Condition Class 1) (USDA 2003). We kept the high and extreme FRID categories separate in our calculations and refer to extreme FRID as ''Condition Class 3+''.

Although preburn Condition Class data used in forest planning were not available for the same assessment in the Manter and Storrie fires, we make some inferences based on previous fire history, the Cal-Veg vegetation type within the burn perimeters, and the Condition Classes that would have been assigned based on the Condition Class criteria used in the Sequoia National Forest.

To determine how Condition Class might relate to fire spread rate—a likely predictor of fire severity that integrates weather, fuel, and topographic influences—we chose to assess BAER fire severity in relation to Condition Class in the McNally fire on days when the spread rate of fire was relatively rapid versus slow. To accomplish this, we plotted the ranked daily extent of total fire progression using data obtained from the Sequoia National Forest. This plot (Figure 3) shows that fire spread was particularly high on 2 days. Rather than analyze severity on just these 2 days, we selected additional days in which at least 2000 ha burned. On all the remaining days, an area equal to 1500 ha or less burned (Figure 3). The total areas on days where at least 2000 ha or 1500 ha or less burned were similar and constituted our relatively rapid- and slow-spread landscapes, respectively.

Data Analysis

We calculated fire-severity proportions in conifer forest vegetation types based on the primary vegetation type indicated in the vegetation map, Cal-Veg, that was used to develop Condition Class. It is a standard planning map used on national forest lands in California. Cal-Veg is a map representing current vegetation that is derived from satellite data. The map version used for the two fires in Sequoia had been updated just prior to the Manter fire, and the one for the Storrie fire had been updated the year before the fire. Updates were based

Table 1. Area of Different Conifer Forest Types Burned in the McNally Fire, Estimated Fire Interval used to Calculate Condition Class, and Percent BAER Severity for each Type

Type of Forest	Area (ha)	Fire Interval for Condition Class (y)	Percent Fire Severity			
			Unburned	Low	Moderate	High
Mixed conifer/fir	10,378	16	20.7	36.9	30.5	11.9
Red fir	10,323	50	38.6	35.1	16.3	10.0
Mixed conifer/pine	4154	16	5.5	33.5	52.1	9.0
Jeffrey pine	39,341	50	5.9	23.5	49.0	21.6
Ponderosa pine	2455	6	9.8	38.6	44.0	7.6
Lodgepole pine	1559	163	49.5	39.7	10.7	0.0
Subalpine conifers	692	163	28.9	60.8	9.9	0.4
White fir	117	16	14.9	47.0	34.4	3.6
Foxtail pine	92	163	70.5	29.5	0.0	0.0
Totals	33,704		23.4	35.1	30.5	10.9

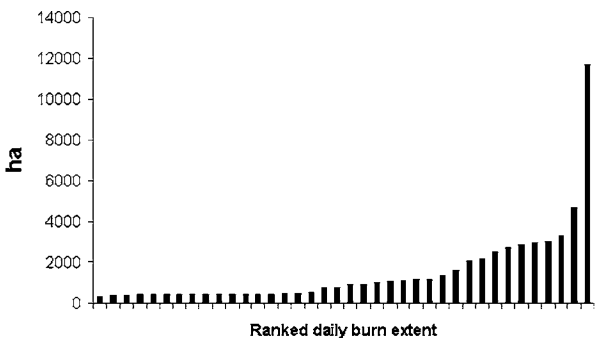

Figure 3. Ranked daily burn extent in the McNally fire as determined from the fire progression data of the Sequoia National Forest.

on accuracy assessments. A detailed description of the Cal-Veg map, and its development and accuracy for Forest Service lands, is at http://www.fs.fed.us/r5/rsl/projects/mapping. The minimum mapping unit is 1 ha. A description of the forest vegetation alliances mapped for the southern and northern Sierra and described in the results can be accessed at http://www.fs.fed.us/r5/rsl/projects/classification/zone-map.shtml.

We excluded pinyon/juniper woodlands and a small amount of open forest on the more arid east side of the Manter fire because it was not in national forest land and was subjected to different mapping protocols. Conversely, we included a small amount of area where the vegetation map indicated a hardwood conifer mix, but where the primary dominant was a conifer forest tree.

A formal statistical approach to testing for differences in severity proportions among Condition Classes by resampling independent, random point locations was not possible (for example, Odion and others 2004) because there was only enough area in some classes to locate a small number of independent points. Therefore, we present the proportions of fire severity by vegetation type and Condition Class and generally evaluate the weight of evidence provided by this information and other descriptors of the current fire regime in the context of the objectives described in the introduction.

Tree mortality was assessed for two diameter-size classes, 25–50 cm and larger than 50 cm. These two classes were compared for differences using a chi-square 2×2 independence test of the hypothesis that smaller trees would suffer greater mortality.

Results

Spatial Patterns of Fire Severity

Most of the conifer forests that burned in the McNally fire (Figure 1) showed characteristics of moderate- or lower-severity fire. High-severity fire accounted for 10.9% of all forest area (Table 1). The highest percentage of high-severity fire occurred in forests dominated by Jeffrey pine (22%), a species that is common on relatively dry and wind-exposed ridges. Most Jeffrey pine forest (83%) burned on the 3rd, 4th, and 5th most extreme-spread days of the McNally fire. Other forest types had much less high-severity fire—in particular, ponderosa pine, mixed conifer/pine, and the relatively small area of forest with long intervals of natural fire (mixed subalpine conifers, lodgepole pine, and foxtail pine). Although the McNally fire burned mostly fir and mixed conifer forests, most of the area that burned in the Manter fire was Jeffrey pine forest. The conifer forests in the Manter fire had more high-severity fire (29%) (Table 2). However, the Manter fire also had a lower

Table 2. Area of Different Conifer Forest Types Burned in the 2000 Manter and Storrie Fires, and the Percent BAER Severity for each Type

	Forest type	Area (ha)	Percent Fire Severity		
			Low	Moderate	High
Manter fire	Jeffrey pine	5,508	24.5	43.6	31.9
	Mixed conifer/fir	1,145	31.9	50.3	17.8
	Red fir	162	68.1	31.9	0.0
	Lodgepole pine	15	0.0	26.7	73.3
Totals		6,829	26.7	44.4	28.9
Storrie Fire	Mixed conifer/fir	7,583	85.8	10.0	4.2
	Mixed conifer/pine	6,577	45.6	26.3	28.1
	Douglas-fir/ponderosa pine	2,986	54.2	35.9	10.0
	White fir	1,511	72.6	5.9	21.6
	Red fir	591	95.8	2.4	1.8
	Jerey pine	128	41.7	52.8	5.6
	Lodgepole pine	7	100.0	0.0	0.0
	Ponderosa pine	2	100.0	0.0	0.0
Totals		19,384	66.3	19.2	14.5

threshold for high-severity fire than the McNally fire (80% versus 90% or more canopy foliage mortality).

For the Storrie fire, severity mapping also used the 80% threshold for high-severity fire. High-severity fire totaled 14.5% among all conifer forests, but the area incurred only about half as much moderate-severity fire as the area burned by the other two fires and consequently considerably more low-severity fire (Figure 2 and Table 2). Of the total area that did burn at high severity (2805 ha), most (1730 ha) of this fire occurred in mixed conifer/pine forests. However, forests dominated by ponderosa and Jeffrey pine had little high-severity fire. Conversely, white fir forests incurred much more high-severity fire than mixed conifer/fir, the most common forest type in the Storrie burn area. Thus, this fire had lower overall severity than the others, and even in different areas mapped with forest types that included many of the same species, the fire nonetheless burned with varying severity.

A few large high-severity patches accounted for much of the total area of high-severity fire in the conifer forests affected by the three burns (Figure 4A–C). However, all three fires produced mostly relatively small patches of high-severity fire. Patches totaling less than 5 ha accounted for 107 of the total of 157 high-severity patches in the McNally fire. They accounted for 28 of a total of 40 in the Manter fire, and 59 of 102 in the Storrie fire.

Many of the pines we monitored that incurred severe burn effects nonetheless produced new foliage from surviving terminal buds in the year after the fire. All surviving trees had either 100% crown scorch and no incineration of foliage or 100% scorch and incineration extending upward to at most 50% of total tree height. For Jeffrey pines incurring these fire effects, 22 of 44 trees survived and there was no difference between the 25–50 cm and greater than 50-cm diameter size classes in terms of the percentage of trees that survived. For the more abundant ponderosa pine, 42 of 83 and 57 of 83 trees in these two size classes survived, and diameter exerted a significant, positive effect (chi-square = 5.6, $P < 0.01$). None of the trees ($n = 90$) with higher levels of crown incineration, survived, indicating that there are significant differences between the effects of crown fire that incinerates foliage and the effects of severe surface fire, which primarily results in the death of foliage due to heat scorch.

Spatial and Temporal Patterns of Fire

For the larger landscape of the national forest in which the McNally and Manter fires occurred, the rotation interval from 1950 to the present for all fire was 185 years. The McNally and Manter fires were responsible for two-thirds of the area that was burned over this time. For both burns combined, the overall percentage proportions of high- and moderate-severity damage in conifer forests was 14% and 33%, respectively. Using these values, the rotation interval in conifer forests was estimated to be about 1330 years, for high-severity fire and about 565 years for moderate-severity fire. Fire has been less common in conifer forests of the Storrie

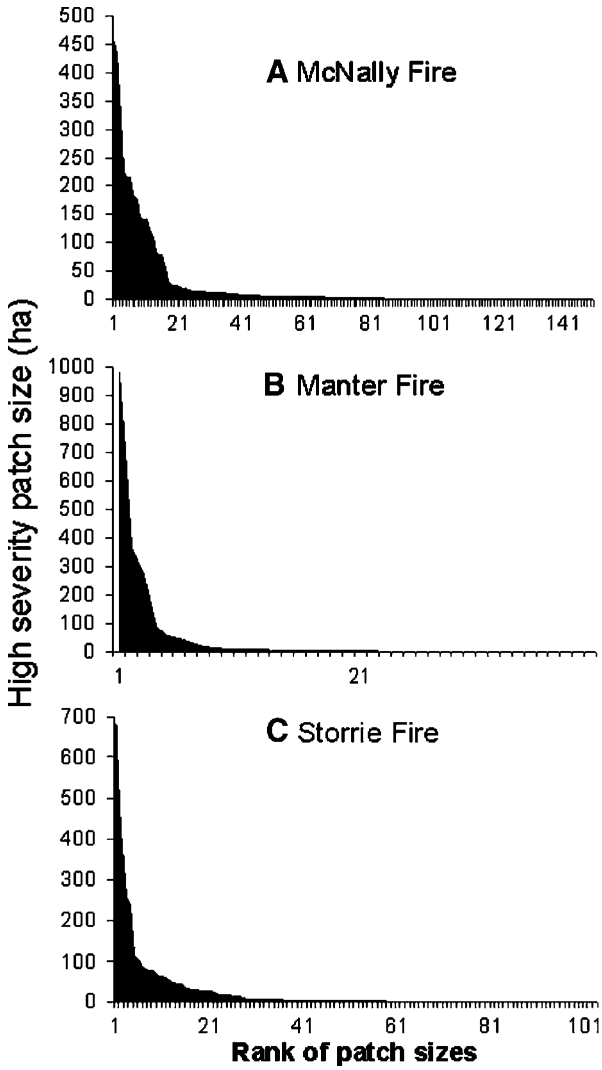

Figure 4. Ranked size of high-severity burn patches in conifer vegetation in the **A** McNally, **B** Manter, and **C** Storrie fires.

fire region. The rotation interval for all fire since 1950 was 507 years. The Storrie fire accounted for about half of all fire in conifer forests over this time period. The estimated rotation interval since 1950 was 3503 years, for high-severity fire and 2460 years for moderate-severity fire in the region in which the Storrie fire occurred.

Severity Patterns and Condition Class

Fire severity proportions by Condition Class under slow- and rapid-spread days in the Sequoia National Forest portion of the McNally fire are shown in Figure 5A–B. The 3939 ha comprising Condition Class 1 forests (2505 ha on slow-spread days plus 1424 ha on rapid-spread days) had almost no high-severity fire. These forests were predominantly comprised of subalpine and other high-elevation forests of red fir, lodgepole pine, and foxtail pine that grow on the relatively flat Kern Plateau.

For Condition Classes 2, 3, and 3+, there were distinctions in degree of severity between rapid- and slow-spread days. In particular, on rapid-spread days, moderate-severity fire was considerably more common, whereas low-severity was less common. The largest area of high-severity fire occurred on rapid-spread days in Condition Class 2 forests (Figure 5A). These forests were comprised mainly of red fir (62%) and Jeffrey pine (22%). Condition Class 3 forests consisted entirely of mixed conifer/fir or pine, whereas Condition Class 3+ forest were ponderosa pine. They had the same proportions of high-severity fire (13%) on rapid- and slow-spread days. This figure was very similar to that for conifer forests throughout the area covered by Condition Class data (Figure 1), which was 11.8%. Condition Class did not appear to have a strong effect in promoting rate of spread because a considerable area of Condition Class 3+ forest burned on slow-spread days (Figure 5).

Applying the McNally Condition Class criteria to the Manter burn area, we find that the 5400 ha of Jeffrey pine and 1145 ha of mixed conifer/fir forests that had no record of previous fire would be Condition Classes 2 and 3+, respectively. Jeffrey pine had 32% high-severity fire, and mixed conifer/fir forests had 17% high-severity. A small area of Jeffrey and lodgepole pine forest (94 ha) that would have been Condition Class 1 had 43% high-severity fire.

Applying the McNally Condition Class criteria to the Storrie fire area and presuming Douglas-fir/ponderosa pine to have an estimated past fire return interval of 16 years, like similar forests (Table 1), we find that there were 792 ha of Condition Class 2 mixed conifer forests. Most of this are burned previously in the 1970s and was primarily forested by Douglas-fir/ponderosa pine. In the Storrie fire, these forests burned with 20% high-severity and 53% moderate severity. Red fir and Jeffrey pine forests (719 ha) had no record of previous fire and would also have been Condition Class 2. They burned at much lower severity than most forests (Table 2). The rest of the forests affected by the Storrie fire had not burned for a long time and would have been condition Class 3+. Collectively they experienced the same severity proportions observed for the burn as a whole—lower than that seen in the Condition Class 2 mixed conifer forests.

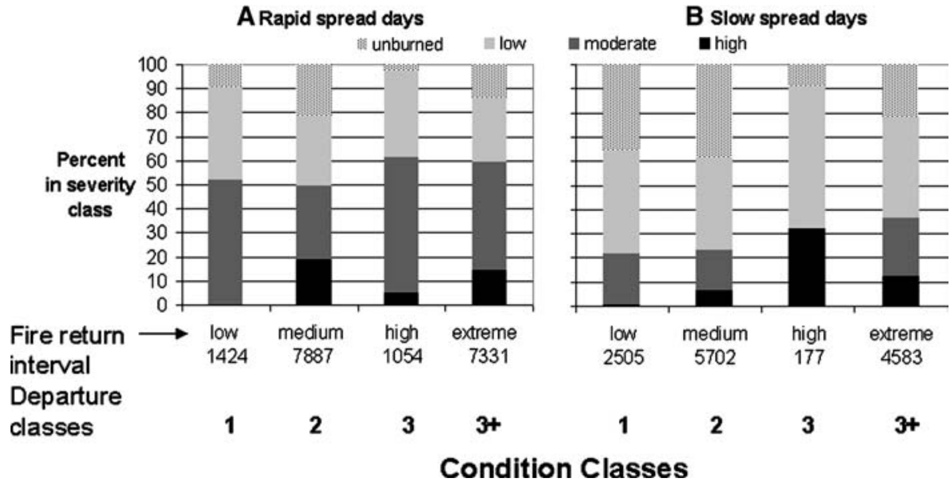

Figure 5. McNally fire severity proportions by Condition Class occurring during **A** days of relatively rapid fire spread (n = 10) and **B** days of relatively slow spread (n = 28). Numbers below columns are hectares burned.

Discussion

Contemporary fire is clearly not almost exclusively high-severity and stand-replacing in long-unburned areas of Sierran conifer forests. In the large area of burned forest that we evaluated, fire severity was highly variable and caused a relatively small amount of high-severity effects. Van Wagtendonk and others (2004) found similar levels of variation and severity proportions in another recent Sierra Nevada burn in the same forest types examined in our study. Our findings are also consistent with the result of recent modeling, which showed that long-unburned Sierran forests unaffected by silvicultural activities would not incur crown fire until temperature, relative humidity, and wind exceeded the 97.5th percentile of their summertime levels (Stephens and Moghaddas 2005).

The burn patterns we observed are also consistent with descriptions and evidence in Sierran forests not influenced by fire suppression and silviculture. There are a number of historical accounts of variability in fire ranging from light understory burning to patchy high-severity fire in Sierran mixed conifer forests, including one by the famed naturalist John Muir (reviewed by Stephenson and others 1991; Stephenson 1999), and another by a forest surveyor John Leiberg (1902). Recent studies using historic photos and field sampling have concluded that patches of high-severity fire have shaped mixed conifer forests in the Sierra Nevada and the adjacent southern Cascades (Russell and others 1998; Beaty and Taylor 2001; Taylor 2002). Show and Kotok (1924), Russell and others (1998), Beaty and Taylor (2001), and Taylor (2002) describe historic high-severity burn patches in the Sierra that are comparable in size to many of the larger patches produced by the three fires we studied. Smaller patches or gaps have also played an important role in determining forest and landscape structure and composition (Stephenson and others 1991; Keeley and Stephenson 2000) and were common in the three fires we studied. Leiberg (1902) and Beaty and Taylor (2001) have also describe the occurrence of large historic fires.

Because the fires we studied burned for 4–5 weeks, mainly in July and August, they were influenced by a range of weather conditions. This may help to explain why they were heterogeneous and qualitatively similar to descriptions of pre–suppression era fires. Most lightning ignitions occur in the Sierra during July and early August (Caprio and Swetnam 1995). Historic lightning ignitions that led to spreading fires would have been driven by the same seasonal patterns of warm, dry weather that typifies the Sierran summers. The large size of the fires we studied likely enhanced their variability by creating both fire-generated winds, which that can make combustion more active, and dense smoke, which can lower temperatures and mitigate fire behavior (Pyne 1984). Thus, it is important to stress that our results apply to fires in the Sierra that burn for long durations and spread over relatively large areas in mid- and late summer. These circumstances are representative of much of the areas burned by contemporary fire, and presumably fire in the past, given the effect of large fire on the cumulative amount of area burned. Much less heterogeneity may result from fires that burn for a shorter time and cover small areas. Our results also apply only to areas in the Sierra where timber harvesting and silvicultural activities have not been common. There are many areas of the Sierra that have been modified considerably by intensive silvicultural activities (SNEP 1996) and where severity is expected to be higher due to increases in available fuel and the loss of fire-resistant trees (Stephens and Moghaddas 2005).

After a long period of reduced fire influence, large, heterogeneous fires can hasten ecological restoration (Baker 1992; Miller and Urban 2000; Fulé and others 2004). They may affect biodiversity by thinning trees and decreasing competitive exclusion processes and by increasing structural and landscape diversity. Fire-created gaps provide opportunities for the natural regeneration of light-demanding conifers such as pines and giant Sequoia (*Sequoiadendron giganteum*) (Stephenson and others 1991; Keeley and Zedler 1998; Stephens and others 1999) whose natural abundance in the Sierra has been reduced (SNEP 1996). There are concerns about the lack of natural regeneration in these species due to the absence of fire severe enough to create openings, consume sufficient duff and litter to facilitate successful germination, and open cones in giant Sequoia (Stephenson and others 1991; Stephens and others 1999). Such fire effects may not only promote the natural reproduction of these conifers, but also favor the relative abundance of these species because they have a greater ability to survive. Large giant Sequoia may survive in areas of crown fire (Stephenson and others 1991), and we found that many medium and large ponderosa and Jeffrey pines can survive severe surface fire. There may be some additional mortality among these trees, but those that survive are likely to experience rapid growth and increased vigor, much like giant Sequoia after severe fire (Stephenson and others 1991). Mature white fir may also be more fire resistant in the Sierra than previously suspected, aided by their ability to produce epicormic branches (Hanson and North 2006). Surviving conifers may serve as sources of seed that help to ensure natural regeneration in high-severity burn patches.

Patches of habitat created by high-severity fire, with their rich array of snags, logs, and nonarborescent vegetation, are among the scarcest habitats in many forested landscapes (Lindenmayer and Franklin 2002). After 50–100 years this early successional habitat can succeed to forest (Russell and others 1998). Thus, based on estimates the area of high-severity fire predicted by our fire rotation analyses for the period since 1950 in the Sequoia and Storrie fire regions, about 4.2% and 1.5% of these landscapes, respectively, may have naturally developed early successional burned forest habitat under the current fire regimes. The maintenance of this habitat in the landscape by fire promotes biodiversity because it supports plant, insect, and wildlife assemblages not found in other Sierran habitats. In addition, there are numerous plant and animal species that have become rare due to their requirements for burned forest habitat. For example, there is some concern over the local extirpation of avian species with these habitat requirements (Kotliar and others 2002). Species such as the black-backed woodpecker (*Picoides arcticus*) may be indicators of whether sufficient, severely burned forest habitat is being maintained for biodiversity (Hutto 1995). These birds require young, severely burned patches of at least 12–25 ha (Saab and others 2002). The three fires we studied created 70 severe-burn patches larger than 12 ha where there had been none or very few due to the lack of fire.

Thus, the effects of the large fires we studied are consistent with the diversity goals of the National Forest Management Act. Elsewhere in the western United States, a number of large fires have also been found to perform the desired ecological functions of fire (for example, Turner and others 2003; Kotliar and others 2003; Fulé and others 2004; Odion and others 2004; Schoennagel and others 2004; Smucker and others 2005). These specific effects may ultimately be necessary for maintaining biodiversity that depends on fire. Prescribed burning can help, but it is limited in extent, severity, and heterogeneity (Baker 1994; Rocca 2004) and may not mimic natural fire (Moritz and Odion 2004). On National Forest Service lands, prescribed burning is often conducted outside the normal fire season, when flaming is subdued but wildlife such as herptofauna are highly vulnerable to smoldering combustion (Bury 2004). Neither these fires, nor the structural modification of forests through mechanical treatments, may provide fire-specific effects for species that require them (for example, flowering plants with fire-dependent seed germination that is sensitive to burn season, conifers with heat-opened cones, and cavity-nesting species that dependent on standing dead trees for nesting and foraging).

Fire Patterns and Condition Class

We found that the proxy for fire suppression effects, Condition Class, was not effective in identifying locations of high-severity fire. Condition

Classes 2, 3, and 3+ in the McNally fire all had similar fire severity proportions. When the same Condition Class criteria were applied to the other two fires, we found that fire severity generally decreased rather than increasing from Condition Class 2 to 3+. In short, Condition Class identified nearly all forests as being at high risk of burning with a dramatic increase in fire severity compared to past fires. Instead, we found that the forests under investigation were at low risk for burning at high-severity, especially when both spatial and temporal patterns of fire are considered.

The lack of an observed relationship between Condition Class and fire severity suggests that exogeneous forces such as weather, climate, topography, and neighboring vegetation (for example, pyrogenic shrubs) largely determine fire-severity patterns in forests. Because fire severity did not increase above Condition Class 2, the combustibility of Sierran forests may reach a maximum at the fire-free intervals indicated by this class (32–48 years for many forest types), (Table 1).

A number of interrelated factors may explain why these forests reach a maximum in combustibility. For example, the total leaf area of a forest reaches a maximum (Waring and Schlesinger 1985). Once forest overstories close in the Sierra, they may exclude pyrogenic shrubs with high light requirements (Show and Kotok 1924), greatly decreasing the potential intensity of understory combustion. The base height of the forest canopy sufficiently dense to propagate fire may also become relatively high in long-unburned forests (Stephens and Moghaddas 2005). In terms of surface fuel beds, those associated with Sierran conifers that increase in abundance with time since fire (for example, fir) are more dense than those found under pine and thus less combustible (van Wagtendonk and others 1998).

Conclusions

Our findings suggest that elevated risk of high-severity fire due to the effects of fire suppression is not the pervasive, predictable ecological problem that it has often been portrayed to be in the Sierran forests we studied. In addition, they provide evidence that fire alone can restore its past influence as a patchwise and stand-thinning disturbance agent as well as a facilitator of species diversity and fire-adapted conifers in these forests. Thus, it appears that management can shift toward process restoration by introducing more fire and increasing the use of wildland fire (Miller 2003). There may be no other effective strategy for restoring and maintaining ecological integrity and for fostering the natural diversity of species dependent on effects specific to fire. The structural modifications of forests cannot mimic the heterogeneous effects of fire. Instituting a policy that allows more fire to burn would require considerable planning and additional efforts to improve human safety, but such efforts are needed under any management scenario.

Both Condition Class and the new LANDFIRE approach are based on mapping any departure in fire regimes from reference conditions. Presuppression reference conditions for fire must be based on retrospective studies. These studies are too methodologically limited to provide a comprehensive description of the spatial extent and variation in the effects of past fires (reviewed by Veblen 2003). As a result, the importance of past surface fire may be overestimated and conversely, past heterogeneity in fire may be underestimated (for example, Minnich and others 2000). To add to the problem of uncertainty about past fire, there may be significant misconceptions about current fire severity that lead to further overestimation of the differences between past and present fire regimes.

By directly assessing existing fire regimes in the context of ecological integrity, we can avoid some of the problems that may arise when current methods for estimating departure in fire regimes are used. A general approach based on the assessment of existing rates and scales of processes in the context of ecological integrity has been recommended for the management of biodiversity as a means of overcoming problems in defining the ''natural'' range of variation in ecological systems (Parrish and others 2003). The direct assessment of fire regimes can be improved by applying more sophisticated mapping of fire severity and performing landscape analyses that provide a clearer link between pattern and process (Wagner and Fortin 2005). In the Sierra Nevada, it is important to distinguish high-severity surface fire from crown fire because the two types of behavior may have very different effects on tree mortality. There is also a need for analyses of fire behavior in areas affected by timber harvesting and silviculture. Finally, better integration of the spatial and temporal components of other forest disturbances in the Sierra Nevada in addition to fire, is needed to determine if their rates and scales are compatible with ecological integrity.

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Examining Historical and Current Mixed-Severity Fire Regimes in Ponderosa Pine and Mixed-Conifer Forests of Western North America

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Abstract

There is widespread concern that fire exclusion has led to an unprecedented threat of uncharacteristically severe fires in ponderosa pine (*Pinus ponderosa* Dougl. ex. Laws) and mixed-conifer forests of western North America. These extensive montane forests are considered to be adapted to a low/moderate-severity fire regime that maintained stands of relatively old trees. However, there is increasing recognition from landscape-scale assessments that, prior to any significant effects of fire exclusion, fires and forest structure were more variable in these forests. Biota in these forests are also dependent on the resources made available by higher-severity fire. A better understanding of historical fire regimes in the ponderosa pine and mixed-conifer forests of western North America is therefore needed to define reference conditions and help maintain characteristic ecological diversity of these systems. We compiled landscape-scale evidence of historical fire severity patterns in the ponderosa pine and mixed-conifer forests from published literature sources and stand ages available from the Forest Inventory and Analysis program in the USA. The consensus from this evidence is that the traditional reference conditions of low-severity fire regimes are inaccurate for most forests of western North America. Instead, most forests appear to have been characterized by mixed-severity fire that included ecologically significant amounts of weather-driven, high-severity fire. Diverse forests in different stages of succession, with a high proportion in relatively young stages, occurred prior to fire exclusion. Over the past century, successional diversity created by fire decreased. Our findings suggest that ecological management goals that incorporate successional diversity created by fire may support characteristic biodiversity, whereas current attempts to "restore" forests to open, low-severity fire conditions may not align with historical reference conditions in most ponderosa pine and mixed-conifer forests of western North America.

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Introduction

In just two days in 1910, 1.2 million ha of forestlands in Idaho and Montana in the western USA burned in a massive fire driven by exceptional winds [1]. In the aftermath, the United States instituted a policy of aggressive fire suppression [2]. Decades of fire suppression activities since 1910 have reduced the extent and number of wildfires in the USA, as well as parts of Canada. There is now widespread concern that fire exclusion has caused vegetation in western North America to be much more susceptible to uncharacteristically severe fire. This concern is greatest in the extensive, often drier forests of the North American Cordillera, especially those dominated by ponderosa pine (*Pinus ponderosa* Dougl. ex. Laws) and Jeffrey pine (*P. jeffreyi* Grev. & Balf.), or those mixed with ponderosa/Jeffrey-pine and other conifer species (hereafter ponderosa pine and mixed-conifer forests of western North America, defined in Table 1 and further described in Methods).

The ponderosa pine and mixed-conifer forests of western North America have traditionally been considered adapted to a low- or low/moderate-severity fire regime (see Tables 1 and 2 for definitions of fire terms) [3–8]. There have been many large mixed-severity fires in western North America in recent years [9] that have helped create widespread concern that fire exclusion has caused an unprecedented threat of uncharacteristically severe fires [6–15]. Concomitantly, however, there has been increasing recognition that fires in ponderosa pine and mixed-conifer forests of western North America were also mixed in severity prior to any significant effects of fire exclusion (Table 2) [16,17]. It has also been increasingly recognized that these forests support biota that are not adapted to low/moderate-severity fire, but rather are dependent on the high-severity fire component of mixed-severity regimes [18–22]. Thus, a better understanding of historical (i.e., generally prior to fire suppression and timber harvesting) fire regimes in these forests is needed to define reference conditions and maintain characteristic ecological diversity.

In recent decades, to address the widespread concerns about uncharacteristically severe fire in western North America, fuel reduction treatments have been implemented on millions of hectares of ponderosa pine and mixed-conifer forests at a cost of billions of dollars [23]. These treatments consist mainly of harvesting smaller trees to reduce forest density [8], but larger trees are typically harvested as well for economic reasons [24]. These treatments can negatively affect fire dependent species. For example, the Black-backed Woodpecker (*Picoides arcticus*), an imperiled fire-dependent species, largely avoids previously thinned forest areas burned at high-severity [18]. Thinning treatments also eliminate/degrade dense forest, which many species need, including the Northern Spotted Owl (*Strix occidentalis caurina*), a Threatened Species under the USA Endangered Species Act [25], and the Pacific fisher (*Pekania pennanti*), a Candidate Species under the USA Endangered Species Act [26]. In addition, forest thinning treatments often require the reopening or construction of access roads, which have many ecosystem impacts [27], and both the thinning treatments and roads promote the establishment of invasive species [27,28]. Thinning ultimately exacerbates fire suppression impacts if it facilitates fire control, or if it becomes a prerequisite for allowing wildfires to burn [13,29]. Thus, there is a need to ensure that actions are ecologically justified.

Most descriptions of the fire regimes that characterize the ponderosa pine and mixed-conifer forests of western North America (e.g., [5–7,11]) emphasize how low-severity fires maintain forests dominated by relatively old and large, fire-resistant trees, with few understory trees, dead or dying trees, or shrubs [3–7,11–13] (Table 2). Park-like conditions and low fuel loads are thought to result from effects of frequent surface fire, which kills young, fire-sensitive trees, while older, fire-resistant trees survive [4,6,7,11,12].

In contrast, mixed-severity fire regimes are characterized by more variable fire and forest structure across a wide range of spatial and temporal scales [17,21] (Tables 1 and 2). The creation of complex early seral vegetation by high-severity fire often occurs in irregular patches across the landscape and at irregular intervals [30]. Over time, the complex early successional vegetation created by fire, if not reburned, transitions to mid- and then late-successional forest, often containing pre-disturbance legacies, such as standing or fallen dead trees and often some fire resistant, large trees that survive fire crown fire (e.g., [31]). Thus, mixed-severity fire regimes create complex successional diversity high beta diversity, and diverse stand-structure across the landscape [17,21,30,32–35].

The concepts and nomenclature used to describe fire regimes in western North America can be ambiguous. Part of the problem with defining fire regimes for the drier forests of western North America is the classification of fire regimes into distinct categories of low-, mixed-, and high-severity [5], or low/moderate-severity

Table 1. Definitions of terms as used in this paper.

Term	Definition
Ponderosa pine and mixed-conifer forests of Western North America	Low- to mid-elevation, montane, non-coastal forests of western North America where a regime of low/moderate-severity fire (see Table 2 for explanation) that limit tree recruitment has traditionally been applied. These extensive forests are dominated by ponderosa pine (*Pinus ponderosa*), Douglas-fir (*Pseudotsuga menziesii*) and fir (*Abies concolor* and *A. grandis*) (see Methods). These forests are drier than coastal forests or most forests at higher elevations, though one region, the Klamath, is more mesic.
Fire dependent	Biota that occur most abundantly after high-severity fire, and which are largely or entirely absent where high-severity fire has not occurred for a long period.
Fire regime	The frequency, size, seasonality, impacts and other characteristics of naturally occurring fires that have occurred in a vegetation type over its lifespan, generally 1–3 millennia [133].
High-severity fire rotation (or moderate to high-severity fire rotation)	The length of time required for an area equal to the area of interest to burn [134]. For high-severity fire, this is calculated as the time period over which high-severity fire (or moderate- and high severity fire combined) is observed, divided by the proportion of the area of interest that burns in that time period at high- or moderate/high-severity.
High-severity fire	Fire that burns on the ground surface, and typically in the overstory canopy (crown fire) as well. Mortality of woody species as measured by basal area is generally >70%. However, sprouting canopy species, such as oaks (*Quercus* spp.) typically survive these fires. High-severity fire mainly occurs in relatively discrete patches under high winds that cause blow ups in fire behavior [108]. These patches range in size from the area occupied by a small group of trees to many thousands of ha in size, as in the case of the 1910 fires.
Low-severity fire	Fire that burns on the ground surface such that relatively little or no mortality of live, standing vegetation occurs. Mortality of woody species as measured by basal area is 0–20%, but is mostly 0–5%. See Table 2 for a detailed explanation of the effects of a regime of low-severity fire.
Moderate-severity fire	Fire that burns only on the ground surface and that has effects that are intermediate between low- and high-severity fire as defined here. Mortality of woody species as measured by basal area is generally 20–70% within a given area.
Mixed-severity fire	Fire that includes low-, moderate-, and high-severity effects. See Table 2 for a detailed explanation of the effects of a regime of mixed-severity fire.
Park-like forest	A forest of widely-spaced live, mature trees and very few, if any, dead trees (snags). The understory is open, often dominated by bunchgrasses, and is mostly lacking woody plants.
Stand age	The age within a stand of the dominant overstory canopy vegetation that recruited more or less as a cohort, typically after a previous disturbance.

These terms may have different meanings in the literature depending on the context in which they are used.
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Table 2. Characteristics of fire regimes in ponderosa pine and mixed-conifer forests of Western North America.

	Low/moderate-severity model	Mixed-severity model
Tree populations	1. Stable. Gap phase recruitment dynamics.	1. Unstable. Gap and stand-level mortality and recruitment. Stand-replacement fires at intervals often shorter than tree lifespans.
	2. Recruitment limited by frequent fire.	2. Recruitment abundant and stimulated by fire.
	3. Resistant to fire (though often described as "fire-resilient").	3. Resilient following fire.
Landscape patterns	1. Successional diversity low.	1. Successional diversity high.
	2. Gradual variation along environmental gradients.	2. Variation along environmental gradients interrupted by sharp boundaries and patchiness.
	3. Low contrast heterogeneity. Intensity/complexity of spatial pattern is low.	3. High contrast spatial heterogeneity. Intensity/complexity of spatial pattern is high.
	4. Low beta diversity.	4. High beta diversity.
Stand structure	1. Does not vary markedly over time.	1. Varies markedly as a function of time since fire disturbance.
	2. Open canopy of mature, medium and large trees; density low.	2. Variable canopy, tree size, and density variable; even-aged cohorts stimulated by fire.
	3. Understory with few trees or shrubs.	3. Understory varies.
Fire behavior	1. Typically low intensity surface fire with flame lengths <3 m; short residence time.	1. Variable intensity surface or crown fire, variable residence time.
	2. Fuel limited. Crown fire cannot initiate.	2. Not necessarily fuel limited. Crown fire can initiate under extreme conditions.
Individual fire canopy mortality	1. Mortality of canopy trees <20% by basal area.	1. Mix of low-, intermediate- and high-severity fire with (0–20%, 20–70%, >70%) mortality of canopy trees by basal area respectively.
Interactive effects of fire on fuels and forest flammability	1. Fires continuously limit fuels and fire sensitive trees.	1. Fires only temporarily lower fuels.
	2. Maintain low flammability and forest mortality over time.	2. Do not maintain low flammability and forest mortality, except initially after fires.
Evolutionary responses	1. Fire resistant trees.	1. Fire resistant and fire-dependent or specialized biota. The latter includes species with reproduction timed to coincide with fire via fire-cued germination, fire "embracer" plant species, and post-fire insect and bird specialists).
Fire exclusion leads to	1. High tree regeneration*.	1. Low tree regeneration.
	2. Greatly increased flammability.	2. Small changes in flammability (vegetation is continuously flammable except initially after fire).
	3. Increased forest susceptibility to mixed-severity fire.	3. Decreased susceptibility to mixed-severity fire.
Carbon storage[1]	1. Low-moderate; considerably lower than carrying capacity.	1. Moderate to high; Near carrying capacity.
Fuel treatments (forest thinning)	1. Restores forest tree structure and fuel loads where infill associated with fire exclusion is removed.	1. May create uncharacteristic structure and composition (reduction in small and intermediate and some overstory trees, shrubs, down wood).
	2. Increase open forest (woodland) biota.	2. Decrease in dense forest biota and post-fire habitat specialists.
	3. May create low contrast heterogeneity.	3. May reduce high-contrast heterogeneity.

[1][135–137].
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and high-severity [9], when nearly all fire regimes include a mix of all three severities. Greater clarity in terminology is needed to improve communication about fire regimes. Tables 1 and 2 document the terminology used herein.

In addition to unclear terminology, other factors create difficulties for identifying which historical (i.e. prior to fire exclusion) fire regime applies to a particular forest region. Where fire has been excluded from a mixed-severity landscape for 100 years, early- and mid-successional patches created by high-severity fire become late-successional patches, making it more likely that these patches, indicative of a mixed–severity regime, will be undetected. For example, high-severity fire patches may be detected in old but not recent aerial imagery [35]. A primary source of data on historical fires are scars in the growth rings of surviving trees damaged by fire, which can provide annually precise dates for past fires at the sampled locations [36–40]. However, these methods cannot effectively determine past occurrence of high-severity fire. Thus, additional evidence is needed to characterize historical fire regimes over more extensive areas.

The US Forest Service Inventory and Analysis (FIA) program provides an extensive dataset that is a probabilistic sample of forest structure in large landscapes. This dataset allows for landscape-scale inference and statistical analyses of forest age and structure parameters consistent with a low- or mixed-severity fire regime.

Using the FIA data, and published sources of landscape-scale (area of inference >25,000 ha) data, our objectives were to address two broad questions: (1) How prevalent were mixed-severity fire regimes historically in ponderosa pine and mixed-conifer forests of western North America; and (2) How have mixed-severity fire patterns in these forests changed with fire exclusion? Consistent with common perceptions and restoration models applied to these forests, we hypothesized that: (1) forest age-class diversity was low, reflecting long-term effects of low/moderate-severity fire regimes (Table 1); and (2) fire exclusion has led to vegetation changes that have increased the prevalence of high-severity fire.

Methods

Study Area

FIA and published sources of landscape-scale (area of inference >25,000 ha) data with inference to pre-settlement fire severity and forest structure were available from the following regions of western North America: Baja California, the Sierra Nevada, the Klamath Region, the eastern Cascades, the northern Rockies, the central Rockies, and the southwestern USA (Figure 1). We used ecoregional class III data from the US Environmental Protection Agency (http://www.epa.gov/wed/pages/ecoregions/level_iii_iv.htm) to define the Sierra Nevada, Klamath, and eastern Cascades regions. The Sierra Nevada was split along the distinct crest of the range into the east and west slopes. The portions of the northern Cascades east of the crest and the main Cascades within California were combined into the eastern Cascades. The Modoc Plateau and eastern Sierra Nevada was also combined with the eastern Cascades. The northern Rockies were in Idaho and Montana, and the central Rockies were in Colorado, Wyoming and South Dakota. The southwestern USA included Arizona and New Mexico.

The dominant conifer over most of the low- to mid-elevation, montane forests in these regions is ponderosa pine, often with lesser amounts of Douglas-fir, white fir (*Abies concolor* (Gord. and Glend.) Lindl.), and/or grand fir (*A. grandis* (Douglas ex D. Don) Lindl.). In the Sierra Nevada and Klamath regions, ponderosa pine is common and may be dominant, especially in low-elevation forests, and mixed-conifer forests generally include components of ponderosa pine, white fir, Douglas-fir, incense-cedar (*Calocedrus decurrens* (Torr.) Florin), sugar pine (*P. lambertiana* Dougl.), California black oak (*Quercus kelloggii* Newb.) and evergreen canyon live oak (*Q. chysolepsis* Liebm.). Mid-elevation forests of the Sierra Nevada and Cascades are often dominated by Jeffrey pine, ponderosa pine, white fir and sugar pine. Low- to mid-montane forests of the eastern Cascades are dominated by ponderosa pine and Douglas-fir, and can include components of white fir, grand fir (*Abies grandis* Dougl. ex D. Don.) Lindl.), and western hemlock (*Tsuga heterophylla* (Raf.) Sarg). Low- and mid-elevation forests of the Rocky Mountains are dominated by ponderosa pine and Douglas-fir. In the northern Rockies, these two dominants may co-occur with white fir and grand fir, and with western hemlock, western redcedar (*Thuja plicata* Donn. Ex D. Don.) and quaking aspen (*Populus tremuloides*). Forests of the southwestern U.S. are heavily dominated by ponderosa pine, with some white fir and Douglas-fir at middle elevations. Precipitation and temperature data for each region in this study are provided in Table 3.

Evidence for Historical Mixed-severity Fire Regimes in Ponderosa Pine and Mixed-conifer Forests

Rotations of high- and moderate-to high-severity fire. We summarized rotations for high-severity fire from published studies with inference to large landscapes (>25,000 ha) in ponderosa pine and mixed-conifer forest landscapes of western NA over a period of 70 or more years. The high-severity fire rotation is equal to the average interval between high-severity fire across the affected landscape (Table 1). Additionally, we summarized other evidence regarding the occurrence of high-severity fire where rotations could not be calculated, but where landscape-scale inference regarding the relative importance of high-severity fire was presented, or where rotations could be calculated but landscapes were <25,000 ha or the time period was <70 years.

Dominant overstory tree age distributions. To assess successional patterns indicative of mixed- vs. low/moderate-severity fire regimes, we analyzed US Forest Service Inventory and Analysis (FIA) stand ages (data available at http://www.fia.fs.fed.us/tools-data/) by region. These data capture the average age of the trees dominating the canopy layer in forest stands (stand age, Table 1) that have been sampled probabilistically, with inference to more extensive landscapes. Because the dominant trees in ponderosa pine and mixed-conifer forests may be several centuries old in the absence of disturbance [e.g., 41,42], we reasoned that the age of relatively young and intermediate-aged stands (e.g. <200 years) reflects the time since a disturbance that shifted dominance from older to younger trees. The FIA database indicated that young stands (generally 0–30 years) were initiated by fire. To determine whether disturbances in other plots were caused by fire, we evaluated the effects of fire exclusion on rates of disturbance, as described below. It is not possible to specify the level of mortality that fire or other disturbances may have caused, but it is possible to determine the extent to which forests were dominated by older age classes, which would be consistent with low−/moderate severity fire, versus stands of more diverse age classes, consistent with mixed-severity fire.

FIA is a monitoring system based on one permanent, random 1-ha plot per ~2400 ha across forested lands in the USA. For tree measurements, the plot area is sub-sampled with four circular plots of 0.1 ha for large trees and 0.017 ha for smaller trees nested within the larger tree plots. Diameter at breast height (dbh) and crown position of each tree and the ring count from cores of the dominant and co-dominant trees (i.e., the main overstory canopy layer) of each tree species are measured in each subplot [43]. The stand-age variable for a "stocked" FIA plot (i.e., one containing trees of any age) is determined from the average of all ring counts from subplot samples of dominant and codominant trees in the size class characteristic of the overstory canopy structure, weighted by cover of sampled trees, and 8 years are added for estimated time to grow to breast height (1.4 m) at which cores are sampled.

We selected FIA data from low- to mid-elevation forest types in Wilderness, Inventoried Roadless Areas, and National Parks to ensure as best we could that stand initiation was not caused by commercial harvesting of trees or other land use (Fig. 1). We had no independent way to confirm that trees were never cut at each plot location, so we interpret the results assuming only that such management was of minor importance, given that Wilderness, Roadless, and National Park designations reflect a lack of past timber harvesting. We selected lands classified as "timberlands" in Pacific states' data sets. In the Rockies and southwestern USA, where there was no such designation, we selected all areas where the potential vegetation was considered capable of >10 percent tree cover.

A small number of plots had different stand ages for different subplots due to disturbances that affected some, but not all, subplots. In FIA split-age plots where both plot ages were ≤100 years, plots were split into two stand ages by FIA if they differed by as little as 1 year. In split-age plots in which both ages were 100–199 years old, plots were split into two stand ages if they differed by as little as 2 years. In split-age plots where both ages were ≥200

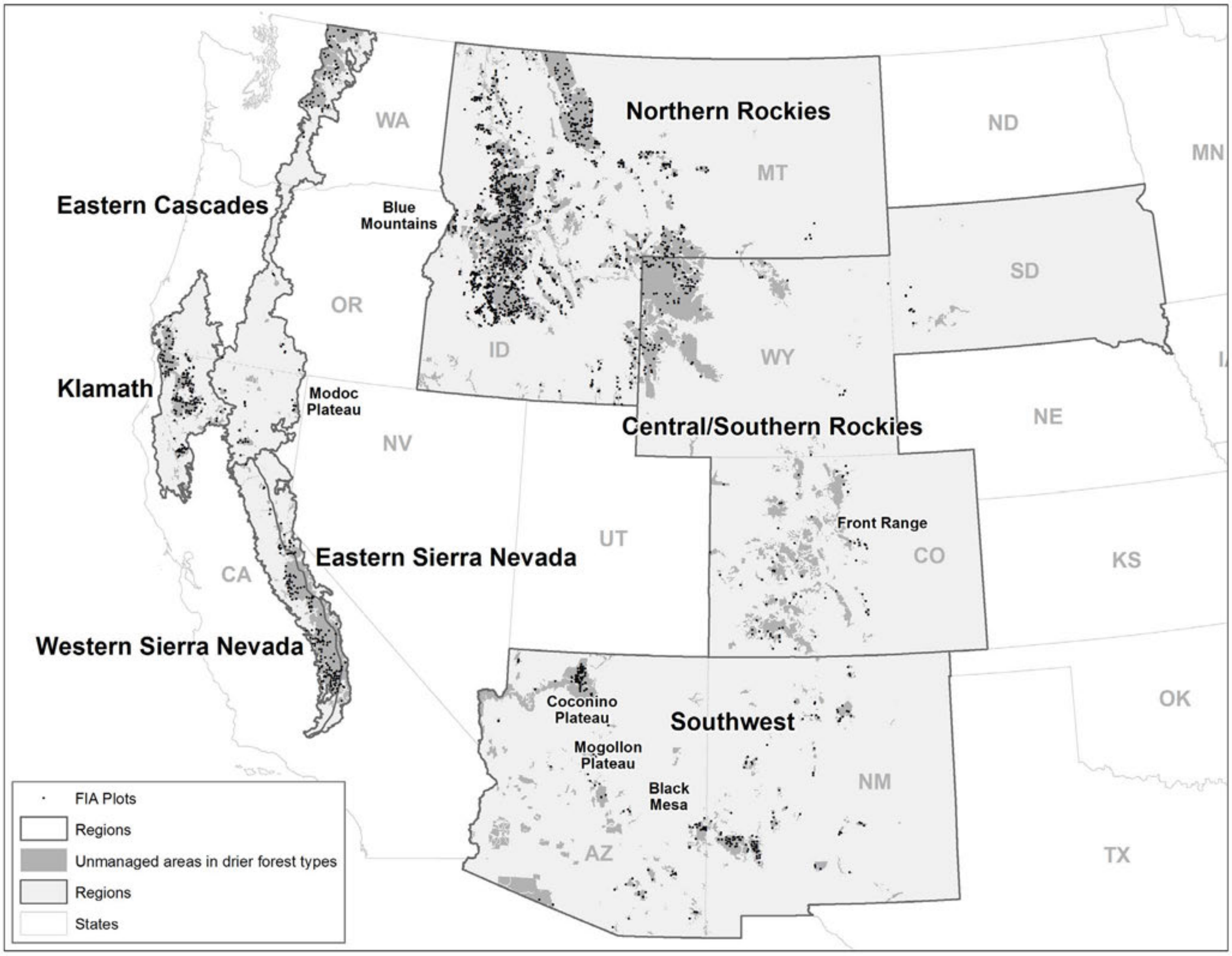

Figure 1. Study area. Dots indicate the general locations of Forest Inventory and Analysis (FIA) plots.
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years, plots were split into two stand ages if they differed by as little as 15 years. To assess the within plot variability in tree ages, we calculated the standard deviation of the trees used to age each plot. We standardized this across the range of stand ages by calculating the standard deviation of the proportional difference between stand age, and the individual trees used to determine stand age in each plot, over the range of stand ages.

We reasoned that, prior to fire suppression, under a low/moderate-severity fire regime, successional, or age-class diversity, would be low, while it would be high under a mixed-severity fire regime. With fire exclusion and greater amounts of uncharacter-

Table 3. Mean annual precipitation, and mean summer maximum and minimum temperatures, in ponderosa pine and mixed-conifer forests in each region.

Region*	Mean annual precipitation (cm)	Mean maximum temperature, June-August (degrees C)	Mean minimum temperature, June-August (degrees C)
Sierra Nevada	104	23	9
Klamath	196	26	11
Eastern Cascades and Eastern Sierra Nevada	113	21	7
Northern Rockies	88	22	6
Central and Southern Rockies	71	22	6
Southwest	58	27	11

*All values are from PRISM data (http://www.prism.oregonstate.edu/normals/) in each 2 km^2 PRISM pixel within which an FIA plot used in the study occurred.
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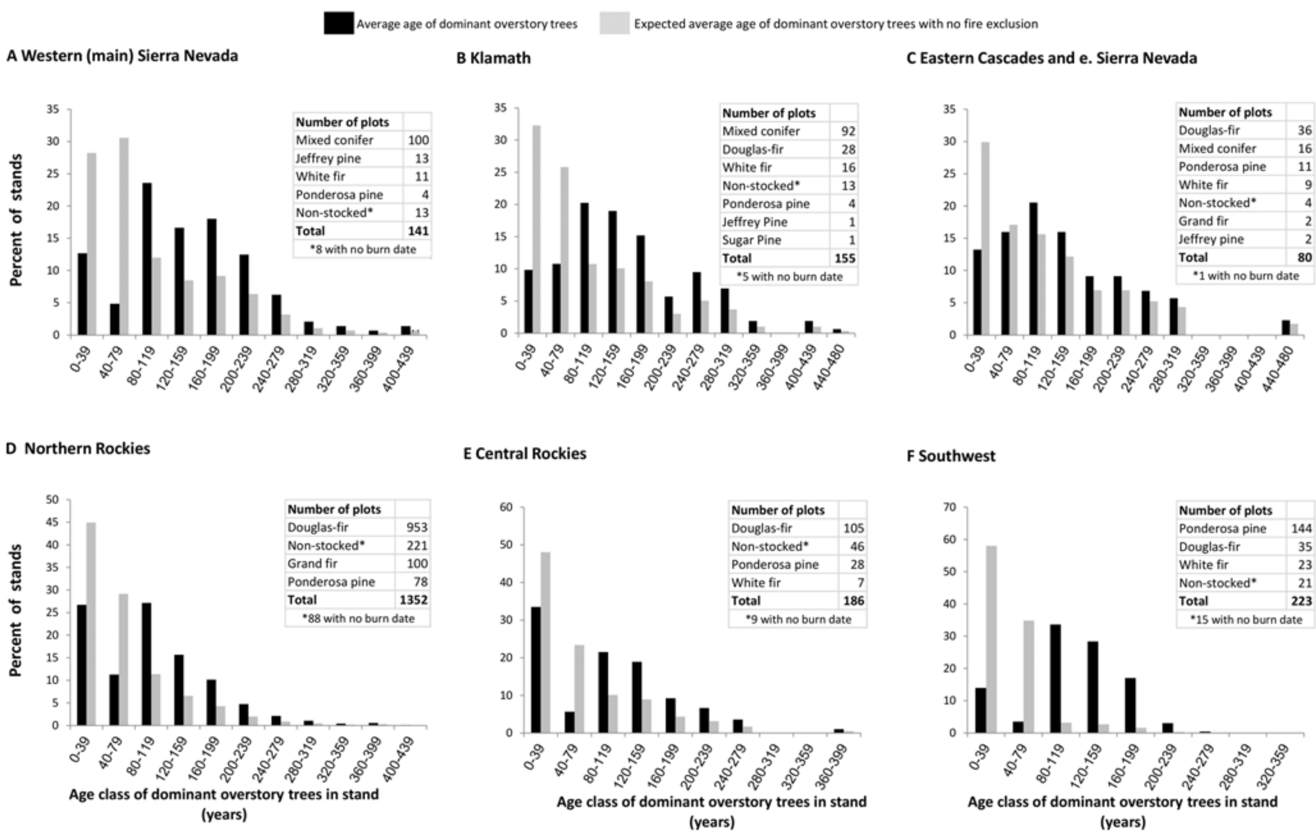
istically severe fire the pattern should reverse in both cases (i.e. increased age-class diversity in low-severity systems and decreased diversity in mixed-severity systems). We used a Chi-square test of proportions [44] to test the null hypothesis that there would be no difference between the actual distribution of stand ages and the distribution based on a hypothetical scenario of no fire exclusion. No effect of fire exclusion would indicate that fire was not a dominant influence on age class diversity. To create a distribution of average dominant stand ages by region that would exist in today's stands had fire exclusion never occurred, we used the distribution of plots with stand ages dating from 1889 or before. This time period was immediately prior to the onset of fire suppression activities by settlers and government agencies [35,45–56]. Because the average tree ages are somewhat imprecise, we binned the data into 40-year age classes for hypothesis testing. In each region, the present age structure for 80 years during effective fire suppression (1930–2009) was compared with the age structure prior to fire suppression (1810–1889). For visual analysis, we shifted the pre-fire suppression (pre-1890) tree age distributions to present (i.e., shifting 1810–1849 to 1930–1969, and shifting 1850–1889 to 1970–2009) to compare with the current age distributions (see Figure 2). This allows a clear, visual comparison of stand ages that currently exist with those that would exist had the same fire regime from 1810–1889 occurred from 1930–2009.

We included only plots where there was one stand age for the full plot because we wanted to evaluate high-severity fire occurrence in patches at least 1 ha in size, rather than include smaller torching of groups of trees. Excluding the split-age plots (27% of plots in the Sierra Nevada, 40% in eastern-Cascades/eastern-Sierra, 26% in Klamath, 14% in northern Rockies, 36% in central/southern Rockies and 14% in southwestern USA) omits some additional evidence for local high-severity fire effects; thus our results may be conservative.

We used FIA data drawn from 2001–2009, comprising 90% of available plots, in our classification of low/mid-elevation forests in the Sierra Nevada, Klamath, and eastern Cascades. In the other regions, FIA plots represented 100% of the data from low- to mid-elevation, montane forests. The number of plots in the 0–39 year age bins may be slight underestimates of the amount of high-severity fire in the last 40 years because severe fire could have occurred subsequent to the sample date (plots were sampled between 1995–2009 in the northern, central and southern Rockies, and southwestern USA and 2001–2009 in the Sierra Nevada, Klamath and eastern Cascades). To estimate the number of plots that burned severely after the sample date, we increased the 0–39 year old bin by a factor of 40/36 in the Sierra Nevada, Klamath and eastern Cascades, 40/34 in the northern Rockies and 40/32.5 in the central/southern Rockies and southwestern USA region. The denominator in these weightings is based on 40



Figures 2. Age class distributions of dominant overstory trees. Data are from US Forest Service Forest Inventory and Monitoring plots from forested areas protected from logging in A. the western (main) Sierra Nevada, B. the Klamath Region, C. the eastern Cascades and Sierra Nevada, D. the northern Rockies, E. the central/southern Rockies, and F. the southwestern USA. Shown in black bars is the current distributions of stand ages. Grey bars show an expected distribution (average age of dominant overstory trees with no fire exclusion), based on projecting the occurrence of the same age distributions that occurred from 1810–1889 into the most recent 80 year time period and rescaling these data. The number of plots by forest type are shown in the imbedded tables. Non-stocked stands are those lacking trees that grew after the fire that could be aged non-destructively.

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minus the mean amount of time in which plots in each region could have burned after being sampled.

We used year of the recent fire disturbance, captured in the disturbance data field, to define the age of very young FIA plots not containing trees that could be aged in a non-destructive way (FIA surveys do not allow trees to be killed). These "non-stocked" plots were relatively rare, as reported in the results, and ages, based on fire dates, all fell within the 0–39 age category. Some non-stocked stands had no disturbance coded. In California, Oregon, and Washington (Pacific states), disturbances were only coded if they were <6 years old. We placed all non-stocked plots where no disturbance was coded in the database into the 0–39 stand age bin.

Next, we considered whether the age distributions as shaped by fire were consistent with mixed- or low/moderate-severity fire regimes. We reasoned that a wide range in the plot stand ages in a landscape would be consistent with age-class diversity created by mixed-severity fire, while stand ages that were evenly distributed in predominantly older age classes would be consistent with a low/moderate-severity fire regime. To test whether stand age distributions were consistent with mixed- or low/moderate-severity fire regimes, we again used a Chi-square comparison of proportions [44]. Specifically, we tested the probability that the actual age distributions differed from an expected stand-age distribution for a low/moderate-severity fire regime. The low/moderate-severity (expected) distribution was based on 12.5% of stands falling into each 40-year age class between 80–399 years (0–319) years at the onset of fire exclusion. Our null hypothesis was that there would be no significant difference between the actual and expected (low/moderate-severity) distributions.

Third, we tested, again using a Chi-square comparison of proportions [44], the hypothesis that there would be less evidence for historical mixed-severity fire in the generally drier ponderosa and Jeffrey pine stands than in the mixed-conifer forests (i.e., the pine forests would be more frequently dominated by older stands).

Results

Evidence for Historical Mixed-severity Fire Regimes in Ponderosa Pine and Mixed-conifer Forests

Rotations of high- and moderate- to high-severity fire. The studies that allow calculation of rotations of high-severity fire over large, ponderosa pine and mixed-conifer forest landscapes of western North America over time periods of at least 70 years include areas ranging from 40,700 to 1,193,200 ha (Table 4). These large landscapes totaled 2.2 million ha in Baja California, the Sierra Nevada, eastern Cascades, northern Rockies (Blue Mountains of Oregon), the Colorado Front Range and Arizona (Black Mesa and the Mogollon Plateau). Most of the evidence presented in these studies was from ponderosa pine forests.

The high-severity fire rotations in Table 4 do not support the hypothesis that low/moderate-severity fire regimes were predominant in the majority of ponderosa pine and mixed-conifer forests of western North America. In all the large, forest landscapes for which data covering at least 70 years exist, high-severity fire rotations ranged from about 217 to 849 years [57], and were mostly ~200–500 years. This is generally less than potential tree lifespans. For combined moderate- and high-severity fires in the eastern Cascades, rotations were 115–128 years (Table 4: [35]), while they were 249 years in the Colorado Front Range (Table 4: [58]). In the Blue Mountains (northern Rockies) and on the Mogollon Plateau in Arizona, high-severity fire rotations of 849 and 828 years, and moderate/high-severity fire rotations of 235 and 319 years, respectively [57], occurred. Where high-severity rotations are relatively long, as they are in these regions, forest structure in portions of the landscape will lack evidence for high-severity fire even though it occurs often enough to create age-class diversity. Thus, while about 40% of the Blue Mountains forests and about 62% of those on the Mogollon Plateau had evidence from GLO surveys of forests shaped by low/moderate-severity fire only [57], similar to the nearby Coconino Plateau [59], structural diversity created by high-severity fire was evident on the remainder of the landscape [57,59].

Numerous other studies that describe historical patterns of fire behavior also have documented or described evidence for mixed-severity fire effects in the ponderosa pine and mixed-conifer forests of western North American, including the occurrence of large high-severity fire patches (Table S1), and high-severity fire occurring over substantial areas of smaller landscapes over a time period of only a few decades prior to fire exclusion (e.g., Klamath region and a transitional area between the Sierra Nevada and eastern Cascades [60–63]).

Previous studies (Table 4) have used evidence of past fire severity from a variety of sources: GLO and other survey data, historical aerial photos; and mapping of vegetation and burns done prior to fire exclusion. The GLO analyses have been formally assessed for accuracy [64]. The methods performed well for addressing general hypotheses about the presence or absence of vegetation shaped by low- or high-severity fire. This was tested using existing vegetation plot data with an error of 14.4–23% [64].

Plot age distributions. A total of 2119 FIA plots representing a sample population of about 5.1 million ha of unmanaged low- to mid-elevation, montane forests in six regions (Figure 1, Table 5) were included in our analysis. Stand ages from ponderosa pine and mixed-conifer forests across the western USA never managed for timber cover areas ranging from 192,200 ha in eastern Cascades-eastern Sierra Nevada to 3,244,800 ha in the northern Rockies. Average stand ages ranged from 0 to 814 years, with the oldest stand in ponderosa pine in the eastern Cascades. The within plot standard deviation of the proportional difference among individual tree ages and stand age across all plots was 0.14 (e.g., for stands 100 years old, one standard deviation would include individual trees ~86–114 years old, and two standard deviations would include trees ~72–128 years old).

The comparison of actual stand ages from 1930–2009 and the rescaled (expected) stand ages from 1810–1889 assuming no effect of fire exclusion are shown in Figs. 2A–F. In all regions, there were highly significant differences between the actual and expected stand age distributions (average ages of dominant trees with no fire exclusion) ($P<0.001$, Fig. 2A–F), indicating that fire was the predominant disturbance prior to effective fire exclusion. The FIA database also indicates that, since the onset of fire suppression, the great majority of stands were initiated by fire. As illustrated by the abundance of plots with stand ages that date to the decades prior to fire exclusion (e.g. 80–160 years old presently), much of the landscapes had young forests, but the rate of establishment decreased dramatically after 1930 (stand ages <80 years are rare). The rate of young forest establishment decreased by a factor of 4 in the Sierra Nevada and southwestern USA, by 3x in the Klamath, and 2x in the eastern Cascades and central and northern Rockies.

Chi-square comparisons between actual stand-age distributions at the onset of fire exclusion versus the expected stand-age distributions for a low/moderate-severity fire had exceptionally low probabilities in all regions ($P \ll .00001$, $n = 61–877$). This was because plots were mostly dominated by young and intermediate aged trees prior to fire exclusion (Figs. 2A–F). The mean stand

Table 4. Rotations for high-severity and moderate-severity fire in low/mid-elevation conifer forests of western North America.

Region	Location	Source	Analysis area (ha)	Forest types	Tree mortality	Time period	Approximate rotation (years)
Pacific states	Northern Baja California	[118][1] analysis of aerial photos	40,700	Mixed conifer and Jeffrey pine	>90% overstory mortality	1925–1991	300
	Northern Sierra Nevada	[51][2] Ground surveys and detailed maps	146,917	Mixed conifer, dominated by ponderosa pine	75% mortality by volume was mapped for patches >32.4 ha	1800–1900	488
	Eastern Cascades (Washington)	[17,35][3, 4]Analysis of historical aerial photos	175,200	Mixed conifer and ponderosa pine	>70% tree mortality[6]	~1830–1930	379–505
					>20% tree mortality[6]	~1830–1930	115–128
	Eastern Cascades (Oregon)	[56][5] Analysis of General Land Office survey data	123,500	Ponderosa pine	>70% tree mortality[7]	~1768–1882	705
			140,400	Dry mixed conifer	>70% tree mortality[7]	~1768–1882	496
Northern Rockies	Oregon (Blue Mountains)	[57][5] Analysis of General Land Office survey data	304,700	Ponderosa pine forests	>70% tree mortality[7]	~1740–1880	849
	Oregon (Blue Mountains)	[57][5] Analysis of General Land Office survey data	304,700	Ponderosa pine forests	Moderate- and high-severity fire	~1740–1880	235
Central Rockies	Central (Colorado Front Range)	[57,73][5] Analysis of General Land Office survey data	65,500	Ponderosa pine forests	>70% tree mortality[7]	~1705–1880	271
	Central (Colorado Front Range)	[58][8] Analysis of General Land Office survey data	624,156	Mostly ponderosa pine and Douglas-fir	Moderate and high-severity fire	1809–1883	249
Southwest (Arizona)	Black Mesa	[57][5] Analysis of General Land Office survey data	151,100	Ponderosa pine forests	>70% tree mortality[7]	~1760–1880	217
	Mogollon Plateau	[57][5] Analysis of General Land Office survey data	452,100	Ponderosa pine forests	>70% tree mortality[7]	~1760–1880	828
	Mogollon Plateau	[57][5] Analysis of General Land Office survey data	452,100	Ponderosa pine forests	Moderate- and high-severity fire	~1760–1880	319
	Black Mesa and Mogollon combined	[57][5] Analysis of General Land Office survey data	603,200	Ponderosa pine forests	>70% tree mortality[7]	~1760–1880	522

Data from General Land Office or mapped data over large areas (>25,000 ha) over >70 or more years prior to fire exclusion.

[1]Study area was dominated by mixed conifer and Jeffrey pine and minimally logged. Fire exclusion only began in the 1970s and has had only a modest impact [138]. Thus, historical and current rates are assumed to be comparable.

[2]Analysis of Leiberg's mapping of high-severity fire areas within unlogged mixed-conifer Sierran stands is found in [139]. According to Leiberg [51], most such fire occurred prior to 1850. In addition, he stated ''If the many small lots [<32 ha] scattered throughout still growing stands were taken into account, the figure [amount of area burned severely] would be considerably increased.''

[3]The numerator was estimated at 100 years based on ponderosa pine in this region [140], whose growth would surpass 30 cm dbh, rendering mixed and high-severity effects indistinguishable (see [35: Table 1]). This calculation is conservative because tree growth to 30 cm dbh in moister forests is faster than 100 years.

[4]High- and mixed-severity fire consistent with a definition of >70% and 20–70% basal area mortality, respectively, was identified from overstory canopy percentage, the overstory size class, the understory size class, and the fire tolerance of the cover type (see [35: Table 1]). Large patch sizes of historical high severity fire (100s to >5000 ha) from this work are reported in [17].

[5]The estimate is from the span of years over which fire effects were distinguishable, using forest structure evident in the Government Land Office historical survey data, divided by the fraction of the forested landscape in which those fires occurred [56]. Rotations for high-severity fire are determined by dividing the observation period (the period of time over which fire effects are distinguishable by stand structure) by the percentage of the landscape experiencing high-severity fire. The methods were found to have 14.4–23% accuracy compared to plot sampling.

[6]High- and mixed-severity fire, consistent with a definition of >70% and 20–70%, respectively, were identified from overstory canopy percentage, the overstory size class, the understory size class, and the fire tolerance of the cover type (see [35: Table 1]).

[7]High-severity consistent with a definition of >70% basal area mortality [35] was identified having a percentage of small trees >50% and a percentage of large trees <20% [56,57,73],

[8]Estimated from the length of General Land Office section lines intercepted by moderate- and high-severity fire. Accuracy tests using the length of section lines intercepted by modern moderate- and high-severity fire yielded a relative error of 15.6%.

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Table 5. Forest Inventory and Analysis (FIA) data.

Region	Number of plots (n) and forest area randomly sampled (ha)	Mean FIA stand age (yrs)		Test for difference in stand initiation since 1930 vs. 1800–1900: Chi-square, *P*
		Current	In 1930	
Sierra Nevada (main)	n = 232 338,400	148	97	86.3, <<0.001
E. Cascades and E. Sierra Nevada	n = 135 192,000	155	114	25.4, <<0.001
Klamath Mountains	n = 251 372,000	157	111	43.9, <<0.001
Central Rockies	n = 276 446,400	105	75	58.9, <<0.001
Northern Rockies	n = 1929 3,244,800	105	70	333.8<<0.001
Southwestern US	n = 319 492,000	116	59	188.2<<0.001

Area of sample population randomly sampled, mean stand age currently, and in 1930, and Chi-square test results.
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ages at the time of onset of fire exclusion were 59–114 years, depending on the region, considerably shorter than current mean ages (105–148 years: Table 5). Therefore, the FIA data were inconsistent with the hypothesis that the ponderosa pine and mixed-conifer forests of western North America, in unmanaged landscapes, were predominantly park-like with low age-class diversity due to the dominant influence of low/moderate-severity fire.

The hypothesis that mixed-severity fire prior to fire exclusion would be lower in the driest (ponderosa and Jeffrey pine) forests than other forests also was not supported. Based on stand-ages (not shown), there was as much as or more mixed-severity fire in the pine forests. In the Pacific states, we found almost identical stand-age distributions from 1800–1900 in ponderosa/Jeffrey pine stands (n = 20 plots) versus all non-ponderosa stands (n = 204 plots). Plots from the time period 1800–1900 accounted for 70% and 73%, respectively, of all plots with dominant trees that established in or before 1900. In the northern and central Rockies, 86% of ponderosa pine stands (n = 66 plots) and 81% of the non-ponderosa pine stands (n = 615 plots) that established in or before 1900 had stand-ages between 1800 and 1900 ($\chi^2 = 0.85$, n = 676, $P > 0.6$). Likewise, in the southwestern USA, 98% of ponderosa pine stands (n = 96 plots) and 92% of the non-ponderosa stands (n = 37 plots) that established in or before 1900 had stand-ages between 1800 and 1900 ($\chi^2 = 1.27$, n = 133, $P > 0.25$). However, when all plots were considered, significantly more stands established from 1800–1900 in ponderosa pine than non-ponderosa forests ($\chi^2 = 11.96$, n = 1038, $P < 0.001$), indicating higher fire disturbance in pine forests.

Comparing the Weight of Landscape-scale Evidence by Region

The consistency of multiple lines of evidence for mixed-severity fire in the ponderosa pine and mixed-conifer forests is an important finding. In all regions, there were tree-age data supporting considerable age-class diversity created by mixed-severity fire, and a paucity of undisturbed park-like forests. The full weight of landscape-scale evidence is greatest in the regions with area-specific rotations of severe fire from GLO data: the eastern Cascades, nearby Blue Mountains in the northern Rockies, central Rockies, and southwestern USA (Table 4). In the Cascades, these data are further supported by analyses of early aerial photography at a regional scale [35], and in small landscapes [61–63] and numerous historical descriptions (see [56]: Table S1). In the northern Rockies, historical documentation (e.g., [45–48,50,53,54]) of mixed-severity regimes has been summarized in regional reviews [16,65,66], and stand-age reconstructions of historical fire regimes indicate mixed-severity fire in ponderosa-pine/Douglas-fir forests [67–69]. In the Colorado Front Range, the findings based on GLO data [57,58] are remarkably consistent with earlier studies based on tree-ring stand reconstructions from broadly distributed samples [70–72]. In the

Table 6. Current high-severity fire rotations.

Region	Source	Forest Types	Time period	Rotation (yrs)
Sierra Nevada, southern Cascades	[132]	All low/mid- and mid/upper elevation conifer forests	1984–2010	645
Klamath (all)	[129]	All low/mid-elevation conifer forests	1984–2005	599
Eastern Cascades (all)	[129]	All low/mid-elevation conifer forests	1984–2005	889
Northern Rockies	[92]	Ponderosa pine forests	1980–2003	500
Central Rockies	[92]	Ponderosa pine forests	1980–2003	714
Central Rockies	[58]	Ponderosa pine forests	1984–2009	431[1]
Southwest	[92]	Ponderosa pine forests	1984–2003	625
Northwest (Eastern Cascades and Blue Mountains)	[92]	Ponderosa pine	1984–2003	1,000

Data cited are from low/mid-elevation conifer forests in western North America.
[1]Higher-severity fire: includes moderate- and high-severity.
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southwestern USA, GLO data are supportive of mixed-severity fire on most of Black Mesa and much of the Mogollon and Coconino Plateaus [57,59], while a number of other studies also describe evidence for mixed-severity fire [9,14,55,73–77].

The remaining forest regions that we assessed lack GLO analyses. However, in the Sierra Nevada and Klamath regions historical surveys and early air photo data describe mixed-severity fire regimes [20,30,49,51,60,78–87] (see Table S1 for descriptions). In all regions except the Klamath, there are multiple lines of evidence from landscape-scale studies, each supporting mixed-severity fire. In contrast, evidence supporting low/moderate-severity fire is confined to relatively small areas (e.g., [88–91]).

Historic vs. Contemporary Fire Regimes

We did not find evidence to support the hypothesis that fire exclusion has greatly increased the prevalence of severe fire in ponderosa pine and mixed-conifer forests (Tables 4–6, and Figs. 2A–F). Comparing current versus historical high-severity fire rotations, we found that current rotations were generally longer (less high-severity fire) in the Sierra Nevada and central Rockies (Tables 4 and 6, Table S1). No direct historical comparison could be made between current and historical high-severity rotations in the Klamath and northern Rockies at the spatial scale required in Table 4, but evidence presented in Table S1 suggests that current rotations of 599 years and 500 years, respectively, may be longer. The estimated rotation of 625 years for recent high-severity fire in the southwestern USA [92] was shorter than the historical estimate of 828 years for the Mogollon Plateau in Arizona. Combining the Mogollon Plateau and Black Mesa to provide a better comparison with fire across the southwestern USA produces a historical high-severity rotation of 522 years [57]. In the eastern Cascades, high-severity fire rotations since 1984 (889 years) were longer than historical rotations (Table 6 vs. Table 4).

Discussion

Historical Fire Regimes

The primary objective of this paper was to address how prevalent mixed-severity fire regimes were historically in ponderosa pine, mixed conifer, and other low- to mid- elevation, montane forests of western North America. We hypothesized that age-class diversity was low, consistent with long-term effects of low/moderate-severity fire regimes (Table 1). We reviewed evidence with inference across both large and smaller landscapes across many forest regions. The majority of the evidence did not support the low/moderate-severity fire hypothesis, but, instead, supported the alternate hypothesis that mixed-severity fire shaped these forest landscapes. This finding applies to Pacific states ponderosa pine, Jeffrey pine, and California mixed-conifer forests, as well as ponderosa pine and mixed-conifer forests in the eastern Cascades, Rockies and southwestern USA, where low/moderate-severity regimes have often been applied. In some areas (Blue Mountains, Mogollon and Coconino Plateaus) high-severity fire occurred at less frequent intervals (rotations of 828–849 years) [57,59]. Even at these rotations, high-severity fire creates considerable age-class diversity in a landscape, and moderate/high-severity fire rotations were 235–319 years, which further enhances diversity (with small groupings of high-severity fire interspersed within moderate-severity fire areas).

FIA stand ages in the unmanaged forests in all regions reflect a pattern of high age-class diversity occurring prior to federal fire suppression policies and reductions in Native American burning (by the early 20th century) with the arrival of settlers [20,46,49,51] Natural disturbances occurred at rates that led to stands numerically dominated mainly by young and intermediate-aged trees. Disturbance processes dramatically declined following the onset of fire exclusion, suggesting fire was the primary disturbance agent [35]. However, in considering the age patterns of dominant trees in the FIA plots, it is essential to also address alternative explanations for the dominance of young- and intermediate-aged stands prior to fire exclusion, such as climate variability and disturbance by insect outbreaks.

While we recognize that climate variability influences rates of tree regeneration generally [93], and may determine success or failure of tree regeneration specifically following disturbance, we believe that the broad patterns of dominant overstory tree ages in the FIA plots mainly reflect the effects of past fire for several reasons. The dominant stand ages of young and intermediate aged trees prior to fire exclusion are consistent with periodic disturbances with significant tree mortality that shifted dominance to a new generation of trees, rather than solely episodic tree establishment due to climatic variation at a multi-decadal scale. This is supported by research in the central Rockies where, at a multi-decadal time scales, large datasets of tree recruitment dates over the past c. 250 years do not correlate with moister climate at the same time scale, but instead correlate with drier climate that was conducive to high-severity fires [70–72,91]. Likewise, studies in the same area show that outbreaks of bark beetles and defoliators result in growth releases of non-host trees rather than even-aged, multi-species tree cohorts [94,95], thus facilitating discrimination from post-fire stand structures [91]. Fire exclusion was likely effective in some areas between 1900 and 1930, which could have led to understory tree recruitment in this time frame. However, research suggests that in some areas the favorable influences of timber harvesting and/or cattle grazing on tree establishment may confound the attribution of tree recruitment to fire exclusion [96]. In addition, the plot age data demonstrate that recruitment was just as common or more common in decades before 1900 as between 1890 and 1930. Lastly, while it is possible that greater mortality in older trees, from competition or insects and pathogens, might explain high levels of recruitment prior to fire exclusion, we do not see this pattern during the suppression era. Thus, higher levels of mortality in older trees seems likely to have been caused by fire.

Our findings illustrate the need for studies with a spatial scale of inference suited to describing patterns across large, heterogeneous landscapes. This is illustrated by three recent studies from old forest stands (one in the Black Hills (500 ha), one in the Sierra Nevada (3,000 ha), and one in the southwestern USA (307 ha)) that reported very little or no historical high-severity fire, and hence low-severity regimes (Table S1: [88–90]). In contrast, broader-scale analyses of historical data for the Sierra Nevada (Table S1: [78]:), Black Hills [65], and southwestern USA [57] suggest fire regimes in the broader landscape within which these three studies occurred were mixed-severity.

A fourth study [97] analyzed 1914–1922 Bureau of Indian Affairs (BIA) timber cruise plot data from within a larger area (38,651 ha), and found relatively low tree densities in ponderosa pine and mixed-conifer forests of the eastern Klamath region in Oregon, and suggested that forests were too open to support any significant crown fire. However, only a subset of the townships surveyed by BIA in these forest types were included in the analysis (Table S1), and the surveys did not include trees 10–15 cm dbh, which comprise ~20% of all trees [97], and most surveys did not include lodgepole pine, which comprise ~10% or more of these forest types in that region within unlogged areas [49]. In addition, historical data indicate that extensive timber harvesting had occurred in the areas analyzed by 1914–1922 (Table S1), and evidence of previous timber harvesting was not among the factors that BIA surveyors were required to note (Table S1). Tree densities in unlogged reference ponderosa pine and mixed-conifer forests in this landscape from the late 19th century and early 20th century indicate much denser and more variable forest conditions (Table S1). Also, USGS surveys conducted in the 1890s within unlogged ponderosa pine and mixed-conifer forests across a larger expanse (310,267 ha) map substantial high-severity fire from 1855–1900 (high-severity rotation of 352 years), suggesting a mixed-severity regime (Table S1).

The absence of evidence for mixed-severity fire in some older forests selected for study may be due to fire exclusion. If the effect of fire exclusion in reducing mixed-severity fire is not accounted for in describing reference conditions, it may lead to shifting baseline syndrome (i.e., a system is not measured against the true baseline, but against one that already has departed from the true baseline [98]). This effect may be caused or compounded by diminishing evidence of age-class diversity. For example, high-severity fire can be mapped at landscape scales from early air photos [9,17,61–63,99], but the same historic fire effects may not be visible from current imagery that can be used for assessing landscape-scale patterns.

Data with greater temporal depth than analyzed here can better capture past variability in the frequency of large fire events. Thus, it is noteworthy that paleoecological studies also support mixed-severity fire regimes for the ponderosa pine and mixed-conifer forests. These studies have found charcoal depositions from major fire episodes in ponderosa pine and interior Douglas-fir forests occurring for millennia in the northern Rockies (central Idaho: [100,101]), Klamath [102], Sierra Nevada [103], eastern Oregon Cascades [104], and southwestern USA [105–107]. These major episodes are generally interpreted as large, severe fire events [101–107].

The occurrence of mixed-severity fire prior to fire exclusion is also well supported by another line of evidence: the potential behavior of wildfire as affected by weather and climate. Based on direct observations of fire behavior, high winds (generally 10 m open wind speeds >32–35 kilometers/hr) may subject virtually any conifer forest, regardless of fuel density, to crown fire [108]. Thus, empirical data call into question a major premise of the low/moderate-severity fire regime: that ponderosa pine and mixed-conifer forests may be completely resistant to crown fire. Fire intensity increases with winds, and at winds of >30 km/hr spot fires may be ignited over 1 km ahead of the fire front [109]. The coalescing of separate spot fires with the fire front can further energize wind-driven fire [110,111]. Severe droughts also intensify fires by reducing fuel moisture to extremely low levels, allowing crown fire under less windy conditions [108,112]. Severe drought years throughout much of western North America occurred from 1856 to 1865, 1870 to 1877 and 1890 to 1896 [113]. The extensive high-severity fires of 1910 (the Big Burn in Idaho and Montana), when large areas of drier forests burned at high severity prior to fire exclusion–much of it in ponderosa pine–illustrate how fire behavior that is rare temporally due to extreme climate and weather can dominate in space [1]. Many fire episodes in the charcoal records that exceed modern fires undoubtedly involve combinations of extreme wind, drought, and mass fire.

The largest patch sizes of high-severity fire likely occurred during the most extreme conditions for fire behavior. While patch sizes of high-severity fire are difficult to document, it follows from commonly observed heavy-tailed distributions of patch sizes created by fire [114,115] that very large patches of high-severity fire (thousands of ha, e.g., [17: Fig. 1, 58]) were a primary reason why considerable area exhibited forest structure consistent with high-severity fire historically in all regions. Large patches, though numerically subordinate, are dominant in terms of total area burned, while the opposite applies to small patches [58].

There is abundant evidence that past forests may not have required extreme weather and climate for mixed-severity fire to have occurred. Younger, more flammable forests [32] appear to have been widespread in dry-forest regions based on dominant stand ages prior to fire exclusion (Fig. 2). In addition, the ranges in fire-free intervals in many low- to mid-elevation forested areas were sufficient to allow for substantial vegetation growth and recovery of fuels between fires (e.g., 20–50+ year rotations [61–63,116–118]). For example, in the Sierra Nevada, fuels may recover to pre-burn levels in nine years [119,120], so fire-free interludes (or fire rotations), more often than not, may have been sufficient to allow growth of significant amounts of high-energy shrub fuels. In describing low/mid-elevation forests throughout the northern Sierra Nevada, Leiberg [51: page 32) states: "There is a great amount of undergrowth in the forest which has attained its present proportions chiefly through the agency of fire. Most of it [undergrowth] consists of species of *Ceanothus*." For mid-elevation forests, he reports (page 37): "Nearly all the type situated at altitudes below 7,000' [2134 m] carries a vast amount of undergrowth. It consists mainly of manzanita [*Arctostaphylos* spp.], ceanothus, and scrub oak [*Quercus chrysolepis, Q.vaccinifolia*]." Similarly abundant shrub fuels were also documented historically in the westside of the central/southern Sierra Nevada [51], in the eastern Oregon Cascades [56: Appendix A] and in Oregon's Blue Mountains [57]. Flame lengths in actively burning manzanita and ceanothus are typically 4–5 times the ~1–2 m height of the shrubs, sufficient to cause ignition of forest canopy tree crowns under favorable burning conditions. Many of these shrub species recruit primarily, if not exclusively, after severe fire, and their occurrence is a further indication of the historical presence of such fire [121].

Changes in Fire Regimes and Stand Age Distributions with Fire Exclusion

We also hypothesized, consistent with existing concerns about unprecedented fire severity in western North America (e.g., [6–9,11,13,15,17,28]), that fire exclusion has greatly increased the prevalence of severe fire in ponderosa pine and mixed-conifer forests. We found little support for this hypothesis. Over the full period of effective fire exclusion in unmanaged forests, average ages of dominant overstory trees in FIA plots suggest there has been about a threefold to fourfold decrease in stand initiation due to fire in the Sierra Nevada, Klamath, and southwestern USA, and about a twofold decrease in the eastern Cascades, central and northern Rockies (Figs. 2A–F). In addition, patch sizes of high-severity fire in the central Rockies have not increased [58]. Our assessment of high-severity rotations based upon existing literature also revealed a generally lower incidence of high-severity fire in these forests in recent decades (Tables 4 and 6, and S1).

Conclusion

The importance of multiple lines of evidence has been stressed in determining whether mixed-severity fire regimes applied historically [122]. Our results illustrate broad evidence of mixed-severity fire regimes in ponderosa pine and mixed-conifer forests of western North America. Prior to settlement and fire exclusion, these forests historically exhibited much greater structural and successional diversity than implied by the low/moderate-severity model (Table 2). Lack of recognition of past variability in fire may be due, in part, to common misclassifications of fire regimes. To improve clarity in communication, we propose that "low/moderate-severity" be applied to those regimes where, as the term implies, high-severity fire is absent. These circumstances appear to be quite rare in the ponderosa pine and mixed-conifer forests of western North America. Therefore, a fire regime with a high-severity component of any amount should not be classified as low/moderate-severity [e.g., 9,17,28].

Our findings suggest a need to recognize mixed-severity fire regimes (Table 2) as the predominant fire regime for most of the ponderosa pine and mixed-conifer forests of western North America. Given societal aversion to wildfires, the threat to human assets from wildfires, and anticipated effects of climate change on future wildfires, many will question the wisdom of incorporating historical mixed-severity fire into management goals. However, focusing fire risk reduction activities adjacent to homes is needed to protect communities [123], and this may expand opportunities for managed wildland fire–away from towns–for ecological benefits of fire-dependent biota. However, a major challenge lies with the transfer of information needed to move the public and decision-makers from the current perspective–that the effects of contemporary mixed-severity fire events are unnatural, harmful, inappropriate and more extensive due to fire exclusion–to embrace a different paradigm [124]. This paradigm would not emphasize a single, appropriate condition, but would explicitly recognize the vital role of variation in fire in maintaining successional diversity and fire-dependent biota [125], and allow natural rates of ecological succession [18,19,126–128]. It would also recognize that these effects have generally diminished, and that more fire, including high-severity fire, where it is in deficit, is an ecologically desirable goal. Of course, while most current research indicates that fire severity is not increasing in ponderosa pine and mixed-conifer forests of western North America [129–132], it will be critical to continually assess fire regimes in a changing climate.

For management, perhaps the most profound implication of this study is that the need for forest "restoration" designed to reduce variation in fire behavior may be much less extensive than implied by many current forest management plans or promoted by recent legislation. Incorporating mixed-severity fire into management goals, and adapting human communities to fire by focusing fire risk reduction activities adjacent to homes [123], may help maintain characteristic biodiversity, expand opportunities to manage fire for ecological benefits, reduce management costs, and protect human communities.

Supporting Information

Table S1 Evidence of historic fire severity in ponderosa pine and mixed-conifer forests of western North America. A summary of published studies and historical documents that provide evidence regarding mixed-severity fire in the ponderosa pine and mixed-conifer forests of western North America, but do not provide sufficient information to estimate high-severity fire rotations, or were conducted in smaller landscapes. Many fire scar studies have also been done in these forests, but fire scars alone are not sufficient to distinguish low-from mixed-severity regimes.
(DOCX)

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Author Contributions

Conceived and designed the experiments: DCO CTH. Performed the experiments: DCO CTH AA WLB DAD RH WK MAM RS TTV MAW. Analyzed the data: DCO CTH AA WLB DAD RH WK MAM RS TTV MAW. Wrote the paper: DCO CTH AA WLB DAD RH WK MAM RS TTV MAW.

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FORMAL COMMENT

Areas of Agreement and Disagreement Regarding Ponderosa Pine and Mixed Conifer Forest Fire Regimes: A Dialogue with Stevens et al.

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Abstract

In a recent PLOS ONE paper, we conducted an evidence-based analysis of current versus historical fire regimes and concluded that traditionally defined reference conditions of low-severity fire regimes for ponderosa pine (*Pinus ponderosa*) and mixed-conifer forests were incomplete, missing considerable variability in forest structure and fire regimes. Stevens et al. (this issue) agree that high-severity fire was a component of these forests, but disagree that one of the several sources of evidence, stand age from a large number of forest inventory and analysis (FIA) plots across the western USA, support our findings that severe fire played more than a minor role ecologically in these forests. Here we highlight areas of agreement and disagreement about past fire, and analyze the methods Stevens et al. used to assess the FIA stand-age data. We found a major problem with a calculation they used to conclude that the FIA data were not useful for evaluating fire regimes. Their calculation, as well as a narrowing of the definition of high-severity fire from the one we used, leads to a large underestimate of conditions consistent with historical high-severity fire. The FIA stand age data do have limitations but they are consistent with other landscape-inference data sources in supporting a broader paradigm about historical variability of fire in ponderosa and mixed-conifer forests than had been traditionally recognized, as described in our previous PLOS paper.

Introduction

The accompanying paper by Stevens et al. [1] is critical of one of the several lines of evidence in Odion et al. (2014) [2] that indicate the traditional reference conditions of low-severity fire regimes are incomplete for most ponderosa pine and mixed-conifer forests of western North America. Specifically, Stevens et al. [1] believe that the stand age attribute in Forest Inventory and Analysis (FIA) data is not a useful descriptor of historical fire regimes in ponderosa pine and mixed-conifer forests.

Here, we first briefly summarize points of agreement between Stevens et al. [1] and us, and then discuss in more detail areas where we disagree, including the analysis and interpretation of FIA stand age data. Authorship of this reply is comprised by those who conducted the FIA portion of Odion et al. (2014) [2], as well as authors of Odion et al. whose contributions and backgrounds were needed to respond to FIA-critique elements by Stevens et al. [1] that went beyond the scope of the FIA analysis in Odion et al. (2014) [2].

Areas of Agreement

High-severity fire is a natural component of ponderosa pine and mixed-conifer fire regimes

In Odion et al. (2014) [2], we presented several lines of converging evidence that high-severity fire was an important part of historical fire regimes in ponderosa pine and mixed-conifer forests. Over three-quarters of our results pertained to lines of evidence other than FIA stand age data. Stevens et al. [1] reviewed this evidence, some of which was based upon studies published by co-authors of Stevens et al., and concluded the following: "High-severity fire was undoubtedly a component of fire regimes in ponderosa pine and drier mixed-conifer forests." This represents a significant shift from perspectives in much of the literature in recent decades, which often mentions only low- or low-moderate severity fire in describing historical fire regimes in ponderosa pine and mixed-conifer forests.

Significant tree recruitment occurs in the absence of fire

We did not intend to suggest that tree recruitment occurred only with fire. Stevens et al. hypothesize that pulsed recruitment in the absence of fire has shaped the age distributions in many FIA plots. We agree that this process occurred historically. There is also agreement that a dominant cohort of trees will establish after high-severity fire, but that later in stand development understory recruitment can happen with favorable climate or following insect outbreaks. This, along with the presence of some trees that pre-date the fire, will create an uneven-aged stand, but there may still be a dominant overstory size class established after fire.

FIA Stand Age Data May Provide Evidence Consistent with Past High-Severity Fire

Stevens et al. [1] report that 42% of the FIA plots used in Odion et al. (2014) [2] had demographic characteristics consistent with a mortality and recruitment event corresponding generally with the FIA stand age. These plots had an estimated 0–10% of the stand basal area in trees that were older than the stand age. The rest of the basal area (all of it in many cases) was from trees that established after (more recently than) the stand age date, even though most of the plots had stand ages < 200 years old. Despite some qualifications, Stevens et al. [1] conclude that it is plausible that these 42% of plots were visited by historical high-severity fire. However, although Stevens et al. recognize high-severity fire as a component of ponderosa pine and mixed-conifer forests, the definition (threshold of mortality) and patch size of high-severity fire remain a matter of considerable debate.

Areas of Disagreement

Appropriate threshold of mortality for high severity fire

Stevens et al. replaced the traditional 70–100% mortality definition for high-severity fire (see, e.g., [3]) that we used with a new 90–100% definition, which means their analysis does not replicate ours and does not refute our findings. Even though this replacement invalidates their analysis of our study, 42% of the FIA plots still have demographic characteristics consistent with high-severity fire with their narrowed definition. Using our original 70–100% mortality definition, there is agreement on 68% of FIA plots regarding demographic characteristics consistent with high-severity fire (and the level of agreement is even higher than this, due to a calculation error in Stevens et al., as discussed below).

Stevens et al. [1] suggest, based on findings of Miller and Quayle (2015) [4], that the high-severity fire definition used by Odion et al. (2014) [2] should be narrowed from 70–100% basal area mortality to 90–100% basal area mortality because Miller and Quayle found that high-severity fire field plots with less than 100% tree mortality were rare. However, 34% of all of their plots with $\geq$75% basal area mortality had live, surviving trees [4]. Thus, surviving trees in high-severity fire plots were not rare based on data that they cite. Further, Miller and Quayle [4] used plots ranging in size from 0.07 ha to 0.63 ha, while FIA plots consist of four subplots spread over an area of 1.0 ha. Thus, the plots of interest here are more likely to contain surviving trees than those of Miller and Quayle [4]. Further, Miller and Quayle (2015) [4] indicate a user and producer accuracy of 11.1 and 19.2 percent for classifying areas with 75–89% percent basal area mortality. Therefore areas with 75–89% mortality were often not identified correctly in their study.

There is also a logical problem: if high-severity fire predominantly caused 90–100% mortality historically, and 70–89% mortality was rare, then there would be very little difference between the number of FIA plots with 90–100% mortality and the number with 70–100% mortality. But, Stevens et al. found a large difference when using these basal area thresholds. Therefore, plots with 70–89% mortality were not rare, and narrowing the fire-severity definition is not supported.

Stevens et al. [1] state that the "minimum threshold of 70% mortality used by Odion et al. [2] to describe a high-severity patch (and the 75% threshold employed by Landfire) was not developed to describe mortality within a stand, but rather mortality across an entire fire." However, the two studies cited by Stevens et al. [1] to support this, Agee (1993) [3] and Barrett et al. (2010) [5], say the opposite (see page 23 of Agee 1993 [3], and page 30 of Barrett et al. (2010) [5].

Plot sizes needed to define high-severity fire

Stevens et al. [1] point out that FIA plot footprints are only 0.4 ha in size in California, Oregon, and Washington, and are only 0.067 ha in size in the other western U.S. states, and use this to suggest that the FIA plots analyzed by Odion et al. [2] were too small to capture true high-severity fire effects. However, Stevens et al. [1] recognize high-severity fire patches as small as 0.4 ha as representing high-severity fire effects. Further, although the total footprint of subplots in FIA plots may be only 0.067 or 0.4 ha, these subplots are representative of a 1.0 ha area. The FIA plots do not capture the size and shape of patches of historical fire, and do not encompass many high-severity patches, which we recognize. But, because they are probabilistic samples, the amount of high-severity fire captured by FIA is a statistical estimate of total amount of high-severity fire. It would be a problem if high-severity fire were rare, or if only a small number of FIA plots were analyzed, but evidence for high-severity fire was abundant, and we analyzed thousands of plots.

Use of diameter-age relationships for reconstructing past basal area of trees

To understand historical forest structure and fire, it is common to reconstruct the size of trees in the 1800s by subtracting tree growth since that time (e.g., [6]). Stevens et al. recognize that the "basal area of the surviving older trees would have increased in the decades between the year implied by the FIA stand age and the measurement date, thus potentially overestimating their past contribution to the stand basal area in the year implied by the FIA stand age." In other words, to the extent that the basal area of surviving trees is overestimated, this translates directly to an under-representation of the potential occurrence of historical high-severity fire. However, Stevens et al. [1] did not subtract the basal area that overestimates the past contributions of surviving trees. The effects can be seen via the following general simulation.

Suppose a plot was burned by high-severity fire 100 years ago with 6.1% basal area surviving fire consisting of 16 m^2 of dead tree basal area. There are 5 live trees of 0.5 m in diameter at breast height (dbh) in the 1-ha FIA plot for a total of 1 m^2 live, surviving basal area. The surviving trees have a higher growth increment in earlier years which decreases as they age. However, when the mean growth rate is calculated using 1594 mature ponderosa pine in dry forests in Oregon [7], the effects of the slower growth at old age is included to give a mean of 0.45 cm dbh/yr. By not considering the growth rates of surviving trees, surviving basal area at the time of the fire would be overestimated by 3.5 times 100 years later. After two hundred years, the age of some FIA plots, the overestimate would be nearly 8 times the actual plot survivorship, with nearly half the basal area incorrectly considered to have survived since prior to the stand age date. Mortality of mature trees after (more recent than) the stand age date could have occurred in some cases, reducing the overestimates by Stevens et al., but this would likely be a small amount compared to the large magnitude of the overestimates. Thus, the potential effects of high-severity fire were greatly underestimated by Stevens et al.

Evidence for historical high-severity fire patches >1,000 ha in size

Stevens et al. [1] suggest that high-severity fire patches >1,000 ha in size in some current fires represent a "departure" from historical conditions. However, DellaSala and Hanson (2015) ([8]: pp. 30–33) present numerous examples of historical data sources documenting high-severity fire patches >1,000 ha occurring before fire suppression in previously unlogged forests in both ponderosa pine and mixed-conifer forest types in every major region of the western U. S. Even though large high-severity patches may have been infrequent, they accounted for most high-severity fire [9].

Combining fire scar data and stand structure data from different plots

Stevens et al. [1] try to test the hypothesis that there would be minimal tree recruitment in the absence of high-severity fire in the FIA plots we studied. However, the locations chosen by Stevens et al. [1] to evaluate recruitment and fire in FIA plots did not actually include any FIA plots. The locations were mostly subjectively selected plots known to not have had severe fire in their long fire-scar history. The plots were up to 1 km away from any FIA plots. Therefore, they do not represent the population of FIA plots we studied.

Fire and tree recruitment

In all six regions we analyzed in Odion et al., the onset of fire suppression about a century ago coincides with a dramatic reduction in the initiation of trees that form the dominant overstory size classes. Thus, the removal of fire had a profound effect on the process of recruitment over vast areas. Recruitment following fire suppression, as hypothesized by Stevens et al., could not account for the pattern of abundant establishment of the dominant size classes of trees before fire suppression. If high-severity fire was a minor process in creating new stand ages, establishment of the dominant overstory trees would not have declined so dramatically with fire suppression.

Stevens et al. claim that "Most" ponderosa pine forests and "many" low/mid-elevation mixed-conifer forests historically were "Low-density" forests with frequent, fuel-limited low/moderate-severity fire regimes is not supported by the evidence

This suggestion by Stevens et al. [1] overstates certain evidence, and does not consider other evidence. The sources cited by Stevens et al. [1] are a biased selection of studies that were mostly conducted at relatively local spatial scales, and were often in old-growth forests that are inherently low-density and by definition have not experienced high-severity fire for centuries. The sources cited by Stevens et al. also include studies of current tree densities that try to determine past tree densities but do not have any way to measure historical trees that died, fell, and decayed, and studies where the past effects of logging or fuel wood cutting (when mining occurred and large amounts of wood fuel was needed) cannot be ruled out [2] or where incomplete historical survey data were used [10]. Additionally, Stevens et al. omit reference to dozens of scientific sources indicating more variable historical ponderosa pine and mixed-conifer forests.

In contrast, Odion et al. (2014)[2] reviewed dozens of historical data sources and reconstructions, finding that historical ponderosa pine and mixed-conifer forests: (1) were highly variable in structure/density; (2) had highly variable fire severity, and most forests were dominated by mixed- and high-severity fire; and (3) consistently had a significant component of open forests dominated by low-severity fire at any given time.

Conclusion

The concern raised by Stevens et al. [1] pertains to only one of the multiple lines of evidence in Odion et al. [2] that together strongly support the historical importance of high-severity fire in ponderosa pine and mixed-conifer forests of the western U.S. Stevens et al.'s comment, specifically on stand age analysis based on Forest Inventory and Analysis field plots, does not refute our study. This is because it is based on a different definition of high-severity fire than the classical definition used by Odion et al. (2014) [2], which is consistent with scientific literature. The new definition proposed by Stevens et al. [1] is based on errors and mischaracterizations of cited sources. Using our definition or theirs of high severity, Stevens et al. [1] found that many FIA plots had demographic structure consistent with a high-severity fire in the 200 years prior to fire suppression and the number of these plots was likely a large underestimate due to the improperly narrow definition of high-severity fire used by Stevens et al., and a major calculation error in their methods.

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Author Contributions

Analyzed the data: DCO CTH. Wrote the paper: DCO CTH WLB DAD MAW.

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Effects of Fire and Commercial Thinning on Future Habitat of the Northern Spotted Owl

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Abstract: The Northern Spotted Owl (*Strix occidentalis caurina*) is an emblematic, threatened raptor associated with dense, late-successional forests in the Pacific Northwest, USA. Concerns over high-severity fire and reduced timber harvesting have led to programs to commercially thin forests, and this may occur within habitat designated as "critical" for spotted owls. However, thinning is only allowed under the U.S. Government spotted owl guidelines if the long-term benefits clearly outweigh adverse impacts. This possibility remains uncertain. Adverse impacts from commercial thinning may be caused by removal of key habitat elements and creation of forests that are more open than those likely to be occupied by spotted owls. Benefits of thinning may accrue through reduction in high-severity fire, yet whether the fire-reduction benefits accrue faster than the adverse impacts of reduced late-successional habitat from thinning remains an untested hypothesis. We found that rotations of severe fire (the time required for high-severity fire to burn an area equal to the area of interest once) in spotted owl habitat since 1996, the earliest date we could use, were 362 and 913 years for the two regions of interest: the Klamath and dry Cascades. Using empirical data, we calculated the future amount of spotted owl habitat that may be maintained with these rates of high-severity fire and ongoing forest regrowth rates with and without commercial thinning. Over 40 years, habitat loss would be far greater than with no thinning because, under a "best case" scenario, thinning reduced 3.4 and 6.0 times more dense, late-successional forest than it prevented from burning in high-severity fire in the Klamath and dry Cascades, respectively. Even if rates of fire increase substantially, the requirement that the long-term benefits of commercial thinning clearly outweigh adverse impacts is not attainable with commercial thinning in spotted owl habitat. It is also becoming increasingly recognized that exclusion of high-severity fire may not benefit spotted owls in areas where owls evolved with reoccurring fires in the landscape.

Keywords: Fire rotation, forest regrowth rate, forest thinning, future habitat, habitat loss, late-successional forest, policy implications, severe fire, spotted owl.

INTRODUCTION

Conservation of the emblematic Northern Spotted Owl (*Strix occidentalis* ssp. *caurina*) in the Pacific Northwest of North America has become a global example of balancing conflicting land management goals (DellaSala and Williams 2006). Concern over degradation of the owl's dense, late-successional forest habitat led to the 1994 Northwest Forest Plan (NWFP). The NWFP shifted management on ~100,000 km^2 of federal USA forestlands from an emphasis on resource extraction to embrace ecosystem management and biodiversity conservation goals. Under the NWFP, ~30% of federal lands traditionally managed for timber production were placed in late-successional reserves that emphasized conservation goals and limited timber harvesting (USFS/USDI 1994).

Over the last decade, managers and policy makers have become increasingly concerned about high-severity fire and reduced timber harvesting in NWFP dry forests (e.g., Spies *et al.* 2006, Power 2006, Thomas *et al.* 2006, Ager *et al.* 2007, USFWS 2011). Forest thinning has been viewed as a solution for controlling fires in dry forests throughout western North America (Agee and Skinner 2005, Stephens and Ruth 2005) and commercial criteria have been included to pursue timber harvest goals (Johnson and Franklin 2009, Franklin and Johnson 2012). Commercial thinning prescriptions currently being implemented under these

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criteria may remove up to one-half of forest basal area, and may also include patch cutting or small clear cuts (USDI 2011). Commercial thinning is now proceeding rapidly without a full understanding of the long-term risks.

For spotted owls, thinning and associated activities often remove or reduce key habitat features in direct proportion to the intensity of the commercial prescription. Key spotted owl habitat features that may be reduced or removed directly or indirectly include high tree density and canopy cover (King 1993, Pidgeon 1995), recently killed pines (*Pinus* spp.) and abundant snags (Pidgeon 1995), multiple tree layers, with abundant medium and small white fir (*Abies concolor*) or Douglas-fir (*Pseudotsuga menziesii*) (King 1993, Pidgeon 1995, Everett *et al.* 1997, Irwin *et al.* 2012), large volume of mature-sized down logs (Pidgeon 1995), shrubs (King 1993, Pidgeon 1995, Irwin *et al.* 2012) and trees with heavy mistletoe infections (Hessburg *et al.* 2008), which are essential for spotted owl nesting (USFWS 2011). Thinning or contemporary harvest near the nest or activity center has been shown to displace Northern Spotted Owls (Forsman *et al.* 1984, King 1993, Hicks *et al.* 1999, Meiman *et al.* 2003). Telemetry studies on California Spotted Owls (*Strix occidentalis* ssp. *occidentalis*) in the Sierra Nevada found that owls avoided Defensible Fuel Profile Zones (an intensive thinning treatment) (USFS 2010). Unoccupied California Spotted Owl territories had a lower probability of re-occupancy after timber harvest, even when habitat alterations comprised <5% of a territory (Seamans and Gutiérrez 2007). In addition, Barred Owls (*S. varia*), which out-compete spotted owls (Dugger *et al.* 2011), use younger and more open forests compared to Northern Spotted Owls (Wiens 2012).

Studies also have found negative impacts of thinning to northern flying squirrels (*Glaucomys sabrinus*), the primary prey of Northern Spotted Owls in most of its range (Waters and Zabel 1995, Waters *et al.* 2000, Carey 2001, Ransome and Sullivan 2002, Gomez *et al.* 2005, Ransome *et al.* 2004, Bull *et al.* 2004, Meyer *et al.* 2007, Wilson 2008, Holloway and Smith 2011, Manning *et al.* 2012). Negative effects may persist for 15 years or longer (Wilson 2008). In addition, openings between trees from thinning may create barriers, due to predator avoidance, for flying squirrels to cross using its gliding locomotion (Manning *et al.* 2012). Thinning has also been found to have negative effects on the abundance of other main prey species for Northern Spotted Owls such as red-backed voles (*Myodes californicus*) (Suzuki and Hayes 2003) and woodrats (*Neotoma cinerea, N. fuscipes*) (Lehmkuhl *et al.* 2006).

Because of the many conflicts between thinning and spotted owl conservation, some authors have recommended that treatments aimed at controlling fire avoid spotted owl habitat and instead treat vegetation elsewhere that is the most flammable and strategic for accomplishing fuel treatment goals (Gaines *et al.* 2010). The 2011 Recovery Plan for the Northern Spotted Owl, the blueprint for management of this species on federal lands in the region (USFWS 2011), contains the proviso that long-term benefits to spotted owls of forest thinning treatments must clearly outweigh adverse impacts (USFWS 2011). The U.S. Fish and Wildlife agency that developed the plan suggested that benefits over time might accrue from a net increase in habitat because fire disturbances would be reduced (USFWS 2011). But whether the benefits would outweigh the impacts remains uncertain due to limitations of previous assessments.

Previous assessments of the efficacy of thinning treatments in reducing fire disturbances in spotted owl habitat (Wilson and Baker 1998, Lee and Irwin 2005, Roloff *et al.* 2005, 2012, Calkin *et al.* 2005, Hummel and Calkin 2005, Ager *et al.* 2007, Lehmkuhl *et al.* 2007) have not incorporated the probability of high-severity fires occurring during the treatment lifespan. The effect of this is to overestimate treatment efficacy in potentially controlling fire or fire behavior (Rhodes and Baker 2008). Nor have the effects of recruitment of dense, late-successional forest that act to offset loss from fire been included in prior assessments. In addition, impacts of the kind of commercial thinning treatments being implemented to address dry forest concerns have not been fully considered for the owl or its prey (e.g., Ager *et al.* 2007, Lehmkuhl *et al.* 2007, Roloff *et al.* 2012). Current commercial thinning prescriptions being implemented in dry forests specifically identify desired future conditions to be maintained (e.g. Johnson and Franklin 2009) that have basal area and other structural targets mostly well below the minimum levels that have been found in spotted owl nesting, roosting and foraging habitat (NRF) in dry forests. For example, basal area targets in a project in southwest Oregon designed to demonstrate the thinning prescriptions in dry forest spotted owl habitat were 13.75-27.5 m^2/ha (USDI 2011), while stands < 23 m^2/ha very rarely support spotted owl nesting territories (Buchanan and Irwin 1995). In addition, the Recovery Plan (USFWS 2011) permits thinning in core areas, but emphasizes treating areas outside of core areas, so there is a need for assessment of impacts outside core areas as well. Areas outside cores may be essential for foraging and be part of the breeding season home range. Furthermore, owls often move outside core areas (USFWS 2011). Lastly, available habitat outside existing cores may become important to owl recovery, particularly if spotted owls are displaced from higher quality habitat by Barred Owls (Dugger *et al.* 2011).

To assess whether benefits of commercial thinning outweigh adverse impacts to spotted owls in dry forests (USFWS 2011), quantitative assessments are needed that allow for direct assessment of the amounts of any dense, mature or late-successional habitat that would be reduced by both commercial prescriptions and severe fire. Accordingly, we calculated these amounts by projecting them over 40 years and incorporated into our calculations the effects of forest regrowth. For our calculations, we used empirical data on fire and forest regrowth from the potential habitat within the two dry forest regions where spotted owls occur, the Klamath and dry Cascades of California, Oregon, and Washington, that are subject to thinning. We analyzed each region separately using region-wide data. Conservation planning for spotted owls commonly occurs at the scale of these regions. For our thinning treatment, we chose a "best" scenario for minimizing the amount of dense, late-successional forest to be treated (Lehmkuhl *et al.* 2007); while we used an optimistic scenario for treatment efficacy, assuming that a 50% reduction in high-severity fire would occur (Ager *et al.* 2007). We also illustrate the effects of varying treatment amount and efficacy. To calculate rotations of severe fire in the forests of the study area, we used available fire data from a time period, 1996-2011, which includes exceptionally large, rare fire events. Our approach may be useful to managers interested in maintaining habitat for other species that rely on dense forests in fire-prone regions (Odion and Hanson 2013).

METHODS

Study Area

We analyzed fire and forest recruitment trends in 19,000 km^2 of dry forests in the Klamath and 18,400 km^2 in the Cascades provinces. As in Hanson *et al.* (2009), we analyzed only late-successional, or "older" forests present in 1995, as mapped by Moeur *et al.* (2005). This is a small fraction of the dry forest regions. Our analysis was further restricted to federal lands. Mapping by Moeur *et al.* (2005) corresponds to mid-montane forest zones where Northern Spotted Owls occur. These montane forest zones include forests dominated mainly by true firs (*A. grandis, A. concolor*), Douglas-fir (*Pseudotsuga menziesii*), and Ponderosa pine (*P. ponderosa*): Other conifers found in the central and northern Cascades in dry forests frequented by spotted owls are western hemlock (*Tsuga heterophylla*), western larch (*Larix occidentalis*), and limited amounts of western red cedar (*Thuja plicata*) and Engelmann spruce (*Picea engelmannii*). Forests in the Klamath are noted for high conifer diversity, with species such as incense cedar (*Calocedrus decurrens*) commonly found in the range of spotted owls. A variety of broad-leaved evergreen trees, such as madrone (*Arbutus menziesii*) and tanoak (*Lithocarpus densiflorus*) are also characteristic of these forests (Whittaker 1960).

Quantifying Future Habitat

We determined existing rates of dry-forest redevelopment following stand initiation in the forests of the study regions as delineated by Mouer *et al.* (2005) using the extensive U.S. Forest Service Forest Inventory and Analysis (FIA) forest monitoring data (http://www.fia.fs.fed.us/tools-data/). FIA is a monitoring system based on one permanent, random plot per ~2400 ha across forested lands. We excluded plots from forests not used by spotted owls (e.g. lodgepole pine, oak forest) and from non-conifer vegetation and non-federal lands. Most of these plots were already excluded by the mapping by Mouer *et al.* (2005) that delineated the study area.

An FIA plot consists of a 1-ha area. For tree measurements, this area is sub-sampled with four circular subplots that are 0.1 ha for large-tree sampling and 0.017 ha for smaller-tree sampling (defined by region). The diameter-at breast-height (dbh) and crown position of each tree and the ring count from two cores from dominant/codominant trees are measured in each subplot (USFS 2010). Stand age for an FIA plot is determined from the average of all ring counts from sub-plot samples, weighted by cover of sampled trees, and 8 years are added for estimated time to grow to breast height (1.4 m). We used live-tree dbh data to prepare regressions with stand age.

FIA data were available from 2001-2009, comprising 90% of the plots available within our study area. A total of 581 plots from the Klamath and 441 from the dry Cascades were considered, representing 13,944 and 10,680 km^2 in each region, respectively. The number would be higher, but we eliminated 139 plots in the Klamath and 141 in the Cascades that had different stand-initiation dates from different subplots of the main FIA plot. This situation occurs throughout the study area due to the patchy nature of mixed-severity fire. Including all the subplots as individual plots creates a larger sample size, but we chose not to do this because some individual locations would be overrepresented. Most importantly, both approaches lead to the same results.

We analyzed fire severity from 1996-2011 in late-successional, or "older" forests mapped by Moeur *et al.* (2005). For 1996-2008, we used the Monitoring Trends in Burn Severity (MTBS) (http://www.mtbs.gov/) data. We used the ordinal classification from MTBS, as MTBS analysts determine for each fire where significant thresholds exist in digital prefire and postfire images, supplemented with plot data and analyst experience with fire effects. In plot data, a composite burn index that sums mortality by vegetation stratum is used to identify high fire severity (see http://www.mtbs.gov/). For 2009-2011, we obtained U.S. Forest Service digital data (http://www.fs.fed.us/postfire-vegcondition) and classified these data following Miller and Thode (2007). We could not use pre-1996 MTBS fire severity data because the pre-burn map of spotted owl forest habitat is from 1995 (Moeur *et al.* 2005). From severity data we calculated high-severity fire rotation (FR^{hs}), the expected time to severely burn an area equivalent to the area of interest once, or the landscape mean interval for severe fire (Baker 2009).

We calculated annual high-severity fire and forest regrowth rates to future proportions for early-, mid- and mature or late-successional forests, denoted herein by "E," "M," and "L," respectively, using annual time steps. We defined late-successional forests by selecting a value, 27.5 m^2/ha. This amount corresponds with the maximum basal area that would be left according to currently implemented thinning prescriptions (USDI 2011). This is somewhat higher than the minimum basal area where spotted owls have been found to nest in dry forests. For example, the mean value minus one standard deviation in all the dry forest stands studied by Buchanan *et al.* (1995) was 23 m^2/ha. However, we did not want to identify the rate of regrowth to the very minimum basal area that constitutes habitat, but regrowth to a basal area more likely to function as habitat. Mid- and early-successional forests were defined as 13.5-27.5 and <13.5 m^2/ha tree basal area, respectively. We separated mid-successional from early-successional forest because, mid-successional forests may be included in thinning treatments, but early-successional forests may not. Thinned forest ("T") was our fourth vegetation state. The forest states are diagramed in Fig. (**1**). The proportion of each state in the landscape at time *t,* defined a vector (P_t^E, P_t^M, P_t^T, P_t^L). Transition probabilities ϕ_t^{rs} equaled the probability that any portion of state *r* at time *t* transitions to

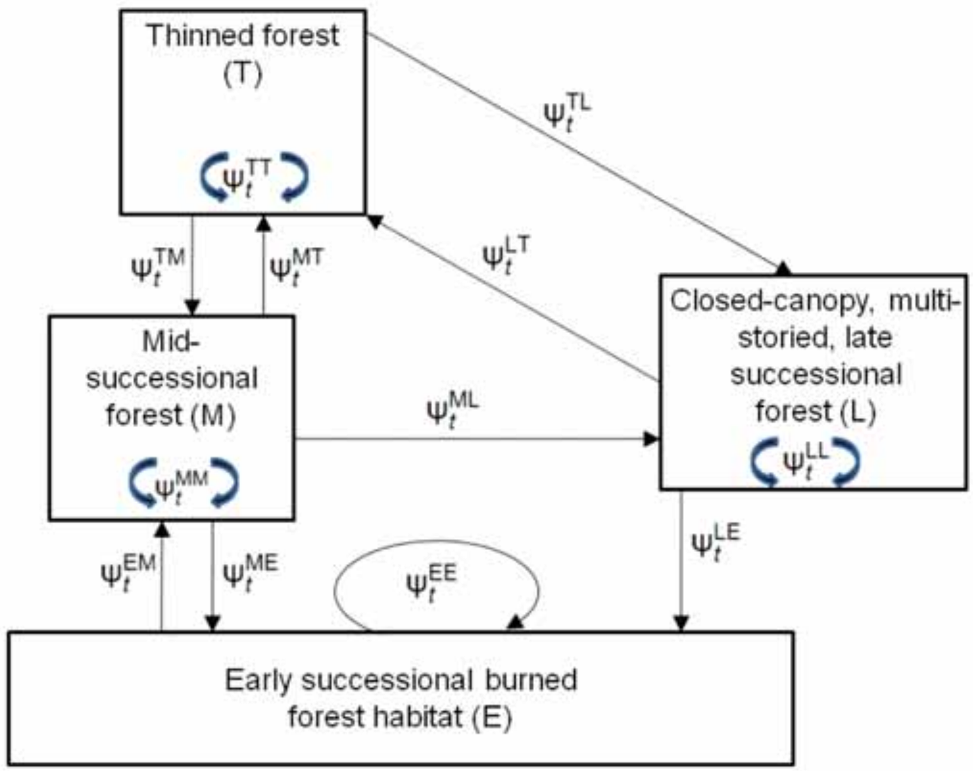

Fig. (1). State (boxes) and transition (arrows) model for dry Pacific Northwest Forest vegetation with fire disturbances and thinning. Variables are the transition rates between states indicated by the associated arrow.

state *s* at time *t* + 1, allowing calculation of future amounts of each forest type using the following equation:

$$\begin{bmatrix} \phi_t^{EE} & \phi_t^{ME} & \phi_t^{TE} & \phi_t^{LE} \\ \phi_t^{EM} & \phi_t^{MM} & \phi_t^{TM} & \phi_t^{LM} \\ \phi_t^{ET} & \phi_t^{MT} & \phi_t^{TT} & \phi_t^{LT} \\ \phi_t^{EL} & \phi_t^{ML} & \phi_t^{TL} & \phi_t^{LL} \end{bmatrix} \begin{bmatrix} P_t^{E} \\ P_t^{M} \\ P_t^{T} \\ P_t^{L} \end{bmatrix} = \begin{bmatrix} P_{t+1}^{E} \\ P_{t+1}^{M} \\ P_{t+1}^{T} \\ P_{t+1}^{L} \end{bmatrix} \quad (1)$$

The initial proportions, $P_{t=0}^{E\text{-}L}$ of the three natural-forest states were from the FIA basal-area analyses, with thinned forests considered zero for simplicity and because of lack of data. The annual transition from mid- and late- to early-successional forest from high-severity fire (ϕ_t^{LE}, ϕ_t^{ME}) was $1/FR^{hs}$. Early-successional forests also burned at this rate (ϕ_t^{EE}). Annual rates of forest redevelopment were from the inverse of the growth period ($1/G^{EM}$) to reach 13.5 m^2/ha live-tree basal area, or to grow from 13.5 to 27.5 m^2/ha live-tree basal area ($1/G^{ML}$), calculated from the regression of live basal area on age (see results). Lower-severity fire can reduce basal area from >27.5 m^2/ha basal area to <27.5 m^2/ha. However, this transition is already considered in the regrowth rate, which also incorporates the effects of lower-severity fires that have occurred on rates of forest redevelopment. Because natural disturbances that may temporarily lower basal area are captured in the transitions from early- to late-successional forest, the transition from late to mid-successional forest was set to zero. Transition rates to thinned forest were based on treatment within 20 years, beginning in year *t* + 1, of the mid- and late-successional forests present at *t* = 0 (see Table **1** for annual rate). Based upon the empirical FIA and MTBS data described above, we used these transitions (Table **1**) and Eq. 1 to project forward 40 years (see sample calculation in the Supplementary Materials). We chose this time interval because it represents one cycle of thinning and forest recovery.

Next, we calculated the effects of varying levels of thinning, and treatment efficacy (in terms of the effect on high-severity fire rotation intervals), over the study period. According to an analysis of a spotted owl landscape by Lehmkuhl *et al.* (2007), a "best" scenario for minimizing the short-term adverse impacts of thinning while reducing fire frequency and severity was one that treated only 22% of the landscape, and limited thinning in nesting, roosting, and foraging habitat to 21% of the area of this habitat. We used this prescription in our calculations to illustrate the effects under a best-case scenario. In our calculations, the amount of mid-successional forest thinning differed between the two regions because amounts of both mid- and late-successional forests were not the same. We also considered the effects of treating from 0 to 45% of forests, holding constant the proportions of treatments that were in late-successional vs. mid-successional forests.

We assumed that there would be no high-severity fire in treated forests over the treatment lifespan. We additionally assumed that thinning 22% of the landscape would lower the amount of high-severity fire in the unthinned landscape by half. This is based on the findings of Ager *et al.* (2007) who simulated the effects of wildfire ignitions following strategic

Table 1. Annual transition probabilities used in transition matrices for each scenario analyzed for dry provinces within the range of the Northern Spotted Owl. FR^{hs} is the high-severity fire rotation. G is the time required for stands to grow from early to mid- (EM) or mid- to late-successional (ML) forest (see Table 2). K = Klamath, C = Cascades. R is the amount that high severity fire is reduced by thinning (50% reduction at 22 percent of late-successional forest thinned).

Transition Probabilities	No Treat	Treat 22% Maintain	Treat 22% Recover
ϕ_t^{LE}	$1/FR^{hs}$	$(1/FR^{hs}\text{-}R)$	$(1/FR^{hs}\text{-}R)$
ϕ_t^{EM}	$1/G^{EM}$	$1/G^{EM}$	$1/G^{EM}$
ϕ_t^{ET}	0	0	0
ϕ_t^{EL}	0	0	0
ϕ_t^{ME}	$2/FR^{hs}$	$2/FR^{hs}$	$2/FR^{hs}$
ϕ_t^{ML}	$1/G^{ML}$	$1/G^{ML}$	$1/G^{ML}$
ϕ_t^{EE}	$1\text{-}1/G^{EM}$	$1\text{-}1/G^{EM}$	$1\text{-}1/G^{EM}$
ϕ_t^{MM}	$1\text{-}1/G^{ML}\text{-}(1/FR^{hs})$	$1\text{-}1/G^{ML}\text{-}(1/FR^{hs}\text{-}R) - \phi_t^{MT*}$	$1\text{-}1/G^{ML}\text{-}(1/FR^{hs}\text{-}R) - \phi_t^{MT*}$
ϕ_t^{MT*}	0	K = 0.033 C = 0.018	K = 0.033 C = 0.018
$\phi_t^{TM\dagger}$	0	0	K = 0.033 C = 0.018
ϕ_t^{TE}	0	0	0
$\phi_t^{TT\dagger}$	0	0	$1\text{-}\phi_t^{TL} - \phi_t^{TM\dagger}$
$\phi_t^{TL\dagger}$	0	0	K = 0.0114 C = 0.0105
ϕ_t^{LM}	0	0	0
ϕ_t^{LT*}	0	K = 0.0114 C = 0.0105	K = 0.0114 C = 0.0105
ϕ_t^{LL}	$1 - 1/FR^{hs}$	$1 - 1/FR^{hs}\text{-}R\text{-}\ \phi_t^{LT}$	$1 - 1/FR^{hs}\text{-}R\text{-}\ \phi_t^{LT}$

*Only in effect for the first 20 years.
†Does not take effect until after 20 years.

thinning treatments in a spotted owl landscape. When <22% of the landscape was affected at any given time (such as any time prior to year 20 when the full treatment would be incomplete, or after one-time treatments began to recover, or for scenarios with <22% of the landscape treated) the same ratio of area treated to reduction in high-severity fire (22% treat: 50% reduction in fire) was used to reduce the area burned at high severity (see Supplementary Material for an illustration). Thus, the amount that fire was reduced by thinning increased with each year as a function of the total area thinned (all other variables were constant). Ager *et al.* (2007) found little additional effect of treatments in reducing wildfires as treatment level increased beyond 20%, so we did not calculate greater reductions in fire as treatment levels went from 22-45%. However, we additionally calculated future habitat amounts as a function of fire rotation to evaluate the effects of varying treatment efficacy, in which case we did calculate the reduced amount of habitat burned severely. This amount is the dependent variable in our summary figures. Treatment lifespan was assumed to be 20 years (Rhodes and Baker 2008) for "one-time thinning," or maintained in perpetuity over the 40 years for "maintained." A sample calculation using the model (equation 1) is presented in the Supplementary Material.

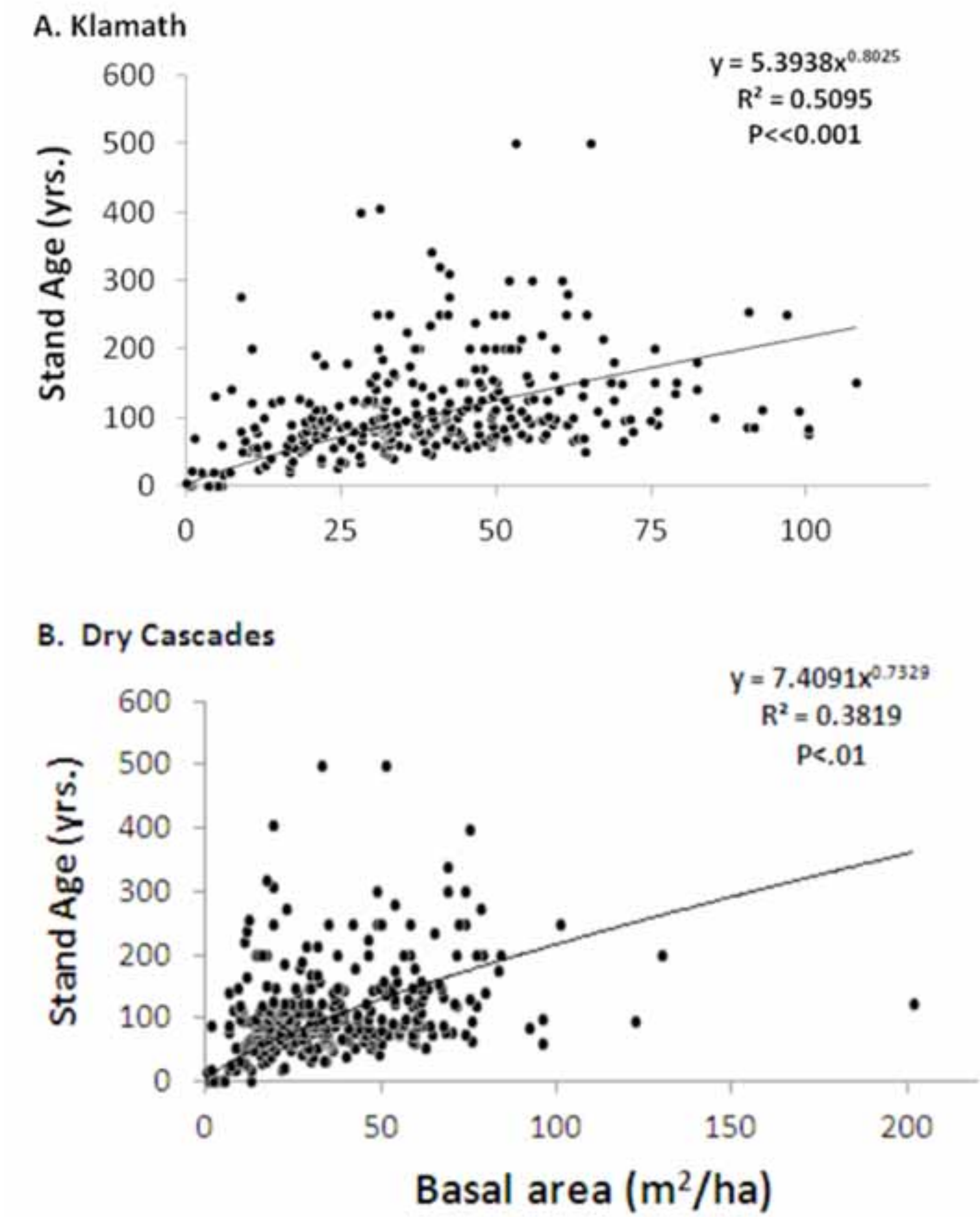

Fig. (2a-b). Scatterplots of live-tree basal area per hectare and stand age from US Forest Service FIA data for the A. Klamath region and B. dry Cascades region.

The only owl habitat we considered for impacts from thinning was suitable nesting, roosting, and foraging (so called NRF habitat). Because treatments aimed at demonstrating the type of thinning to be implemented in spotted owl habitat reduce basal area down to 13.75-27.5 m^2/ha, mostly well-below the minimum amounts for NRF habitat (Pidgeon 1995, Buchanan and Irwin 1998, LeHaye and Gutiérrez 1999), and because treated forests also have reduced amounts of key habitat features like multi-canopy structure, down wood, small firs and mistletoe infections, the area affected by these treatments will largely correspond to the amount of habitat lost. Thinning may also render adjacent, unthinned forest unsuitable or less suitable (Seamans and Gutiérrez 2007), but we did not account for this effect. The lifespan for thinning treatments that we used was 20 years for one-time thinning (Rhodes and Baker 2008), and 40 years for maintained treatments. Transition from late- to early-successional vegetation due to high-severity fire also was considered habitat loss. This may overestimate the impacts of fire on Northern Spotted Owl foraging habitat (Bond *et al.* 2009, USFWS 2011), but the assumption is largely irrelevant due to the low rates of high-severity fire in both study regions in relation to forest regrowth, as described next.

RESULTS

We found a highly significant relationship between live-tree basal area and stand age in both regions (Figs. **2a-b**, Klamath n = 442, dry Cascades n = 304). Much of the variance in the plot data was caused by a modest number of relatively old stands that had much lower basal area for their

Table 2. Forest Inventory and Analysis (FIA) plot parameters for the Klamath and dry Cascades provinces, California, Oregon, and Washington, based on most recent survey data from 2001-2009. Also shown are the amounts of time after fire that is takes forest to regrow to the specified live basal area (BA) thresholds using the regression equations shown in Figs. (2a-b).
[a]These plots have 2 or more stand ages associated with them due to different disturbance histories within the main FIA plot.

Entity	Klamath	Dry Cascades
Number of plots (total)	581	445
Number of plots excluded from analysis†	139	141
Initial (P^{E}_{t+0}) early-successional forest (%)	9	14.5
Initial (P^{M}_{t+0}) mid-successional forest (%)	14.4	26.9
Initial (P^{L}_{t+0}) late-successional forest (%)	76.6	55.6
Regrowth period, 0-13.5 m^2/ha live BA (yrs)	44	53
Regrowth period, 13.5-27.5 m^2/ha live BA (yrs)	32	36
Regrowth period, 0-27.5 m^2/ha live BA (yrs)	76	89
High-severity fire rotation	362	913

†These plots have 2 or more stand ages associated with them due to different-aged sub-plots within the main FIA plot.

age than did other plots. The amount of time following disturbance needed for regenerating forests to reach live-tree basal area >27.5 m^2/ha was 77 and 90 years, respectively, for the Klamath and dry Cascades (Table **2**).

Using the MTBS data, the rotation for high-severity fire from 1996-2011 was 362 to 913 years in the Klamath and dry Cascades, respectively (Table **2**). At these rates, a total of 1,221 and 325 km^2 of high-severity fire would occur in Klamath and dry Cascades late-successional forests, respectively, in 40 years. With annual regrowth rates of late-successional forests that were 4.5 to >10 times greater than the rates of fire disturbances (i.e. (1/77)/(1/362) for the Klamath and (1/89)/(1/913) for the dry Cascades, and no disturbances other than fire, late-successional forests would eventually come to occupy 83% of the potential forested area in the Klamath and 91% in the Cascades. Thus, over 40 years, late-successional forests in the Klamath increased slightly over their current amount of 77% of the forested landscape FIA plots to 81% or from about 10,668 km^2 to 11,335 km^2 (Fig. **3a**). In the dry Cascades, where late-successional forests were 59% of the forested landscape FIA plots, they increased relatively rapidly to 77% of the forested landscape, or from 6,253 km^2 to 8,234 km^2 in 40 years (Fig. **4a**).

Simulated thinning of 21% of dense, late-successional forest of the Klamath landscape meant that a total of 2,225 km^2 would be reduced, while treatments in mid-successional forests would cover 840 km^2 to reach a treatment level of 22% of the whole landscape. After the one-time thinning, late-successional forests returned to slightly lower amounts than occurred without thinning after 40 years (Fig. **3a**). The net effect of the one-time thinning was to reduce late-successional habitat by 10.7% over the 40-year period, or from an average of 11,086 km^2 to 9,996 km^2 over 40 years (i.e., 1,090 km^2 less each year on average, Fig **3b**). The amount of dense, late-successional forest that was prevented from burning at high severity was 16 km^2/year, resulting in 320 km^2 of dense, late-successional forest, which would otherwise have been transformed into early-successional forest, in each year on average over the 40-year period. Therefore, in this scenario, thinning reduced 3.4 times more late-successional forest than it increased. The maintained treatment reduced habitat by 15.3%, from 11,086 km^2 on average over 40 years to 9,396 km^2 (i.e., 1,690 km^2 less each year on average, Fig. **3c**). In both cases, 13% of the habitat loss was from thinning in mid-successional forest that prevented or slowed these forests from developing into dense, late-successional forest. The amount of dense, late-successional forest that was prevented from burning at high severity was 20 km^2/year, resulting in 400 km^2 of dense, late-successional forest, which would otherwise have been transformed into early-successional forest, in each year on average over the 40-year period. Therefore, the combination of thinning and maintenance reduced 4.2 times more late-successional forest than it increased.

In the Cascades, to treat 22% of the landscape, the thinning scenario targeted 1,313 km^2 of dense, late-successional forest, and 1,036 km^2 of mid-successional forest. After the one-time thinning, late-successional forests again returned to slightly lower amounts than occurred without thinning after 40 years (Fig. **4a**). The net effect of the one-time thinning treatment over 40 years was to reduce dense, late-successional forest by an average level of 11.1% (836 km^2 less each year on average, Fig. **4b**). The amount of dense, late-successional forest that was prevented from burning at high severity from the one time treatment was 3.5 km^2/year, resulting in 140 km^2 of dense, late-successional forest, which would otherwise have been transformed into

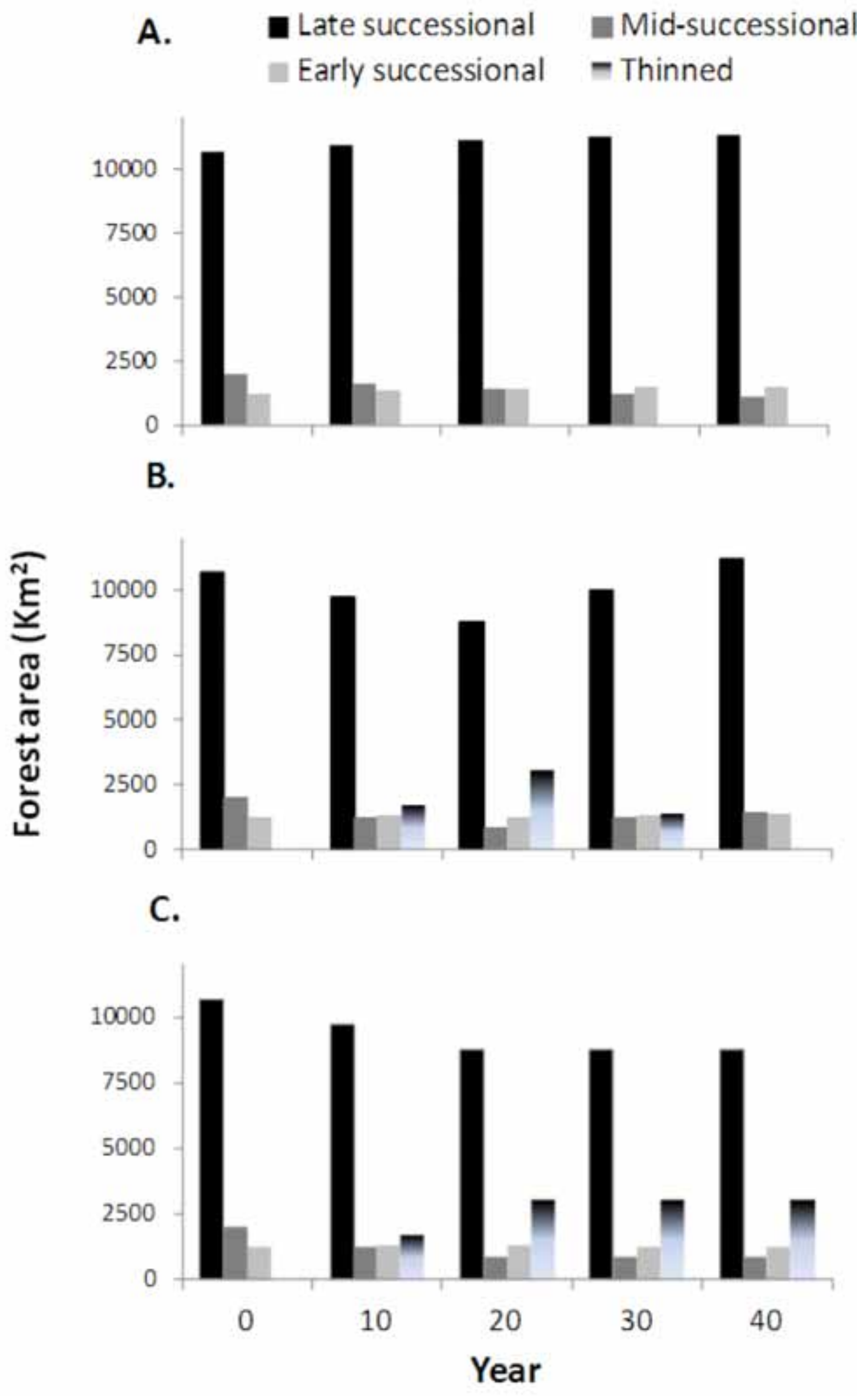

Fig. (3a-c). Amounts of the four forest types (early-, mid-, late-successional, and thinned) in the landscape over a 40-year period based on the states shown in (Fig. **1**) and transition rates (Table **2**) for the Klamath province, California, and Oregon, and the following scenarios: A) no treatment; B) one-time treatment of 21% of late-successional forests (>27.5 m^2/ha live-tree basal area) and 42% of mid-successional forests (= total of 22% of landscape treated) followed by recovery in 20 years to late-successional forest; C) treatment of 21% of late-successional forests (>27.5 m^2/ha live-tree basal area) and 42% of mid-successional (= total of 22% of landscape treated) forests with future maintenance. We converted proportions of forest types from modeling output to km^2 using the area estimate from FIA for the Klamath study region.

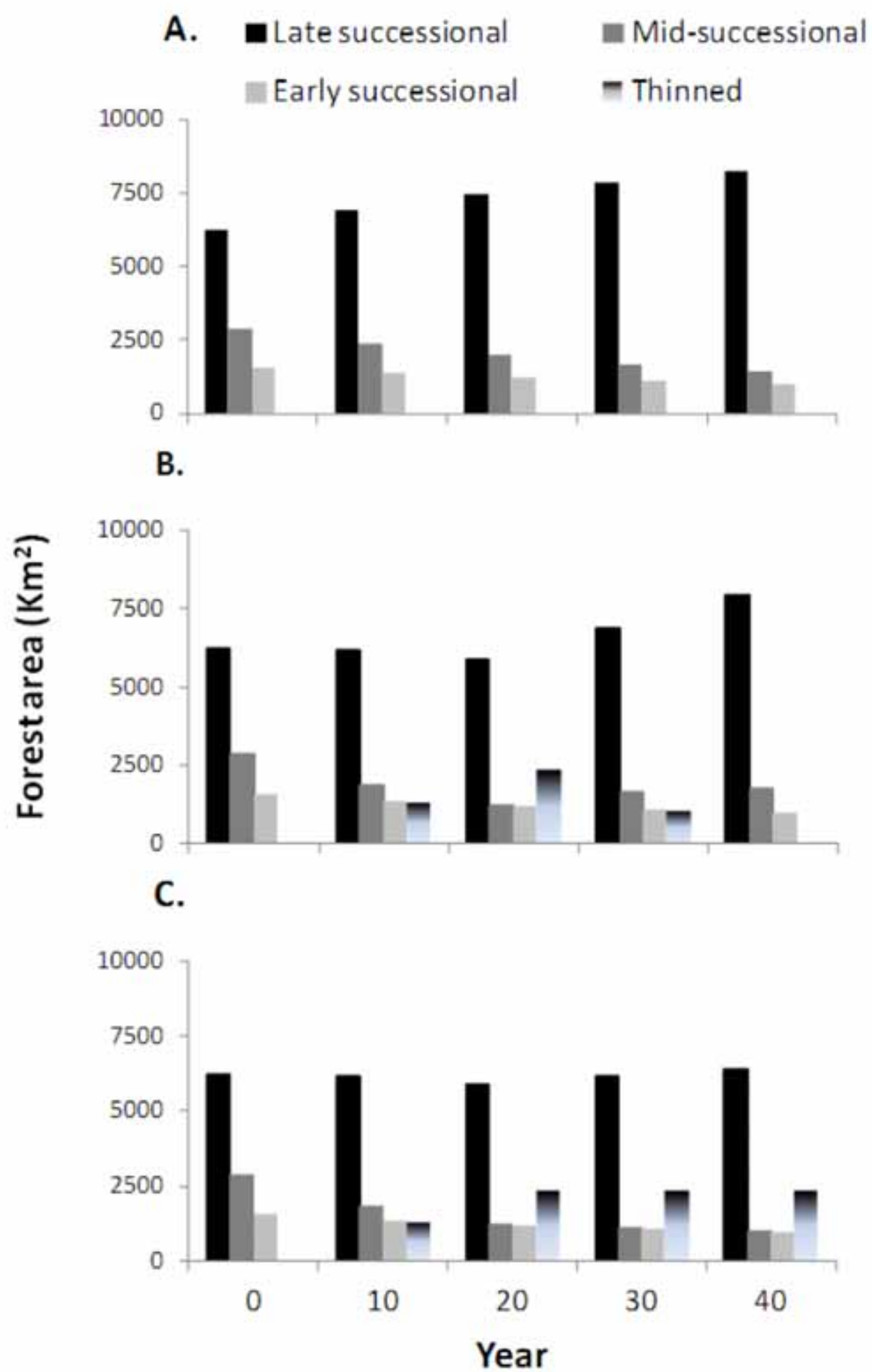

Fig. (4a-c). Amounts of the four forest types (early-, mid-, late-successional, and thinned) in the landscape over a 40-year period based on the states in (Fig. **1**) and transition rates (Table **2**) for the dry Cascades province, California, Oregon, and Washington and the following scenarios: **A**) no treatment; **B**) one time treatment of 21% of late-successional forests (>27.5 m^2/ha live tree basal area) and 36% of mid-successional forests (=22% of landscape treated) followed by recovery in 20 years to late-successional forest; **C**) treatment of 21% of late-successional forests (>27.5 m^2/ha live tree basal area) and 36% of mid-successional forests (=22% of landscape treated) in perpetuity. We converted proportions of forest types from modeling output to km^2 using the area estimate from FIA for the dry Cascades study region.

early-successional forest, in each year on average over the 40-year period. Therefore, thinning reduced 6.0 times more late-successional forest than it increased. The maintained treatment reduced dense, late-successional forest by an average of 16.4% (1,212 km^2less each year on average, Fig. **4c**). Of this reduction, 30% was from the indirect effect of thinning in mid-successional forests, more of which were treated in the Cascades scenario. The amount of dense, late-

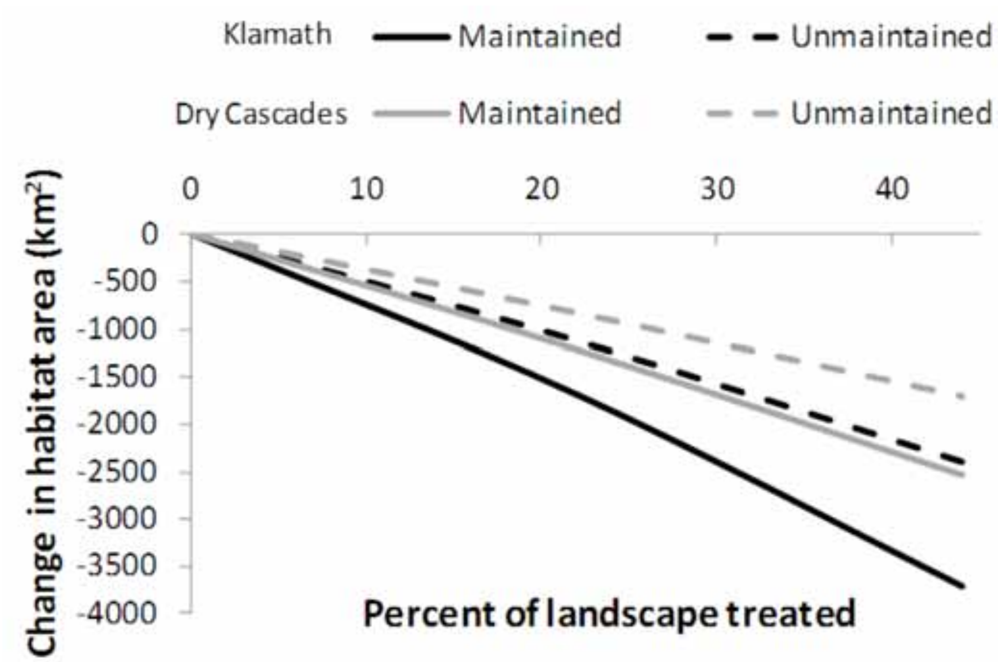

Fig. (5). Net amount of habitat lost over 40 years compared to the no-treatment scenario as a function of treatment of 0-45% of the landscape. The amount of late-successional forest treated was held constant at 21% of the area of this forest, except at very low levels of treatment. The amount of mid-successional forest treated varied from zero at very low treatment levels, to a large proportion of the mid-successional forests when 45% of the landscape was treated, particularly in the Klamath region.

successional forest that was prevented from burning at high severity from the maintained treatment scenario was 4.5 km^2/year, resulting in 180 km^2 of dense, late-successional forest, which would otherwise have been transformed into early-successional forest, in each year on average over the 40-year period. Therefore, the combination of thinning and maintenance reduced 6.7 times more late-successional forest than it increased.

As treatment level increased from 11 to 22%, habitat loss doubled (Fig. **5**). With 22% of the landscape treated, the effect of reducing fire by 50% in the rest of the landscape was reached, and there was no further reduction in fire with increasing treatment amount. With less fire prevented per km^2 treated, the rate of habitat loss increased as treatment went from 22 to 45% of the landscape.

We also assessed the effect of holding treatment level constant and varying the efficacy of treatments. Even if treatment efficacy was considerably greater than we assumed and rotations of high-severity fire substantially longer than twice their current length, the amount of dense, late-successional forest habitat that would be reduced due to thinning would only be slightly lower (Figs. **6a-b**). With complete elimination of fire over 40 years as a result of treatments, the amount of dense, late-successional forest would be 9-10% less than with no treatment. This becomes a large amount of habitat loss over time.

DISCUSSION

We found that the habitat recruitment rate exceeded the rate of severe fire by a factor of 4.5 in the Klamath and 10 in the dry Cascades, leading to a deterministic increase in dense forest habitat over time, assuming no other disturbance events. In contrast, previous published assessments of fire on spotted owls have not explicitly considered fire and forest regrowth rates (Wilson and Baker 1998, Lee and Irwin 2005, Roloff *et al.* 2005, 2012, Calkin *et al.* 2005, Hummel and Calkin 2005, Ager *et al.* 2007, Lehmkuhl *et al.* 2007). Not including the probability of high-severity fire, which is low, leads to highly inflated projections of the effects of thinning versus not thinning on high-severity fire (Rhodes and Baker 2008, Campbell *et al.* 2012).

Our calculations of thinning effects included rates of forest regrowth along with high-severity fire. The calculations illustrate how the requirement that the long-term benefits of thinning clearly outweigh adverse impacts (USFWS 2011) is not attainable as long as treatments have adverse impacts on spotted owl habitat. This is because the amount of dense, late-successional forest that might be prevented from burning severely would be a fraction of the area that would be thinned. Under our "best case" scenario, thinning reduced dense, late-successional forest by 3.4 and 6.0 times more than it prevented such forest from experiencing high-severity fire in the Klamath and dry Cascades, respectively, similar to findings in a recent unpublished report by U.S. Forest Service scientists from the Pacific Northwest Research Station (Raphael *et al.* 2013). This would not be a concern if thinning effects were neutral, but the commercial thinning prescriptions being implemented call for forests with basal area reduced by nearly half to 13.5-27.5 m^2/ha, which is mostly well below the minimum level known to function as nesting and roosting habitat (ca. 23 m^2/ha) (Buchanan *et al.* 1995, 1998). Thus, if dense forests are subjected to these treatments, much of the impacted area would no longer have minimum basal area needed to function as nesting and roosting habitat. Even an immediate doubling of fire rates due to climate change or

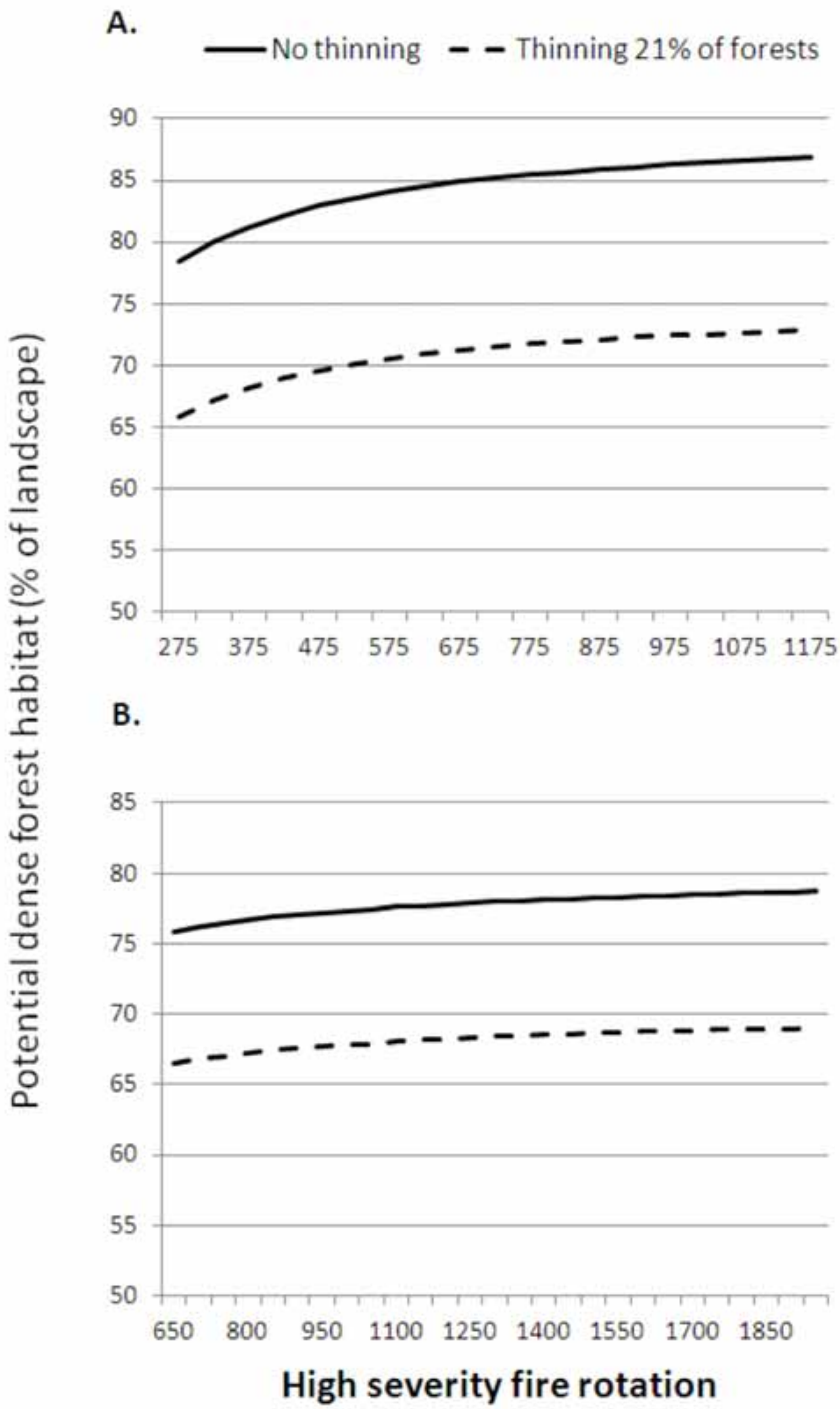

Fig. (6a-b). Amount of forest habitat in the range of the Northern Spotted Owl in the A. Klamath, and B. dry Cascades 40 years in the future as a function of the average high severity rotation over that time period, and longer rotations.

other factors would result in far less habitat affected by high-severity fire than thinning. In addition, much of the high-severity fire might occur regardless of thinning, especially if the efficacy of thinning in reducing high-severity fire is reduced as fire becomes more controlled by climate and weather (Cruz and Alexander 2010). Clearly, the strategy of trying to maintain more dense, late-successional forest habitat by reducing fire does not work if the method for reducing fire adversely affects far more of this forest habitat than would high-severity fire, and the high-severity fire might occur anyway because it is largely controlled by climate and weather.

There may be silvicultural treatments that can be done in spotted owl habitat that may reduce adverse impacts. For example, thinning that maintains at least 23-27.5 m^2 ha basal area. However, given that key habitat elements such as small trees, down wood, and likely some intermediate-sized trees are going to be targeted in any forest fuel reduction treatment, it appears unlikely that any conventional fuels reduction treatment in spotted owl habitat would not have at least some adverse impacts. This is supported by research on thinning that was often less intensive than commercial thinning prescriptions. This research showed negative impacts on spotted owls or their prey, as summarized in our introduction (Waters and Zabel 1995, Waters *et al.* 2000, Carey 2001, Ransome and Sullivan 2002, Gomez *et al.* 2003, Suzuki and Hayes 2003, Ransome *et al.* 2004, Bull *et al.* 2004, Lehmkuhl *et al.* 2006, Meyer *et al.* 2007, Wilson 2010, Holloway and Smith 2011, Manning *et al.* 2012), and how spotted owls have been displaced by even very limited amounts of thinning or contemporary harvest near the nest or activity center (Forsman *et al.* 1984, King 1993, Hicks *et al.* 1999, Meiman *et al.* 2003, Seamans and Gutiérrez 2007). Even if adverse impacts were quite modest, the amount of dense, late-successional forest that might be prevented from experiencing high-severity fire is so much smaller than the area that would be treated in an effort to accomplish this reduction in fire, that the net impact of the thinning would still be much greater. In addition, it is becoming increasingly less clear whether a reduction in high-severity fire below current rates would necessarily be beneficial to spotted owls. The dry forests in which spotted owls are found were historically characterized by mixed-severity fires (see Hessburg *et al.* (2007), Baker (2012), and Odion *et al.* (2014) for historic fire in the dry Cascades of Washington and Oregon, Beaty and Taylor (2001) and Bekker and Taylor (2001, 2010) for the California Cascades, and Wills and Stuart (1994), Taylor and Skinner (1998, 2003), and Odion *et al.* (2014) for the Klamath). Recent research suggests that this historic fire may have neutral and beneficial effects to spotted owls.

Studies on the effects of fire on spotted owls are few and often focused on other owl subspecies and some studies are confounded by post-fire logging effects (Clark *et al.* 2013). Nonetheless, it has long been known that fire in woody vegetation causes an increase in small rodent populations and consequently raptor populations (Lawrence 1966), and studies on spotted owls and fire where no logging occurred suggest that high-severity fire at current rates may confer benefits or be neutral. Bond *et al.* (2009) found that California Spotted Owls in the Sierra Nevada preferentially foraged in severely burned forests more than unburned forests within about 1.5 km of a core-use area. The percentage of high-severity fire in burned Mexican Spotted Owl (*Strix occidentalis* ssp. *lucida*) sites had no significant influence (Jenness *et al.* 2004). Roberts *et al.* (2011) found no support for an occupancy model for California Spotted Owls that distinguished between burned and unburned sites in unmanaged forests; the mean “owl survey area” that burned at high-severity was 12%, with one survey area experiencing up to 52% high-severity fire, which is almost three times the current amount of severe fire in owl habitat, according to the MTBS data. In a longer-term (1997-2007) study of California Spotted Owl site-occupancy dynamics throughout the Sierra Nevada, high-severity fire that burned on average 32% of forested vegetation around nests and core roosts had no significant effect on extinction or colonization probabilities, and overall occupancy probabilities were slightly higher in mixed-severity burned areas than in unburned forest (Lee *et al.* 2012), while other research found no significant difference in home range size between mixed-severity fire areas and unburned forest (Bond *et al.* 2013). Studies on reproduction in occupied sites of all three spotted owl subspecies indicated no difference between unburned sites and mixed-severity burned sites (excluding burn out areas created by fire suppression operations) (Jenness *et al.* 2004), or in some cases reproduction may have been greater in burned sites (Bond *et al.* 2002, Roberts 2008). The longer-term value of fire disturbances is in the creation of landscape heterogeneity with inclusions of young stands, improving habitat at the landscape scale. Fire also plays a vital role in creating snags, large down logs, and other key elements of the highest quality spotted owl habitat at the territory scale (Franklin *et al.* 2000). No assessments of fire and thinning effects on spotted owls, including this one, have accounted for any potential beneficial effects of mixed-severity fire, nor the potential negative effects of lack of mixed-severity fire in treated areas.

While much of the concern about fire and thinning in dry forests of the Pacific Northwest has focused on spotted owls, it may also apply to other biota associated with dense, old forests, including species of conservation concern, such as Pacific fisher (*Martes pennanti pacifica*), which research indicates may benefit from mixed-severity fire (Hanson 2013), the Northern Goshawk (*Accipiter gentilis*), and, following fire, the Black-backed Woodpecker (*Picoides arcticus*), which depends upon higher-severity fire in dense, older forest (Odion and Hanson 2013). Like the spotted owl, studies have documented that this woodpecker is also negatively affected by thinning (Hutto 2008). Also, like the spotted owl, the Back-backed Woodpecker, Pacific Fisher and Northern Goshawk occur in forests where the historic fire regime was not low-severity. Modeling for the fisher, similar to modeling for the spotted owl, has not used the actual rates of high-severity fire and forest regrowth to assess possible impacts of fire, and has assumed that fire represents a loss of fisher habitat (Scheller *et al.* 2011), contrary to more recent empirical findings (Hanson 2013). Not including the actual probability of fire leads to considerably inflated projections of the effects of thinning vs. not thinning in reducing high-severity fire (Rhodes and Baker 2008, Campbell *et al*. 2012). Our findings highlight the need to be cautious about conclusions that thinning treatments are needed for species found in dense forest and that they will not have unintended consequences (e.g., Stephens *et al.* 2012) until long-term, cumulative impacts are better understood. As we found with spotted owls, long-term and unintended consequences may be substantial for species that rely on dense, late-successional forests, especially when these species are sensitive to small amounts of thinning in their territory.

CONCLUSION

We used a quantitative approach that, unlike others, accounted for rates of high-severity fire and forest recruitment, allowing assessment of future amounts of spotted owl habitat at current rates of fire, with and without thinning. We found that the long-term benefits of commercial thinning would not clearly outweigh adverse impacts, even if much more fire occurs in the future. This conclusion applies even if adverse impacts of treatments are quite modest because of the vastly larger area that would need to be treated compared to area of high-severity fire that might be reduced by thinning. Moreover, our results indicate that, even if a longer time interval is analyzed (e.g., 100 years), the declines in dense, late-successional habitat due to thinning would not flatten, as long as thinning is reoccurring. Thus, where spotted owl management goals take precedence, the best strategy for maintaining habitat will be to avoid thinning treatments that have adverse impacts in spotted owl habitat or potential habitat (Gaines *et al.* 2010). There is ample area outside of existing or potential spotted owl habitat where managers wishing to suppress fire behavior or extent may focus their efforts without directly impacting spotted owls (Gaines *et al.* 2010), such as in areas adjacent to homes or in dense conifer plantations with high fuel hazards (Odion *et al.* 2004). In addition, there are management approaches that may be more effective than thinning in helping accomplish these fire prevention goals, such as controlling human-caused fire ignitions (Cary *et al.* 2009). Lastly, emerging research suggests that fire is not the threat it has been assumed to be for spotted owls, suggesting that, rather than management that focuses on suppressing fire behavior, other, no regrets active management may be more appropriate (Hanson *et al.* 2010). Research is needed to determine if these findings might apply to other species that are characteristic of dense forests, particularly given the widespread and growing emphasis on thinning as a management tool for suppressing wildland fires.

CONFLICT OF INTEREST

The authors confirm that this article content has no conflict of interest.

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SUPPORTIVE/SUPPLEMENTARY MATERIAL

Supplementary material is available on the publishers Web site along with the published article.

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Land use strategies to mitigate climate change in carbon dense temperate forests

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Strategies to mitigate carbon dioxide emissions through forestry activities have been proposed, but ecosystem process-based integration of climate change, enhanced CO_2, disturbance from fire, and management actions at regional scales are extremely limited. Here, we examine the relative merits of afforestation, reforestation, management changes, and harvest residue bioenergy use in the Pacific Northwest. This region represents some of the highest carbon density forests in the world, which can store carbon in trees for 800 y or more. Oregon's net ecosystem carbon balance (NECB) was equivalent to 72% of total emissions in 2011–2015. By 2100, simulations show increased net carbon uptake with little change in wildfires. Reforestation, afforestation, lengthened harvest cycles on private lands, and restricting harvest on public lands increase NECB 56% by 2100, with the latter two actions contributing the most. Resultant cobenefits included water availability and biodiversity, primarily from increased forest area, age, and species diversity. Converting 127,000 ha of irrigated grass crops to native forests could decrease irrigation demand by 233 billion $m^3 \cdot y^{-1}$. Utilizing harvest residues for bioenergy production instead of leaving them in forests to decompose increased emissions in the short-term (50 y), reducing mitigation effectiveness. Increasing forest carbon on public lands reduced emissions compared with storage in wood products because the residence time is more than twice that of wood products. Hence, temperate forests with high carbon densities and lower vulnerability to mortality have substantial potential for reducing forest sector emissions. Our analysis framework provides a template for assessments in other temperate regions.

forests | carbon balance | greenhouse gas emissions | climate mitigation

Strategies to mitigate carbon dioxide emissions through forestry activities have been proposed, but regional assessments to determine feasibility, timeliness, and effectiveness are limited and rarely account for the interactive effects of future climate, atmospheric CO_2 enrichment, nitrogen deposition, disturbance from wildfires, and management actions on forest processes. We examine the net effect of all of these factors and a suite of mitigation strategies at fine resolution (4-km grid). Proven strategies immediately available to mitigate carbon emissions from forest activities include the following: (*i*) reforestation (growing forests where they recently existed), (*ii*) afforestation (growing forests where they did not recently exist), (*iii*) increasing carbon density of existing forests, and (*iv*) reducing emissions from deforestation and degradation (1). Other proposed strategies include wood bioenergy production (2–4), bioenergy combined with carbon capture and storage (BECCS), and increasing wood product use in buildings. However, examples of commercial-scale BECCS are still scarce, and sustainability of wood sources remains controversial because of forgone ecosystem carbon storage and low environmental cobenefits (5, 6). Carbon stored in buildings generally outlives its usefulness or is replaced within decades (7) rather than the centuries possible in forests, and the factors influencing product substitution have yet to be fully explored (8). Our analysis of mitigation strategies focuses on the first four strategies, as well as bioenergy production, utilizing harvest residues only and without carbon capture and storage.

The appropriateness and effectiveness of mitigation strategies within regions vary depending on the current forest sink, competition with land-use and watershed protection, and environmental conditions affecting forest sustainability and resilience. Few process-based regional studies have quantified strategies that could actually be implemented, are low-risk, and do not depend on developing technologies. Our previous studies focused on regional modeling of the effects of forest thinning on net ecosystem carbon balance (NECB) and net emissions, as well as improving modeled drought sensitivity (9, 10), while this study focuses mainly on strategies to enhance forest carbon.

Our study region is Oregon in the Pacific Northwest, where coastal and montane forests have high biomass and carbon sequestration potential. They represent coastal forests from northern California to southeast Alaska, where trees live 800 y or more and biomass can exceed that of tropical forests (11) (Fig. S1). The semiarid ecoregions consist of woodlands that experience frequent fires (12). Land-use history is a major determinant of forest carbon balance. Harvest was the dominant cause of tree mortality (2003–2012) and accounted for fivefold as much mortality as that from fire and beetles combined (13). Forest land ownership is predominantly public (64%), and 76% of the biomass harvested is on private lands.

Significance

Regional quantification of feasibility and effectiveness of forest strategies to mitigate climate change should integrate observations and mechanistic ecosystem process models with future climate, CO_2, disturbances from fire, and management. Here, we demonstrate this approach in a high biomass region, and found that reforestation, afforestation, lengthened harvest cycles on private lands, and restricting harvest on public lands increased net ecosystem carbon balance by 56% by 2100, with the latter two actions contributing the most. Forest sector emissions tracked with our life cycle assessment model decreased by 17%, partially meeting emissions reduction goals. Harvest residue bioenergy use did not reduce short-term emissions. Cobenefits include increased water availability and biodiversity of forest species. Our improved analysis framework can be used in other temperate regions.

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Data deposition: The CLM4.5 model data are available at Oregon State University (terraweb.forestry.oregonstate.edu/FMEC). Data from the >200 intensive plots on forest carbon are available at Oak Ridge National Laboratory (https://daac.ornl.gov/NACP/guides/NACP_TERRA-PNW.html), and FIA data are available at the USDA Forest Service (https://www.fia.fs.fed.us/tools-data/).

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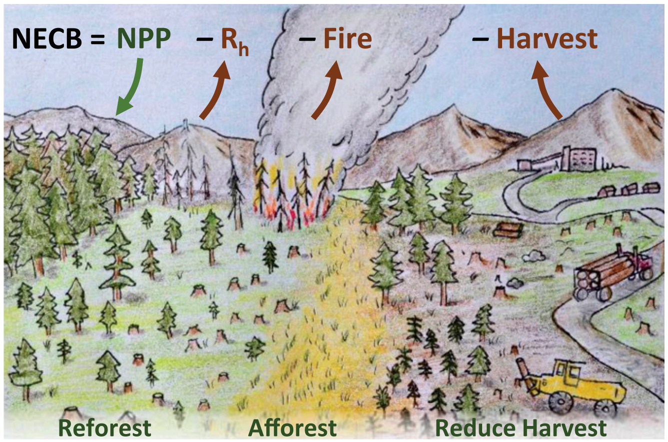

Fig. 1. Approach to assessing effects of mitigation strategies on forest carbon and forest sector emissions. NECB is productivity (NPP) minus Rh and losses from fire and harvest (red arrows). Harvest emissions include those associated with wood products and bioenergy.

Many US states, including Oregon (14), plan to reduce their greenhouse gas (GHG) emissions in accordance with the Paris Agreement. We evaluated strategies to address this question: How much carbon can the region's forests realistically remove from the atmosphere in the future, and which forest carbon strategies can reduce regional emissions by 2025, 2050, and 2100? We propose an integrated approach that combines observations with models and a life cycle assessment (LCA) to evaluate current and future effects of mitigation actions on forest carbon and forest sector emissions in temperate regions (Fig. 1). We estimated the recent carbon budget of Oregon's forests, and simulated the potential to increase the forest sink and decrease forest sector emissions under current and future climate conditions. We provide recommendations for regional assessments of mitigation strategies.

Results

Carbon stocks and fluxes are summarized for the observation cycles of 2001–2005, 2006–2010, and 2011–2015 (Table 1 and Tables S1 and S2). In 2011–2015, state-level forest carbon stocks totaled 3,036 Tg C (3 billion metric tons), with the coastal and montane ecoregions accounting for 57% of the live tree carbon (Tables S1 and S2). Net ecosystem production [NEP; net primary production (NPP) minus heterotrophic respiration (Rh)] averaged 28 teragrams carbon per year (Tg C y^{-1}) over all three periods. Fire emissions were unusually high at 8.69 million metric tons carbon dioxide equivalent ($tCO_2e\ y^{-1}$, i.e., 2.37 Tg C y^{-1}) in 2001–2005 due to the historic Biscuit Fire, but decreased to 3.56 million $tCO_2e\ y^{-1}$ (0.97 Tg C y^{-1}) in 2011–2015 (Table S4). Note that 1 million tCO_2e equals 3.667 Tg C.

Our LCA showed that in 2001–2005, Oregon's net wood product emissions were 32.61 million tCO_2e (Table S3), and 3.7-fold wildfire emissions in the period that included the record fire year (15) (Fig. 2). In 2011–2015, net wood product emissions were 34.45 million tCO_2e and almost 10-fold fire emissions, mostly due to lower fire emissions. The net wood product emissions are higher than fire emissions despite carbon benefits of storage in wood products and substitution for more fossil fuel-intensive products. Hence, combining fire and net wood product emissions, the forest sector emissions averaged 40 million $tCO_2e\ y^{-1}$ and accounted for about 39% of total emissions across all sectors (Fig. 2 and Table S4). NECB was calculated from NEP minus losses from fire emissions and harvest (Fig. 1). State NECB was equivalent to 60% and 70% of total emissions for 2001–2005 and 2011–2015, respectively (Fig. 2, Table 1, and Table S4). Fire emissions were only between 4% and 8% of total emissions from all sources (2011–2015 and 2001–2004, respectively). Oregon's forests play a larger role in meeting its GHG targets than US forests have in meeting the nation's targets (16, 17).

Historical disturbance regimes were simulated using stand age and disturbance history from remote sensing products. Comparisons of Community Land Model (CLM4.5) output with Forest Inventory and Analysis (FIA) aboveground tree biomass (>6,000 plots) were within 1 SD of the ecoregion means (Fig. S2). CLM4.5 estimates of cumulative burn area and emissions from 1990 to 2014 were 14% and 25% less than observed, respectively. The discrepancy was mostly due to the model missing an anomalously large fire in 2002 (Fig. S3*A*). When excluded, modeled versus observed fire emissions were in good agreement ($r^2 = 0.62$; Fig. S3*B*). A sensitivity test of a 14% underestimate of burn area did not affect our final results because predicted emissions would increase almost equally for business as usual (BAU) management and our scenarios, resulting in no proportional change in NECB. However, the ratio of harvest to fire emissions would be lower.

Projections show that under future climate, atmospheric carbon dioxide, and BAU management, an increase in net carbon uptake due to CO_2 fertilization and climate in the mesic ecoregions far outweighs losses from fire and drought in the semiarid ecoregions. There was not an increasing trend in fire. Carbon stocks increased by 2% and 7% and NEP increased by 12% and 40% by 2050 and 2100, respectively.

We evaluated emission reduction strategies in the forest sector: protecting existing forest carbon, lengthening harvest cycles, reforestation, afforestation, and bioenergy production with product substitution. The largest potential increase in forest carbon is in the mesic Coast Range and West Cascade ecoregions. These forests are buffered by the ocean, have high soil water-holding capacity, low risk of wildfire [fire intervals average 260–400 y (18)], long carbon residence time, and potential for high carbon density. They can attain biomass up to 520 Mg C ha^{-1} (12). Although Oregon has several protected areas, they account for only 9–15% of the total forest area, so we expect it may be feasible to add carbon-protected lands with cobenefits of water protection and biodiversity.

Reforestation of recently forested areas include those areas impacted by fire and beetles. Our simulations to 2100 assume regrowth of the same species and incorporate future fire responses to climate and cyclical beetle outbreaks [70–80 y (13)]. Reforestation has the potential to increase stocks by 315 Tg C by 2100, reducing forest sector net emissions by 5% by 2100 relative to BAU management (Fig. 3). The East and West Cascades ecoregions had the highest reforestation potential, accounting for 90% of the increase (Table S5).

Afforestation of old fields within forest boundaries and nonfood/nonforage grass crops, hereafter referred to as "grass crops," had to meet minimum conditions for tree growth, and crop grid cells had to be partially forested (*SI Methods* and Table S6). These crops are not grazed or used for animal feed. Competing land uses may decrease the actual amount of area that can be afforested. We calculated the amount of irrigated grass crops (127,000 ha) that could be converted to forest, assuming success of carbon offset programs (19). By 2100, afforestation increased stocks by

Table 1. Forest carbon budget components used to compute NECB

Flux, Tg C·y^{-1}	2001–2005		2006–2010		2011–2015		2001–2015
NPP	73.64	7.59	73.57	7.58	73.57	7.58	73.60
Rh	45.67	5.11	45.38	5.07	45.19	5.05	45.41
NEP	27.97	9.15	28.19	9.12	28.39	9.11	28.18
Harvest removals	8.58	0.60	7.77	0.54	8.61	0.6	8.32
Fire emissions	2.37	0.27	1.79	0.2	0.97	0.11	1.71
NECB	17.02	9.17	18.63	9.14	18.81	9.13	18.15

Average annual values for each period, including uncertainty (95% confidence interval) in Tg C y^{-1} (multiply by 3.667 to get million tCO_2e).

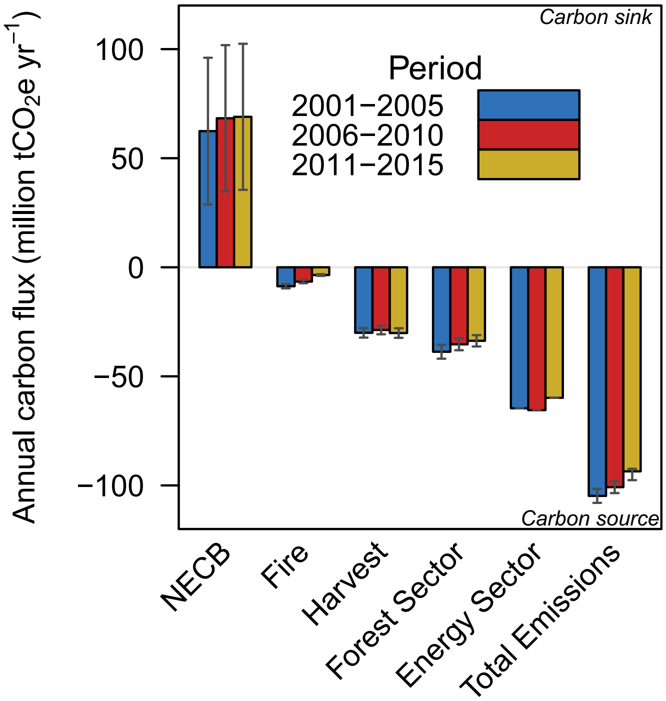

Fig. 2. Oregon's forest carbon sink and emissions from forest and energy sectors. Harvest emissions are computed by LCA. Fire and harvest emissions sum to forest sector emissions. Energy sector emissions are from the Oregon Global Warming Commission (14), minus forest-related emissions. Error bars are 95% confidence intervals (Monte Carlo analysis).

94 Tg C and cumulative NECB by 14 Tg C, and afforestation reduced forest sector GHG emissions by 1.3–1.4% in 2025, 2050, and 2100 (Fig. 3).

We quantified cobenefits of afforestation of irrigated grass crops on water availability based on data from hydrology and agricultural simulations of future grass crop area and related irrigation demand (20). Afforestation of 127,000 ha of grass cropland with Douglas fir could decrease irrigation demand by 222 and 233 billion $m^3 \cdot y^{-1}$ by 2050 and 2100, respectively. An independent estimate from measured precipitation and evapotranspiration (ET) at our mature Douglas fir and grass crop flux sites in the Willamette Valley shows the ET/precipitation fraction averaged 33% and 52%, respectively, and water balance (precipitation minus ET) averaged 910 $mm \cdot y^{-1}$ and 516 $mm \cdot y^{-1}$. Under current climate conditions, the observations suggest an increase in annual water availability of 260 billion $m^3 \cdot y^{-1}$ if 127,000 ha of the irrigated grass crops were converted to forest.

Harvest cycles in the mesic and montane forests have declined from over 120 y to 45 y despite the fact that these trees can live 500–1,000 y and net primary productivity peaks at 80–125 y (21). If harvest cycles were lengthened to 80 y on private lands and harvested area was reduced 50% on public lands, state-level stocks would increase by 17% to a total of ~3,600 Tg C and NECB would increase 2–3 Tg C y^{-1} by 2100. The lengthened harvest cycles reduced harvest by 2 Tg C y^{-1}, which contributed to higher NECB. Leakage (more harvest elsewhere) is difficult to quantify and could counter these carbon gains. However, because harvest on federal lands was reduced significantly since 1992 (NW Forest Plan), leakage has probably already occurred.

The four strategies together increased NECB by 64%, 82%, and 56% by 2025, 2050, and 2100, respectively. This reduced forest sector net emissions by 11%, 10%, and 17% over the same periods (Fig. 3). By 2050, potential increases in NECB were largest in the Coast Range (Table S5), East Cascades, and Klamath Mountains, accounting for 19%, 25%, and 42% of the total increase, whereas by 2100, they were most evident in the West Cascades, East Cascades, and Klamath Mountains.

We examined the potential for using existing harvest residue for electricity generation, where burning the harvest residue for energy emits carbon immediately (3) versus the BAU practice of leaving residues in forests to slowly decompose. Assuming half of forest residues from harvest practices could be used to replace natural gas or coal in distributed facilities across the state, they would provide an average supply of 0.75–1 Tg C y^{-1} to the year 2100 in the reduced harvest and BAU scenarios, respectively. Compared with BAU harvest practices, where residues are left to decompose, proposed bioenergy production would increase cumulative net emissions by up to 45 Tg C by 2100. Even at 50% use, residue collection and transport are not likely to be economically viable, given the distances (>200 km) to Oregon's facilities.

Discussion

Earth system models have the potential to bring terrestrial observations related to climate, vulnerability, impacts, adaptation,

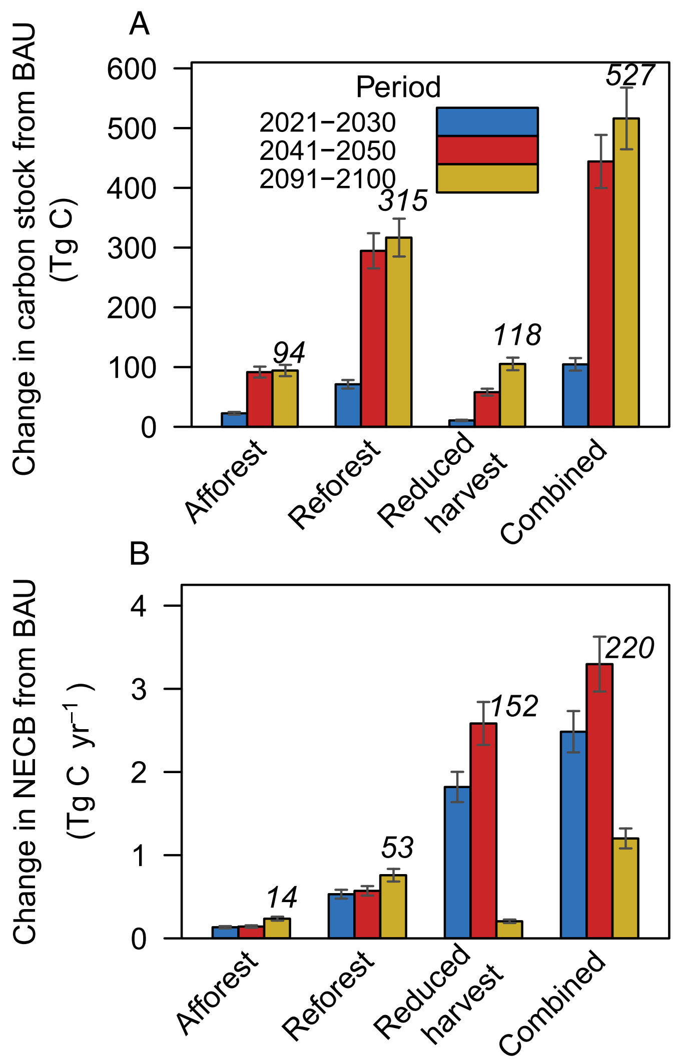

Fig. 3. Future change in carbon stocks and NECB with mitigation strategies relative to BAU management. The decadal average change in forest carbon stocks (*A*) and NECB relative to BAU (*B*) are shown. Italicized numbers over bars indicate mean forest carbon stocks in 2091–2100 (*A*) and cumulative change in NECB for 2015–2100 (*B*). Error bars are ±10%.

and mitigation into a common framework, melding biophysical with social components (22). We developed a framework to examine a suite of mitigation actions to increase forest carbon sequestration and reduce forest sector emissions under current and future environmental conditions.

Harvest-related emissions had a large impact on recent forest NECB, reducing it by an average of 34% from 2001 to 2015. By comparison, fire emissions were relatively small and reduced NECB by 12% in the Biscuit Fire year, but only reduced NECB 5–9% from 2006 to 2015. Thus, altered forest management has the potential to enhance the forest carbon balance and reduce emissions.

Future NEP increased because enhancement from atmospheric carbon dioxide outweighed the losses from fire. Lengthened harvest cycles on private lands to 80 y and restricting harvest to 50% of current rates on public lands increased NECB the most by 2100, accounting for 90% of total emissions reduction (Fig. 3 and Tables S5 and S6). Reduced harvest led to NECB increasing earlier than the other strategies (by 2050), suggesting this could be a priority for implementation.

Our afforestation estimates may be too conservative by limiting them to nonforest areas within current forest boundaries and 127,000 ha of irrigated grass cropland. There was a net loss of 367,000 ha of forest area in Oregon and Washington combined from 2001 to 2006 (23), and less than 1% of native habitat remains in the Willamette Valley due to urbanization and agriculture (24). Perhaps more of this area could be afforested.

The spatial variation in the potential for each mitigation option to improve carbon stocks and fluxes shows that the reforestation potential is highest in the Cascade Mountains, where fire and insects occur (Fig. 4). The potential to reduce harvest on public land is highest in the Cascade Mountains, and that to lengthen harvest cycles on private lands is highest in the Coast Range.

Although western Oregon is mesic with little expected change in precipitation, the afforestation cobenefits of increased water availability will be important. Urban demand for water is projected to increase, but agricultural irrigation will continue to consume much more water than urban use (25). Converting 127,000 ha of irrigated grass crops to native forests appears to be a win–win strategy, returning some of the area to forest land, providing habitat and connectivity for forest species, and easing irrigation demand. Because the afforested grass crop represents only 11% of the available grass cropland (1.18 million ha), it is not likely to result in leakage or indirect land use change. The two forest strategies combined are likely to be important contributors to water security.

Cobenefits with biodiversity were not assessed in our study. However, a recent study showed that in the mesic forests, cobenefits with biodiversity of forest species are largest on lands with harvest cycles longer than 80 y, and thus would be most pronounced on private lands (26). We selected 80 y for the harvest cycle mitigation strategy because productivity peaks at 80–125 y in this region, which coincides with the point at which cobenefits with wildlife habitat are substantial.

Habitat loss and climate change are the two greatest threats to biodiversity. Afforestation of areas that are currently grass crops would likely improve the habitat of forest species (27), as about 90% of the forests in these areas were replaced by agriculture. About 45 mammal species are at risk because of range contraction (28). Forests are more efficient at dissipating heat than grass and crop lands, and forest cover gains lead to net surface cooling in all regions south of about 45° latitude in North American and Europe (29). The cooler conditions can buffer climate-sensitive bird populations from approaching their thermal limits and provide more food and nest sites (30). Thus, the mitigation strategies of afforestation, protecting forests on public lands and lengthening harvest cycles to 80–125 y, would likely benefit forest-dependent species.

Oregon has a legislated mandate to reduce emissions, and is considering an offsets program that limits use of offsets to 8% of

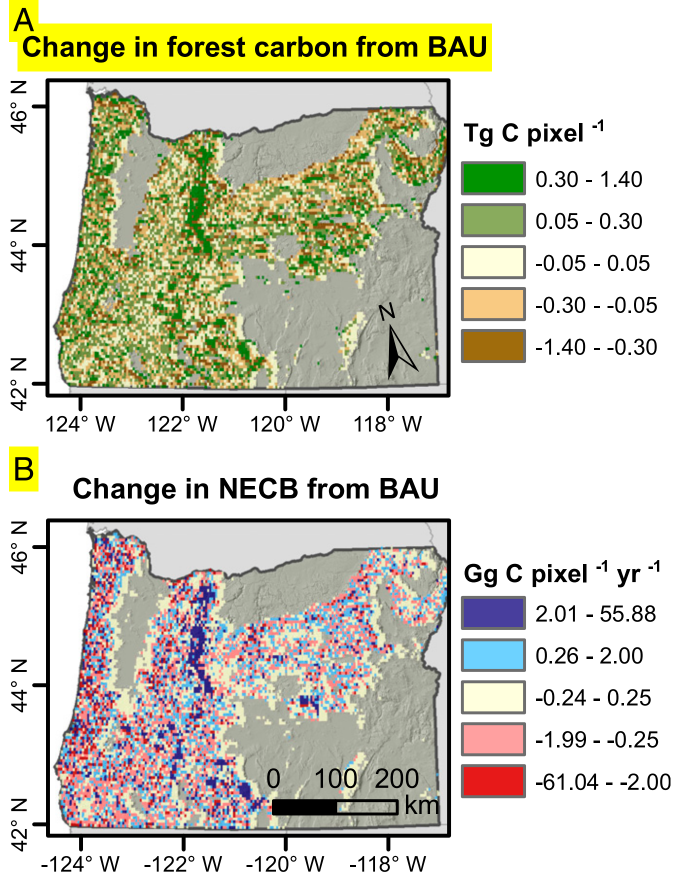

Fig. 4. Spatial patterns of forest carbon stocks and NECB by 2091–2100. The decadal average changes in forest carbon stocks (*A*) and NECB (*B*) due to afforestation, reforestation, protected areas, and lengthened harvest cycles relative to continued BAU forest management (red is increase in NECB) are shown.

the total emissions reduction to ensure that regulated entities substantially reduce their own emissions, similar to California's program (19). An offset becomes a net emissions reduction by increasing the forest carbon sink (NECB). If only 8% of the GHG reduction is allowed for forest offsets, the limits for forest offsets would be 2.1 and 8.4 million metric tCO_2e of total emissions by 2025 and 2050, respectively (Table S6). The combination of afforestation, reforestation, and reduced harvest would provide 13 million metric tCO_2e emissions reductions, and any one of the strategies or a portion of each could be applied. Thus, additionality beyond what would happen without the program is possible.

State-level reporting of GHG emissions includes the agriculture sector, but does not appear to include forest sector emissions, except for industrial fuel (i.e., utility fuel in Table S3) and, potentially, fire emissions. Harvest-related emissions should be quantified, as they are much larger than fire emissions in the western United States. Full accounting of forest sector emissions is necessary to meet climate mitigation goals.

Increased long-term storage in buildings and via product substitution has been suggested as a potential climate mitigation option. Pacific temperate forests can store carbon for many hundreds of years, which is much longer than is expected for buildings that are generally assumed to outlive their usefulness or be replaced within several decades (7). By 2035, about 75% of buildings in the United States will be replaced or renovated, based on new construction, demolition, and renovation trends (31, 32). Recent analysis suggests substitution benefits of using wood versus more fossil fuel-intensive materials have been overestimated by at least an order of magnitude (33). Our LCA accounts for losses in product substitution stores (PSSs) associated with building life span, and thus are considerably lower than when no losses are assumed (4, 34). While product substitution reduces the overall forest sector emissions, it cannot offset the losses incurred by frequent harvest and losses associated with product transportation, manufacturing, use, disposal, and decay. Methods for calculating substitution benefits should be improved in other regional assessments.

Wood bioenergy production is interpreted as being carbon-neutral by assuming that trees regrow to replace those that burned. However, this does not account for reduced forest carbon stocks that took decades to centuries to sequester, degraded productive capacity, emissions from transportation and the production process, and biogenic/direct emissions at the facility (35). Increased harvest through proposed thinning practices in the region has been shown to elevate emissions for decades to centuries regardless of product end use (36). It is therefore unlikely that increased wood bioenergy production in this region would decrease overall forest sector emissions.

Conclusions

GHG reduction must happen quickly to avoid surpassing a 2 °C increase in temperature since preindustrial times. Alterations in forest management can contribute to increasing the land sink and decreasing emissions by keeping carbon in high biomass forests, extending harvest cycles, reforestation, and afforestation. Forests are carbon-ready and do not require new technologies or infrastructure for immediate mitigation of climate change. Growing forests for bioenergy production competes with forest carbon sequestration and does not reduce emissions in the next decades (10). BECCS requires new technology, and few locations have sufficient geological storage for CO_2 at power facilities with high-productivity forests nearby. Accurate accounting of forest carbon in trees and soils, NECB, and historic harvest rates, combined with transparent quantification of emissions from the wood product process, can ensure realistic reductions in forest sector emissions.

As states and regions take a larger role in implementing climate mitigation steps, robust forest sector assessments are urgently needed. Our integrated approach of combining observations, an LCA, and high-resolution process modeling (4-km grid vs. typical 200-km grid) of a suite of potential mitigation actions and their effects on forest carbon sequestration and emissions under changing climate and CO_2 provides an analysis framework that can be applied in other temperate regions.

Materials and Methods

Current Stocks and Fluxes. We quantified recent forest carbon stocks and fluxes using a combination of observations from FIA; Landsat products on forest type, land cover, and fire risk; 200 intensive plots in Oregon (37); and a wood decomposition database. Tree biomass was calculated from species-specific allometric equations and ecoregion-specific wood density. We estimated ecosystem carbon stocks, NEP (photosynthesis minus respiration), and NECB (NEP minus losses due to fire or harvest) using a mass-balance approach (36, 38) (Table 1 and *SI Materials and Methods*). Fire emissions were computed from the Monitoring Trends in Burn Severity database, biomass data, and region-specific combustion factors (15, 39) (*SI Materials and Methods*).

Future Projections and Model Description. Carbon stocks and NEP were quantified to the years 2025, 2050, and 2100 using CLM4.5 with physiological parameters for 10 major forest species, initial forest biomass (36), and future climate and atmospheric carbon dioxide as input (Institut Pierre Simon Laplace climate system model downscaled to 4 km × 4 km, representative concentration pathway 8.5). CLM4.5 uses 3-h climate data, ecophysiological characteristics, site physical characteristics, and site history to estimate the daily fluxes of carbon, nitrogen, and water between the atmosphere, plant state variables, and litter and soil state variables. Model components are biogeophysics, hydrological cycle, and biogeochemistry. This model version does not include a dynamic vegetation model to simulate resilience and establishment following disturbance. However, the effect of regeneration lags on forest carbon is not particularly strong for the long disturbance intervals in this study (40). Our plant functional type (PFT) parameterization for 10 major forest species rather than one significantly improves carbon modeling in the region (41).

Forest Management and Land Use Change Scenarios. Harvest cycles, reforestation, and afforestation were simulated to the year 2100. Carbon stocks and NEP were predicted for the current harvest cycle of 45 y compared with simulations extending it to 80 y. Reforestation potential was simulated over areas that recently suffered mortality from harvest, fire, and 12 species of beetles (13). We assumed the same vegetation regrew to the maximum potential, which is expected with the combination of natural regeneration and planting that commonly occurs after these events. Future BAU harvest files were constructed using current harvest rates, where county-specific average harvest and the actual amounts per ownership were used to guide grid cell selection. This resulted in the majority of harvest occurring on private land (70%) and in the mesic ecoregions. Beetle outbreaks were implemented using a modified mortality rate of the lodgepole pine PFT with 0.1% y^{-1} biomass mortality by 2100.

For afforestation potential, we identified areas that are within forest boundaries that are not currently forest and areas that are currently grass crops. We assumed no competition with conversion of irrigated grass crops to urban growth, given Oregon's land use laws for developing within urban growth boundaries. A separate study suggested that, on average, about 17% of all irrigated agricultural crops in the Willamette Valley could be converted to urban area under future climate; however, because 20% of total cropland is grass seed, it suggests little competition with urban growth (25).

Landsat observations (12,500 scenes) were processed to map changes in land cover from 1984 to 2012. Land cover types were separated with an unsupervised K-means clustering approach. Land cover classes were assigned to an existing forest type map (42). The CropScape Cropland Data Layer (CDL 2015, https://nassgeodata.gmu.edu/CropScape/) was used to distinguish nonforage grass crops from other grasses. For afforestation, we selected grass cropland with a minimum soil water-holding capacity of 150 mm and minimum precipitation of 500 mm that can support trees (43).

Afforestation Cobenefits. Modeled irrigation demand of grass seed crops under future climate conditions was previously conducted with hydrology and agricultural models, where ET is a function of climate, crop type, crop growth state, and soil-holding capacity (20) (Table S7). The simulations produced total land area, ET, and irrigation demand for each cover type. Current grass seed crop irrigation in the Willamette Valley is 413 billion $m^3 \cdot y^{-1}$ for 238,679 ha and is projected to be 412 and 405 billion m^3 in 2050 and 2100 (20) (Table S7). We used annual output from the simulations to estimate irrigation demand per unit area of grass seed crops (1.73, 1.75, and 1.84 million $m^3 \cdot ha^{-1}$ in 2015, 2050, and 2100, respectively), and applied it to the mapped irrigated crop area that met conditions necessary to support forests (Table S7).

LCA. Decomposition of wood through the product cycle was computed using an LCA (8, 10). Carbon emissions to the atmosphere from harvest were calculated annually over the time frame of the analysis (2001–2015). The net carbon emissions equal NECB plus total harvest minus wood lost during manufacturing and wood decomposed over time from product use. Wood industry fossil fuel emissions were computed for harvest, transportation, and manufacturing processes. Carbon credit was calculated for wood product storage, substitution, and internal mill recycling of wood losses for bioenergy.

Products were divided into sawtimber, pulpwood, and wood and paper products using published coefficients (44). Long-term and short-term products were assumed to decay at 2% and 10% per year, respectively (45). For product substitution, we focused on manufacturing for long-term structures (building life span >30 y). Because it is not clear when product substitution started in the Pacific Northwest, we evaluated it starting in 1970 since use of concrete and steel for housing was uncommon before 1965. The displacement value for product substitution was assumed to be 2.1 Mg fossil C/Mg C wood use in long-term structures (46), and although it likely fluctuates over time, we assumed it was constant. We accounted for losses in product substitution associated with building replacement (33) using a loss rate of 2% per year (33), but ignored leakage related to fossil C use by other sectors, which may result in more substitution benefit than will actually occur.

The general assumption for modern buildings, including cross-laminate timber, is they will outlive their usefulness and be replaced in about 30 y (7). By 2035, ∼75% of buildings in the United States will be replaced or renovated, based on new construction, demolition, and renovation trends, resulting in threefold as many buildings as there are now [2005 baseline (31, 32)]. The loss of

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the PSS is therefore PSS multiplied by the proportion of buildings lost per year (2% per year).

To compare the NECB equivalence to emissions, we calculated forest sector and energy sector emissions separately. Energy sector emissions ["in-boundary" state-quantified emissions by the Oregon Global Warming Commission (14)] include those from transportation, residential and commercial buildings, industry, and agriculture. The forest sector emissions are cradle-to-grave annual carbon emissions from harvest and product emissions, transportation, and utility fuels (Table S3). Forest sector utility fuels were subtracted from energy sector emissions to avoid double counting.

Uncertainty Estimates. For the observation-based analysis, Monte Carlo simulations were used to conduct an uncertainty analysis with the mean and SDs for NPP and Rh calculated using several approaches (36) (*SI Materials and Methods*). Uncertainty in NECB was calculated as the combined uncertainty of NEP, fire emissions (10%), harvest emissions (7%), and land cover estimates (10%) using the propagation of error approach. Uncertainty in CLM4.5 model simulations and LCA were quantified by combining the uncertainty in the observations used to evaluate the model, the uncertainty in input datasets (e.g., remote sensing), and the uncertainty in the LCA coefficients (41).

Model input data for physiological parameters and model evaluation data on stocks and fluxes are available online (37).

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LETTER

Do insect outbreaks reduce the severity of subsequent forest fires?

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Abstract

Understanding the causes and consequences of rapid environmental change is an essential scientific frontier, particularly given the threat of climate- and land use-induced changes in disturbance regimes. In western North America, recent widespread insect outbreaks and wildfires have sparked acute concerns about potential insect–fire interactions. Although previous research shows that insect activity typically does not increase wildfire likelihood, key uncertainties remain regarding insect effects on wildfire severity (i.e., ecological impact). Recent assessments indicate that outbreak severity and burn severity are not strongly associated, but these studies have been limited to specific insect or fire events. Here, we present a regional census of large wildfire severity following outbreaks of two prevalent bark beetle and defoliator species, mountain pine beetle (*Dendroctonus ponderosae*) and western spruce budworm (*Choristoneura freemani*), across the US Pacific Northwest. We first quantify insect effects on burn severity with spatial modeling at the fire event scale and then evaluate how these effects vary across the full population of insect–fire events ($n = 81$ spanning 1987–2011). In contrast to common assumptions of positive feedbacks, we find that insects generally reduce the severity of subsequent wildfires. Specific effects vary with insect type and timing, but both insects decrease the abundance of live vegetation susceptible to wildfire at multiple time lags. By dampening subsequent burn severity, native insects could buffer rather than exacerbate fire regime changes expected due to land use and climate change. In light of these findings, we recommend a precautionary approach when designing and implementing forest management policies intended to reduce wildfire hazard and increase resilience to global change.

1. Introduction

Forest ecosystems play a vital role in the biosphere, but anthropogenic climate change and shifting disturbance regimes threaten to destabilize the ecosystem services that forests provide from local to global scales (Kurz *et al* 2008, Littell *et al* 2010, Seidl *et al* 2011, Turner *et al* 2013). Indeed, the indirect effects of climate change on forests via disturbances (including wildfires, insect outbreaks, introduced species, and pathogens) are expected to exceed the direct but more gradual effects of warmer temperatures (Ayres *et al* 2014, Hart *et al* 2015). In an era of rapid, nonlinear changes in the Earth system, understanding the causes, consequences, and feedbacks of forest disturbances is a crucial scientific and policy frontier.

Disturbance interactions—when one disturbance influences the likelihood, extent, or severity of another (Paine *et al* 1998, Simard *et al* 2011, Buma 2015, Meigs *et al* 2015a)—are a particularly important example of feedbacks that could be reinforced under novel climatic conditions (e.g., persistent drought (Turner *et al* 2013, Harvey *et al* 2014b, Hart *et al* 2015)). In western North America, insect outbreaks and wildfires are the two most ecologically and economically significant natural forest disturbances (Westerling *et al* 2006, Kurz *et al* 2008, Hicke *et al* 2013). Both disturbances have been widespread in recent decades and are projected to increase in response to climate and land use change (Hessburg *et al* 2000, Westerling *et al* 2006, Raffa *et al* 2008, Bentz *et al* 2010, Littell *et al* 2010, Ayres *et al* 2014). By killing trees and redistributing forest fuels, insect outbreaks influence fire regimes in many parts of the world, and recent large outbreaks have sparked acute societal concerns about potential insect–fire interactions and impaired ecosystem resilience (Hicke *et al* 2012, Harvey *et al* 2014b, Jenkins *et al* 2014). For example, based on the implicit assumption that insect outbreaks increase wildfire hazard by generating abundant dead fuels, the 2014 US Farm Bill designated \$200 million annually to support fuel reduction activities across 18 M ha of US National Forest lands affected by diseases and insects (Agricultural Act of 2014, Hart *et al* 2015).

Despite concerns about altered fire regimes and insect–fire interactions, recent studies indicate that insect outbreaks generally do not increase wildfire likelihood (Lynch and Moorcroft 2008, Kulakowski and Jarvis 2011, Flower *et al* 2014, Hart *et al* 2015, Meigs *et al* 2015a). When they do overlap, however, key uncertainties remain regarding the influence of insect outbreaks on subsequent wildfire severity (Hicke *et al* 2012, Harvey *et al* 2014b, Hart *et al* 2015). Specifically, although insect-caused tree mortality may increase the flammability of canopy fuels at fine scales in time and space (Jolly *et al* 2012), a pivotal question in contemporary environmental management is whether these insect-altered fuels increase burn severity (i.e., ecological impact; a major fire regime component) at broader spatiotemporal scales. If insect outbreaks do amplify subsequent fire effects, the resultant compound impacts may hasten climate-induced shifts in disturbance regimes toward more severe fire and altered ecosystem structure and function. Conversely, if insects buffer subsequent fire effects by redistributing fuel density and/or availability, recent widespread outbreaks may bolster ecosystem resistance to shifting fire regimes. Empirical studies that identify particular time lags and locations where insect-altered fuels either exacerbate or dampen fire effects on surviving trees are directly applicable to time-sensitive management activities (e.g., post-insect salvage logging, fuel reduction at the wildland–urban interface) as well as broader policy discussions of forest health in a time of shifting disturbance regimes.

Due in part to data paucity, computational limitations, and the relative rarity of insect–fire co-occurrence, recent empirical assessments of insect effects on burn severity have been limited to specific insect outbreaks, fire events, or insect–fire time lags (e.g., Crickmore 2011, Harvey *et al* 2013, Harvey *et al* 2014b, Prichard and Kennedy 2014). These studies suggest that burn severity is either unaffected by or weakly positively associated with outbreak severity, that insect effects are context-dependent, or that factors like fuel treatments, topography, and weather are stronger predictors of fire effects. To further elucidate general system behavior and inform regional management strategies, it is essential to investigate numerous fire events spanning multiple insect types (e.g., bark beetle versus defoliator), insect and burn severities, and time lags. Here, we leverage recent advances in remote sensing of forest disturbance dynamics (Kennedy *et al* 2010, Meigs *et al* 2015b) to conduct a burn severity census of large wildfires following recent outbreaks of the two most prevalent native forest insects across a large forested region, the US Pacific Northwest (PNW; 40 M ha; Oregon and Washington; figure 1). We focus on all large fire events (⩾400 ha) with substantial overlap of fire perimeters with prior outbreaks of either mountain pine beetle (MPB) [*Dendroctonus ponderosae* Hopkins (Coleoptera: Curculionidae: Scolytinae); a bark beetle] or western spruce budworm (WSB) [*Choristoneura freemani* Razowski (Lepidoptera: Tortricidae); a defoliator] (total $n = 81$; table S1). Our specific objectives are (1) to quantify the fine-scale (30 m) effects of recent insect outbreaks on subsequent burn severity with spatial modeling at the fire event scale and (2) to evaluate the role of insect type, time since outbreak, insect and fire extent, fire season, and interannual drought across the full population of insect–fire events.

2. Methods

2.1. Study area and recent insect dynamics

Conifer forests of the PNW vary across gradients of climate, topography, soil, and management history (Franklin and Dyrness 1973, Hessburg *et al* 2000, Meigs *et al* 2015a). Despite climatic variability, a common feature is that low precipitation during summer months (Franklin and Dyrness 1973) yields conditions conducive to periodic insect and wildfire disturbances, particularly in mixed-species conifer forests east of the crest of the Cascade Range (Meigs *et al* 2015a). In general, these forests occur in remote, mountainous terrain and are managed by US federal agencies for multiple resource objectives. Given the extent of similar geographic conditions, vegetation types, and anthropogenic pressures, recent PNW insect and wildfire patterns are broadly representative of contemporary disturbance dynamics in conifer forests of western North America.

Bark beetles, especially MPB outbreaks, have altered forest composition and structure across tens of millions of hectares of North American forests in recent decades (Raffa *et al* 2008, Bentz *et al* 2010). MPB adults attack pine tree stems [*Pinus spp.*, particularly mature lodgepole pine (*Pinus contorta* Douglas ex Louden)], inducing variable but relatively rapid tree mortality during major outbreaks (Raffa *et al* 2008, Meigs *et al* 2011). In contrast, WSB larvae typically consume the current year's foliage of host trees {particularly true firs [*Abies spp.*], spruces [*Picea spp.*], and Douglas-fir [*Pseudotsuga menziesii* (Mirb.) Franco]},

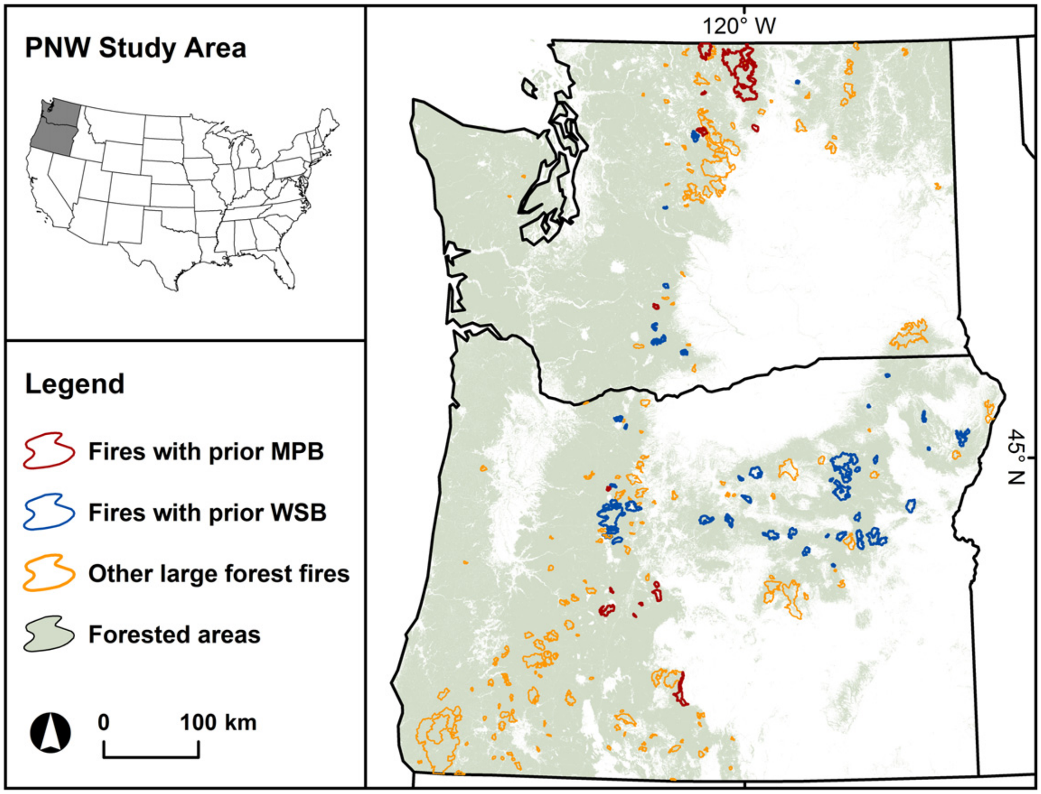

Figure 1. Distribution of large forest fires affected by prior outbreaks of mountain pine beetle (MPB; red; $n = 19$) and western spruce budworm (WSB; blue; $n = 62$) within the US Pacific Northwest study area (Oregon and Washington; inset). Note that 29% of large wildfires during the study period (⩾400 ha; 1987–2011; $n = 277$) had ⩾10% prior insect damage, highlighting that these potential compound disturbance events are limited to drier, interior conifer forests. We include only large wildfire perimeters with ⩾50% forest cover (see methods). Oregon and Washington encompass ca. 40 M ha, half of which is forested (forest cover from Ohmann *et al* 2012).

and multiple years of WSB defoliation can result in tree mortality, often in conjunction with secondary bark beetles (Hummel and Agee 2003, Meigs *et al* 2011). Across the PNW, both insects have erupted in multiple outbreaks since 1970, with WSB exceeding MPB in cumulative extent and tree mortality (Meigs *et al* 2015b). Importantly, WSB host forests are more widespread and occur in relatively warmer, more productive locations than MPB host forests in the study area.

2.2. Insect and fire census data

Recent advances in remote sensing of forest dynamics across the PNW (Kennedy *et al* 2010, Meigs *et al* 2015b) provide an unprecedented opportunity to investigate relationships between insect outbreaks and wildfire severity in a retrospective, empirical, census-based framework. We used regional maps of insect and fire effects developed with LandTrendr time series analysis, which is described in detail by Kennedy *et al* (2010). Briefly, we acquired georectified images from the USGS Landsat archive and applied a series of steps —pre-processing (atmospheric correction, cloud masking), processing (temporal segmentation), and analysis (disturbance attribution, regional mosaicking)—to reduce multiple sources of uncertainty and assess trajectories of vegetation change (Kennedy *et al* 2010, Meigs *et al* 2015b).

We accounted for insect activity with LandTrendr-based maps of the cumulative magnitude, cumulative duration (count of years), and time since onset of MPB and WSB outbreaks developed by Meigs *et al* (2015b). These insect maps improve on regional aerial surveys by capturing fine-scale variation of insect impacts (30 m) and constraining maps to locations with durable vegetation change in known insect host forests from 1985 to 2012. The maps also quantify the impacts of MPB and WSB in consistent units of spectral change as seamless mosaics across the PNW study area (including all or part of 35 Landsat satellite scenes (Meigs *et al* 2015b)).

We accounted for burn severity by combining LandTrendr-based regional mosaics of spectral change (Kennedy *et al* 2010) with fire perimeters from a database of large wildland fires in the western US (⩾400 ha; 1985–2012 (available online: http://mtbs.gov)). We first compiled annual time series (temporally stabilized at the pixel scale) of the normalized burn ratio (NBR; which combines near-infrared and mid-infrared wavelengths of the Landsat TM/

ETM + sensor (Miller and Thode 2007)). Importantly, the Landsat time series are anchored in time near the median date of each scene (generally 1 August), which reduces seasonal variability associated with phenology and sun angles. We then computed the relative differenced normalized burn ratio (RdNBR (Miller and Thode 2007)) in two-year intervals to ensure pre- and post-fire coverage for all pixels within a given fire event. By capturing the relative change in dominant forest vegetation, RdNBR enables the assessment of burn severity across numerous fire events spanning heterogeneous vegetation (Miller and Thode 2007, Cansler and McKenzie 2014) or variable prefire disturbances (including insect outbreaks (Harvey *et al* 2013, Prichard and Kennedy 2014)). Although remotely sensed spectral change indices such as RdNBR have inherent limitations and do not measure very fine-scale fire effects and responses (e.g., tree charring, forest floor combustion, or postfire regeneration (Harvey *et al* 2014b)), they provide the only spatially and temporally consistent metric of burn severity encompassing all fires since 1985. Furthermore, because NBR is at the core of many current fire monitoring protocols (e.g., Key and Benson 2006), our RdNBR-based analysis is directly applicable to contemporary fire research and management.

We conducted a regional insect–fire severity census by focusing on large fire events with the following characteristics: total fire extent $\geqslant$400 ha; $\geqslant$10% of fire extent affected by prefire insect outbreaks (either MPB or WSB); $\geqslant$50% forest cover (30 m resolution (Ohmann *et al* 2012)). Because this forest cover map targets conditions in the year 2000 and classifies some previously burned areas as non-forest, we manually included several fires ($n = 8$) with mapped forest cover <50%. To avoid potential confounding effects, we excluded fire polygons with prior outbreaks of both MPB and WSB ($n = 8$), fires in 1986 with only one full year of prefire insect data ($n = 5$), fires in 2012 without postfire imagery for RdNBR calculations, and one fire classified as a prescribed fire. With these criteria, we refined the total population of forest fires ($n = 425$ spanning 1985–2012) to our final census of large wildfires with prefire insect activity ($n = 81$ spanning 1987–2011; figure 1).

2.3. Statistical analysis

We developed a hierarchical framework to investigate insect effects on burn severity within and among all wildfires in our census (i.e., at the individual and population level). Within each large insect–fire event, we assessed fine-scale (30 m) insect effects on burn severity with sequential autoregression (SAR), a powerful spatial modeling approach advanced recently for wildfire analysis (e.g., Wimberly *et al* 2009, Prichard and Kennedy 2014). SAR incorporates the inherent spatial autocorrelation in dependent and independent variables with a spatial error term (Haining 1993, Wimberly *et al* 2009). This spatial error term also accounts for spatially autocorrelated variables not included explicitly, resulting in more robust inferences than traditional approaches like ordinary least squares regression (Wimberly *et al* 2009, Prichard and Kennedy 2014).

We conducted all analyses in the R statistical environment (R Core Team 2015), constructing SAR models with the spautolm function in the spdep package (Bivand *et al* 2013) in the form:

$$Y = X\beta + \lambda W(Y - X\beta) + \varepsilon,$$

where Y is the vector of the dependent variable, X is the matrix of independent variables, β is the vector of parameters, λ is the autoregressive coefficient, W is the spatial weights matrix, and ε is the uncorrelated error term. W is based on the spatial structure of the dependent and independent variables and is defined by an inverse distance rule that assigns a weight of zero to all pixels outside the focus pixel neighborhood and weights equal to the inverse of the distance within the focus pixel neighborhood. We determined the most parsimonious inverse distance rule of W by selecting the neighborhood that minimized both the Akaike information criterion (AIC) and residual spatial autocorrelation of the SAR model (Moran's I) (Kissling and Carl 2008, de Knegt *et al* 2010). Specifically, we ran SAR models with all dependent and independent variables (described below) across seven neighborhood distances (30–210 m in 30 m increments) for a subset of fires ($n = 15$) spanning the range of conditions in the large fire census. We then calculated AIC and Moran's I of the SAR residual values (moran.test function in Bivand *et al* 2013), which indicated an optimal neighborhood distance of 30 m, consistent with previous SAR burn severity modeling in the study area (Prichard and Kennedy 2014) and typical for a spreading disturbance phenomenon such as fire.

Following these initial steps, we quantified insect effects on subsequent burn severity at the individual fire level (Objective 1) by running a SAR model for each large insect–fire event in the regional census ($n = 81$). We used all 30 m pixels within each fire perimeter to predict burn severity (RdNBR) with the same set of independent variables related to forest fuels and topography (table 1). We included the insect damage and duration variables described above as well as pixel-level estimates of prefire biomass from annual Landsat time series and nearest neighbor imputation with forest inventory data derived for a regional analysis of carbon trajectories (http://lemma.forestry.oregonstate.edu/projects/cmonster).

Although our primary focus was insect effects on burn severity via their impacts on vegetation/fuels, we recognize that topography and weather are fundamental drivers of fire behavior and effects. We thus included a set of five topographic variables (aspect, elevation, slope, and topographic position index at 150 and 450 m; derived from a 30 m digital elevation model) associated with burn severity in the region

Table 1. List of variables used in sequential autoregression modeling of burn severity (RdNBR spectral index) of all fire events affected by prior mountain pine beetle or western spruce budworm. All data were compiled as regional mosaics encompassing the Pacific Northwest study area (figure 1) and processed at 30 m resolution.

Variable	Description	Source
Burn severity (response)	Relative differenced normalized burn ratio (RdNBR, two year interval)	(Miller and Thode 2007)
Prefire insect damage	Cumulative prefire vegetation change due to insect activity from Landsat time series (NBR)	(Meigs *et al* 2015b)
Prefire insect duration	Count of years with prefire insect activity from Landsat time series (y)	(Meigs *et al* 2015b)
Prefire biomass[a]	Prefire tree biomass from imputation mapping (kg ha^{-1})	
Aspect[b]	Cosine transformed aspect (°)	
Elevation[b]	Elevation (m)	
Slope[b]	Slope steepness (%)	
Topographic position index (150 m)[b]	Difference between a pixel's elevation and the mean elevation of pixels within 150 m	
Topographic position index (450 m)[b]	Difference between a pixel's elevation and the mean elevation of pixels within 450 m	

[a] Annual biomass maps were derived from Landsat time series and nearest neighbor imputation with forest inventory data as part of a regional analysis of carbon trajectories (http://lemma.forestry.oregonstate.edu/projects/cmonster).

[b] Topographic variables derived from 30 m digital elevation model.

Table 2. List of population-level predictor variables used to assess drivers of insect effects on burn severity across all fire events affected by prior mountain pine beetle or western spruce budworm.

Variable	Description	Source
Insect type	Mountain pine beetle (bark beetle) or western spruce budworm (defoliator)	(Meigs *et al* 2015b)
Time since outbreak	Time since onset of insect outbreak according to Landsat time series (y)	(Meigs *et al* 2015b)
Area affected by insect	Area of fire extent affected by prior mountain pine beetle or western spruce budworm according to Landsat time series (cumulative %)	(Meigs *et al* 2015b)
Fire size	Extent of fire event (ha)	http://mtbs.gov
Fire season	Day of year of fire ignition	http://mtbs.gov
Interannual drought	Palmer drought severity index (mean June–August PDSI) by fire year and state	http://www.wrcc.dri.edu/wwdt/time/

(Thompson *et al* 2007, Dillon *et al* 2011, Prichard and Kennedy 2014). Unlike these spatially static covariates, fire weather is a dynamic variable that needs to match SAR model resolution in both space and time. Recent advances in the development of gridded meteorological data (e.g., Abatzoglou 2013) have great potential for such analysis but must be combined with accurate fire progression maps to assign fire weather conditions to each pixel for the day it burned. Because consistent fire progression maps are a recent development in North American wildfire monitoring, they are not available for most fires in our census, precluding the use of fire weather covariates in our SAR analyses. Nevertheless, a major strength of SAR is that the spatial error term captures unmeasured but spatially structured variables at the pixel scale (Haining 1993, Wimberly *et al* 2009), including fire weather.

To evaluate key drivers of insect–fire effects at the population level (Objective 2), we assessed the distribution of SAR regression coefficients derived for each fire event with a set of predictor variables not included in the SAR models (table 2). Because the large variability and range of the independent SAR variables precluded direct comparison across model coefficients, we first standardized the coefficients by calculating *z*-scores based on the standard deviation of the mean across all SAR models. We then investigated whether insect effects on burn severity (*z*-scores of prefire insect damage coefficients) varied with insect type (MPB versus WSB), time since outbreak, total area affected by prior insect outbreaks (%), fire size (total extent), fire season (inferred from fire ignition date), or drought condition of each fire year (Palmer drought severity index; PDSI). We derived these predictor variables from the insect and fire census data described above, with the exception of state-level PDSI values (available online: http://www.wrcc.dri.edu/wwdt/time/), which we assigned to each fire, averaging June–August after Heyerdahl *et al* (2008). We estimated time since onset of insect outbreak at the fire event scale as the majority year of first detection in the Landsat-based insect atlas (Meigs *et al* 2015b), recognizing that actual insect activity begins one year before vegetation changes are detected (Meigs *et al* 2015a) and that outbreak initiation varies within a given fire perimeter, depending on outbreak and fire extent. Finally, we computed linear models to assess univariate relationships between these population-level predictors and the insect–fire coefficients.

We evaluated uncertainty in the SAR models for each fire event as well as the distribution of model accuracy across all fire events (table S1). Specifically, we

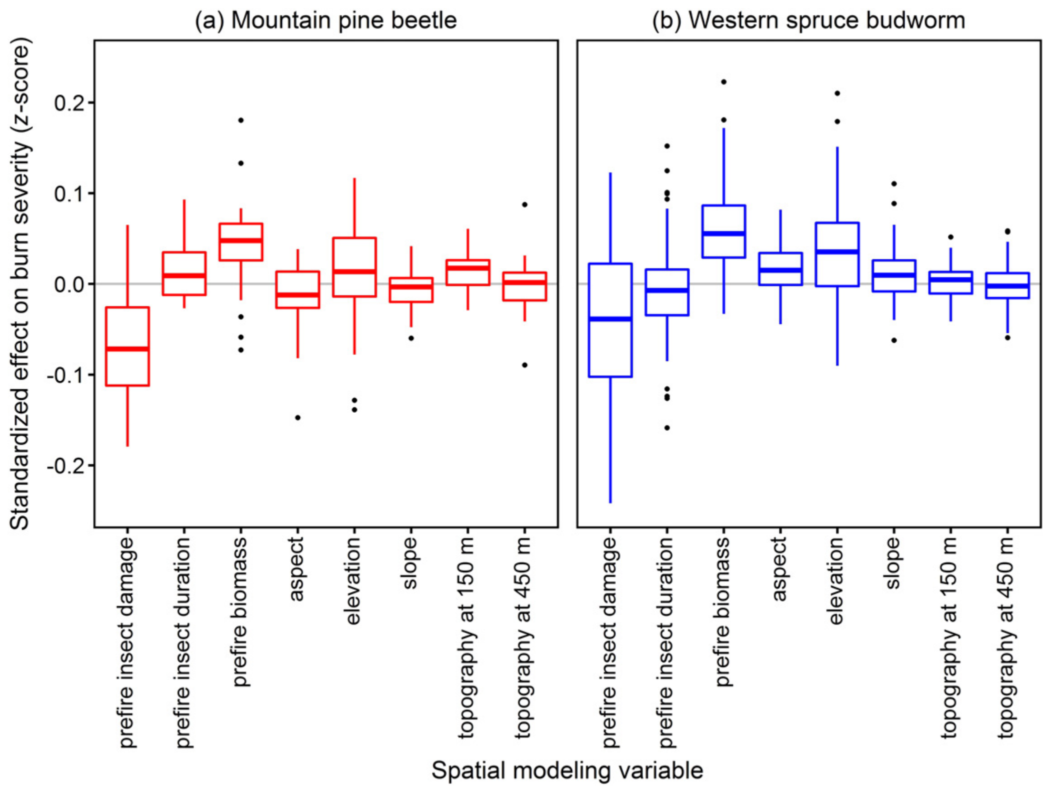

Figure 2. Effect of insect, vegetation, and topography drivers on remotely sensed burn severity across fire events with prior mountain pine beetle (a, $n = 19$) or western spruce budworm (b, $n = 62$). The standardized z-scores from spatial modeling (sequential autoregression; SAR) of each large fire event indicate that burn severity (RdNBR spectral index) is lower in forests with higher cumulative prefire insect damage and lower prefire biomass. The SAR models also account for prefire insect outbreak duration and five potential topographic drivers of severity (table 1). The box and whisker plots show the median as a horizontal line, 25th and 75th percentiles as box edges, 1.5 times the interquartile range as whiskers, and outliers as points.

graphed SAR model coefficients of determination (R^2) by insect type and across the same key predictor variables used in the population-level analysis. Recognizing additional uncertainties inherent to these spatial datasets, we emphasize general patterns across the regional census and the relative effects of insect outbreaks. For example, because the insect outbreak year is offset by one year and uncertain for any given pixel within a fire perimeter, we focus on the relative time since insect outbreak across all fires rather than the specific time lag for a given fire event.

3. Results

Our census of recent insect–fire events across Pacific Northwest forests reveals that, after accounting for prefire biomass and topography, burn severity is generally lower in forests with higher cumulative prefire insect damage (figure 2). Notably, this negative effect of prior insect damage on burn severity is strong enough to emerge without directly accounting for weather conditions at the time of burning.

Following both MPB and WSB outbreaks, burn severity is lower across most time lags (figure 3). The two insects exhibit divergent temporal trajectories, however, revealing differential insect effects on tree mortality, vegetation response, and associated fuel dynamics. Specifically, whereas burn severity decreases with time following MPB outbreaks (figure 3(a)), severity increases with time following WSB outbreaks, eventually recovering to a neutral effect within 20 years (figure 3(b)).

In addition, insect effects on burn severity do not depend on the other population-level predictor variables. Specifically, the insect–fire coefficients are not associated with the proportion of fire extent affected by insects (%), total fire extent (ha), fire season (ignition date), or interannual drought condition (PDSI) for either insect species (figures S1–S4). This lack of association underscores the importance of time-since-outbreak as an emergent predictor of fine-scale insect effects on burn severity (figure 3).

In general, SAR model accuracy is high (MPB mean $R^2 = 0.64$; WSB mean $R^2 = 0.72$; table S1), indicating that the insect, vegetation, and topography variables—as well as the inherent spatial patterning represented by the spatial error term—explain a large proportion of variation in estimated burn severity. In addition, the coefficients of determination are generally evenly distributed across the regional predictor

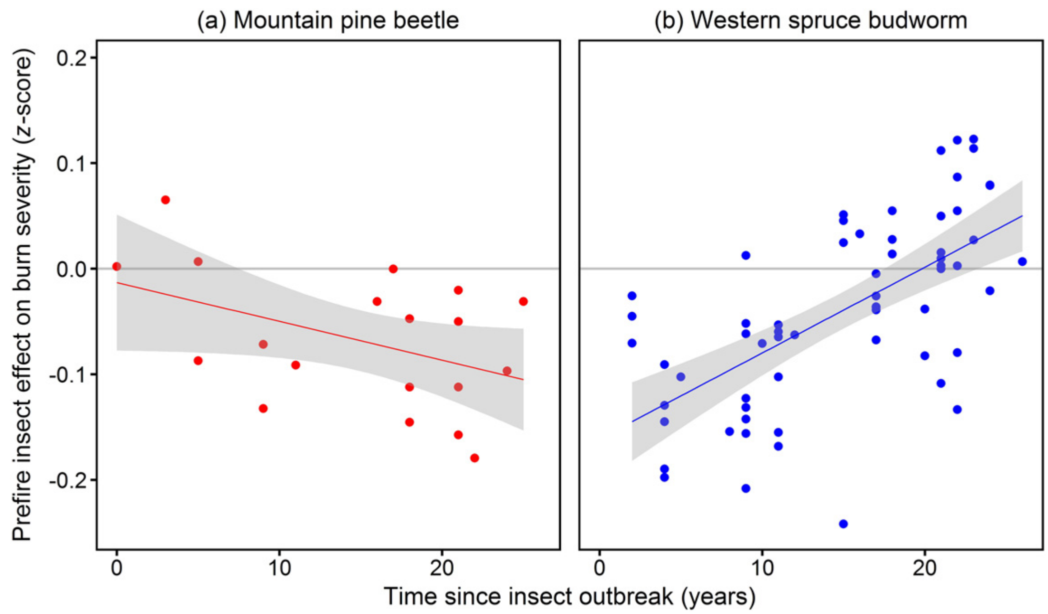

Figure 3. Relationship between prefire insect effect on burn severity (RdNBR spectral index) and years since onset of insect outbreak across fire events (individual points) with prior mountain pine beetle (a) or western spruce budworm (b). For both insects, burn severity is lower in forests with higher prefire insect damage across most time lags, although the two insects exhibit divergent trajectories. Whereas burn severity decreases with time following mountain pine beetle outbreaks ($R^2 = 0.19, P = 0.06, n = 19$), severity increases with time following western spruce budworm outbreaks ($R^2 = 0.40, P < 0.0001, n = 62$), recovering to a neutral effect within 20 years.

variables, which encompass a broad range of insect, fire, and drought conditions (figures S5–S9). Finally, other recognized drivers of burn severity that we did not model explicitly, particularly fire weather and firefighting response at the event scale, contribute to the spatially autocorrelated variance captured indirectly by the SAR spatial error term associated with each fire.

4. Discussion

By quantifying the fine-scale effects of insect outbreaks on burn severity within all large insect–fire events across a heterogeneous forest region, this study demonstrates a general pattern of lower burn severity following outbreaks of both bark beetles and defoliators, in contrast to recent findings that burn severity is either unaffected by or weakly positively associated with outbreak severity (e.g., Crickmore 2011, Harvey *et al* 2013, 2014a, 2014b, Prichard and Kennedy 2014). We suggest that higher severity insect outbreaks reduce the abundance of live vegetation susceptible to wildfire while altering vertical and horizontal fuel distributions, particularly as trees defoliate, die, and transition from canopy to surface fuels (Hummel and Agee 2003, Simard *et al* 2011, Hicke *et al* 2012, Cohn *et al* 2014, Harvey *et al* 2014a).

In the case of MPB, this forest thinning effect results in a lasting reduction of fire impacts on residual vegetation (figure 3(a)). Moreover, the continuing decline in post-beetle burn severity indicates that the thinning effect may persist until vegetation and fuel distributions recover to pre-insect conditions. Because there were relatively few fire events within the first few years following MPB outbreak in our census (figure 3(a)), future studies should continue to investigate the transient yet highly flammable red stage of outbreak (Jolly *et al* 2012). Nevertheless, our finding of generally lower burn severity in forests affected by MPB outbreaks—as well as the relative rarity of red-stage fire events in recent decades despite major beetle outbreaks in the study region (Meigs *et al* 2015b)—highlights the need for discretion in forest and fuel management following beetle outbreaks.

In the case of WSB defoliation, lower initial burn severity is consistent with reduced potential fire behavior and effects due to fine-scale canopy thinning and mortality dynamics (Cohn *et al* 2014). The relatively rapid increase of the budworm-fire coefficient with time (figure 3(b)) indicates that the thinning effect on fuel profiles is less persistent for the defoliator (WSB) than for the bark beetle (MPB). In addition to relatively lower per-unit-area tree mortality impacts (Meigs *et al* 2011), WSB affects host forests that are more productive than those affected by MPB in the study region (Meigs *et al* 2015b), leading to more rapid accumulation of live overstory and understory vegetation. Thus, as time elapses following WSB outbreaks, fuel density and connectivity likely increase in multiple strata, including dead surface fuels (Hummel and Agee 2003) and total live biomass, the latter of which is associated with higher burn severity (figure 2). The potentially synergistic budworm-fire effects in older outbreaks have important implications for current forest management in the US Pacific Northwest, where regional WSB outbreaks peaked 25–30 years ago, exceeding recent MPB outbreaks in cumulative extent and impacts (Meigs *et al* 2015b).

Very few studies to date have assessed post-insect burn severity in an empirical, spatially explicit manner, and our census of numerous large fire events occurring up to 26 years following bark beetle and defoliator outbreaks provides a broader context for assessments of specific insect outbreaks, wildfires, locations, and time lags. In so doing, our analysis demonstrates generally negative feedbacks, in comparison with the neutral or relatively transient positive effects quantified with field observations in wildfires occurring up to 15 years following MPB outbreaks in Northern Rocky Mountain forests (Harvey *et al* 2014a, 2014b). In addition, our results differ from the positive MPB-fire feedbacks identified via SAR for the 2006 Tripod Fire Complex in northern Washington (Prichard and Kennedy 2014). Finally, analyses of fire effects following WSB defoliation have been especially rare. The post-budworm temporal trend suggests a neutral effect ca. 18–23 years post-outbreak (figure 3(b)), consistent with the lack of association between budworm damage and the severity of the 2003 B&B Fire Complex in central Oregon (18 years post-outbreak (Crickmore 2011)).

Our core finding that insect outbreaks actually dampen wildfire severity across numerous large insect–fire events has direct applications to natural resources management. Specifically, policies based on the assumption that recent insect outbreaks increase the hazard of subsequent wildfires might be unjustified (Hart *et al* 2015). Furthermore, given that insects also can reduce wildfire likelihood (Lynch and Moorcroft 2008, Meigs *et al* 2015a), these findings illustrate the role that a biotic disturbance (i.e., insect outbreak) can play in limiting both the occurrence and impacts of an abiotic disturbance (i.e., wildfire). Because bark beetle and defoliator effects on burn severity appear to diverge over time, however, forest management strategies should recognize the differential and dynamic effects of each insect on fuel conditions and associated fire potential.

Although our regional census reveals negative insect effects on burn severity across a range of conditions that has not been assessed to date, numerous uncertainties and research questions remain, particularly regarding the mechanistic linkages among insects, fuels, and other known drivers of fire behavior and effects. Specifically, our inference is limited to the locations and years captured by the available spatial datasets, and future studies could investigate insect–fire severity relationships over broader spatiotemporal scales. Future studies also could combine our spatially extensive methods with the temporally rich insights provided by tree ring analysis (e.g., Flower *et al* 2014). Such a fusion approach would enable forest researchers and managers to determine whether recent insect and fire patterns represent a departure from historic disturbance regimes. In addition, because our census uses remotely sensed relative spectral change (RdNBR) as a proxy for fire effects, we cannot directly address causal relationships, fine-scale ecological impacts and responses (e.g., soil heating, tree regeneration), fire behavior (e.g., fire intensity, crowning), or operational fire management (e.g., firefighter safety, suppression tactics) (Thompson *et al* 2007, Harvey *et al* 2014b, Jenkins *et al* 2014, Hart *et al* 2015). Moreover, although the SAR spatial error term indirectly captures the effects of missing variables (Haining 1993, Wimberly *et al* 2009), future studies could explicitly address the effects of other key drivers like fire weather on a subset of events where fine-scale, consistent, and accurate weather and fire progression data are available (e.g., Harvey *et al* 2014b, Prichard and Kennedy 2014). Similarly, topography and climate are known drivers of burn severity in the western US (Dillon *et al* 2011), and future research could further investigate the generally positive association between elevation and burn severity in our SAR modeling (figure 2) and lack of association between drought and insect–fire effects across this census, which spans a range of drought conditions (figure S4). Finally, our analysis is limited to the relatively rare events where wildfires occur within the initial decades following insect outbreaks, and future studies should continue to evaluate the pervasive ecological and economic impacts of these and other disturbance agents separately (e.g., Westerling *et al* 2006, Kurz *et al* 2008, Hicke *et al* 2013).

5. Conclusion

Contrary to common assumptions of positive feedbacks, recent forest insect outbreaks actually dampen subsequent burn severity at multiple time lags across the US Pacific Northwest. Indeed, by altering forest structure and composition from forest stand to regional scales (Raffa *et al* 2008, Flower *et al* 2014, Meigs *et al* 2015b), these native insects contribute to landscape-scale heterogeneity, potentially enhancing forest resistance and resilience to wildfire. Because insect outbreaks do not necessarily increase the severity of subsequent wildfires, we suggest a precautionary approach when designing and implementing forest management policies aimed at reducing wildfire hazard in insect-altered forests.

In addition, by dampening subsequent burn severity, insect outbreaks could buffer rather than exacerbate some fire regime changes expected due to global change (e.g., climate warming, drought, invasive species (Littell *et al* 2010, Ayres *et al* 2014)) and forest response to land use (e.g., fire exclusion, timber harvest, livestock grazing (Hessburg *et al* 2000)). However, each of the disturbances assessed here (bark beetle, defoliator, wildfire) influences more forest area separately than in combination (Meigs *et al* 2015a), and it will remain a high priority to monitor and adaptively manage their individual impacts on forest health and ecosystem services. Given projected increases in the activity of both wildfires and insects (Raffa *et al* 2008, Bentz *et al* 2010, Littell *et al* 2010), the potential for disturbance interactions will continue to increase, as will the potential for ecological surprises like the negative feedbacks apparent in this census.

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Learning to coexist with wildfire

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The impacts of escalating wildfire in many regions — the lives and homes lost, the expense of suppression and the damage to ecosystem services — necessitate a more sustainable coexistence with wildfire. Climate change and continued development on fire-prone landscapes will only compound current problems. Emerging strategies for managing ecosystems and mitigating risks to human communities provide some hope, although greater recognition of their inherent variation and links is crucial. Without a more integrated framework, fire will never operate as a natural ecosystem process, and the impact on society will continue to grow. A more coordinated approach to risk management and land-use planning in these coupled systems is needed.

Fire is unique among the natural hazards that affect human communities and the ecosystems on which we depend[1]. Although humans sometimes intentionally ignite and manage fires, our main focus is on fighting them. For other natural hazards, such as earthquakes, hurricanes and floods, there is much more emphasis on identifying vulnerabilities and adaptations. The 'command and control' approach[2] typically used in fire management neglects the fundamental role that fire regimes have in sustaining biodiversity and key ecosystem services[3–6]. Unless people view and plan for fire as an inevitable and natural process, it will continue to have serious consequences for both social and ecological systems.

Over the past two decades, wildfires around the world have increasingly affected human values (for example, lives, views or sacred environments) and assets (for example, damage to homes or public infrastructure) and ecosystem services (for example, air quality and long-term carbon storage). The growing list of negative outcomes and their financial effects have complex causes and consequences[7]. The natural range of fire sizes and resultant frequencies, timings and intensities — the 'fire regime' — varies greatly among ecosystems, as do the ways in which human activities have altered them (for example, through timber harvesting, fire suppression, urban or agricultural encroachment, novel ignition patterns and invasive species). Not surprisingly, policy strategies to address wildfires often emphasize fuel reduction[8,9]. However, even where strategies recognize interacting cultural, environmental and economic dimensions of wildfire[10–12], few tackle the difficult land-use issue of where and how humans choose to build their communities in the first place. The prospect of widely increasing fire activity with climate change[13] intensifies the need for a new path forward.

Viewing fire-related problems in the context of coupled socioecological systems (SESs)[14], which explicitly recognize links between humans and their natural environments, provides insights into achieving a more sustainable coexistence with wildfire. We have learned a great deal about fire as an essential ecosystem process and the human dimensions of living on fire-prone landscapes. Synthesis of this knowledge through a coupled systems approach can highlight specific vulnerabilities and trade-offs, and facilitate adaptation strategies across widely varying public and private landscapes (Fig. 1). In this Review, we summarize research on fire-prone ecosystems and fire effects on human communities through the lens of SESs, identify links in these coupled systems, and discuss recommendations for greater resilience. We emphasize insights from three regions (Fig. 2) where major fire-related losses have occurred in recent decades: the Mediterranean basin, the western United States and Australia.

Socioecological systems and fire

Sustainable solutions to most environmental problems will be impossible if the links and interdependencies between humans and ecosystems are ignored[14]. In the context of wildfire, the most well-developed SES research that incorporates this coupling concerns climate-change effects on Alaskan boreal forest ecosystems and rural indigenous communities[15,16]. Case studies in rural communities of New Zealand[17] and California[18] also exist. Remarkably, a coupled wildfire SES framework has yet to be adopted for the more densely developed wildland–urban interface (WUI; area in which communities intermix with or abut natural vegetation), where most of the human fatalities, home losses and fire-suppression expenditures occur.

The complexity of how wildfire operates in different ecosystems and how humans interact with it indicates that place-based hazards and risks should be addressed as a coupled SES[16,19]. Reframing the problem to minimize harmful effects as the climate changes and humans increasingly inhabit fire-prone landscapes identifies an integrated set of coupled SES linkages (Fig. 1). Importantly, this allows us to recognize how the geographic context of the coupling itself contributes to impacts and losses of assets throughout the wildfire SES. Local characteristics of the WUI, and the components on either side of it, will largely determine the degree to which fire may be accommodated and how communities will be affected. The spatial scale of the coupling may also be broad in some cases, such as when fires compromise recreation values (for example, trail access, camping facilities or fishing habitat) and water supplies of distant urbanized areas, or when concerns over human exposure to drifting smoke influence management decisions about fires that are burning relatively far away. Although this framing does not intrinsically address connections between fire and global-scale climate change mitigation[13,15,20], it helps to

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reveal geographically relevant solutions for decreasing harmful effects and increasing the positive benefits of fire on the landscape. The institutional complexity that underlies many aspects of this coupled SES framework — agency mandates, property rights, building ordinances, indigenous governance, economic subsidies and political pressures — will also feed into a particular set of solutions, often creating challenging constraints.

Sustainable coexistence with wildfire is both a process and a long-term goal, such that policy, planning and management are adapted and refined through time (Fig. 1). Responsibility must be shared between governments and the people at risk, and the approach integrates building, planning, fuel management, suppression capability, and knowledge of fire and ecosystem dynamics at different scales. Coexistence with wildfire should ultimately allow ecologically appropriate fire regimes to operate on landscapes near and far from the WUI, with relatively low risks to people, property and resources, while also allowing us to enjoy ecosystem services enhanced by fire (for example, habitat maintenance, potential hazard reduction, natural hydrologic functioning, and carbon and nutrient cycling). This outcome should also reduce the costs of fire suppression and the need to put firefighters at risk.

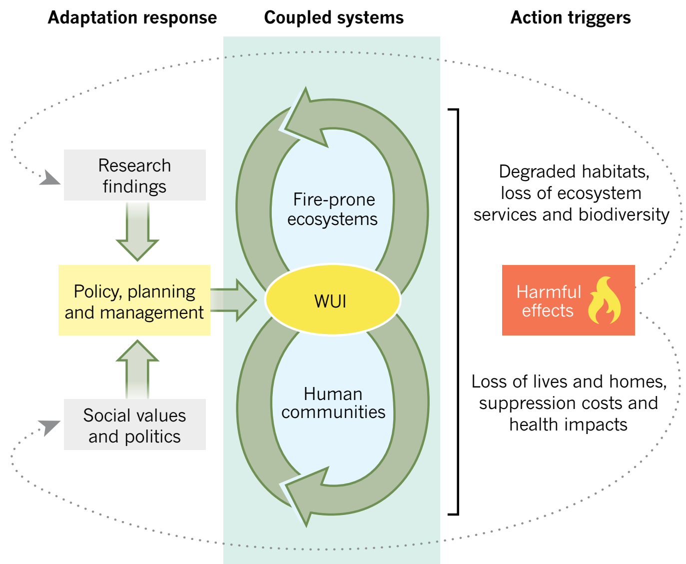


Figure 1 | Links and pathways to resilience in coupled socioecological systems affected by fire. Coexistence with wildfire is strongly influenced by the type of natural fire regimes that operate on a given landscape, and the degree to which communities can reduce exposure and vulnerabilities there. The wildland–urban interface (WUI) is the spatial manifestation of the coupling, and the most proximate scale of exposure and risk mitigation. To learn from and minimize the harmful effects of fire in both the ecosystem and the community, links between systems and scales of interactions must be recognized. Doing so will trigger, through research and in response to changing social values and political context, further adaptation and change in policy, planning and management.

Fire and ecosystems

The role of fire in different ecosystems varies by the degree of current landscape modification, relative to natural or historical patterns and processes. Some regions have large expanses of semi-wilderness where maintenance or restoration of certain fire regimes is crucial to ongoing habitat characteristics or ecosystem services (for example, the western United States and Australia). Here the links between fire characteristics and ensuing ecological effects, or fire 'severity', are often emphasized. Other regions have been so completely altered for various human needs that what is 'natural' is no longer a clear consideration (for example, the Mediterranean basin). Furthermore, climatic controls on fire regimes (for example, frequency of droughts or high-wind events, or length of fire season) tend to dominate in some ecosystems, whereas local controls (for example, topography, fuel loads and ignitions) strongly influence others. Fire resilience is thus context-dependent, varying with the biophysical environment and desired future conditions. Accordingly, our capacity to avoid ecosystem degradation and catastrophic shifts[21] (Fig. 1) depends on the ecosystem in question and how climate change will manifest there.

Mediterranean basin

Mediterranean landscapes are mosaics of various shrublands and oak- and pine-dominated woodlands intermixed with extensive pastures, cultivated lands and abandoned agricultural fields[22]. Despite fire's ecological influence there[4], no reference conditions exist for fire management or restoration, and traditional use of fire for rangeland and game management has strongly influenced historical landscape dynamics[23]. Pronounced biophysical and land-use gradients have recently resulted in contrasting fire and vegetation dynamics. The southern and eastern regions are subject to land over-exploitation and reduction in vegetation cover that increases the risk of desertification and loss of ecosystem services. By contrast, socioeconomic drivers are increasing fire hazards and losses over Mediterranean Europe (northern region) owing to rural depopulation, increased WUI exposure and land-cover changes that are sometimes promoted through afforestation policies[24]. Most shrublands and woodlands in the northern region are becoming dense enough to support climate-driven high-intensity 'crown' fires[22,25].

Wildfire in European Union countries is addressed in national and regional forest policy plans, but consensus on fire and ecosystem management is lacking. In spite of large expenditures, increased preparedness and greater firefighting abilities, extreme fire-weather conditions have caused devastating fires in several Mediterranean countries[26]. A new framework to regulate and promote traditional fire practices, accommodating diverse territorial contexts and operational use of fire, has thus been advocated[27]. Currently limited to local management, prescribed burning is increasing across Europe as a tool that aims to reduce fuel loads and diminish the risk of high-intensity fires[28]. Modest changes to regional and national wildfire policies have therefore included long-term preventive actions, but fire management is still primarily centred on short-term fuel- and suppression-oriented measures[8]. There are concerns over the ecological consequences of recent fire patterns[29], but human-centred fire exclusion generally prevails on most Mediterranean-basin landscapes.

Western United States

Fire management in many western US ecosystems is informed by research on the historical role of fire[30], especially through dendrochronology[31] and landscape reconstructions[32]. Before modern management, different types of fire occurred among vegetation types and maintained important natural structures and functions, with great variation geographically[5,32–35].

In western US forests, high-severity fires that kill overstory trees are typical of cool, high-elevation, subalpine environments[36,37]. Although severe fires may seem catastrophic from a human perspective, in these forests they stimulate vegetation regeneration, promote landscape diversity in terms of vegetation types, provide habitat for many species and sustain other ecosystem services[5]. The many organisms and propagules that may survive the fire, combined with heterogeneity in age, structure and species composition across landscapes, confer resilience against shifts to non-forest types. High-severity fires predominate across about 30% of western US forests, naturally mixing with low-severity fires through time and space across another ~45%[36]. Key regional controls of high-severity fire regimes are extreme drought and high winds[37], and local (for example, topographic) influences on severity patterns can emerge during less dry conditions[38]. Fuels tend to be naturally abundant in these ecosystems, so modern fire suppression may have decreased historical levels of landscape fragmentation, but it has not increased fuel loads[5,39].

By contrast, many dry and mesic, low-elevation and mid-montane forests historically experienced more frequent low-severity fires that maintained relatively open forest structures of fire-resistant

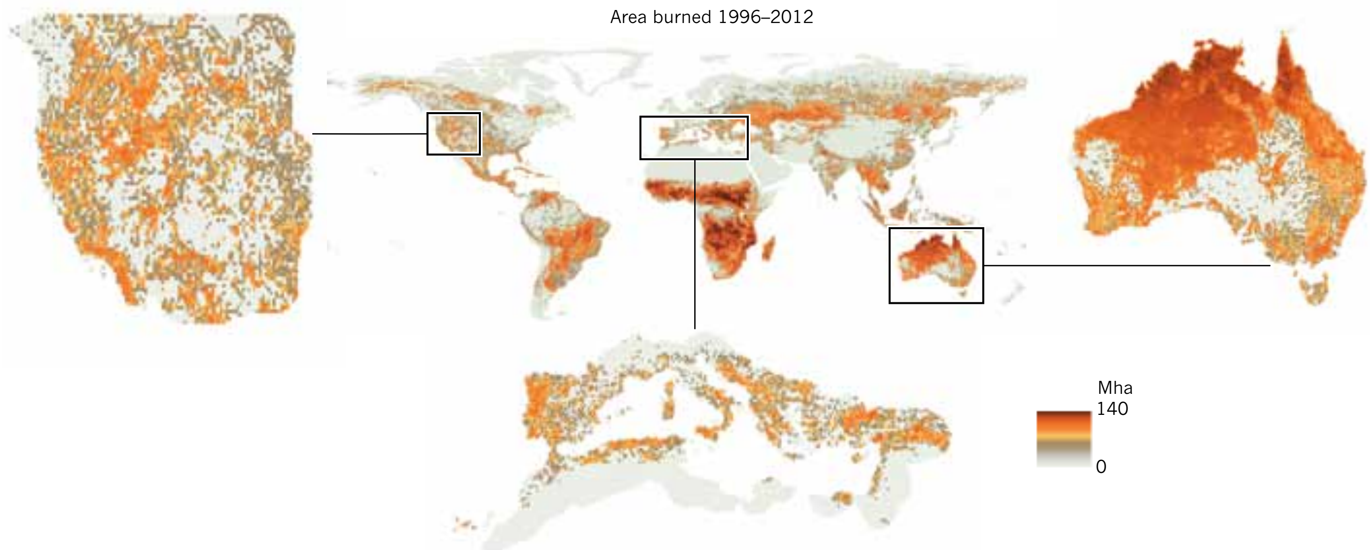


Figure 2 | Area burned patterns and locations of fire-prone regions. The cumulative area burned between 1996 and 2012 in millions of hectares (Mha) per mapped cell. The western US region consists of the 11 western states in the conterminous United States (left), the Mediterranean basin (middle) contains the Mediterranean-climate biomes and the Australian region (right) encompasses the entire continent (see Supplementary Information).

trees[33,34,40], across about 25% of western US forests[36]. Ignition patterns, vegetation structure and fuel amount exert a strong control on regimes of frequent low-severity fire, making them more sensitive to modern human perturbations and also more amenable to fuel-management techniques[33,39–41]. Unlike high-severity fire regimes, timber harvesting and decades of fire suppression in drier forests have lengthened intervals, increased densities of smaller trees and shifted regimes of mostly low-severity fires to include more high-severity, stand-replacing fires. The extent to which this has happened is a topic of debate, raising questions about how widespread 'mixed severity' fire regimes were prehistorically[32,35,42]. Regardless, reducing accumulated fuels in these forests is often a high management priority. Only where such departures from natural fire regimes have led to denser, multilayered, fire-intolerant forests, however, may fuel-reduction treatments restore more characteristic forest structure and function (Box 1).

There is a general consensus regarding the importance of fire, including the need for prescribed burning, to maintain native grasslands and open woodlands. Woody plant encroachment in many ecosystems with sparse tree cover, driven by a lack of fire and replacement of native herbivores, has reduced plant biodiversity, altered vegetation structure and threatened the fauna that depend on those habitats[43,44]. Fire also plays a crucial part in regeneration for some of the vast shrublands of the western United States, especially California's densely urbanized chaparral ecosystems. Similar to high-elevation forests, fire in chaparral is stand-replacing and under strong climatic control (patterns of drought and extreme fire weather)[45], meaning that fuel-reduction efforts have limited effect except in strategic locations[46,47]. Increased fire frequencies, due to abundant human ignitions and non-native grasses that support rapid reburning, threaten to convert many native shrublands to degraded habitats[48]. Invasive grasses also cause very frequent and often large fires across parts of the Great Basin in the western United States[34,49], driven by the 'grass-fire cycle' positive feedback[50] and bringing serious management challenges even to fire-sensitive desert ecosystems[51].

Australia

Fire is ubiquitous in Australian ecosystems, including deserts and tropical forests, and a wide range of fire regimes have been mapped using remote sensing[52]. Annual pulses of relatively intense fire dominate the extensive savannahs of northern Australia, with less frequent, massive fires in the arid zone occurring after above-average rainfall[53]. By contrast, large fires in the temperate forests of the south, although intense, are less extensive and also less regular (decadal occurrence). Biophysical models of fire-regime controls[54] and analysis of trade-offs in fuel characteristics and fire types[52] confirm the primary role of climate, especially the gradient in summer monsoonal precipitation. Thus, fire frequencies tend to vary with latitude, decreasing towards the south and especially the arid interior. Most fire activity on the Australian continent is in grass fuels and of relatively low intensity.

Although palaeo-charcoal deposits document fire's very long history in Australia[55], fine-scale understanding of fire-regime variability through dendrochronology is generally lacking, hindering detailed perspectives on long-term variations in fire regimes. Comprehensive fire management initiatives focus on key environmental objectives, such as biodiversity conservation[20] and emissions reduction[56], as a function of local context. Maintenance of contemporary fire regimes for biodiversity conservation is a priority in most regions, as opposed to the emphasis on restoration that dominates western US approaches.

Australia's productive eucalyptus forests, which can burn at very high intensities and low–moderate frequencies, are largely restricted to southern and eastern edges of the continent. Although these forests are characteristically Australian, their proximity to urbanized areas has probably fed the continent's reputation for high-intensity fire events (see 'Where do people live?'). Debates over the degree to which fuel reduction, whether by mechanical or prescribed fire treatment, can alter the probabilities of high-intensity events[57,58] are similar to those that occur for western US forests.

Prescribed burning in Australia is extensive, but controversial. Fuel reduction burning can partially reduce risk to human life and economic assets, although trade-offs with risks to environmental assets such as biodiversity and ecosystem services are not well understood[3,59]. However, functional responses of species to fire frequencies, sizes, timings and intensities provide a measurable basis for predicting how ecological diversity will respond to management and climate change[60,61].

Resilience and climate change

Ecosystem managers in the three regions covered here (Fig. 2) may have limited ability to alter the numbers, sizes and characteristics of fires occurring in different ecosystems[5,34,39,59]. As already discussed, this is because coarse-scale climatic influences tend to control fire regimes in many ecosystems, especially those that are naturally prone to large and high-severity fires. Except under the most extreme conditions, fire regimes typically constrained by more local-scale controls, such as ignition frequencies and biomass accumulation rates, may respond more strongly to prescribed fire and mechanical fuel reductions. This characterization of two opposing types of fire regimes is, however, a vast over-simplification — idealized end points along a spectrum of variation within and between fire-prone ecosystems[62] — and management prescriptions need to somehow accommodate such complexity. Furthermore, fire-related sensitivities and responses vary among plant and animal species, so fire management for the persistence of one important group of organisms may not favour that of the others.

The potential for climate change to cause 'novel' or 'no analogue' environmental conditions in some ecosystems presents new challenges for management, policy and planning. An obvious goal is to have ongoing fire regimes that minimize the risk of biodiversity loss[59]. Yet, what adaptation responses are appropriate (Fig. 1) if we do not know how future climates and related biophysical processes will differ from the recent past? These uncertainties have resulted in somewhat similar recommendations about fire and ecosystem resilience[63–65]. Heterogeneity in vegetation types, stand structures and successional age classes at all spatial scales and environmental settings is emerging as a strategy for enhancing ecosystem resilience to climate change. This essentially facilitates diverse initial conditions for multiple future ecological trajectories, the most likely and successful of which will not be known for decades. The role of diverse topography in creating microclimate refugia, or 'holdouts'[66], as well as in influencing fire sizes and severity characteristics within large fires[38,67], comprises the physical template for resilience in more mountainous regions. In ecosystems with a recent paucity of burning, fire management that fosters burning under diverse conditions may be useful for achieving this desired heterogeneity and reducing fuel accumulations[41]. Not all fire-generated heterogeneity is ecologically significant, however, so understanding the effects of specific types of 'pyrodiversity' is important[68].

Where do people live?

The WUI is the most proximate spatial manifestation of the coupling in a wildfire SES (Fig. 1). Understanding and addressing vulnerabilities related to the WUI in fire-prone areas is therefore crucial to long-term solutions. As distances between urbanized areas and those protected from development decrease globally[69], a growing WUI will expand the scope of coupling in wildfire SESs worldwide. Negative fire effects that were once due to 'distant' fires (for example, the impacts of smoke on human health) will be increasingly common, making coexistence with wildfire much more challenging.

The current WUI of the western United States is relatively well characterized, with over 60% expansion since 1970 (ref. 70) and about 70% in private ownership[71]. The WUI in this region also predominantly occurs where fire severities are high[70]. Only 14% of private land in the western US WUI is developed, so substantial increases in human exposure to fire may occur as the remaining portions become populated[72]. Although less well characterized, there is growing awareness of expanding WUI in Mediterranean Europe[24,73,74] and Australia[19,75].

Global systematic analyses of human settlement in fire-prone environments is important, but lacking[76]. Coarse-scale characterization of how population densities relate to various fire-prone environments (Fig. 3) provides some insight. Although often characterized as a 'forest fire' problem, western US patterns indicate that highly fire-prone locations with large numbers of people tend to be associated with sparse or no tree cover (for example, the chaparral shrublands of southern California); locations with both high population densities and denser forests exhibit the least area burned (Fig. 3, left). Australia exhibits greater area burned over a broader range of environments, with intermediate population densities being more fire-prone regardless of the amount of forest cover (Fig. 3, middle). The Mediterranean basin is unique because the greatest area burned coincides with the highest population densities (Fig. 3, right), although this too occurs in locations with relatively low forest cover (for example, abandoned agricultural lands[26]).

Acknowledging the diversity of the fire-prone environments and vegetation types where people live is important, because it has implications for the types of fuel treatments that may or may not work to mitigate fire hazards within or near the WUI, and it could help to guide future resource allocation decisions (for example, among vegetation removal, evacuation planning and home vulnerability retrofits)[77]. Awareness of the institutional and social diversity of different human communities is also important, as we discuss in the next section, because it influences their capacity for preparation and mitigation of hazards such as wildfires[18].

Fire and human communities

This section reviews research on how fires affect human communities and is organized by the scale of coupling in a wildfire SES (Fig. 1), ranging from individuals to landscapes. Social science research on wildfire, primarily undertaken in Australia and the United States,

BOX 1

What can 'thinning' of fuels achieve?

There is intense pressure on land-management agencies to reduce fire hazards (for example, rates of spread or flame lengths if a fire occurs). Treatments should be prioritized, however, where they may help to protect communities or reduce fuel loads in the areas that are most likely to experience uncharacteristically severe burns[36,71]. Mechanical fuel-reduction treatments are most suited to certain dry and fire-prone mesic forests[34,39–41,77], where thinning the density of smaller understory trees and removing surface fuel residues (non-merchantable tree tops and limbs) created by these treatments can reduce fire intensities and rates of spread[40]. Not treating the additional surface-fuel by-products can actually increase fire intensity and severity when a wildfire does occur[41].

Some of the most basic trade-offs that limit the widespread use of mechanical fuel reductions involve their economic viability. Often, larger commercial trees will be harvested to help offset operational costs, but this typically generates more surface-fuel residues. Moreover, opening up the overstory canopy and increasing sunlight penetration can increase growth of highly flammable understory vegetation. Controlling this growth response is an ongoing endeavour, the economic feasibility of which is unknown.

Uncertainty about when and where treatments might actually perform as desired must also be considered. Although there are many examples of fuel treatments reducing fire behaviour when conditions are not extreme, recently treated forests can experience a stand-replacing crown fire when wind speeds exceed 30 km h^{-1} and when fuel moisture is low[102]. When the probability of fire occurring in a particular area is relatively low, the odds of a fuel treatment influencing the behaviour of a wildfire there, within the time frame that treatments are effective, is also low[103]. The degree of protection provided by a particular mechanical treatment may thus depend on uncertain parameters (for example, ignition patterns and extreme wind frequencies).

In many areas, ecological restoration and fuel-management goals may be best balanced and accomplished through fire[4,41], which creates natural heterogeneity and provides for fire-dependent species.

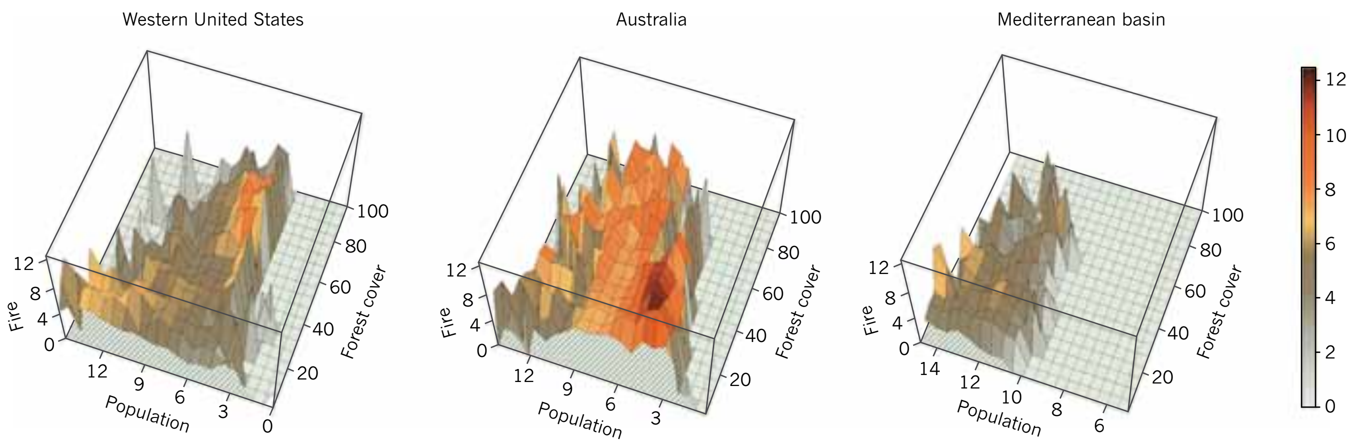


Figure 3 | Relationship between forest cover, population density and area burned in fire-prone regions. Locations with both higher human populations and greater amounts of burning tend not to be consistently characterized by high forest cover. Patterns vary greatly among regions, reflecting the different contexts in which each side of the wildfire socioecological system have intersected. (Data were aggregated from original sources (see Supplementary Information) to 0.25° resolution cells and plotted as density surfaces.) Forest cover is the percentage area covered by trees (>5 m height) per cell in 2000; population is number of people per cell (log transformed) in 2000; and fire is total area burned in hectares per cell (log transformed) between 1996 and 2012. The colour scale for fire is to help differentiate higher peaks in area burned.

is relatively sparse and not easily generalized. Work in the United States emphasizes social acceptance of techniques to mitigate fire risk (for example, fuel reduction on public and private lands) and, more recently, public response during and after fires[78]. In Australia, where many people do not evacuate during fires, risk perception, homeowner preparedness and response during fires, and community safety[79] are key areas of research. We also include studies outside the social sciences that have examined the role of vegetation and fuel treatments linked with losses and the built environment itself.

Risk perception and public response

Public response to wildfire is shaped by numerous factors, such as local context and individual personality and experience, so simple explanations for action or inaction do not exist. For instance, many researchers and managers assume that individuals do not understand fire risk. But US studies show that most people living in high-fire-risk areas understand their exposure, but there is a tenuous link between understanding risk and taking action to mitigate it; whereas recognizing risk might be necessary to consider mitigation, perceived efficacy of mitigation and resource constraints can be more influential[80]. Similarly, whereas around 80% of people in the fire risk areas of Victoria, Australia, know they are in a hazardous area[81], this does not necessarily translate to safer actions. After the devastating 2009 Black Saturday fires in Victoria, most people in high-fire-risk areas were aware of what new fire warnings meant and how to ensure their safety, but few acted on the knowledge when the highest-level warning was issued[81]. A deeper understanding of the influences on preparedness, evacuation decisions and support for hazard mitigation is needed.

Specific cultural and institutional systems affect public response to wildfire, as do psychological and social dynamics. For example, institutional structures in the United States and Australia are quite different, but key social dynamics have many similarities. In both countries, trust is a key factor shaping public support for agencies, whether they provide information or engage in fire-management activities[82]. US studies of public acceptance of prescribed fire reveal that trust in the personnel implementing the burn, along with familiarity with the practice, are associated with higher acceptance levels[83]. In terms of the US public response during fires, evacuating has long been the norm, often with mandatory evacuation orders; until Black Saturday, Australians were urged to either prepare to stay and protect their properties, or to leave early, on the basis that either option was safer than leaving late[79]. Despite this difference, the range of public behaviours in both countries is similar, with some residents leaving early, some staying to defend and a substantial number waiting to see how the situation develops. Furthermore, individual actions do not necessarily reflect a consistent response, as some household members may leave and some stay, while others go back and forth to check on property, animals or those who stay[84]. Although historically 'stay or go' seems to have worked reasonably well in Australia[79], the approach was questioned after the Black Saturday fires, as it was widely seen to have contributed to many of the 173 deaths. However, roughly half the people (around 3,000 households) in the burnt areas seemed to have stayed and defended their properties successfully and about half left, almost as the fire front was approaching. Most were satisfied with their decision and said they would do the same thing again[84]. Most also stated that they would like to be better prepared. The post-fire effort naturally concentrated on fatalities, with official advice after Black Saturday inquiries shifting to leaving early.

When the public response is to evacuate, key elements to success include environmental conditions (especially fire-weather severity), patterns of roads, neighbourhoods and topography. In Australia, public warnings have been based on a fire-weather danger scale, which was revised after Black Saturday to capture the most extreme conditions, along with altered warning messages and advice for these extremes. There is some public understanding of the reclassification, but little evidence of altered behaviour[81] or understanding that weather conditions well below the extreme level are still dangerous. Analogous fire-weather warnings are issued regularly in other parts of the world, but are not standardized and rarely trigger evacuation orders. Similar to many regions, fatalities during evacuations in the Mediterranean basin tend to occur during the most severe weather conditions, when fires have already begun and people choose to evacuate too late[85]; in addition, such extreme events seem to be on the rise[26]. A growing public safety challenge associated with evacuating people from fire-prone communities in mountainous terrain is limited road access. For example, housing densities are increasing in many WUI regions of the western United States without commensurate increases in the road network to support their evacuation[86]. Emergency planning, including preparation of structures and training for those who choose to stay or simply cannot evacuate safely[87], is thus increasingly important to the resilience of many communities in the regions reviewed here.

Structures and surrounding vegetation

To mitigate the risk of structure losses during wildfires, there is increasing evidence from many regions that it is best to focus on the house first and move outward from there[77]. Most structure losses are due to ember attack[88,89], when flaming or smoldering plant material is lofted by winds and blown inside or against the building or adjacent elements, often long before the flaming front arrives. Embers can cause structure ignition by entering through gaps as small as

2 mm[90] or accumulating outside against flammable building (or surrounding) features. Once ember ignition is addressed through structural design or retrofitting, less prevalent modes of structure loss are important, such as radiant heat and flame exposure. To address these, both building design and surrounding vegetation management are normally considered in unison[19], with the balance of these treatments being site specific. Similar to evacuation success, an understanding of the local fire-weather conditions and expected types of fires is required[91]. Hence, the building design strategy is to either consider all possible extremes and the weakest link in the system[88] or to pick a threshold level beyond which the structure may not survive. By relating these to a corresponding fire-weather severity, the occupant has the information for deciding when it is necessary to leave early. As a contingency, egress paths from the building interior to another building or area of minimal fuel could improve safety, but preparation for such a fallback is needed long before a wildfire arrives.

Vegetation reduction is most effective immediately adjacent to structures[88,92–94], as it can eliminate the most immediate sources of combustible material. Vegetation overhanging the structure[91] and ornamental plants[95] have been strongly associated with structure loss. Vegetation clearances more than about 30 m away, however, seem to provide no significant additional benefit in shrubland environments of southern California, even on steep slopes[94], reflecting an important trade-off between hazard reduction and habitat values (for organisms dependent on the vegetation removed). Although these findings may only apply to similar shrubland environments, a similar distance to heavily vegetated areas has also been identified for some forested environments, based on radiant heat exposure to structures[77,96]. In Australia, however, a distance from forest edges of more than 30 m was found to influence home losses[93], indicating that this buffer distance may vary substantially (for example, with fuels, weather and construction types). Another key reason to reduce vegetation near the home is to provide a relatively safe place to engage in structure protection, in case home owners or firefighters are present. It is notable, however, that some species of well-maintained trees (litter removed and high foliar moisture) near the home can actually provide protection, screening embers[19] and acting as a heat sink[96] for an approaching wildfire.

Landscape-scale patterns

Although fuel treatments seem to provide the greatest protection when located near human communities[19,88,93,94,97], landscape-scale characteristics of the WUI itself are important. For this reason, a long-term

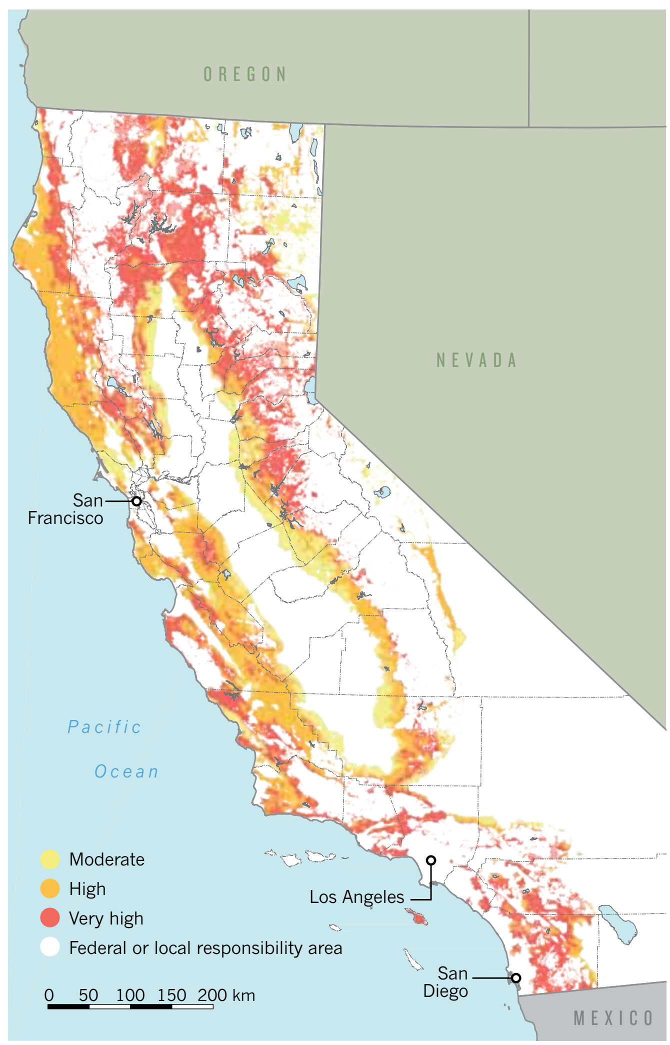
BOX 2

Adaptation measures and fire-hazard mapping

Regardless of the surrounding ecosystem conditions, all communities can better coexist with fire by taking several steps: retrofitting homes against ember attack, effectively managing fuels around homes, developing household and community plans for evacuation compared with stay-and-defend decisions, and participating in risk awareness continuing education. For existing high-hazard wildland–urban interface (WUI) areas, landowners may need to take primary responsibility for pursuing the optimal combination of adaptation measures, based on their local vulnerabilities and wildfire exposure. For development of new communities in high-hazard WUI areas, governments need to take a leadership role in planning. Regardless of responsibility, however, all of these efforts will be guided by better mapping of the fire hazard itself.

The fire hazard severity zone (FHSZ) maps (Box Fig.) of California are an official product of the state Department of Forestry and Fire Protection based on a consistent statewide methodology for estimating potential fire behaviour under a set of relatively dry and high wind conditions. Variables that affect modelled fire behaviour include local topography and potential fuel loads, although weather conditions in the current iteration of maps are not tailored to local extremes. Future updates to the FHSZ methodology will incorporate locally varying wind patterns, better reflecting conditions that cause the worst fire-related losses of lives and homes[45,98].

Fire-resistant residential construction standards are determined by the FHSZ rating of the location in question. In addition, FHSZ classifications must be disclosed at the time of home sales; although this may not deter a sale, it can affect the cost of insuring the home against fire losses. FHSZ maps are thus an incremental but important step towards treating fire like other natural hazards (for example, land-use restrictions associated with flood-plain and earthquake fault maps). Similar mapping methods and codes are produced in Victoria, Australia. Such maps do not explicitly restrict development from occurring — a constraint that should be considered in extremely hazardous locations. Comprehensive approaches should, however, help to better design communities within a complex matrix of both risk and resilience that such maps could reflect spatially. (See Supplementary Information).



approach involving land-use planning offers great potential for reducing wildfire impacts in human communities. A greater understanding is needed concerning building configuration in the WUI and how it relates to risk of losses and fatalities in various environments[73,74]. In some shrubland-dominated landscapes, the arrangement and location of homes have been the most important factors for explaining structure loss: landscape factors such as low housing density, isolated clusters of residential development and long distances to major roads are better predictors of house loss than local factors such as defensible space, fuel or terrain[94,98]. Whether these findings apply to fire-prone landscapes in general or whether there are variations between development patterns and fire regimes needs further research. Although isolated clusters of development and low housing density mean that homes are embedded within, and more exposed to, a matrix of wildland vegetation[19], ignition-prone homes that are closely spaced in neighbourhoods can also facilitate the spread of house-to-house fire, especially during extreme fire weather.

Achieving a sustainable coexistence with wildfire

A coupled SES view of wildfire highlights the variation in each half of the SES, as well as how they come together at the WUI, to create many permutations of hazards and vulnerabilities for both human and natural systems. As such, there will be different thresholds for how harmful effects trigger action before, during and after wildfires, and competing societal pressures will influence the degree to which scientific findings are able to guide adaptive responses (Fig. 1). Despite such complexity, some priorities for future work emerge from the extensive research reviewed here.

Context-specific and place-based approaches will be needed to address many existing and future coupled wildfire SES problems. This is because certain fire regimes are inherently more amenable to management activities than others, and also due to the institutional and social diversity that influences human capacity for mitigating risks to individuals and their communities. It is possible, however, that the permutations mentioned above collapse into characteristic typologies that could inform more systematic analyses. If so, are there mutually resilient combinations that are well matched or somehow compatible? Some fire regimes might dictate the degree to which evacuations should be mandatory or how resources might be allocated (for example, training homeowners to protect homes compared with fuel reduction or structure retrofits). A deeper understanding of the variation, links and scales of causes and effects in coupled wildfire SESs is therefore vital.

Governments have a primary responsibility in the long-term evolution of the WUI and the degree to which it limits or amplifies trans-boundary threats in coupled wildfire SESs, so much greater attention to land-use planning is warranted. Land-use regulations to guide fire-related building codes (Box 2) or restrict development in the most fire-prone locations[2,26,99,100] are clearly important steps that government agencies could take to manage the coupling in a wildfire SES. Agencies have a deeper role, however, in the growth of these trans-boundary threats. For example, the ‘safe development paradox’ applied to flood and hurricane protection demonstrates that making hazardous areas safer for human habitation in the short term actually increases the potential for severe losses over longer time scales[101]. Given that government agencies around the world have focused on reducing fire hazards (for example, through subsidized fire suppression and/or fuel reduction), much less attention has been paid to the ways in which vulnerable WUI development might have been designed from the start. As further development occurs and the WUI expands, so does the need for increased hazard reduction. A perverse consequence of the typical human reaction to fire — to fight it instead of accommodate it — thus contributes to a deepening of coupled wildfire SES problems.

Strategically addressing threats at the WUI maximizes the potential for both effective risk mitigation within developments and management for sustainable fire regimes over the broader sweep of landscapes. Ultimately, trade-offs and sacrifices must be made to balance these competing demands, but concentration of management effort for risk mitigation in the WUI minimizes the area where adverse effects on environmental assets are likely. Better maps of fire hazards, ecosystem services and climate change effects are thus important for assessing these and other related trade-offs. Addressing all social, economic and environmental assets at risk will necessarily focus on separating those that require exclusion of fire from those where fires of some sort are desirable or inevitable. However, it is unlikely that any planning or management regime will completely exclude fires from vulnerable developments on many landscapes (considerable residual risk to people and property will endure). The capacity for communities to cope with the inevitability of fire, as well as its effects at multiple scales, will therefore be essential.

There is a great deal of research to support better policy, planning and management in all aspects of the coupled wildfire SES problem. Viewing fire as a natural and inevitable hazard should be central to most solutions, so we can anticipate its important positive and negative effects on both human and natural systems. Given that combustion is one of the most basic and ongoing natural processes on Earth, we must continue to learn from our experiences to achieve a sustainable coexistence with wildfire. ■

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The Holocene record of fire and erosion in the southern Sacramento Mountains and its relation to climate

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Introduction

As highlighted in this issue's Gallery of Geology on page 24, large, severe wildfires have become part of the New Mexico late spring and early summer experience in the last few decades. Such fires have considerable relevance to geomorphologists, as erosion rates in mountainous landscapes are often dramatically increased in the several years after severe fires. Erosion and sediment transport often take place during major debris flows and flash floods (Fig. 1). These events are most commonly triggered by intense thunderstorm rainfall, as in New Mexico's summer monsoon, and very rapid runoff from slopes devoid of vegetation or forest litter. Although water-repellent soils formed by fire effects are often cited as the primary cause of increased runoff, the creation of extensive bare, smooth soil surfaces alone is more than sufficient—for example, consider the erosion that would occur on a plowed, smooth farm field at slope angles of 15–30° or more! The extreme flows that are generated can entrain huge volumes of sediment as they course down slopes and channels. Events of this nature affected a number of small, steep drainages in the Sacramento Mountains southeast of Cloudcroft after the 2002 Peñasco fire. Large quantities of mud- to boulder-sized sediment may be deposited on alluvial fans along the valley margins, and in some cases deep gullies are also cut in the fans. Major damage to roads, buildings, and property resulted in several locations in the Peñasco fire area, as valley-side alluvial fans are common sites for residential and other development.

Along with their importance in understanding geologic hazards and watershed impacts, sediments deposited on alluvial fans by postfire debris flows and floods also provide a means of assessing the timing and spatial distribution of past forest fires, and relations between fire and climate, in particular episodes of severe drought. These sediments are often rich in charcoal fragments from the burned area, which allows radiocarbon dating of fire-related sedimentation events thousands of years into the past, providing an important supplement to the more commonly available tree-ring fire histories. Tree-ring dating has provided a wealth of information on low-severity surface fires that scar trees, but leave them living. Such fires swept through the understory of many southwestern forests every few years to a few decades before European settlement and intensified grazing, logging, and fire suppression in the late 1800s (e.g., Brown et al. 2001). However, tree-ring fire-scar records extend back about 500 yrs at most, and do not provide data on severe fires that kill large stands of trees. Stand-destroying fires can be dated via the ages of living trees that germinated after fire, but this reveals the last such fire only, and again is limited to about the last 500 yrs. Therefore, alluvial sediments can greatly extend fire histories, albeit with greater uncertainty in ages. Climatic change on time scales of a thousand years or more has strongly affected Earth environments over the Holocene Epoch, the ~12,000 yrs since the last episode of continental glaciation. Thus, the sensitivity of fire activity to climate change over such time scales is of great interest. It is also critical to understanding the potential impacts of future droughts on New Mexico's mountain forests, given that predicted warming over the next century has no precedent on the short time scales covered by tree-ring fire chronologies.

FIGURE 1—Deposits of a debris flow from a tributary basin of lower Cox Canyon in the Sacramento Mountains. The tributary basin was severely burned in the 2002 Peñasco fire. Debris-flow deposits partially filled and dammed the main valley arroyo at this location (the arroyo wall is visible across the center of the photo). The surface deposits lack mud because of reworking by later flood flows.

Fire and alluvial history in the Sacramento Mountains

South of Cloudcroft, the Sacramento Mountains are essentially a broad eastward-dipping cuesta, where the range crest and ridges on the eastern slope are capped by resistant limestone and dolomite of the Permian San Andres Formation. Below the San Andres Formation, the highly erodible Permian Yeso Formation forms slopes that have contributed large volumes of fine-grained Holocene alluvium to valleys. Abundant exposures of these alluvial sediments are present in both deep main valley arroyos that predate the Peñasco fire (Fig. 2), and in gullies cut in alluvial fans by recent postfire debris flows and floods (Fig. 1). Deposits of modern postfire events helped us to define criteria for recognition of fire-related sediments in the Holocene alluvium. Conifer forests of the Sacramentos range from spruce and fir near the range crest, through ponderosa pine and mixed conifers at middle elevations, to piñon-juniper stands near the lower forest border.

We focused our investigations along the valleys of the Rio Peñasco and lower Cox Canyon in the eastern Sacramentos (Frechette and Meyer 2009), and in Caballero Canyon west of the range crest (New 2007). Fire-related deposits dating to as much as 8000 yrs ago were discovered, interbedded with deposits with no clear evidence of an origin in a burned area. The most active period of fire-related sedimentation occurred from about 6000 to 4000 yrs ago, as is highlighted in Figure 3. Although it does not stand out markedly as a dominant peak in the fire-related sedimentation curve, this interval saw fans build rapidly with several meters of accumulated sediment including thick, charcoal-rich debris-flow deposits (Fig. 2, inset). This evidence for severe fires is consistent with a generally warmer middle Holocene climate, characterized by widespread and persistent drought conditions in the Southwest and in the interior western United States in general (Buck and Monger 1999; Shuman et al. 2009). However, it may also reflect a higher variability in precipitation and (or)

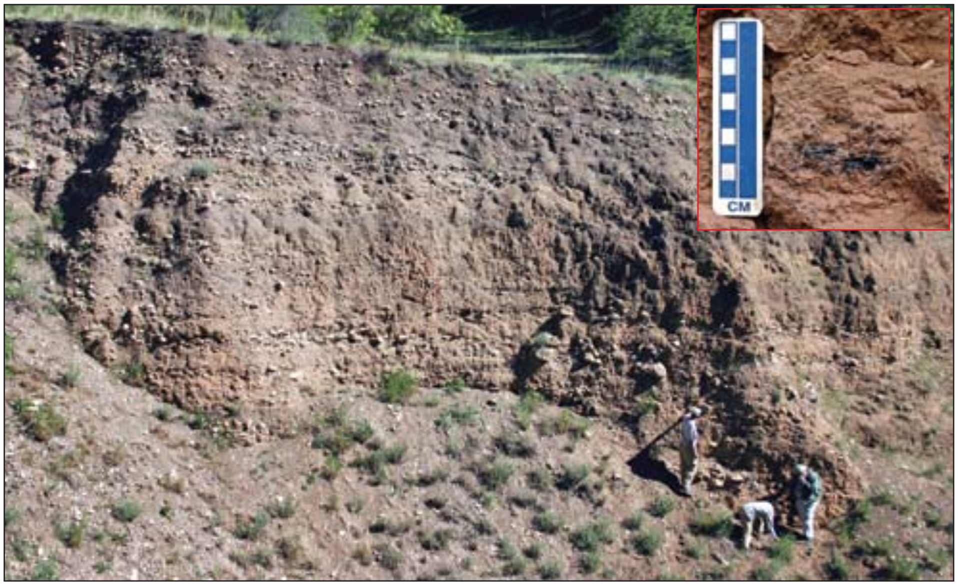

FIGURE 2—More than 10 m (32 ft) of sediment in the toe of an alluvial fan is exposed in this main valley arroyo along the Rio Peñasco. The fan deposits range in age from about 8000 to 650 yrs before present. Darker layers represent organic-enriched soils that developed during times of slow sediment accumulation. The middle part of the section with little soil development starting above the persons' heads dates to about 6000–4000 yrs ago, when rapid fire-related aggradation occurred. Inset shows abundant charcoal fragments in a muddy, poorly sorted fire-related debris-flow deposit from this section.

temperatures at this time (e.g., Asmerom et al. 2007), which would allow forests to grow more dense during wetter intervals that minimize the occurrence of surface fires. Severe crown fires would then be more likely during major droughts.

Evidence for severe fires in charcoal-rich debris flows in the Sacramento Mountains became substantially less common after 4,000 yrs ago, probably because of the climatic cooling associated with declining summer solar radiation in the Northern Hemisphere, and advances of mountain glaciers in the western USA and elsewhere known as the Neoglacial. However, episodic fire-related sedimentation punctuated this time interval, most notably around 650 yrs ago (Fig. 3). This peak in fire-related sedimentation is associated with the "Great Pueblo Drought" of AD 1276–1297, a period of persistent, severely dry conditions that was noted in some of the earliest climate reconstructions using tree rings (Douglass 1929). This megadrought centered on the Four Corners area but affected a much larger region of the Southwest (Cook et al. 2007). It came at the end of a period of generally warmer climate in the western USA from about AD 900–1300 known as the Medieval Climatic Anomaly (Fig. 3). There is also substantial evidence from tree rings, lake levels, and other paleoclimatic records of unusually large fluctuations between wet and dry conditions during this time, which could again promote dense growth of conifer stands followed by extensive severe fires. Prior work in Yellowstone National Park and in central Idaho has also shown the Medieval Climatic Anomaly to be a time of major fire-related debris-flow activity and building of alluvial fans (Meyer et al. 1995; Pierce et al. 2004), illustrating the widespread effects of drought in this interval. The most prominent episode of postfire debris flows in these areas, however, is correlated with the AD 1140–1162 megadrought, considered to be the worst in North America in the last two millennia (Cook et al. 2007). Fire-related deposits dating to the time of the mid-1100s megadrought are found in the Sacramento Mountains (Fig. 3), but are less common than those emplaced after fires in the late AD 1200s. Overall, the occurrence of major droughts, severe fires, and debris flows across the interior western USA is consistent with an inference of generally higher temperatures during the Medieval Climatic Anomaly.

The effects of warmer climate and severe drought notwithstanding, fire-related sedimentation in the Sacramentos over most of the Holocene is characterized by relatively small and sporadic events, consistent with a regime of low-severity surface fires, with the occasional patch of higher-severity crown fires. Likewise, tree-ring fire-scar records in the Sacramentos show that frequent, low-severity surface burns dominated fire activity throughout the range of forest types, from the beginning of the record about AD 1580 to the late 1800s (Brown et al. 2001). This period falls within the generally cooler and effectively moister climate of the Little Ice Age (e.g., Cook et al. 2007) (Fig. 3). A regime of low-severity fires makes sense during the Little Ice Age, as reduced temperatures and evapotranspiration would promote grass growth to fuel surface fires, as well as limit the potential for reducing moisture levels in the forest canopy to the point where extensive, high-severity crown fires could occur. Since the late 1800s, fire suppression and other land uses greatly limited surface fire activity, and the resulting denser forest stands—along with a warming climate, especially in the last several decades—have created conditions that are ripe for extensive stand-destroying crown fires.

An interesting aspect of Sacramento Mountains forests are the large, dense stands of Gambel oak that are especially prominent on the upper western slopes of the range. Professor Thomas Swetnam of the University of Arizona Laboratory of Tree-ring Research has hypothesized that these brushy patches represent areas where severe crown fires destroyed conifer stands and Gambel oak recolonized. Identification of charcoal fragments found in soils under oak brush in Caballero Canyon suggest that some past fires at these sites were indeed in conifer stands (New 2007), but further work is necessary to test this hypothesis.

Another question we considered in our Sacramento Mountains work is whether the deep arroyos found along many valley reaches may, at least in part, stem from land uses such as railroad logging and intensive grazing, especially along the eastern flank of the range. The main valley arroyos predate modern severe wildfires, but the timing of their initiation is largely unknown. At lower elevations in the Southwest and on the Colorado Plateau, most valley-filling alluvial sequences show clear evidence of alternating episodes of arroyo cutting and valley filling, especially in the last 4000 yrs (e.g., Waters and Haynes 2001; Love 1983). We found no clear evidence of past arroyo incision in the exposures examined for fire-related deposits in the eastern Sacramento Mountains. We also found no place where gravelly deposits filled a deep paleochannel in finer-grained valley fill, as occurred in modern times after the Peñasco fire (Fig. 1). Such a relationship should have been obvious despite imperfect exposures along many present arroyos. Some evidence exists for paleo-arroyo cutting in Caballero Canyon on the west slope, however. The lack of clear precedence for modern arroyos in the eastern Sacramentos suggests that 19–20th century land use may have been an important factor. However, we again need to investigate this question further, including dating the initiation of the present episode of arroyo cutting, and conducting a focused search for paleochannels.

Implications of long-term fire-climate relations

As in other studies of Holocene fire-climate relations (e.g., Pierce et al. 2004), our work in the Sacramento Mountains shows that fire behavior is highly sensitive to relatively modest climatic change. With the high probability of increased temperatures and episodes of severe drought over the next century, catastrophic wildfires and their accompanying debris flows and flash floods will become even more likely in New Mexico's mountain forests. Thinning of over-dense stands that have resulted from fire suppression, especially in ponderosa pine forests, can reduce

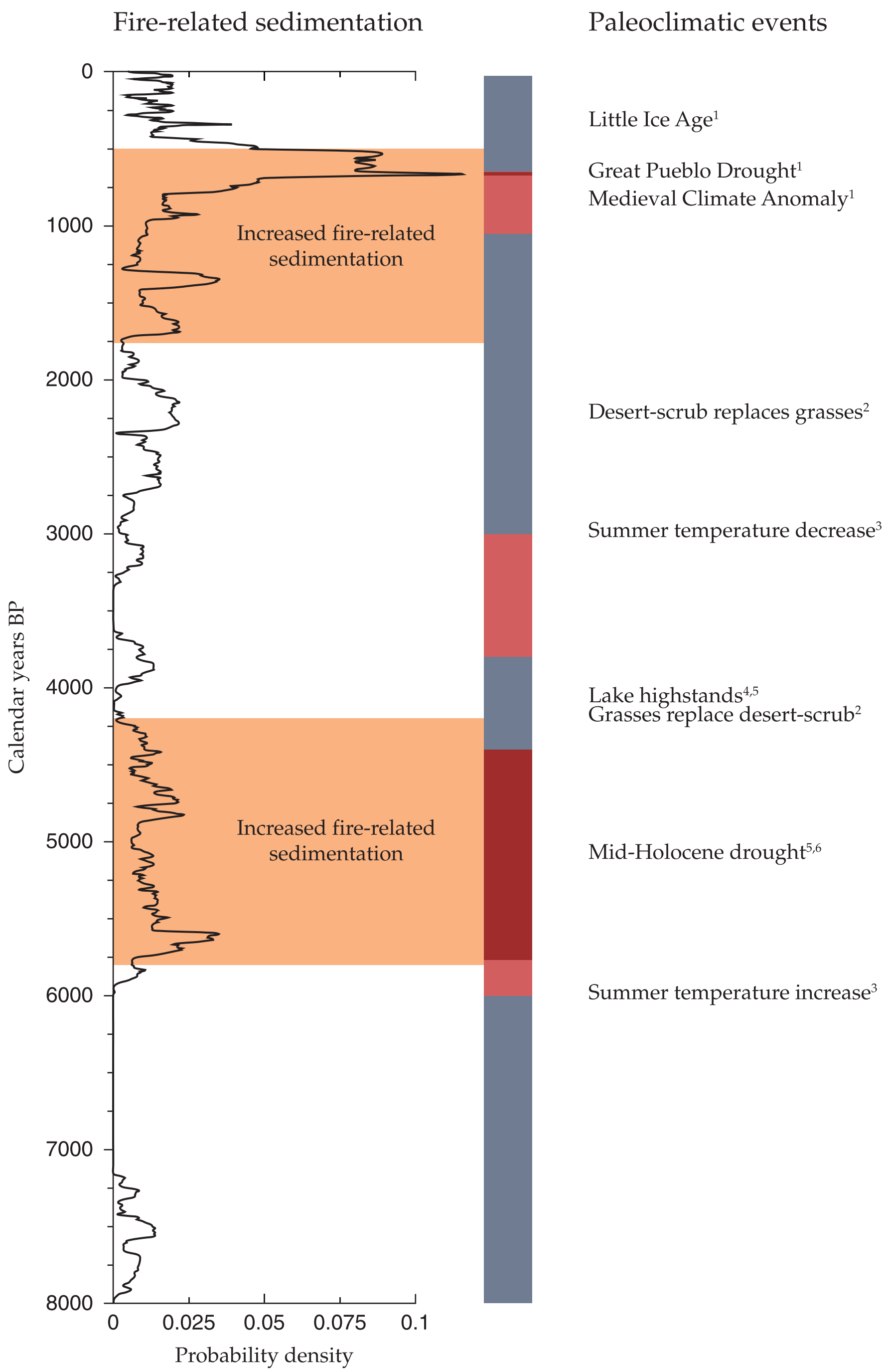

FIGURE 3—Record of fire-related sedimentation in the Sacramento Mountains compared to simplified Holocene paleoclimatic events from the western USA. The curve at left shows the relative probability of fire-related sedimentation, and was constructed by summing probability distributions for calibrated radiocarbon ages on individual fire-related sedimentation events. Periods of increased fire-related sedimentation are inferred based on both radiocarbon age distributions and the nature of stratigraphic evidence for fires. References for paleoclimatic information: [1]Cook et al. 2007, [2]Buck and Monger 1999, [3]Viau et al. 2006, [4]Castiglia and Fawcett 2006, [5]Shuman et al. 2009, and [6]Davis and Shafer 1992.

the impact of wildfires, but this is a very large and expensive task that can have impacts of its own, for example, in roadless areas, and there is limited commercial value to the small-diameter trees that must be removed. Public awareness of the hazards that stem directly from development in fire-prone forests, as well as those from postfire debris flows and flash floods on alluvial fans, is key to reducing risks to life and property.

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Recent Forest Insect Outbreaks and Fire Risk in Colorado Forests: A Brief Synthesis of Relevant Research

Photo: Jeff Hicke

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Introduction

Extensive outbreaks of tree-killing insects are occurring in many parts of the West, including Colorado. In combination with recent high-intensity forest fires, these insect outbreaks are raising concerns about the health of our forests and our ability to deal with these issues. The visual impact of a high-severity bark beetle outbreak or fire may give the impression that we are in a crisis situation and that we must take dramatic steps to deal with this "emergency". However, recent scientific research on the ecology of forest disturbances, by scientists in Colorado and elsewhere, leads us to interpret these recent events in a much more nuanced manner.

We believe that the responses to insect outbreaks and fires will not produce beneficial results unless those responses are consistent with the basic ecology of the affected forest ecosystems. Hence, we have written this brief synthesis of the current state of knowledge about forest insects and fires in Colorado to help inform effective management options. Our emphasis is on the ecological aspects of the insect outbreaks now affecting thousands of acres in the state. We do not deal extensively with other dimensions of insect outbreaks and fires, although we acknowledge that aesthetics, economics, wildlife management, recreation, watersheds, and fuels are all important considerations in making decisions about forest policy and management.

This report is organized into two sections. The first section addresses nine key questions about the basic ecology of insect outbreaks in Colorado forests; the second section evaluates six possible treatment options. We do not advocate any particular policy or management treatment, but instead describe the likely ecological effects of each potential option. We also provide a very brief synopsis of each answer or treatment option in italics at the beginning of each section. Our hope is that the information summarized here will aid managers and policy-makers in making decisions about how to deal (or not deal) with different kinds of insect outbreaks occurring in different contexts. As will become clear below, not all forests and not all insects are alike. The authors all have training and research experience in forest ecology or hydrology, both in Colorado and elsewhere.

Questions about the Basic Ecology of Forest Insects

Question #1: Which insects are killing trees across large areas in Colorado?

Summary: *The major insects killing trees in Colorado today include bark beetles (mountain pine beetles, spruce beetles, and piñon ips beetles) and defoliators (notably western spruce budworm). All of these insects are native to Colorado and have co-existed with their host tree species for thousands of years (Figure 1).*

Two major groups of insects have been responsible for killing large numbers of trees over extensive areas under outbreak conditions in Colorado: bark beetles and defoliators (Schmid and Mata 1996). Adult bark beetles bore through a tree trunk and lay eggs within the inner bark. The eggs hatch and the beetle larvae eat the inner bark, killing the tree. After the larvae mature, the new adults fly to new trees, bore through the bark, and continue the cycle. There are several species of bark beetles, each of which feeds on one or several species of trees. For example, the mountain pine beetle feeds on ponderosa, lodgepole, and limber pine; the spruce beetle feeds on Engelmann spruce; and the piñon Ips beetle feeds on piñon pine.

Defoliators are a group of insects having a life cycle very different from the bark beetles. The adult defoliators are tiny moths that lay their eggs in the buds of trees. The eggs hatch into caterpillars that feed on the emerging new leaves in spring and early summer. When numerous, the caterpillars may eliminate essentially all of a tree's annual production of leaves or needles. Small trees, or trees that are stressed by other factors, may die after a few years of defoliation, though usually most of the trees in a stand survive the outbreak of defoliators. The most important defoliator in Colorado forests is the western spruce budworm which feeds on Douglas-fir, white fir, subalpine fir, and spruce. Douglas-fir tussock moth is a less frequent but locally significant defoliator of Douglas-fir, white fir, and spruce. Aspen trees may be defoliated by tent caterpillars and large aspen tortrix.

These insects are usually present in a forest in very low numbers, killing only the occasional weak tree. Such low numbers are referred to as

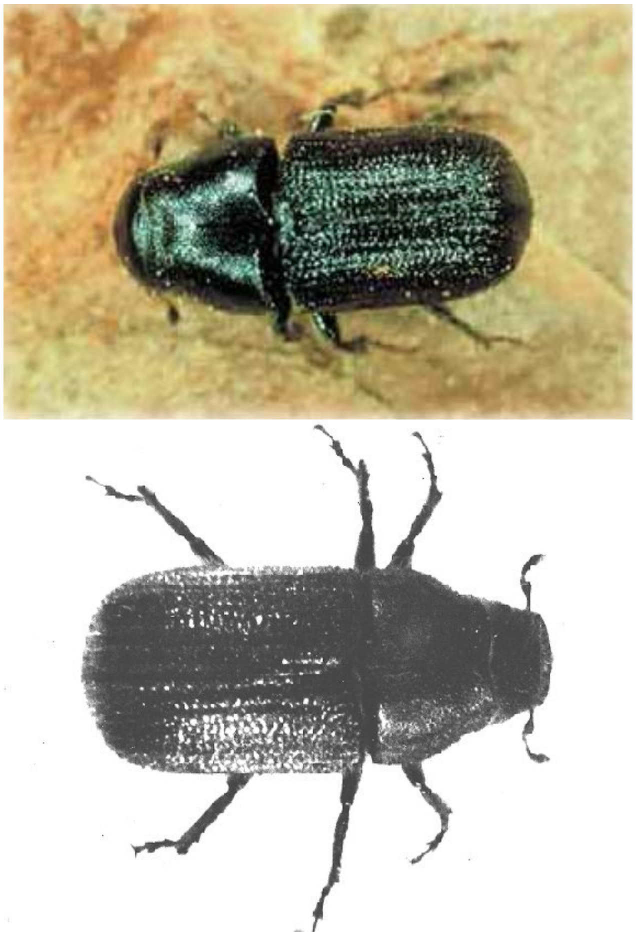

Figure 1. The major insects killing trees in Colorado today include several species of bark beetles (such as the mountain pine beetle above and the spruce beetle below) as well as various species of defoliators. All are native to Colorado and have co-existed with their host tree species for thousands of years. Mountain pine beetle photo from Colorado State Educational Extension Service. Spruce beetle photo from USDA National Agricultural Library.

"endemic" populations. Periodically, however, insect populations grow rapidly and kill large numbers of trees over large areas. This is referred to as an "outbreak" or "epidemic" population. Outbreaks of all of the insect species described above have occurred recently, and have caused extensive mortality events in their respective tree hosts. It is important to note, however, that the trees of Colorado and the Rocky Mountains have coexisted with these native bark beetles and defoliators for thousands of years.

Question #2: Are the insect outbreaks now occurring in Colorado unprecedented in the ecological history of this region, or are they “natural” events similar to outbreaks that occurred in the past?

Summary: *There is no evidence to support the idea that current levels of bark beetle or defoliator activity are unnaturally high. Similar outbreaks have occurred in the past (Figure 2).*

There is no evidence to support the idea that current levels of bark beetle or defoliator activity in Colorado’s lodgepole pine and spruce-fir forests are unnaturally high. The outbreaks now taking place in Colorado are similar in intensity and ecological effects to previously documented outbreaks in the Rocky Mountains. For example, mountain pine beetle outbreaks killed millions of lodgepole pine trees over thousands of square miles in the Cascade and Rocky Mountains during the 1960s, 1970s, and early 1980s (Lynch 2006; chapter 4); and a spruce beetle outbreak in the 1940s killed spruce trees over much of the White River Plateau in western Colorado. Historic photos and tree-ring evidence also document extensive insect outbreaks prior to the 20th century (Baker and Veblen 1990, Veblen et al. 1991, Veblen et al. 1994, Swetnam and Lynch 1998, Eisenhart and Veblen 2000, Veblen and Donnegan 2006). Thus, insect outbreaks are a natural occurrence in almost all of the different kinds of forests in Colorado. Outbreaks do not occur very frequently; the time interval between successive outbreaks in any given area is usually measured in decades. Nevertheless, outbreaks can be expected periodically in almost any place in the state where forests are found.

It is true that bark beetle outbreaks are now

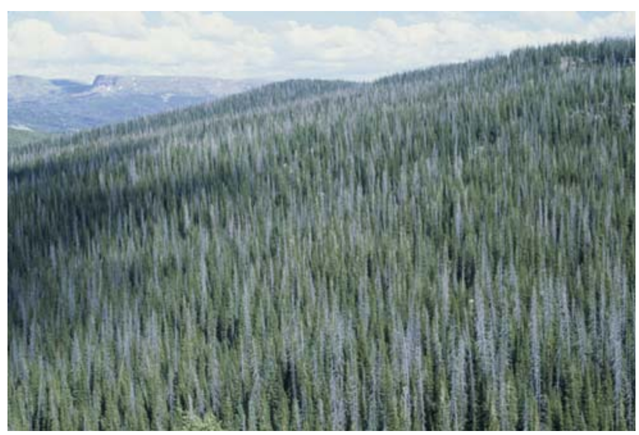

Figure 2. The insect outbreaks now occurring in Colorado are similar in extent and severity to outbreaks of the past. For example, spruce beetles killed millions of trees over thousands of acres in the White River National Forest in the late 1940s and early 1950s. The dead trees (above) are still visible. (Photo by T. T. Veblen).

occurring in parts of Colorado where such extensive insect activity had not been seen at any time during the previous hundred years (e.g., in the Fraser Valley). However, in the absence of tree-ring reconstructions or other spatially detailed information on historical mountain pine beetle outbreaks in Colorado, it is not known if similar outbreaks occurred in the same locations or habitats in the past several centuries. Given the naturally long intervals between recurrent bark beetle outbreaks in Rocky Mountain forests, there is nothing unusual about a hundred-year period of low activity followed by an extensive outbreak. It also is true that mountain pine beetles now are killing trees at unusually high elevations (Wayne Shepperd, personal communication). This may be a significant departure from previous outbreaks. However, it is difficult to know if the current insect activity at high elevations is truly unprecedented, given the lack of data on precise spatial patterns of prehistoric outbreaks. The occurrence of outbreaks today at high elevations, where the insects ordinarily are limited by cold temperatures, is not surprising considering the warm temperatures we have experienced during the past decade, as we discuss in the next question.

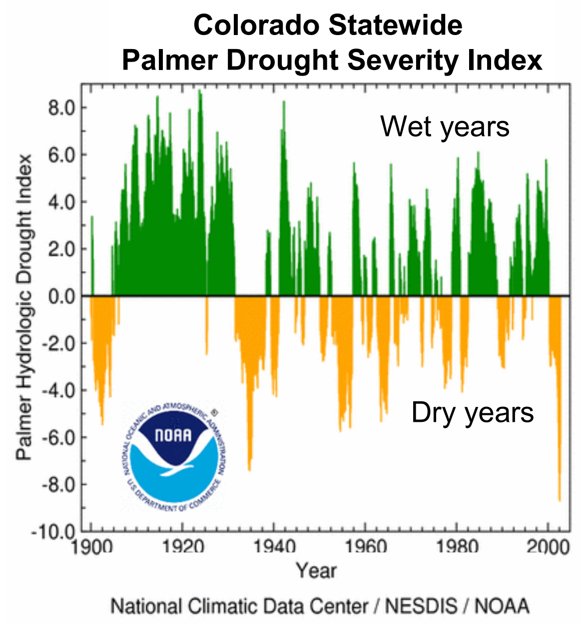

Figure 3. The reason why bark beetle outbreaks are so extensive and severe in Colorado today is because of four interacting ecological factors: (i) long-term drought, as shown above, that stresses trees; (ii) warm summers and (iii) warm winters, which enhance beetle growth and survival; and (iv) abundant food sources (trees) for beetles.

Question #3: Why are the insect outbreaks so severe and so widespread at this time?

<u>Summary</u>: *The ecological factors that control insect populations are complex. Recent bark beetle outbreaks in Colorado probably are a result of four interacting factors: (i) long-term drought, which stresses trees and makes them more vulnerable to insects, (ii) warm summers, which further stress the trees and may accelerate growth of the insects, (iii) warm winters, which enhance survival of insect larvae, and (iv) abundant food (trees) for the insects in Colorado's extensive and often dense forests (Figure 3).*

The factors that control the initiation, spread, and termination of insect outbreaks are complex, and involve a combination of climatic conditions and characteristics of forest stand structure. The relative importance of climate vs. stand structure in any given outbreak is not fully worked out, and in fact may vary from place to place and among the various insect and tree species. Nevertheless, the following is what we know about the interacting influences of drought, temperature, and stand conditions on insect outbreaks.

Evidence from observational, laboratory, and modeling studies indicates that climate is a major controlling factor of bark beetle outbreaks (Bentz et al. 1991, Logan et al. 2003, Carroll et al. 2004, Breshears et al. 2005). The initiation of a bark beetle outbreak is often associated with drought. It is thought that the dry conditions stress the trees and make them less able to defend themselves against the beetles (Carroll et al. 2004). For some insects, the end of the drought usually means the end of the outbreak. However, with mountain pine beetles and spruce beetles, once the beetles have killed a large number of trees and produced abundant offspring, their numbers may become so great that they can overwhelm even healthy trees. If this point is reached, continued drought is not so important: the beetle population continues to grow until it is checked either by a prolonged period of bitter cold weather or until they exhaust their food supply. Low temperatures (around –40 degrees F for about a week), especially in late fall or early spring, may kill the beetle larvae in the trunks of the trees, and thereby terminate the outbreak at any stage in its development.

A warming climate during the last 100 years, particularly in the last few decades, also appears to have played a role in driving recent insect outbreaks. Higher temperatures and a longer frost-free period subject the trees to additional water stress, and may accelerate the growth and development of the beetle larvae. The warming trend of the past few decades (Westerling et al. 2006) may have contributed to the current outbreak of mountain pine beetle in Colorado, as well as recent outbreaks that have occurred outside of Colorado in historically marginal environments for bark beetles, such as at the northern extent of their range in Canada (Carroll et al. 2004) or in high elevations of the northern Rockies (Logan and Powell 2001, Hicke et al. 2006). Furthermore, changing climate conditions are thought to have been responsible for a very severe mortality event in the piñon trees of southern Colorado and adjacent states. Between 2002 and 2004, extensive piñon mortality occurred during a severe drought and an accompanying outbreak of Ips bark beetle (Breshears et al. 2005). Although a more intense drought actually occurred in the 1950s, piñon mortality was far more severe and widespread in 2002 - 2004, apparently because the unusually warm conditions that accompanied the recent drought put additional stress on the trees and allowed more extensive outbreaks of the piñon Ips beetle. Breshears et al. (2005) documented elevated maximum and minimum temperatures at numerous weather stations throughout the Four Corners region during the past decade.

Stand structure also is important in bark beetle outbreaks. The inner bark of very small trees usually is not thick enough to support beetle larvae, and consequently the adult beetles tend to select larger trees to lay their eggs. The minimum tree size for the mountain pine beetle is around four to five inches diameter, but is different for other beetle species (Furniss and Carolin 1977). Thus, stands with large trees are more susceptible to bark beetle outbreaks than are stands with smaller trees. In addition, trees in old or dense stands may be less vigorous and therefore more susceptible to beetles than trees in young or less dense stands, because of competition among trees for limited water and nutrients (Shore and Safranyik 1992). At the landscape scale, if most of the forest is of similar age and has a structure conducive to bark beetle outbreaks, it is likely that outbreaks will be widespread -- if climate conditions are also appropriate. Although fire suppression in the lodgepole pine zone probably reduced opportunities for establishment of young stands since about 1940, young stands have established after timber harvests during this period. The main influence on lodgepole pine age structure in Colorado, however, is widespread burning in the late 1800s that resulted in extensive cohorts of relatively similar age that now are entering a stage that is susceptible to bark beetle outbreaks.

So, why have recent insect outbreaks been so extensive and severe in Colorado? We believe the answer is as follows. The past decade has brought severe drought to many parts of the state (Pielke et al. 2005, Figure 3), accompanied by relatively warm temperatures in both summer and winter (Westerling et al. 2006). The combination of drought and hot summers probably stressed the trees and made them more susceptible to bark beetles; the warm summers may have accelerated the growth and reproduction of some bark beetle species (e.g., spruce beetles and piñon Ips); and the mild winters produced very little mortality of beetle larvae. These climatic conditions probably are the major reason why insect outbreaks have gotten started in many different regions of the state. Once the outbreaks began, the beetles found an abundant food supply (trees) in most of Colorado's forests. Many stands are densely stocked with trees because they have not been disturbed for a very long time by fire, insects, or harvest. All of these factors have combined to create a "perfect storm" of bark beetle outbreaks across much of Colorado.

Question #4: Are the dense forest stands that we see in Colorado today the unnatural consequence of past fire suppression and lack of timber harvesting?

Summary: *The answer to this question depends on the type of forest and its geographic location, as explained below. For example, high density in lodgepole pine and spruce-fir forests is not related to fire suppression; it is simply a natural* *ecological feature of these subalpine forests. It is important to note that not all forests have been affected in the same way by past fire suppression and other human activities (Figure 4).*

Many Colorado forests are very dense, but not all dense forest stands are the unnatural consequence of past fire suppression and lack of timber harvesting. For example, high tree density is a natural condition of most high-elevation forests, including lodgepole pine and spruce-fir. On the other hand, some ponderosa pine forests (but not all) do have unnaturally high tree densities -- higher than would have been seen prior to Euro-American settlement of the region. Thus, it is necessary to distinguish among different forest types in Colorado and elsewhere in the West when considering the effects of past fire suppression and timber harvest (or lack thereof) on current stand density.

Ponderosa Pine Forests Summary: *Tree densities have increased significantly in dry ponderosa pine forests in parts of Arizona, New Mexico, and southern Colorado, largely as a result of fire suppression and other human activities. Ponderosa pine in northern Colorado has been affected to a lesser extent, because fires were historically less frequent in this region than farther south, and the historical landscape was a mosaic of dense and open stands. The proportion of dense vs. open stands is greater in some areas of the Front Range today than historically, in part because of fire suppression, but also because of recovery from 19th century disturbances and because 20th century climate was generally favorable for tree growth.*

Dry ponderosa pine forests in the Southwest were formerly characterized by frequent, low-intensity surface fires, and it is primarily in these forests where fire suppression has contributed to unnaturally dense stands and increased fire severity today (Covington and Moore 1994, Mast et al. 1999, Moore et al. 1999, Allen et al. 2002). Although fire suppression is part of the reason for very dense stands of ponderosa pine in the Southwest, previous grazing, logging, and climate have also contributed to this change in forest structure (Allen et al. 2002). For example, abundant recruitment of pine seedlings typically occurs during moist climatic periods, and the twentieth century has been characterized by several such periods (Savage et al. 1996, Brown and Wu 2005). In the Colorado Rockies, a

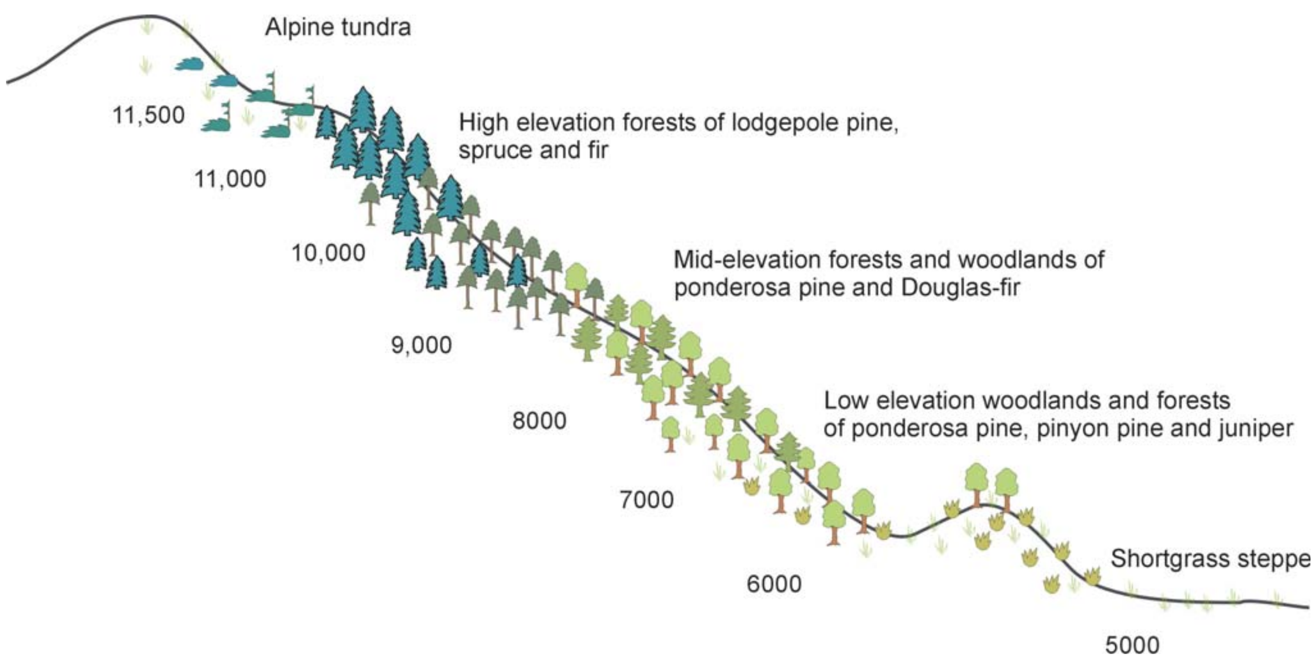

Figure 4. Colorado's forests and woodlands are diverse, ranging from piñon-juniper woodlands in the foothills and basins, to ponderosa pine and Douglas-fir forests at middle elevations, to lodgepole pine and spruce-fir forests at the highest elevations. The natural frequency and effects of forest fires are equally diverse. Tree density in some ponderosa pine forests is greater today than historically because of fire suppression, grazing, and logging during the past century. In contrast, dense stands in high-elevation forests are not related to 20th century fire suppression or land use history; they are simply natural features of these forests where fires have always occurred infrequently. (Figure prepared by L. Huckaby)

model similar to that in the Southwest -- suppression of formerly frequent low-severity fires followed by increased tree density -- applies to some but not all ponderosa pine forests. This "Southwestern ponderosa pine" model appears to be most applicable towards lower elevations and more southerly portions of Colorado.

The Southwestern ponderosa pine model generally does not apply throughout the moister, cooler forests in northern Colorado and at higher elevations, even though ponderosa pine may still dominate (Kaufmann et al. 2006, Baker et al. 2006). For example, in ponderosa pine forests of the Colorado Front Range, tree-ring and other evidence demonstrates that the historical fire regime included both low-severity fire (i.e., surface fires that thin the forests) and high-severity fires (i.e., fires that kill canopy trees and often result in dense regeneration) (Mast et al. 1998, Brown et al. 1999, Kaufmann et al. 2000, Veblen et al. 2000, Huckaby et al. 2001, Ehle and Baker 2003, Sherriff 2004, Kaufmann et al. 2006). In fact, less than 20% of the ponderosa pine zone in the northern Colorado Front Range appears to have been characterized mainly by frequent, low-severity fires. Instead, most of the ponderosa pine zone was characterized by a variable–severity fire regime that included a significant component of high-severity fires (Sherriff, 2004).

The high-severity fires of Front Range ponderosa pine forests tend to occur less frequently than the low-severity fires, and forests naturally grow dense during the long intervals between successive fires. These dense stands are interspersed with more open stands, creating a complex mix of forests. Thus, we conclude that the dense ponderosa pine forests seen in some parts of Colorado's northern Front Range are only partly due to 20th century fire suppression and low rates of timber harvest in recent decades. In contrast to some forests in the Southwest, dense stands of ponderosa pine have always been a component of the Front Range landscape. The proportion of dense vs. more open pine stands has shifted towards more dense stands during the past half-century in many areas, in part because of fire suppression, but also because of climatic conditions conducive to tree growth and natural recovery of forests that were burned or logged in the late 19th century.

Lodgepole Pine Forests <u>Summary</u>: *Dense lodgepole pine stands are not an artifact of fire suppression. These forests have always burned infrequently (intervals of many decades or centuries between fires) and at high intensity, and these fires are naturally followed by development of a dense young stand. Fire suppression has not significantly altered the natural frequency or ecological effects of fire in most lodgepole pine forests.*

Dense stands historically were the norm in lodgepole pine and other high-elevation forests throughout the Rocky Mountain region (Parker and Parker 1994, Kashian et al. 2005, Schoennagel et al. 2004). In these forests in Colorado, fires occur infrequently (on the order of many decades or a century or more between successive fires in any given stand) and naturally tend to be high-intensity fires, usually crown fires, that kill the majority of the trees (Buechling and Baker 2004, Sibold et al. 2006, Veblen and Donnegan 2006). This type of natural fire behavior contrasts strikingly with the frequent surface fires of dry, low-elevation ponderosa pine forests: rather than thinning forests by killing primarily small, fire-intolerant individuals, the naturally severe fires of high-elevation forests typically kill all of the forest canopy and stimulate regeneration of the stand. Post-fire regeneration of lodgepole pine often results in a dense stand, especially where a large proportion of the trees have serotinous cones. Serotinous cones remain sealed by resin until the heat of a fire melts the resin and releases the seeds; thus, even though the adult trees are killed by the fire, they have stored huge numbers of seeds in their cones and those seeds are released into an optimal seed bed created by the fire.

The effect of fire suppression on the structure of individual stands and on the characteristics of stands across the landscape has been relatively minimal in lodgepole pine and other high-elevation forests in Colorado and throughout the Rocky Mountains (Schoennagel et al. 2004). The remote mountainous areas where these forests grow were generally difficult to access for fire-fighting, especially prior to the 1950s. Furthermore, the length of time that fire has been effectively excluded (~50 to 80 years) is short relative to the natural fire return interval (measured in centuries). As a consequence, fire exclusion has not significantly lengthened fire intervals in lodgepole pine forests. Note that this is in marked contrast to frequent, low-severity fire regimes such as Southwestern ponderosa pine.

It is true that a large proportion of the lodgepole pine stands in Colorado are more than 100 years old today (e.g. as reflected in stand age data from USDA Forest Service). However, this pulse of tree establishment was mainly due to widespread severe fires during the second half of the 19th century when climate was conducive to fires in the subalpine zone (Sibold and Veblen 2006). Tree-ring data show that similar pulses of establishment of lodgepole pine followed similar episodes of widespread fire in the 17th and 18th centuries across the subalpine zone of northern Colorado (Kulakowski and Veblen 2002, Kulakowski et al. 2003, Sibold et al. 2006). Thus, the tree-ring record of fire and tree establishment in subalpine forests indicates a high degree of variability in fire extent and stand initiation at time scales of 100 years. This variability included periods of extremely rare fires over 100-year periods of climate unfavorable to fire spread, so that long fire-free intervals such as in the 20th century are not outside the historical range of variability for these forests. Thus, age structures similar to the current dominance of the 100+ year old age class are typical of the historical conditions of lodgepole pine forests.

Because of the natural disturbance regime in lodgepole pine forests, characterized by infrequent but periodically large severe fires and insect outbreaks, these high-elevation forests do not exhibit a static or consistent average age class over time. We know that fires before 1900 in this forest type were infrequent but could grow to very large size under very dry weather conditions (Schoennagel et al. 2004, Sibold et al. 2006). It follows that we should expect large fires in lodgepole pine in the future, and that these future large fires should not be viewed as abnormal from an ecological standpoint. The key point about lodgepole pine forests is that they were dense and burned infrequently historically, and they are dense and burn infrequently today. High density in lodgepole pine forests is not related to fire suppression in any way; on the contrary, it is a natural feature of their ecology.

Spruce-Fir Forests Summary: *Dense spruce-fir stands are not artifacts of fire suppression either. Spruce-fir forests have always burned infrequently (intervals of centuries between fires) and at high intensity, and these fires are naturally followed by development of a dense young stand. Fire suppression has not significantly altered the natural frequency or ecological effects of fire in most spruce-fir forests.*

As in lodgepole pine forests, dense stands are also normal in spruce-fir forests (Veblen and Donnegan 2006). Prior to the beginning of fire suppression efforts in the 20th century, these forests were primarily shaped by large and severe fires that occurred in a given stand, on average, only once per several hundred years (Kulakowski et al. 2003, Buechling and Baker 2004, Sibold et al. 2006). Natural patterns of post-fire stand development resulted in high tree densities. Since long fire-free periods were normal in these forests prior to fire suppression efforts, it is very unlikely that several decades of fire suppression have fundamentally changed the natural fire regime or have resulted in forest structures that could be considered unnaturally dense. Instead, the dense spruce-fir forests today are very much like they have been in past centuries.

Question #5: Are recent wildfires in some of Colorado's dense forest stands unusually severe compared to pre-20th century fire severity?

Summary: *Recent fires have been more severe than historically in some forests, notably dry ponderosa pine forests in parts of Arizona, New Mexico, and southern Colorado. However, recent fires have behaved just as they did historically in most of Colorado's high-elevation forests, such as lodgepole pine and spruce-fir. Large intense fires are the normal fire behavior in these latter kinds of forests, and 20th century fire suppression has not caused them to be unnaturally severe (Figure 5).*

Again we stress the importance of distinguishing among forest types. Recent fires have been more severe, for example, in dry ponderosa pine forests in the Southwest, including some of the forests in southwestern Colorado. However, recent fires clearly are not

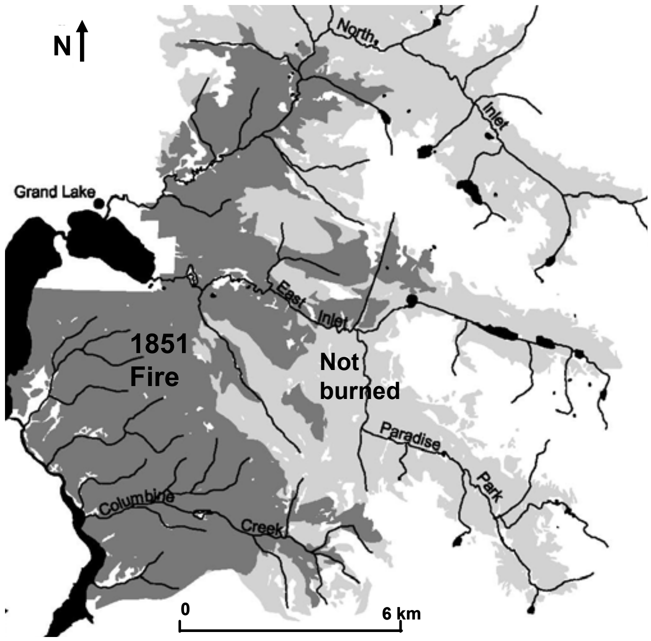

Figure 5. Large, intense forest fires are a natural feature of high-elevation forests in Colorado. For example, much of the country around Grand Lake, Colorado, burned in 1851. Most of the burned area now is covered by 150-year old lodgepole pine forests. (Figure from J. Sibold, 2005 Ph.D. Dissertation, CU Boulder). Some recent fires in ponderosa pine forests have been more severe than would have occurred historically, because of fuels changes associated with fire suppression, grazing, and logging during the past century. However, recent fires in lodgepole pine and spruce-fir forests also have been intense -- but no more intense than occurred historically.

more severe in lodgepole pine or in spruce-fir than fires that occurred in previous centuries. Even in the case of ponderosa pine forests in Colorado, not all areas follow the Southwestern pattern of increased stand densities following the near elimination of fires by grazing and fire suppression in the late 19th and 20th centuries. As noted above, in the Colorado Front Range the ponderosa pine zone was characterized by an historical mixed-severity fire regime in which some areas burned at low severity (as in the Southwest) but other areas, often large, were burned severely and regenerated to dense stands (Mast et al. 1998, Brown et al. 1999, Kaufmann et al. 2000, Veblen et al. 2000, Huckaby et al. 2001, Ehle and Baker 2003, Sherriff 2004, Kaufmann et al. 2006).

Question #6: Do outbreaks of mountain pine beetles and other forest insects increase the risk of severe wildfires?

Summary: *Although it is widely believed that insect outbreaks set the stage for severe forest fires, the few scientific studies that support this idea report a very small effect, and other studies have found no relationship between insect outbreaks and subsequent fire activity. Theoretical considerations suggest that bark beetle outbreaks actually may reduce fire risk in some lodgepole pine forests once the dead needles fall from the trees. It is true that severe fires have occurred recently in some forests following insect outbreaks (e.g., in spruce-fir forests of western Colorado). However, these fires burned under very dry weather conditions, and severe fires are the norm for these kinds of forests even without insect activity. Based on current knowledge, the assumed link between insect outbreaks and subsequent forest fire is not well supported, and may in fact be incorrect or so small an effect as to be inconsequential for many or most of the forests in Colorado (Figure 6).*

Our focus here is on active crown fires, i.e., fires that move from tree crown to tree crown under dry windy conditions. Surface fires also are significant; they can affect soils and understory plants, cause major damage to homes and other structures, and can be difficult to control, especially when burning in heavy fuels. However, in this discussion we emphasize crown fires because these often are the most fast-moving fires, they are the fires that typically cause the most damage to homes and other vulnerable structures, and they are almost impossible to control even with modern fire-fighting technology. It is important to realize that active crown fires do not burn only the dead fuels. On the contrary, crown fires are propagated through both live fuels (needles and small twigs) and dead fuels. Tree-killing insects do not really increase the amount of fuels in a forest stand; what they do is shift some of the live fuels into the dead fuel category. Both live and dead fuels can carry fire under very dry weather conditions.

Although more research is needed to confidently predict the effects of insect outbreaks on subsequent fires in Colorado forests, we offer the following interpretation based on theoretical considerations. Whether beetle-caused mortality enhances fire risk and severity compared to an

Figure 6. Lodgepole pine and spruce-fir forests typically burn at high intensity even without previous insect activity. It is widely believed that insect outbreaks set the stage for intense forest fires, but there is little scientific evidence for such a connection. Some recent Colorado fires have burned intensely in lodgepole pine and spruce-fir forests where insect outbreaks had occurred from a few to 50 years previously (e.g., in the Routt and White River National Forests). However, these fires occurred during extremely dry weather conditions, and forests unaffected by bark beetle outbreaks burned in similar fashion. (Photo by W. H. Romme)

unaffected stand very likely depends on time since outbreak. Post-outbreak stand development and associated fire risk may proceed through three stages. (i) Immediately following an outbreak, when trees are dead and dry needles remain on the trees, the chance of a crown fire getting started may be greater than for live trees. However, the dead needles may not significantly change the likelihood of a crown fire spreading from tree to tree, because crown fire spread is controlled not just by dead fuel quantity, but also by live fuel moisture, wind speed, and canopy bulk density (total amount of live and dead fuels in the canopy). This first stage lasts a relatively short time, because the dead needles usually fall within about two years of a tree's death. (ii) Once the needles fall off the dead trees, the likelihood of both crown fire initiation and spread actually may be reduced in comparison to an unaffected stand, since the dead trees create gaps in the canopy and reduce canopy bulk density. It is known that reducing canopy continuity and bulk density through mechanical thinning or harvesting can reduce crown fire risk (Graham et al. 2004), and it is likely that reductions in canopy continuity and bulk density resulting from insect-caused mortality would have a similar effect. (iii) After the dead snags fall, typically one to several decades after the insect outbreak, it is expected that the risk of crown fire initiation and spread may increase once again through two mechanisms. First, the fallen snags may fuel an intense surface fire, with heat and flame lengths that reach into the crowns of the trees. Second, small trees, which generally survived the outbreak and grew more rapidly in the more open conditions resulting from death of canopy trees, create “ladder fuels” that can carry a surface fire into the canopy. In sum, crown fire risk may be elevated for a brief time during and immediately after the peak of the outbreak, while the trees retain their dead needles; then fall to lower levels for the next few decades while the bare snags remain standing; and finally return to pre-outbreak levels some 20 – 50 years after the outbreak when the snags have fallen and a fast-growing understory has created ladder fuels between the heavy surface fuels and the canopy.

We emphasize again that the interpretation just presented is primarily theoretical and requires further study before definitive conclusions can be drawn. We also stress that this analysis focuses on effects of insect-caused mortality within a single stand. The impact on subsequent fire behavior will be different depending on the proportion of the trees killed in the stand. Moreover, it is important to recognize that a large forest landscape is composed of many individual stands. Substantial changes in stand structure and fire behavior within just one or a few stands may have little influence on fire spread and fire severity across the entire landscape.

A few empirical studies have evaluated subsequent fire activity in areas across the West that have been affected by major insect outbreaks, as summarized below. The general conclusion of these studies has been that the outbreak had no effect or only a small effect on subsequent fire occurrence or severity. However, more research of this kind is needed before we can make definitive statements about insects and fire.

Spruce beetle in subalpine spruce-fir forests. It is well established that in spruce-fir forests, extensive fires are highly dependent on infrequent, severe droughts (Buechling and Baker 2004, Sibold and Veblen 2006). Under those extreme drought conditions, dead fuels from insect outbreaks or other causes appear to play only a minor role, if any, in increasing fire risk. For example, following the 1940s spruce beetle outbreak that resulted in dead-standing trees over most of the subalpine zone of White River National Forest of western Colorado, there was no increase in the numbers of fires compared to unaffected subalpine forests (Bebi et al. 2003). Large fires did not occur in these forests until the drought of 1980, when 10,000 acres burned in the Emerald Lake Fire, and in the very severe drought year of 2002 (Pielke et al. 2005) when 31,000 acres burned in the Big Fish and Spring Creek fires. The 2002 fires in western Colorado affected extensive areas of spruce-fir and lodgepole pine forests that were previously affected by outbreaks of spruce beetle and of mountain pine beetle. Yet despite the expectation that these outbreaks (both the 1940s and an ongoing post-1998 outbreak) would have led to an increased risk of severe fires, the forests that were affected by the outbreaks generally did not burn more extensively or more severely than forests that were not affected (Bigler et al. 2005; Kulakowski and Veblen 2006).

Mountain pine beetle outbreaks in lodgepole pine forests. Turner et al. (1999) evaluated the influence of beetle outbreaks that had occurred 5-15 years previously on the behavior of the 1988 Yellowstone fires in lodgepole pine forests. They found that the likelihood of crown fire was increased somewhat where beetle-caused tree mortality had been high (perhaps because the fallen trees created heavy fuel loads), but was reduced where beetle-caused mortality was only moderate (perhaps because the dead trees interrupted the horizontal continuity of the canopy). Lynch (2006; chapter 3) also examined the influence of previous beetle activity on the 1988 Yellowstone fires by testing whether fire was more likely where the beetles had killed trees than in areas unaffected by the beetles. She found that beetle-affected areas did have a higher probability of burning, but that the increase was only about 11% compared with areas unaffected by beetles.

Spruce budworm defoliation. Massive outbreaks of western spruce budworm affected the Douglas-fir forests of the northern Colorado Front Range in the late 1970s and 1980s, but there is no evidence that they resulted in increased fire occurrence. Widespread fires have occurred recently in these forests, but these fires were associated with the extreme drought of 1998-2002. Therefore, if there was any potential increase in fire risk associated with the spruce budworm outbreaks, that potential was not realized until at least 25 years later when weather conditions were conducive to extreme fire behavior even in the absence of insect effects. In Ontario, Canada, Fleming et al. (2002) found a significant increase in probability of fire 3-9 years after an outbreak (perhaps because of increased vertical fuel continuity between fuels on the forest floor and fuels in the canopy), but probability of fire was not continuously elevated after the outbreak. However, in British Columbia, Canada, Lynch (2006; chapter 2) reported a significant decrease in risk of forest fire for nine years following a spruce budworm outbreak.

The upshot of these few studies of insect effects on subsequent fire risk is that the relationships are complex, and that no simple statements can be made about how outbreaks do or do not increase the risk of fire. One reason for the lack of clear-cut patterns is that spruce-fir and lodgepole pine forests naturally burn very infrequently, and only under very dry weather conditions. When the weather conditions are right for a big fire in spruce-fir or lodgepole pine, fire behavior is naturally intense, whether affected by previous insect activity or not. If insect outbreaks do in fact increase the likelihood of fires getting started or burning intensely through these kinds of forests, the magnitude of increase probably is small and difficult to detect, because fire is so strongly controlled by weather in these forests, and because they naturally burn at high intensity.

Question #7: Are forests with large amounts of insects and dead trees “unhealthy?”

<u>Summary</u>: *"Forest health" is an ambiguous concept, one that is not well defined scientifically. The presence of dead or dying trees does not necessarily mean that the forest ecosystem as a whole is not functioning appropriately, even when such trees are numerous. In fact, dead trees and fallen logs perform some important ecological functions in forests, such as providing wildlife habitat and returning nutrients and organic matter to the soil. Nevertheless, dead trees are unattractive and unappealing to many people, and it can be quite painful to lose trees that have special meaning to an individual, such as large pines surrounding one's home (Figure 7).*

Although it may be relatively easy to ascertain whether an individual tree is healthy or not, the concept of “forest health" is very ambiguous. The presence of unhealthy trees does not necessarily imply that the forest as a whole is unhealthy. On the contrary, standing dead trees and fallen logs (coarse wood) play important roles in wildlife habitat, soil development, and nutrient cycling, and are a defining characteristic of old-growth forests. Bark beetle outbreaks rarely kill all of the trees in a stand, because they preferentially attack the larger trees and generally ignore the smaller trees. These smaller trees may be hidden by the red needles of the large killed trees during the peak of the outbreak, such that one often has an impression of total tree mortality. However, once those needles fall it usually becomes apparent that many small and moderate sized trees survived the outbreak. These smaller trees may grow two to four times more rapidly after the outbreak than they did before, because they are no longer competing with the big trees for light, water, and nutrients (Romme et al. 1986). In mixed forests of lodgepole pine and aspen, the aspen may grow more vigorously after beetles kill the dominant pine trees. Even when all of the trees are killed, as in a severe forest fire, the result usually is stand regeneration, as described above for lodgepole pine. Thus, from a purely ecological standpoint, dead and dying trees do not necessarily represent poor “forest health." They may instead reflect a natural process of forest renewal.

Figure 7. "Forest health" is an ambiguous concept. The presence of dead and dying trees does not necessarily mean that the forest ecosystem as a whole is not functioning appropriately. Dead trees and fallen logs perform important ecological functions, such as providing wildlife habitat and returning nutrients and organic matter to the soil. (photo by W. H. Romme)

Nevertheless, dead trees are unattractive and unappealing to many people, especially when those dead trees are abundant, and it can be quite painful to lose trees that have special meaning to an individual, such as large pines surrounding one's home. The change in the appearance of the forest after an insect outbreak also can have negative economic consequences for a community. Over time, the visual impacts are lessened as aspen and small pines grow larger and more abundant, and the gray trunks of the beetle-killed trees gradually fall to the ground. Nevertheless, the visual evidence of an insect outbreak may persist for a decade or more after the outbreak subsides.

Question #8: Does a large insect outbreak constitute an "emergency?"

Summary: *Forests naturally change slowly, almost imperceptibly, over long periods of time. But periodically this slow process of change is punctuated by rapid change via insect outbreak, fire, or other natural disturbance. The sudden death of thousands of trees may be an emergency for people and communities whose amenities, economic activities, and management plans were based on the slowly changing forest that used to occupy the area. From an ecological perspective, however, insect outbreaks are part of the natural rhythm of change in forest ecosystems, and are followed by a gradual re-development of the forest through natural ecological processes. Where aspen was present before the outbreak, the death of the pines may lead to an increase in the aspen component of the forest (Figure 8).*

The normal development of forests involves very slow changes that continue over decades or centuries. A large-scale insect outbreak or forest fire changes a forest rapidly, over a period of a few weeks or years. Such a rapid change often generates great concern about the health and future of the forest and landscape. Is this an emergency? The sudden death of thousands of trees may be an emergency for people and communities that are accustomed to the slowly changing forest that used to occupy the area. Recreational opportunities and values suddenly change, and long-term plans that relied on only slow changes in the forest (such as estimations of annual wood yield) no longer apply. Thus, these may be emergencies from certain standpoints.

From an ecological perspective, we recognize that the forest will slowly re-develop through natural processes. Many montane landscapes in central Colorado are well suited for both conifers (lodgepole pine, spruce, and fir) and aspen, and several of these species commonly occur in the same forest. A century of forest development without any major disturbance typically leads to decreasing abundance of aspen as the conifers increase in dominance. A bark beetle outbreak that kills many of the conifers may be beneficial to the aspen. Old aspen trees will likely grow faster, and new aspen will become established. An increase in aspen will occur only where aspen clones were present before the beetle outbreak. If there was not aspen already present, then composition of the forest will not change; the surviving conifers (mostly smaller individuals and non-susceptible species) will increase their growth rates and replace the large conifer trees that were killed by beetles.

The terms "ecological emergency" and "insect emergency" suggest that insect outbreaks are unforeseen events. However, insect outbreaks, even extensive ones that kill canopy trees over hundreds of thousands of acres, are natural events in forest ecosystems throughout the Rocky Mountains, and have been occurring for thousands of years (e.g., Swetnam and Lynch 1998, Lavoie 2001). The insects have long been natural components of these forest ecosystems. Therefore, from a purely ecological perspective, an insect outbreak generally would not be regarded as an "emergency," but as an infrequent but normal episode of rapid change within an ecosystem that most of the time is changing only slowly.

Figure 8. Forests naturally change slowly, almost imperceptibly, over long periods of time. But period-ically this slow process of change is punctuated by rapid change via insect outbreak, fire, or other natural disturbance. From an ecological perspective, insect outbreaks are part of the natural rhythm of change in forest ecosystems, and are followed by a gradual re-development of the forest through natural ecological processes. (photo by Dominik Kulakowski)

Question #9: How do insect outbreaks affect streamflow and water quality?

Summary: *An insect outbreak, or any disturbance that reduces the total area of leaf surface in a forest, can potentially increase streamflow by reducing the amount of interception and transpiration. No increase in streamflow is likely when the total annual precipitation is less than 18-20 inches. In areas with more than 18-20 inches of annual precipitation, an increase in streamflow generally will not be detectable unless at least 15-20% of the forest canopy is killed. By themselves, insect outbreaks are unlikely to cause erosion or degrade water quality because they do not disturb the forest soil. Unpaved roads and high-severity wildfires can cause much greater effects on runoff, erosion, and water quality (Figure 9).*

The hydrologic effects of insect infestations vary with the type of forest, the number and size of trees that are killed, and the amount and type of precipitation. The likely effects of a given change in forest density and structure can be predicted with a relatively high degree of confidence because of the long history of plot, process, and watershed scale studies in Colorado and elsewhere (MacDonald and Stednick, 2003). Over the last decade there has been a sharp increase in our understanding of how wildfires, prescribed fires, and thinning affect runoff and erosion rates in Colorado (e.g., Moody and Martin, 2001; Benavides-Solorio and MacDonad, 2005; Kunze and Stednick, 2006).

Removal of all or a part of the forest canopy may potentially increase streamflow via two mechanisms. First, the forest canopy intercepts a portion of incoming precipitation, and this intercepted rain or snow simply evaporates or sublimates back into the atmosphere without ever reaching the soil. A reduction in the forest canopy generally reduces the amount of water that is intercepted and thereby increases net precipitation (but see below for other complicating factors). Second, live trees take up water from the soil and transpire that water into the atmosphere.

Several principles determine whether a particular insect infestation or management action will significantly alter the amount and timing of runoff. First, removing the forest cover from areas that receive less than about 18-20 inches of annual precipitation will have little effect on the amount and timing of runoff as long as there are no significant changes to the infiltration rate of the soil. The primary reason for this lack of change is that any reductions in interception and transpiration are negated by an increase in soil evaporation and transpiration by any remaining vegetation (Bosch and Hewlett, 1982). Once annual precipitation exceeds about 18-20 inches, the reduction in interception and transpiration due to forest harvest or dieback will increase annual runoff, and this increase generally will be proportional to the amount of annual precipitation. Second, at least 15-20% of

Figure 9. An insect outbreak can potentially increase streamflow by reducing the amount of water transpired by trees. However, the increase probably will not be detectable unless total annual precipitation is greater than 18-20 inches and at least 15-20% of the forest canopy is killed. By themselves, insect outbreaks generally do not cause erosion or degrade water quality, because they usually do not disturb the soil. (photo by J. A. Hicke)

the forest canopy has to be killed or removed before there will be any measurable increase in annual runoff. Removing a smaller proportion of the forest cover may still increase the amount of runoff, but this increase probably will not be statistically detectable. Third, the increase in annual runoff due to forest harvest or tree death is roughly proportional to the amount of the forest canopy that is removed or killed. Fourth, the absolute changes in streamflow will be much smaller in dry years than wet years, and become harder to detect as spatial scale increases (MacDonald and Stednick, 2003).

Extrapolation of paired-watershed studies in snow-dominated areas of Colorado and Wyoming indicates that removing the forest canopy from 100% of a watershed will increase mean annual water yields as follows: by a little over 1 inch or about 18% of the mean annual runoff when the mean annual precipitation is 21 inches (Bates and Henry, 1928); by 8 inches or roughly 90% when the mean annual precipitation is 30 inches (Troendle and King, 1985); and by over 12 inches or about 70% when the mean annual precipitation is 34 inches (Troendle et al., 2001). Nearly all of this increase in water yield will come on the rising limb of the snowmelt hydrograph in May-June. Complete removal of the forest canopy can be expected to increase the size of the mean annual peak daily flow by about 40% while having minimal effect on the timing of the annual peak flow (MacDonald and Stednick, 2003).

The hydrologic effects of insect outbreaks are similar in many respects to the effects of forest harvest, but there also are some important differences (MacDonald and Stednick, 2003; Uunila et al., 2006). One difference is that under natural conditions the insect-killed trees remain in place, and this residual canopy will still intercept a portion of the incoming rain and snow, especially while the needles and fine twigs are still in place. This means that the water yield increase due to bug-killed trees will be smaller than the water yield increase due to a comparable amount of forest harvest. A second important difference is that although the insects may kill most or all of the trees within small patches of a few acres, outbreaks never kill all of the trees across a large watershed or landscape; thus, the increases in water yield following insect outbreaks will be smaller than the values listed in the previous paragraph for complete tree harvest (Schmid et al., 1991). Finally, any increase in runoff will decay over time with forest re-growth, and the time to hydrologic recovery may be shorter for an insect outbreak as compared to forest harvest. Studies in Colorado indicate that the time needed for hydrologic recovery after a clearcut varies from about 60 years in the spruce-fir and lodgepole pine zones to around half this time in aspen stands (MacDonald and Stednick, 2003). Insect outbreaks usually kill a portion of the trees, and the surviving trees may grow two to four times faster than they did before the outbreak. Therefore, canopy basal area may return to pre-outbreak levels within a shorter period of time, and this will reduce the potential increase in water yields relative to timber harvest.

Several studies have attempted to evaluate or predict the hydrologic effects of insect outbreaks in Colorado and elsewhere, but most of these studies were been hampered by the lack of a well-controlled design and the available statistical tools. After the 1939-1946 spruce beetle epidemic in the White and Yampa River basins, Love (1955) claimed that annual streamflow in the White River increased by about 2.3 inches or 22%, but this was refuted by Bue et al. (1955). Bethlahmy (1974, 1975) conducted more extensive analyses using different techniques and claimed that the beetle epidemic increased annual water yields by up to 2.0 inches in the White River basin and 2.4 inches in the Yampa River basin, and that the water yield increases were still present after 25 years. A more recent modeling study predicted that water yields would increase in the North Platte River basin by 2.2 inches if 30-50% of the trees were killed by insects (Troendle and Nankervis, 2000). While none of these studies can be considered definitive, the general results are consistent with the principles and values outlined in this section.

In terms of water quality, forested areas typically have very high infiltration rates and rarely generate surface runoff. The death of trees by insects should not compact the soil or cause a loss of the protective litter layer. In the absence of any compaction or ground disturbance, there should be minimal change in soil infiltration rates or the soil moisture storage capacity. Hence an insect outbreak should not induce overland flow or increase erosion rates, even on steep slopes. On the other hand, the increased duration of high flows due to forest harvest or dieback can increase watershed-scale sediment yields by increasing the stream's sediment transport capacity (Troendle and Olsen, 1994). In practical terms this is of little significance because the sediment yields from forested areas are typically very low (MacDonald and Stednick, 2003). In many forested areas, unpaved roads are a primary source of sediment (Libohova, 2004), and the number, location, and design of forest roads is a key control on whether thinning or harvest activities will affect water quality and aquatic ecosystems (MacDonald and Stednick, 2003; Libohova, 2004). Forest harvest and bug kill can reduce slope stability as a result of the decay in root strength (Sidle et al., 1985), but the increased susceptibility to landslides and debris flows is rarely an issue in Colorado.

Although insect outbreaks usually produce little or no soil erosion, and may have minimal impact on runoff, other disturbances may have significant impacts on soils and runoff. The effects of wild and prescribed fires on runoff and erosion depend primarily on fire severity as well as the timing and cause of peak flows. Low severity fires have minimal effects on runoff and erosion rates because these do not remove the protective litter layer and generally do not kill the larger and more mature trees. In contrast, high severity fires consume all of the protective organic layer, kill most or all of the vegetation, and can induce a water repellent layer at or near the soil surface (Huffman et al., 2001; Benavides-Solorio and MacDonald, 2005; Pietraszek, 2006). In areas with summer convective storms, peak flows and erosion rates can increase by several orders of magnitude after a high-severity fire (Moody and Martin, 2001; Libohova, 2004; Benavides-Solorio and MacDonald, 2005), and the combination of ash and sediment can severely degrade water quality (Moody and Martin, 2001; Kunze and Stednick, 2006). A series of studies in the ponderosa pine zone in the Colorado Front Range suggests that long-term sediment delivery rates from unpaved roads may be similar in magnitude to rates from periodic high-severity fires, while forest thinning has no detectable effect on runoff or erosion rates (MacDonald and Larsen, in press). In snowmelt-dominated areas high-severity fires may have a much smaller effect because soils are not water repellent under wet conditions (MacDonald and Huffman, 2004), and the number and intensity of summer thunderstorms may be lower than in mid-elevation forests. Hence the hydrologic effects of fires in the higher-elevation forests may be more similar to the effects of forest harvest, but there are few data from these higher-elevation sites.

Potential Treatment Options

Even though the insect outbreaks now occurring in Colorado generally cannot be regarded as *ecological* emergencies, there is no denying that the extensive stands of dead and dying trees do affect the aesthetic and economic attributes of many forests. Moreover, forest fires may cause serious damage to property and may even threaten human lives – whether or not previous insect activity has caused those fires to be more severe than they would be otherwise. Therefore, efforts to reduce the impacts of insects and fires are warranted in many areas. The following sections describe and evaluate the likely effects of a range of treatments that have been used or proposed to ameliorate the effects of insect outbreaks and fires.

Option #1: Spraying with Insecticide

<u>Summary</u>: *This can be an effective means of saving high-value trees in localized areas, but is not feasible over large landscapes (Figure 10).*

Spraying trees with an appropriate insecticide can be an effective means of preventing bark beetle attack or reducing defoliator damage. County extension agents and personnel of the Colorado State Forest Service and USDA Forest Service can recommend the best products to use against a particular insect in a particular area.

This may be the best means available for protecting high-value trees around homes, in town parks, or other localized places. However, there are limits to what can be accomplished by spraying insecticides. Annual spraying, or even spraying several times in a single year, is required to prevent attacks by each successive

Figure 10. Spraying with insecticide can be an effective way to preserve high-value trees, such as around a home. However, spraying is not feasible or effective in stopping insect outbreaks over large landscapes. (photo by W. H. Romme)

generation of insects. Spraying is not feasible at the scale of an entire forest landscape because of cost and difficulty of hitting all of the places where insects may be present. In addition, insecticides are not entirely species-specific: a broad-scale spraying of insecticides will kill many harmless and beneficial insects, such as pollinators and butterflies, in addition to the target bark beetles and defoliators. In general, bark beetle preventive sprays have less impact on non-target insects than do insecticide sprays used to control defoliatiors, because the former sprays are targeted to the trunk of the tree whereas the latter sprays need to cover entire tree canopies.

Option #2: Preventing or controlling outbreaks through forest management

Summary: *Removing stressed or unhealthy trees, and thinning to prevent crowding and competition among trees, can effectively reduce the risk of an insect outbreak getting started in a forest stand. Forest management is unlikely to prevent all outbreaks, however, because (i) it will never be feasible to intensively manage all of the forests of Colorado, and (ii) drought and warm temperatures are also important causes of outbreaks. Once an outbreak has begun, management generally cannot stop it, because the insects are numerous enough to overcome even healthy trees (Figure 11).*

Because outbreaks may initiate in stressed or unhealthy trees, intensive forest management focused on regular removal of old or unhealthy trees may reduce the likelihood of an insect outbreak getting started in a stand. Thinning may reduce tree-to-tree competition, increase tree vigor, and thus provide an enhanced ability of trees to defend against an attack (Amman and Logan 1998, Schmid and Mata 2005). If periodic harvest removes large trees and maintains a preponderance of small-diameter trees, this too may help prevent the start of a bark beetle outbreak, since bark beetles (but not defoliators) prefer larger trees. Thus, careful forest management, including appropriate timber harvest, may help locally to prevent the onset of an outbreak (Cole et al. 1976).

By itself, however, forest management probably cannot prevent all insect outbreaks -- for two reasons. First, it is unlikely that all stands in Colorado landscapes will be managed intensively enough to remove all of the stressed trees in which an outbreak can get started; in fact, the public values “unmanaged” forests that contain large and old trees. Second, drought and warm temperatures are major causes of bark beetle outbreaks, and forest management by itself cannot entirely overcome these climatic effects. And it is important to recognize that once an extensive bark beetle outbreak has started, it is unlikely that timber management can stop it. Under outbreak conditions, the beetles can overwhelm even the healthiest trees, so selective removal of weak or stressed trees will

Figure 11. Removing stressed or unhealthy trees, and thinning to prevent crowding and competition among trees, can effectively reduce the risk of an insect outbreak getting started in a forest stand. Forest management is unlikely to prevent all outbreaks, however, because it will never be feasible to intensively manage all of the forests of Colorado, and drought and warm temperatures are also important causes of outbreaks. (photo by W. H. Romme)

likely have little impact. Most entomological evidence indicates that once an outbreak has started, there is nothing that can be done to stop it. The outbreak ends when there are no more suitable trees for the beetles, or when unusually cold conditions kill beetle populations. Intensive even-aged management was applied to lodgepole pine forests in the Targhee National Forest, along the western boundary of Yellowstone National Park, from the 1960s through 1980s; yet a mountain pine beetle outbreak that swept through the region in the 1970s and early 1980s appeared to affect the managed Targhee stands as severely as the unmanaged stands in Yellowstone Park (Romme et al. 1986). Similarly, the lodgepole pine forests of British Columbia, Canada, are now being affected by a very extensive and severe mountain pine beetle outbreak, despite a long history of intensive forest management in this province.

Figure 12. Removing dead trees and other fuels can effectively reduce the risk of fire damage at a local scale, e.g., in the immediate vicinity of a home or community. However, the effectiveness of harvest in reducing fire risk to homes and communities over larger areas, e.g., a forest landscape, is less clear. (photo by W. H. Romme)

Option #3: Harvesting insect-killed trees to reduce wildfire risk

Summary: *Removing dead trees and other fuels can effectively reduce the risk of fire damage at a local scale, e.g., in the immediate vicinity of a home or community. However, the effectiveness of harvest in reducing fire risk over larger areas, e.g., a forest landscape, is less clear. Conventional timber harvest may do little to reduce fire risk at any scale if it removes primarily large trees, because smaller trees, brush, and dead fuels often are the major carriers of a spreading fire. Harvesting smaller trees and removing small fuels may more effectively reduce fire risk (Figure 12).*

As with the spraying and forest management options, the effectiveness of this option varies with the scale at which it is applied. Removing dead trees – plus other flammable material (including wood roofs and decks, woodpiles and burnable vegetation) from the immediate vicinity of a home or other vulnerable structure -- has been shown to be effective in protecting the structure from wildfire (Cohen 2000). The local characteristics of a home's external materials and adjacent fuels are the primary determinant of home ignitability -- not spatially extensive wildland fuel conditions. For example, the heat released even from intense crown fires will not ignite wooden walls at distances greater than 40 meters away (Cohen 2000). Fuel reduction around a home needs to focus not just on the dead fuels (e.g., the insect-killed trees), but often needs to include some of the live fuels (living trees and shrubs) which also carry fire under severe fire weather conditions. Specific guidelines for reducing fire risk around a home can be found at the Firewise website (Firewise.org) or from extension agents or the Colorado State Forest Service.

Moving up to a broader scale, however, the effectiveness of harvesting insect-killed trees to reduce fire risk across an entire forest landscape is far less certain than the effectiveness of Firewise techniques to protect an individual home. This is especially true in high-elevation forests such as lodgepole pine and spruce-fir. Commercial tree harvest typically involves the removal of large fuels (tree trunks) rather than smaller fuels (branches and needles) due to economic and logistical constraints. These smaller fuels contribute to ignition and spread of fire (e.g., to start a campfire one begins with tinder and kindling). Smaller surface and ladder fuels are important precursors to crown fire initiation (Agee and Skinner 2005). Hence, harvesting tree trunks has little effect on the risk of fire ignition or spread. It is true that if tree harvest also results in reduced canopy bulk density, this may make it more difficult for crown fires to spread. Nevertheless, it is the fine fuels (on the ground or in the canopy) that have the greatest influence on fire initiation and spread, not the large pieces of wood. Thus, management of fine surface or ladder fuels (which is usually time-consuming and expensive) would have the greatest impact on fire spread and potential high-severity crown fire.

It is important to acknowledge that traditional timber management usually is not designed or intended to reduce crown fire risk, but to produce wood fiber in an economically sustainable manner. Although anything that thins the canopy without greatly increasing the amount of fine fuels can reduce fire spread and intensity during moderate weather conditions (Graham et al. 2004), the most damaging wildfires typically occur under extreme conditions of wind and drought. Most traditional harvesting techniques (including overstory removal and individual tree selection) do not effectively reduce fire severity under extreme fire weather conditions (Stephens and Moghaddas 2005). In the 2002 Hayman fire, pre-fire harvesting where residual fuels (small, non-merchantable material) had not yet been removed, actually contributed to higher severity fire compared to unmodified areas (Omi and Martinson, 2002). If the goal is to reduce fire risk, removal of small trees either via mechanical thinning or prescribed fire (or a combination of both), plus retention of large, old-growth trees, can lower expected fire severity (Stephens and Moghaddas 2005, Agee and Skinner 2005). For example, portions of the 2002 Rodeo-Chediski fire in Arizona experienced lower fire severity where prescribed burning and other management activities during the previous decade had reduced fine fuels and small trees, but had left larger trees intact (Finney et al. 2005). Much of the research on thinning and underburning effects on subsequent wildfire severity has primarily been conducted in low-elevation, dry-forest types: similar effects cannot be assumed in high-elevation forests.

A single thinning treatment cannot maintain lowered wildfire risk over the long-term, because thinning typically stimulates rapid growth of the vegetation that is not taken (Graham et al. 2004). Research shows, for example, that past timber harvesting in ponderosa pine forests is responsible in part for the high densities we witness today (Kaufmann et al. 2000, Gruell et al. 1982, Baker et al. 2006). Although low-intensity prescribed burns reduce fine fuels in the short-term, they also contribute to subsequent dead fuels by killing understory trees, which can result in fuel levels that exceed pre-burn levels within a decade (Agee 2003). Therefore, repeated or staged prescribed fire or mechanical thinning treatments are essential for maintaining lower forest densities; otherwise, a one-time thinning may facilitate dense tree establishment.

Thus, it may be possible to reduce fire intensity and to obtain some control of fire spread patterns across a forest landscape by strategic placement of appropriate timber harvest activities, which may need to focus more on removal of small trees than of commercially valuable sawtimber (Finney 2001, Stratton 2004, Graham et al. 2004). Research is underway to develop specific prescriptions for effective use of vegetation management to alter wildfire intensity and spread at the scale of an entire forest landscape, e.g., at the U.S. Forest Service's fire laboratory in Missoula, MT (Mark Finney, personal communication). Another recently developed tool is the Fuel Treatment Evaluator, a web-based program that uses standard U.S. Forest Service inventory data to identify locations offering the greatest opportunities for hazardous fuel reduction activities (Wayne Shepperd, personal communication). However, this research is still in the early stages, and most has been conducted in only a few forest types (notably drier, lower-elevation forests like ponderosa pine). Thus, it is difficult at this time to make confident predictions of how a specific forest treatment will affect fire behavior under a range of forest types and fire weather conditions.

A major source of uncertainty about the effectiveness of landscape-level fuel treatments in altering fire behavior, is the fact that extreme fire weather can over-ride fuel effects (as seen, for example, in Hayman 2002, Routt National Forest 2002, and Yellowstone 1988). In the Hayman fire, most of the vegetation treatments that had been implemented prior to the fire had very little impact on the severity or direction of the fire during the extreme weather conditions of June 9th and 18th, which were the two days when the majority of the area burned (Finney et al. 2003). It should be noted that not all previous vegetation treatments in the Hayman area had been designed to mitigate fire behavior, but were implemented for other objectives such as timber stand improvement -- further illustrating the point that not all timber harvest activities can be assumed to reduce fire hazard. In the 1988 Yellowstone fires, once fuels reached critical moisture levels, the spatial pattern of burning was largely controlled by weather (wind direction and velocity), rather than by fuels (Minshall et al. 1989, Turner et al. 1994). A study of the 2002 fires in Routt National Forest in Colorado found that previous salvage logging had no detectable influence on fire extent or severity during the extreme drought conditions (Kulakowski and Veblen 2006).

In sum, there is no doubt that Firewise activities in the immediate vicinity of vulnerable structures can increase their survivability in a forest fire (though it must be recognized that the risk of fire damage can never be reduced to zero). However, it is far less certain how effective fuel reduction treatments at greater distances from homes will be in protecting those homes. We also note that timber harvest may be conducted for more purely ecological objectives rather than or in addition to protection of homes. In some types of forests, notably Southwestern ponderosa pine, thinning of overly dense small trees can reduce the risk of stand-replacing wildfire and also contributes to a larger goal of forest restoration (Friederici 2003, Schoennagel et al. 2004). But in other forest types, notably lodgepole pine and spruce-fir, thinning of small trees does not represent restoration of more natural conditions, because these kinds of forests are naturally dense and naturally burn at high intensities; fuel management also has less influence on fire behavior in these ecosystems where climate so strongly controls fire occurrence and severity (Schoennagel et al. 2004). We emphasize again the importance of distinguishing among forest types in evaluating the opportunities and impacts of forest management for wildfire mitigation and ecological restoration.

Option #4: Salvaging insect-killed trees to improve overall forest health

<u>Summary</u>: *From a purely ecological standpoint there usually is little or no need to remove insect-killed trees. However, many people do not like to see great numbers of dead trees surrounding their communities or places they like to visit. If the dead trees have a negative impact on aesthetic preferences or local economics, then it may be desirable to remove them (Figure 13).*

As discussed above, “forest health” is an ambiguous concept. From a purely ecological standpoint there usually is little or no need to remove insect-killed trees. In fact, standing snags and fallen logs actually contribute to a number of ecological and aesthetic values in forests, including maintenance of "natural" forest structures and processes, protection of soils and water quality, and preservation of species at risk from the effects of roads, exotic species, and habitat alteration. For example, the three-toed woodpecker feeds on bark beetles in dead and dying trees, and nests most successfully in areas of recent fire or beetle outbreak. Withdrawing all or most of the large dead trees after a fire or insect outbreak will reduce habitat quality for this and other species.

At the same time, there is a widespread public perception that a forest filled with dead or

Figure 13. From a purely ecological standpoint there usually is little or no need to salvage insect-killed trees in the interest of improving forest health. However, if the dead trees have a negative impact on aesthetic preferences or local economics, or if timber production is an important goal in an area, then it may be desirable to remove the dead trees. (photo by J. A. Hicke)

Figure 14. Salvage of insect-killed trees may be a preferred option in some areas because of the economic value of the timber product that can be obtained. In these situations, especially where lodgepole pine trees have been killed by mountain pine beetles, the dead trees must be harvested as soon as possible, because the wood quality deteriorates rapidly after the trees die. (photo by D. Binkley)

dying trees is “unhealthy,” and many people do not like to see great numbers of dead trees surrounding their communities or in places that they like to visit. Whether or not this perception is consistent with what we know about forest ecology, it nevertheless has an impact on aesthetic preferences and local economics. Visitors may choose not to come to a resort surrounded by dead trees; home buyers may avoid locations where the view is one of sick and dying trees. For these and other reasons, efforts to reduce tree mortality (options 1 and 2) and to remove the unsightly results of that mortality (this option), will be the preferred response to insect outbreaks in some locations.

Option #5: Salvaging insect-killed trees for economically valuable products

Summary: *Salvage of insect-killed trees may be a preferred option in some areas because of the economic value of the timber product that can be obtained. In these situations, the trees usually must be harvested as soon as possible, because the wood deteriorates rapidly after the trees die (Figure 14).*

Although salvage of insect-killed trees usually is not necessary for the normal development of the forest, it may be a preferred option in some areas because of the economic value of the timber product that can be obtained. Harvest of large trees for economic reasons can be done in ways that minimize adverse ecological impacts, e.g., by laying out harvest units in spatial patterns that mimic the patterns created by natural disturbances such as fire (e.g., Kohm and Franklin 1997, Friederici 2003, Romme et al. 2003, Perera et al. 2004). If ponderosa pine or lodgepole pine killed by mountain pine beetles are to be salvaged for their timber value, they must be harvested as soon as possible, because the wood deteriorates rapidly after the trees die. However, spruce trees killed by spruce beetles may remain merchantable for decades (Wayne Shepperd, personal communication).

Option #6 -- No treatment

Summary: *Natural ecological processes generally lead to the development of new forests after insect outbreaks, so a "no treatment" option can be a form of responsible forest management (Figure 15).*

Natural ecological processes generally lead to the development of new forests after insect outbreaks and fires, without salvage logging or other operations, so post-outbreak or post-fire treatment usually is unnecessary from a purely ecological perspective. Other choices may be made for other reasons, such as including a logging program to salvage economic value from dead trees or to create more desirable visual conditions (options 4 and 5 above).

Figure 15. Natural ecological processes generally lead to the development of new forests after insect outbreaks, as in this lodgepole pine forest 30 years after a bark beetle outbreak killed more than 50% of the canopy. Thus, a "no treatment" option can be a form of responsible forest management. (photo by W. H. Romme)

Nevertheless, a "no treatment" option can be a form of responsible forest management.

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forest policy

Implementing the 2012 Forest Planning Rule: Best Available Scientific Information in Forest Planning Assessments

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National forests and grasslands in the United States are governed by land and resource management plans that should be updated every 15 years to reflect changing social, economic, and environmental conditions and to address new priorities. A new forest planning rule finalized in 2012 introduces new planning approaches and requirements, and several forests have completed the forest assessment phase of their planning process. Using document analysis and interview data, we analyzed four completed forest assessments to gain insights into early forest planning efforts under the 2012 rule. We found that forest assessments address the required topics, although the organization and depth of treatment varies across cases; government sources and academic publishers are relied on most often as sources of scientific information; and approaches to best available scientific information rely on peer-reviewed information, agency technical reports and syntheses, and personal expertise and judgement.

Keywords: early adopter, expertise, US Forest Service

Management of the 154 national forests and 20 grasslands in the United States is governed by land and resource management plans (also called forest plans), as required by the National Forest Management Act of 1976 (NFMA; 16 U.S.C. 1604). The forest plan functions as a guiding document that outlines goals, objectives, and strategies for management of the unit. Periodically, the rule related to forest planning is revised to reflect societal changes, new approaches and technologies, and scientific discoveries. For many years the US Forest Service (USFS), which manages the system of national forests and grasslands, has operated under a planning rule finalized in 1982 (47 FR 43026) despite several efforts (2000, 2005, and 2008) to revise and improve the rule (Schultz et al. 2013). A new planning rule issued in April 2012 (77 FR 21161) introduces several significant changes, including a renewed emphasis on collaboration, improved transparency, and a strengthened role for public involvement throughout the planning process. Of interest for our study is the requirement to use the best available scientific information (BASI) to inform the assessment, plan revision decisions, and monitoring program.

To date, little research has addressed implementation of the 2012 planning rule. Schultz et al. (2013) examined approaches to wildlife conservation planning under the new rule, raising concerns regarding potential extirpation of species. Another study analyzed public participation processes in 12 national forests (University of Montana 2015), and Schembra (2013) explored the role of standards and guidelines and how they are used in planning activities. Forest planning under the 2012 rule consists of three phases (assessment, plan development, and monitoring). The assessment phase is important, as it assembles relevant scientific information that planners will rely on to make decisions on forest management in the plan development phase. Our study contributes to this growing body of knowledge by examining the assessment phase of the forest planning process.

Eight "early adopter" national forests, along with several other forests, are currently developing their forest plans using the 2012

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rule. These forests were designated as early adopters because they provide important benefits, had strong existing collaborative networks in place, and needed to revise their forest plans (USDA Forest Service 2012a). The eight early adopter forests are: Cibola (NM), Chugach (AK), El Yunque (PR), Nez Perce and Clearwater (ID), and three forests that are coordinating planning on a regional basis: Inyo, Sequoia, and Sierra (CA).

Although implementation is still in early stages, several of the early adopter forests have completed their forest assessments and draft forest plans, which presents an opportunity to study implementation of the planning process under the new rule. One forest (the Francis Marion in SC) has completed the full plan revision process as of this writing. We examined four forests that have completed their assessments, including three forests identified by the agency as early adopters and one forest that is keeping pace with this group. The study explored three questions: 1) What does the 2012 planning rule require regarding the structure, content, and process for forest assessments? 2) How have forests implemented the directives related to forest assessments under the 2012 planning rule? 3) How are forests approaching the requirement for the use of best available scientific information in their assessments?

Forest Planning under the 2012 Rule

The 2012 planning rule suggests an adaptive approach to forest planning, instructing managers to 1) *assess* forest conditions; 2) *revise or amend* plans if the assessment indicates a need for change; and 3) *monitor* plan implementation (36 CFR 219.5). The process is cyclical, with monitoring data feeding back into the assessment of conditions in the management unit (USDA Forest Service 2012b). During the assessment phase, planners are expected to "rapidly evaluate existing information about relevant ecological, economic, and social conditions, trends, and sustainability, and their relationship to the land management plan within the context of the broader landscape" (36 CFR 219.5(a)(1)). The second phase of the planning process is plan development, amendment, or revision, where planners use the results of the assessment to establish a need for change and generate planning alternatives (36 CFR 219.5(a)(2)), and the public has the greatest opportunity for input. The plan development phase includes environmental impact assessment, public input, and plan publication (36 CFR 219.5(a)(2)). The third phase (monitoring) is an opportunity to track and measure management effectiveness over time (36 CFR 219.5(a)(3)). The planning *process* under the 2012 rule is similar to the process specified under the 1982 rule, but differs in terms of the specific elements required for the *assessment* (2012 rule) and the *analysis of the management situation* (the assessment's counterpart in the 1982 rule).

We focused our study on the assessment phase of the planning process. The assessment phase is important because it requires the forest to assemble and synthesize the most recent, relevant, and highest-quality science on social, ecological, and economic conditions to inform the plan development. Not only does this provide planners an opportunity to evaluate changes in biophysical and socio-economic conditions based on the latest monitoring data, it also represents a chance to reflect on new concepts, models, and methods that result in new scientific information about the local forest environment. Under the 2012 planning rule, the assessment phase identifies existing conditions, trends, risks, uncertainties, and information gaps that are relevant to land and resource management issues in the unit (36 CFR 219.5–219.6). In the assessment phase, the planning unit is not required to generate new studies or information, but is expected to obtain pre-existing information that is publicly available or voluntarily provided (36 CFR 219.6). Information can come from government and nongovernment sources, and the rule instructs the Forest Supervisor to provide opportunities for stakeholders to provide information for the assessment (36 CFR 219.6). The primary product of the assessment phase is an assessment document that evaluates existing information for 15 specific topic areas (Figure 1). Although the general topic areas are mandated by the 2012 rule, the Forest Supervisor has discretion to determine the scope, scale, and timing of the assessment, assuming the other requirements in the planning rule are followed (36 CFR 219.6).

Role of Science in Natural Resource Management

Historically, natural resource management in the United States was guided by the idea of scientific management and Progressive-era approaches (Taylor 1896). In particular, Samuel Hays's "gospel of efficiency" relied on a rational and scientific method of making decisions through a single, central authority. The thought was to avoid conflict via a scientific approach to social and economic issues (Hays 1959, p. 267). The US Forest Service exemplifies the approach of technical rationality and empirical science as the basis for sound resource management practices (Wellman 1987; Kaufman 1960). Foresters and natural resource managers

Management and Policy Implications

Although implementation of the US Forest Service's 2012 planning rule is still in the early stages, several national forests have completed the assessment phase and moved on to the next phase of forest planning. Our analysis of forest assessments from several "early adopter" forests illustrates that forest planners are making serious efforts to address required topics and rely on the best available scientific information. Assessment reports were disproportionately heavy in science related to terrestrial and aquatic ecosystems, and more limited in treatment of infrastructure, land ownership and access patterns, cultural heritage, and areas of tribal importance. Ensuring that assessment teams include broad and diverse disciplinary experts will help address this challenge, recognizing that some forests may not have access to necessary disciplinary specialists. It is also possible that some of the topics (e.g., ecosystem services, tribal and cultural resources, land status and use patterns) simply do not have as much relevant and available information as other topics. Assessment teams may want to consider additional ways to interact with scientists and others to create functioning communities of practice related to science exchange for forest planning. In the same way, agency scientists may consider forging new and enduring relationships with planners and managers that could generate new science that is of immediate relevance. We found similarities across all forests in the most common approaches to identifying BASI in addition to other approaches such as data sharing meetings, a wiki review site, and requests for a science synthesis. Information from non-peer-reviewed sources was more difficult for planners to assess and evaluate. Sharing best practices, along with revised guidance for planning rule implementation, may help national forest planners improve the utility, efficiency, and quality of forest assessments.

- Terrestrial ecosystems, aquatic ecosystems, and watersheds
- Air, soil, and water resources and quality
- System drivers, including ecological processes, disturbance regimes, and stressors
- Baseline carbon stocks
- Threatened, endangered, proposed and candidate species; potential species of concern
- Social, cultural, and economic conditions
- Benefits people obtain from the planning area (ecosystem services)
- Multiple uses and their contributions to economies
- Recreation settings, opportunities, and access, and scenic character
- Renewable and nonrenewable energy and mineral resources
- Infrastructure (recreational facilities, transportation and utility corridors)
- Areas of tribal importance
- Cultural and historic resources and uses
- Land status, ownership, use, and access patterns
- Existing designated areas including wilderness and wild and scenic rivers; need and opportunity for new designations

Figure 1. Topics for forest plan assessments (36 CFR 219.6)

are expected to incorporate state-of-the-art scientific knowledge to manage public lands (Lachapelle et al. 2003). However, the role of science in natural resource decision-making has become much more complex (Mills and Clark 2001). Recent literature acknowledges that no important policy issue or decision is purely technical, that established practices are problematic, and that politics are unavoidable (Brunner et al. 2005). In spite of this, numerous policies reflect the scientific management paradigm in their calls for best available science.

In the United States, many policies and statutes contain references to best available science, including the Marine Mammal Protection Act of 1972, the Endangered Species Act of 1973, and the Safe Drinking Water Act of 1974. Despite references to the concept of best available science, these policies do not include specific definitions of its properties, standards, or practical application in the decision-making process (Doremus 2004; Smallwood et al. 1999), leading to different definitions of what it means. Ryder et al. (2010) identify attributes of best available science from published literature that span topics such as endangered species legislation, protection of conservation areas, forest management, water resource management, and ocean fisheries. The paper highlights the diversity of attributes assigned to best available science, and demonstrates that no single attribute is common to all studies, suggesting that best available science is context specific (Ryder et al. 2010). Moreover, as Lowell and Kelly (2016) observe, the ability to use best available science may be inhibited by institutional constraints within particular agencies limited by time or organizational capacity. Other literature has attempted to assign descriptors to the concept. For example, "best" often connotes scientific information with the greatest degree of excellence and authenticity based on sound logic (Moghissi et al. 2010), or that there is no better scientific information, and suggests the use of the most relevant and contemporary data and methods (National Research Council 2004). "Available" connotes scientific information that is accessible and attainable (Moghissi et al. 2010), or that decisions can be consistent with the scientific information that is available even though data gaps exist (National Research Council 2004). "Science or Scientific information" is defined as knowledge that emerges from a process of observation, identification, description, and testing of explanatory hypotheses about fundamental principles that govern cause-and-effect (National Research Council 2004). The National Research Council report includes guidelines for effectively using best available science, including concepts of relevance, inclusiveness, objectivity, transparency and openness, timeliness, and peer review. Finally, Charnley et al. (2017) analyzed a science synthesis for three national forests and suggest criteria for evaluating "best available *social* science," which may be different from the criteria used to evaluate best available biophysical science.

A key aspect of the 2012 planning rule is that it requires the planning process to draw on the best available scientific information (36 CFR 219.3). The preamble to the planning rule notes that there is a range of information that can be considered BASI, stating:

> In some circumstances, the BASI would be that which is developed using the scientific method, which includes clearly stated questions, well-designed investigations and logically analyzed results, documented clearly and subjected to peer review. However, in other circumstances the BASI for the matter under consideration may be information from analyses of data obtained from a local area, or studies to address a specific question in one area. In other circumstances, the BASI also could be the result of expert opinion, panel consensus, or observations, as long as the responsible official has a reasonable basis for relying on that scientific information as the best available. (77 FR 21192 [April 9, 2012])

Planning Directives are agency guidance documents that direct implementation of rules such as the 2012 planning rule, and directives for assessments are in Chapter 10 of the Land Management Planning Handbook (USDA Forest Service 2015a). The definition of BASI is contained in the "zero code" chapter of the handbook and specifies three primary criteria for determining BASI: accuracy, reliability, and relevance (FSH 1909.12.07.12), in addition to referencing the Data Quality Act (PL 106–554) for guidance on evaluating available information (Figure 2). Available is defined as information that currently exists in a form useful for the planning process without further data collection, modification, or validation (FSH 1909.07.01).

The directives also provide guidance regarding sources of scientific information. The sources mentioned in the guidance include peer-reviewed articles, scientific assessments, other scientific information (expert opinion, panel consensus, inventories, or observational data), data prepared and managed by the Forest Service

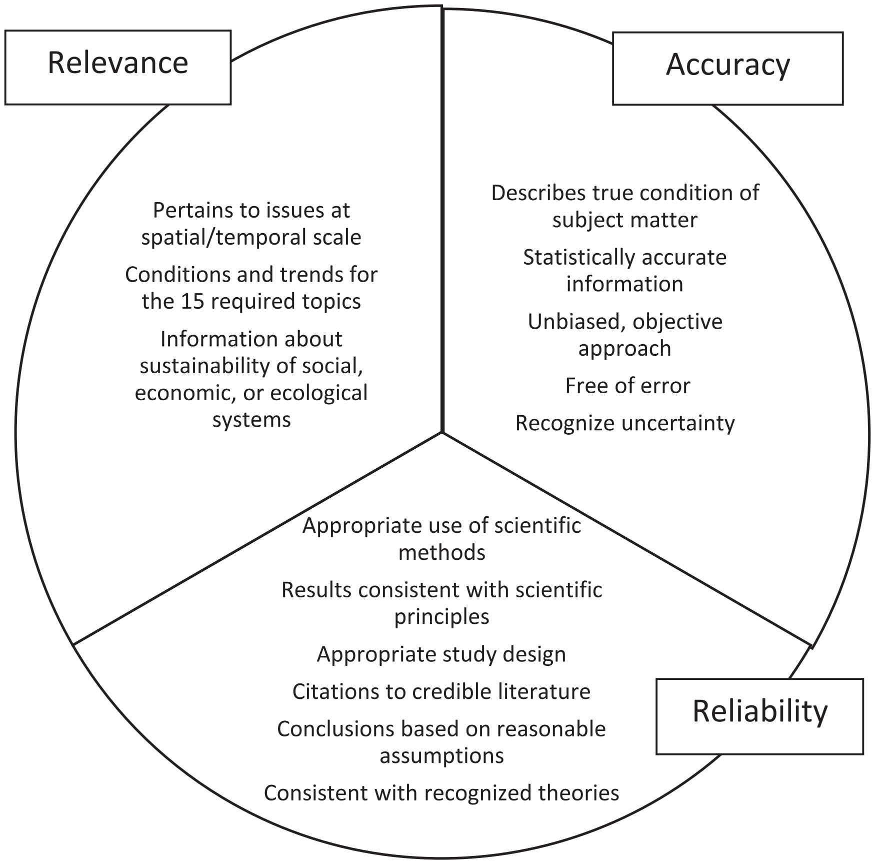

Figure 2. Criteria for determining best available scientific information (BASI). Source: Forest Service Handbook 1909.12.07.12

or other federal agencies, information prepared by universities, national research networks, and other reputable scientific organizations, and data or information from public and governmental participation (FSH 1909.12.07.13).

At the US Forest Service, two regional science synthesis efforts were initiated to assist forest planners in identifying BASI for their assessments. The first synthesis included the Sierra Nevada, southern Cascades, and Modoc plateau areas of California, and informed plan revisions on three national forests (Long et al. 2014). The second synthesis is currently underway as part of the Northwest Forest Plan area planning process, which covers 17 national forests and five Bureau of Land Management units across parts of the Cascade and coastal ranges of Washington, Oregon, and northern California. Once drafted, the synthesis report underwent independent third-party peer review, in addition to public review, and is currently under revision (Spies et al. 2017). Science synthesis efforts represent a noteworthy approach to developing BASI for use in forest assessments, creating a role for public engagement, and for employing a bioregional approach to assembling the latest science for use by multiple forests.

Methods

We used an exploratory case study approach to examine four national forest planning units that were revising their forest plans under the 2012 rule. Information on the USFS website helped us determine the planning status of each national forest as of spring 2015. The primary selection criterion was completion of the assessment process by spring 2015. We also strove to select national forests from different regions. Based on these criteria, we selected the Chugach National Forest (Alaska), Cibola National Forest (New Mexico), Inyo National Forest (California), and the Nantahala and Pisgah National Forests (North Carolina). Table 1 displays characteristics of each national forest planning unit in our sample.

Our research approach relied on content analysis of documents and interview data. We began by conducting a chapter-by-chapter analysis of each forest's assessment report to identify and characterize the information presented. We recorded page counts for each of the 15 assessment topics specified in the 2012 rule. In some cases, the chapters directly aligned with the required topics (Figure 1). In other cases, we had to make a more subjective characterization of the chapter contents. We also noted and analyzed any references to the use of best available science.

Second, as part of the document review, we analyzed data sources used in the assessment. For each assessment report, we identified all of the items cited in the reference section. We then coded each cited item according to the type of publishing entity and the type of document. Every cited item was placed in one category for each coding exercise. For each cited item, we determined the appropriate categories by examining the information in the citation entry and (when necessary) directly reviewing the item or gathering information on the publishing entity. We grouped publishing entities into five types: government; non-government; scientific, scholarly, or peer-reviewed; universities; and unknown or other (Table 2). This categorization approximates the rigor of scientific review, but there is overlap in categories. Most scholarly journals require a double-blind peer-review process, where reviewers and authors are

Table 1. Characteristics of national forests in the study.

Management unit(s)	Geography	Total acreage* (millions of acres)	Notes on use and resources	Designated early adopter?	Most recent previous plan revision	Notes on current plan revision
Chugach National Forest Alaska Region (R10)	Southcentral Alaska: major geographic areas are Cooper River, Prince William Sound, and eastern Kenai Peninsula	6.26	Subsistence, timber, recreation, mining. Human use concentrated in Kenai area. Very limited road coverage and use in other areas. Habitat for all 5 Pacific salmon species	Yes	2002	Managed by a planning team housed within unit
Cibola National Forest Southwest Region (R3)	West-Central New Mexico: Eight noncontiguous parcels organized around distinct mountainous areas known as "sky islands"	2.11	Recreation, timber, cultural heritage, range. Surrounding region experiencing population growth and demographic changes. Pinyon-juniper & ponderosa pine are predominate vegetation types	Yes	1985	Managed by a planning team housed within unit. Does not include 4 associated national grasslands
Inyo National Forest Pacific Southwest Region (R5)	Eastern California & West Nevada: Two noncontiguous parcels at intersection of Sierra Nevada, Great Basin, and Mojave Desert areas	2.07	Water supply, hydroelectricity, recreation, timber, range. Nearly 47% of total area is wilderness. Focus on wildland fire management. Substantial variation in vegetation type, habitat, and elevation	Yes	1988	One of three early adopters in R5. Coordination through a regional planning team, with separate planning teams for each unit. Each unit releases its own assessment & forest plan. Joint EIS for 3 units
Nantahala & Pisgah National Forests Southern Region (R8)	Western North Carolina: Blue Ridge region of Appalachian Mountains	2.48	Timber, recreation, cultural/historical heritage, water development. Located in Blue Ridge National Heritage Area. Hardwood forest with high species diversity	No	1987	Both units will use same revised plan. Managed by planning team housed at NF in NC headquarters

*Total acreage includes NFS-owned land and acreage under other ownership within each unit. Source: USDA Forest Service 2015b.

Table 2. Categories for coding type of publishing entity.

Publishing entity	Description of coding criteria
Government	Federal, tribal, state, or local governments in the United States; foreign governments; international intergovernmental groups such as the United Nations and affiliates. Includes peer-reviewed and non-peer-reviewed materials
Non-government	Materials not published by a government agency, university, or peer-reviewed entity. Includes businesses, consulting firms, and advocacy groups
Scientific scholarly or peer reviewed	Associations, societies, journal publishers, university presses, or other entities that produce peer-reviewed scientific or scholarly material
Universities	Materials from universities that may or may not be subject to rigorous academic peer review. Includes university or college departments, programs, laboratories, and centers, and theses and dissertations from universities
Unknown or other	News organizations or other undefined groups; disposition of publisher could not be determined

unknown to each other. University and government agency scientific documents often require peer review, but the level of rigor of the review may be variable. It was not possible to discern the level or type of peer review or scientific rigor for each category.

For the type of document, we sorted the references into 12 categories: academic book; non-academic book; conference proceeding; correspondence; database; scientific journal; news; technical report; statute or regulation; thesis or dissertation; website; and unknown (Table 3).

Our final data collection activity was qualitative interviewing with members of the planning teams at three of the forests in our study. We conducted nine semi-structured interviews (nine people in total; three interviews each from three forests). Unfortunately, we were not able to recruit interview participants from the Cibola planning effort. Potential interview participants were identified through the list of preparers included in each assessment document. Interviewees were subject matter experts who had contributed material to the assessment reports, along with planning staff officers or coordinators. Interview questions explored the overall structure of the assessment process, the role of the planning directives, the overall organization of the forests' plan revision efforts, and approaches to identification and use of best available science. Interviews were audio-recorded, transcribed, and analyzed using content analysis with a coding framework developed by the study team. Content analysis is a method that uses codes, or labels that assign meaning to descriptive or inferential data collected during a study (Miles et al. 2014). The codes are used to retrieve and organize similar data and aid the researcher in relating data to research questions, theoretical concepts, and themes (Araujo 1995; Miles et al. 2014).

Results

We present results of our analysis in three sections: 1) required topics; 2) sources and types of information; and 3) identifying and using BASI.

Required topics in the forest assessment

The number and percent of pages devoted to each required topic is presented in Table 4. We did not include introductory front matter in the page counts. A 0* entry means that the assessment report did not

Table 3. Categories for coding type of document.

Document type	Description of coding criteria
Academic book	An item printed, bound, distributed as a book, or released as an e-book by a peer-reviewed/scholarly entity
Non-academic book	An item printed, bound, distributed as a book, or released as an e-book by an entity whose primary orientation is not peer reviewed/scholarly
Conference proceeding	Papers, abstracts, and talks presented at a conference and published in a conference proceeding collection
Correspondence	Letters or emails written by individuals of any affiliation
Database	Raw data or data analysis tools/software; online databases
Scientific journal	A peer-reviewed article in a scholarly journal
News	Articles in newspapers (print or online) and news magazines
Technical report	Technical and research reports, white papers, policy papers, fact sheets, briefings
Statute, regulation, and planning documents	Federal, state, or local laws and rules; EISs; management plans; strategic plans
Thesis or dissertation	Advanced degree projects and papers
Website	One or more webpages on a non-database website, including encyclopedias with narrative entries
Unknown	The type of document could not be discerned

Table 4. Page counts and percentages of total pages for 15 required assessment topics.

		Number of pages (pct. of total pages in report)				Pct.
Topic #	Assessment topics (per 36 CFR 219.6)	Chugach	Cibola	Inyo	N&P	Avg.
1	Terrestrial ecosystems, aquatic ecosystems, and watersheds	66 (22.9%)	51.5 (11.2%)	38.5 (21.0%)	29 (15.7%)	**17.7**
2	Air, soil and water resources and quality	17 (5.9%)	88 (19.2%)	9 (4.9%)	19 (10.3%)	**10.1**
3	System drivers (processes, disturbance regimes, and stressors)	40 (13.9%)	21 (4.6%)	15 (8.2%)	7 (3.8%)	**7.6**
4	Baseline carbon stocks	7 (2.4%)	6 (1.3%)	4 (2.2%)	7 (3.8%)	**2.4**
5	Threatened, endangered, candidate species; potential species of conservation concern	12 (4.2%)	36 (7.9%)	24 (13.1%)	4 (2.2%)	**6.8**
6	Social, cultural, and economic conditions	21 (7.3%)	71 (15.5%)	14 (7.7%)	8 (4.3%)	**8.7**
7	Benefits obtained by people (ecosystem services)	49 (17.0%)	0* (0.0%)	2.5 (1.4%)	4 (2.2%)	**5.1**
8	Multiple uses and their contributions to economies	0* (0.0%)	26 (5.7%)	15 (8.2%)	17 (9.2%)	**5.8**
9	Recreation settings, opportunities, and access, and scenic character	29 (10.0%)	39 (8.5%)	15.5 (8.5%)	21 (11.4%)	**9.6**
10	Renewable and nonrenewable energy and mineral resources	17 (5.9%)	18 (3.9%)	3.5 (1.9%)	8 (4.3%)	**4.0**
11	Infrastructure	2 (0.7%)	12 (2.6%)	9.5 (5.2%)	10 (5.4%)	**3.5**
12	Areas of tribal importance	2 (0.7%)	13 (2.8%)	4.5 (2.5%)	3 (1.6%)	**1.9**
13	Cultural and historical resources and uses	3.5 (1.2%)	40 (8.7%)	7 (3.8%)	23 (12.4%)	**6.6**
14	Land status and ownership, use, and access patterns	8 (2.8%)	17 (3.7%)	7 (3.8%)	9 (4.9%)	**3.8**
15	Designated areas, potential/need for new designations	15 (5.2%)	20 (4.4%)	14 (7.7%)	16 (8.7%)	**6.5**
	TOTAL	**288.5**	**458.5**	**183**	**185**	**100**

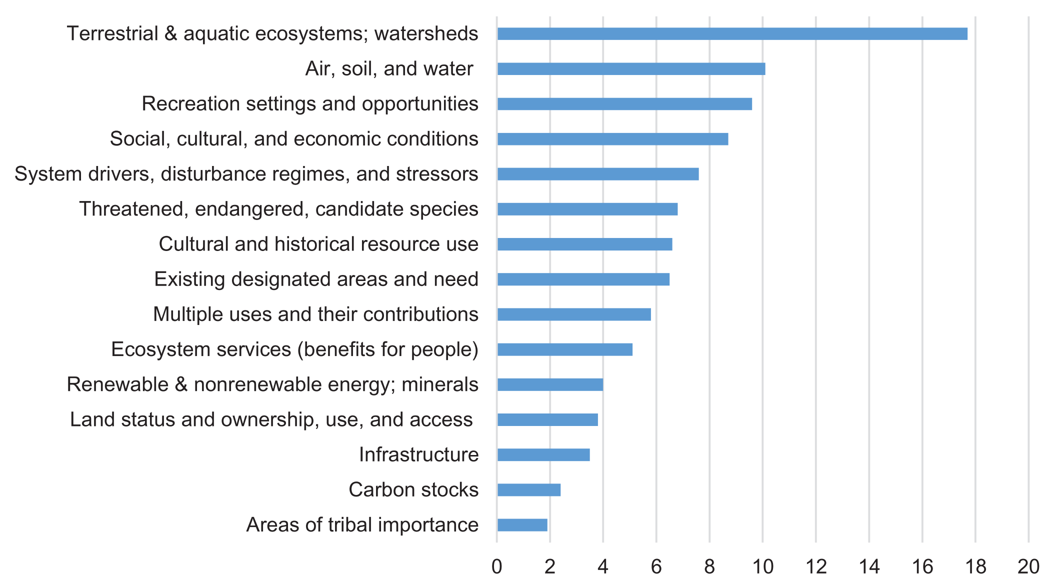

Figure 3. Average percentage of pages devoted to each topic in each forest assessment for all forests combined

have any pages that were specifically devoted to the topic, but references to the topic were instead interspersed throughout the report and it was too difficult to separate them from other topic page counts.

Two of the national forests (Inyo and Nantahala-Pisgah) published assessment reports that consisted of 15 chapters that directly reflected each of the required topics. Meanwhile, the Chugach and Cibola took a different approach; some of the chapter topics aligned with the topic requirements in the 2012 rule, but other required topics were broken up and distributed among multiple chapters. For example, the Chugach had one chapter for areas of tribal importance and one chapter for land status and ownership, but divided the terrestrial and aquatic ecosystems and watersheds

Table 5. Percent allocation of predominant topics among four forest assessments.

Rank	Chugach topics	Pct.	Cibola topics	Pct.	Inyo topics	Pct.	N&P topics	Pct.
1	Terrestrial and aquatic ecosystems	23%	Air, soil, and water	19%	Terrestrial and aquatic ecosystems	21%	Terrestrial and aquatic ecosystems	16%
2	Benefits obtained by people (ecosystem services)	17%	Social, cultural, and economic conditions	16%	Threatened and endangered species	13%	Cultural and historic resources	12%
3	System drivers, disturbance regimes, and stressors	14%	Terrestrial and aquatic ecosystems	11%	Recreation settings and opportunities	9%	Recreation settings and opportunities	11%
4	Recreation settings and opportunities	10%	Cultural and historic resources	9%	System drivers, disturbance regimes, and stressors	8%	Air, soil and water	10%
5	Social, cultural, and economic conditions	7%	Recreation settings and opportunities	9%	Multiple uses	8%	Multiple uses	9%
Total		71%		63%		59%		59%

into five chapters, one each for watersheds, fish, wetlands, vegetation, and wildlife, and these chapters were integrated with material discussing soils and carbon stocks. Two forests did not have any pages specifically devoted to one required topic each (benefits obtained by people for the Cibola, and multiple uses for the Chugach), but these subjects were still referenced in the context of the other topics.

For all four assessments combined, the required topic with the largest average percentage of pages was terrestrial and aquatic ecosystems and watersheds (17.7%), followed by air, soil, and water resources (10.1%) and recreation opportunities (9.6%) (Figure 3).

Terrestrial and aquatic ecosystems and watersheds comprised the largest section of the assessment for three of the four forests. Air, soil, and water was especially prominent for the Cibola National Forest, and all of the forest assessments covered recreation evenly. In contrast, the three required topics with the smallest page counts, on average, were areas of tribal importance (1.9%), carbon stocks (2.4%), and infrastructure (3.4%). Benefits obtained by people (ecosystem services) had the most variable coverage, with one of the shortest sections for three of the four forest assessments, but the second longest topic for the Chugach National Forest. In all four assessment documents, benefits obtained by people were mentioned throughout the document in sentences or paragraphs at too fine a scale for this analysis to count.

We found some variation among the forest assessments in terms of the extent to which a forest focused on a particular topic (Table 5).

For the Chugach National Forest, the top five topics comprised more than 70% of the assessment, with the bulk emphasizing terrestrial and aquatic ecosystems, which reflects the importance of salmon habitat. The Chugach was the only forest to emphasize ecosystem services as a predominant framework to capture benefits obtained by people. However, other forests may have captured this topic under the category of multiple uses. Disturbance regimes (fire and invasive species) were also important for the Chugach. The Cibola National Forest was unique in their emphasis on air, soil, and water as well as social, cultural, and economic conditions and cultural and historic sites. Because water access is very important in the southwest, the predominance of this topic is not surprising. For the Inyo National Forest, the topic of threatened and endangered species was prominent, while topics related to recreation and disturbance regimes (fire, invasive species, and other ecosystem stressors) were also important. Meanwhile, cultural and historical resources were prominent in the Nantahala and Pisgah National Forests, along with recreation.

Although the 2012 rule provides a list of 15 distinct required topics, these topics overlap and are not discussed in complete isolation from one another. As we found in our analysis, it is difficult to discuss multiple uses without also discussing benefits obtained by people; air, soil, and water resources; recreation; and terrestrial and aquatic ecosystems and watersheds. In our analysis, we often found that an assessment chapter devoted to a required topic also contained information that closely resembled material discussed elsewhere. In particular, we found the chapters on multiple uses and benefits obtained by people to be largely redundant, given the other topics that were also included in the report.

Sources and types of information in the forest assessment

To understand the sources and types of information used in the assessments, we conducted a systematic examination and tally of citations by publication source and type. Overall, government sources were the most commonly cited information source (51.8%), followed by scientific scholarly publications (30.7%) (Table 6).

Table 6. Citations based on information source for forest assessments.

	Count (Percent)				
Publishing entity	Chugach	Cibola	Inyo	Nantahala & Pisgah	TOTAL (Mean)
Government	239 (53.6%)	159 (49.8%)	131 (49.8%)	109 (54.0%)	**638 (51.8%)**
Scientific scholarly or peer reviewed	155 (34.8%)	82 (25.7%)	82 (31.2%)	63 (31.2%)	**382 (30.7%)**
Non-government	21 (4.7%)	39 (12.2%)	24 (9.1%)	18 (8.9%)	**102 (8.7%)**
Universities	30 (6.7%)	39 (12.2%)	19 (7.2%)	11 (5.5%)	**99 (7.9%)**
Unknown or other	1 (0.2%)	0 (0%)	7 (2.7%)	1 (0.5%)	**9 (0.9%)**
TOTAL	446	319	263	202	**1230**

Table 7. Citations based on document type for forest assessments.

Document type	Count (Percent)				
	Chugach	Cibola	Inyo	Nantahala & Pisgah	**TOTAL**
Technical report	174 (39.0%)	121 (37.9%)	108 (41.1%)	73 (36.1%)	**476 (38.5%)**
Scientific journal article	129 (28.9%)	47 (14.7%)	63 (24.0%)	48 (23.8%)	**287 (22.8%)**
Academic book	28 (6.3%)	36 (11.3%)	20 (7.6%)	15 (7.4%)	**99 (8.2%)**
Statute, regulation, or planning document	43 (9.6%)	26 (8.2%)	23 (8.8%)	12 (5.9%)	**104 (8.1%)**
Website	33 (7.4%)	42 (13.2%)	3 (1.1%)	13 (6.4%)	**91 (7.0%)**
Database	17 (3.8%)	25 (7.8%)	17 (6.5%)	18 (8.9%)	**77 (6.8%)**
Conference proceeding	10 (2.2%)	3 (0.9%)	6 (2.3%)	18 (8.9%)	**37 (3.6%)**
Non-academic book	4 (0.9%)	9 (2.8%)	10 (3.8%)	0 (0.0%)	**23 (1.9%)**
Correspondence	0 (0.0%)	7 (2.2%)	5 (1.9%)	4 (2.0%)	**16 (1.5%)**
Thesis or dissertation	8 (1.8%)	2 (0.6%)	2 (0.8%)	1 (0.5%)	**13 (0.9%)**
News	0 (0.0%)	1 (0.3%)	3 (1.1%)	0 (0.0%)	**4 (0.4%)**
Unknown	0 (0.0%)	0 (0.0%)	3 (1.1%)	0 (0.0%)	**3 (0.3%)**
TOTAL	446 (100.0%)	319 (100.0%)	263 (100.0%)	202 (100.0%)	**1230 (100.0%)**

A large portion of the government sources included US Forest Service publications (average of 28%), which were more commonly cited than other federal government sources (average of 12%) or state and local governments (average of 11%). Some variation exists among the forests in our sample, but the trends were consistent in terms of reliance on government sources and scholarly peer-reviewed publishers for the majority of citations (82.5% combined average for both categories). The Chugach relied to a greater degree on scholarly publications than other forests. The Cibola had the highest proportion from non-governmental organizations and trade groups (12.2%). The Inyo and the Nantahala and Pisgah mirrored the group average.

Next, we explored citations by the type of document referenced. We found that technical reports were the most common type of document cited in the assessments, with an average of 38.5% (Table 7).

The technical report classification is broad and includes technical and scientific reports, policy briefings, white papers, and other types of information (sometimes referred to as gray literature). All four forests were consistent in the ratio of technical reports cited. The second most common document type was the scientific journal article, with an average of 23%, although the Cibola assessment featured far fewer than the other forests. All of the forests cited a wide variety of regulations, statutes, and planning documents, (e.g., water quality regulations, county comprehensive plans, environmental impact statements, state resource management plans, and forest plans). The Cibola assessment featured the greatest variety of document types, relying on websites and academic books more than the other forests. The Nantahala and Pisgah assessment relied more heavily on conference proceedings. The least commonly cited document types, on average, were news articles (0.4%), theses or dissertations (0.9%), and correspondence (1.5%). Although there is a separate category for websites, documents in many of the other categories were readily available online.

Table 8. Approaches to identifying and using BASI from interview data.

BASI approach	Chugach	Nantahala/ Pisgah	Inyo
Literature review (e.g. Google Scholar for scholarly literature)	x	x	x
Forest Service reports, monitoring data	x	x	x
Personal expertise/training/judgement	x	x	x
Existing dataset/database	x		x
Nearby Forest Service research station		x	x
Nearby university		x	
Host data sharing meeting (partners and stakeholders)		x	
Meet with scientists		x	
Post draft documents on wiki site for public review/editing			x
Other public review opportunity		x	
Gray ("non-peer-reviewed") literature, traditional knowledge			x

Identifying and using best available scientific information in the forest assessment

In interviews, respondents were asked how they identified and obtained BASI for their assessment. Table 8 displays the different approaches used by three of the four forests.

Literature reviews and searches, Forest Service reports and datasets, and personal scientific expertise were mentioned by all nine respondents as primary ways that they identified and obtained BASI. Literature reviews focused on identifying peer-reviewed journals, conference proceedings, or agency reports. Existing datasets and nearby Forest Service research stations and universities were also relied upon. The Sierra Nevada science synthesis effort, which informed the Inyo National Forest assessment, took nearly 18 months to complete (Long et al. 2014). The Inyo also posted draft documents on a wiki site for public review and editing. All nine interviewees stated that their assessment team used the Draft Planning Directives, but also mentioned that the directives were not clear, save for the focus on organizing around the 15 topics. No respondent mentioned specific guidance beyond the draft directives on how to identify BASI. The final directives do specifically address the definition of BASI, as discussed above (Figure 2). Gray literature and traditional knowledge presented challenges, as it at times conflicted with peer-reviewed information. Two respondents mentioned that they wanted to incorporate this type of information, but were unsure how to do so.

Assessments must document what information was determined to be BASI, explain the basis for that determination, and explain how the information was applied to the issues considered (36 CFR

219.3). Our analysis of the assessment documents reveals that all documents discuss the use of high-quality and valid scientific information, citing criteria such as clearly defined and well-developed methodology; standardized methodology; logical conclusions; and reasonable inferences (Chugach National Forest 2014; Inyo National Forest 2014; Nantahala and Pisgah National Forests 2014; Cibola National Forest and National Grasslands 2015). The assessments for all forests mention their reliance on information relevant to their specific forests and issues. Only the Nantahala-Pisgah assessment presented a hierarchy of information sources, with peer-reviewed journal articles the highest, followed by government documents and reports, monitoring datasets, theses and dissertations from universities, and expert opinion where facts were not known through the other sources.

Discussion

The 2012 forest planning rule requires that each national forest or grassland conduct a scientific assessment to guide plan development. We found that assessment reports were disproportionately heavy in science related to terrestrial and aquatic ecosystems, and more limited in treatment of infrastructure, land ownership and access patterns, cultural heritage, and areas of tribal importance. Recreation was the only topic to receive consistent attention across all four forests, although the topic was overshadowed by terrestrial and aquatic ecosystems. We may only speculate about why terrestrial and aquatic ecosystem information was the most prevalent in all four forests, but it is consistent with agency administrative hiring practices since the 1980s that have emphasized recruitment of ecologists, biologists, and other biophysical scientists, compared to social scientists, for example (Thomas and Mohai 1995). The abundance of agency specialists in these topic areas may reinforce the relative importance of terrestrial and aquatic ecosystems compared to other topic areas, such as recreation, social science, or cultural resource management. This has been confirmed by a national assessment of interdisciplinary planning team composition (Cerveny et al. 2011). Ensuring that assessment teams include broad and diverse disciplinary experts will help address this challenge, recognizing that some forests may not have access to necessary disciplinary specialists. It is also possible that some of the topics (e.g., ecosystem services, tribal and cultural resources, land status and use patterns) simply do not have as much relevant and available information as other topics.

The benefits obtained by people (ecosystem services) topic received little or no explicit coverage in all but one assessment. The limited coverage of ecosystem services may make sense because it was not even considered an area of research until the late 1990s, so there would be less existing information on certain important ecosystem service topics (e.g., pollination, stormwater attenuation, medicinal resources, and spiritual and historical significance) compared to recreation, threatened and endangered species, and other traditional assessment topics (Blahna et al. 2017). Previously, "forest benefits to people" were considered elements of "multiple use" and planners might have addressed these benefits under the "multiple use" topic. Ecosystem services (ES) are often categorized into four classes: provisioning, regulating, cultural, and supporting. Timber, recreation, wildlife, and other traditional forest planning topics all fall into one of these four classes. Another reason for lack of coverage of ecosystem services may be that planners could not differentiate the normal assessment topics from the ecosystem service classes. Efforts to help planning team members understand ecosystem services approaches and how they can be used to inform the planning process may be warranted, and the rule's current requirement for only using existing data in assessments may need to be revisited (Blahna et al. 2017). For example, implementation teams working on ecosystem services may consider the benefits of providing specific tools, frameworks, and guidelines for integrating ecosystem services models into the forest planning process. In addition, critical issues and topics (e.g., newly listed threatened or endangered species, or changing recreation behaviors) that forest plans need to address may change from one planning cycle to the next.

The specific required topics may not be universally appropriate for every planning unit. Planners felt obligated to address all 15 topics, but the lack of coverage for some topics suggests that the topic was not deemed relevant or meaningful for their plan, there was no available data on the topic, or it was unclear how the topics could be covered. Variability in application of the directives, and acknowledgment of local context and conditions, is consistent with the overall Forest Service approach toward decentralized decision-making (Kaufman 1960; Tipple and Wellman 1991; Koontz 2007) and localized interpretation by planning teams, similar to "street-level" bureaucrats who create de facto policy through everyday practice (Sabatier et al. 1995; Lipsky 2010; Trusty and Cerveny 2012). Kaufman (1960) observes the traditional Forest Service practice of maintaining control of heterogeneous and geographically dispersed management units by issuing centralized directives that provide parameters (or "side boards") within which line officers have some leeway to make decisions. This tendency toward uniformity and "pre-formed" decisions may result in some inefficiencies and omissions. The implied obligation to cover all 15 topics may have resulted in some assessments that distract from the most important management issues for the unit. This will be especially important during the next stage of planning—revision or amendment—where the assessment data will be used to analyze different management scenarios. Approaches for identifying and analyzing the most relevant assessment data that address the key environmental problems or social conflicts that confront each planning unit will be needed (Blahna et al. 2017). This is especially important for topics like human benefits (ecosystem services) and multiple uses, which cut across all of the other topical areas and are not as easily categorized in assessments. Recent efforts to engage the public in science synthesis efforts in support of forest planning suggest that there may be an important role for the public to help prioritize forest assessment topics.

The most common sources of information were government sources, followed by scholarly academic sources. Many of the agency sources were peer-reviewed scientific studies, which appear to be especially useful because of the topical specificity or geographic focus (relevance). Although not all technical reports are peer reviewed, they may be more accessible and usable compared to scholarly journal articles, which may require planning team members to interpret the findings and make inferences for relevance to local conditions. This finding is consistent with previous research examining the information needs and sources of Forest Service fire managers (Ryan and Cerveny 2011) and recreation managers (Ryan and Cerveny 2010). Fire managers relied heavily on agency information sources. Although managers in the study noted the availability of high-quality, relevant information, they faced significant barriers in terms of time, funding,

and personnel to access and use that information. Similarly, recreation managers also relied on agency information sources, but indicated strong preferences for enhanced interactions with agency scientists, including collaborative research, conferences, and a desire for agency researchers to reach out more directly to managers to ensure their research was relevant and useful. With regard to forest assessments, engagement with scientists is particularly important for topics where little research is available. Assessment teams may want to consider additional ways to interact with scientists and others to create functioning communities of practice related to science exchange for forest planning. In the same way, agency scientists may consider forging new and enduring relationships with planners and managers that could generate new science that is of immediate relevance.

The 2012 planning rule and its directives provide criteria for BASI, and we found similarities across all forests in the most common approaches to identifying BASI, in addition to other approaches, such as data sharing meetings, a wiki review site, and requests for a science synthesis. Information from non-peer-reviewed sources was more difficult for planners to assess and evaluate, and it is not clear how this information was incorporated into each assessment. Teams may not have the capacity to separately evaluate and assess the many different types and sources of information, and so they rely on hierarchical ranking approaches (peer-reviewed sources being highest rank) to streamline the evaluation. Planning teams clearly value peer-reviewed and agency-generated information, and it may be that they are simply identifying information that is "available" and using the "best" of that based on their judgments. This may result in situations where the science expertise on each team could influence BASI decisions. As discussed above, consideration of the makeup and membership of the assessment team is important here, as well as increased transparency regarding the process for determining science relevance and quality.

Conclusion

Implementation of the US Forest Service 2012 planning rule is still in its early stages. Our study illustrates that forest planners use a variety of approaches to address required topics, and do rely on BASI as they develop their forest assessments. While each national forest assessment included the 15 required topics, we found considerable variation in coverage, which suggests that planners may emphasize topics most relevant to their forest, or that variation exists in terms of what science or planning team expertise is available or deemed desirable. The predominance of science related to terrestrial and aquatic ecosystems in the assessments compared to other topics warrants further inquiry in order to learn whether this asymmetry is based on policy, availability of information, existing expertise, or other factors. Efforts to include the public in the process of prioritizing topics for the assessments could also be evaluated. The reliance on government sources for scientific information suggests that agency-supported science is either more accessible or more relevant to the planning team. It also suggests that there may be benefits to bolstering "communities of practice" for key topical areas covered by forest assessments that bring together university and agency scientists with managers.

The appearance of science in an assessment report is important, but the actual *use* of science in planning may be more important. Although our findings are not generalizable to all national forests, they do provide an understanding of plan assessment activities for those in the early phases of forest planning, whose efforts are likely to inform and influence other national forests. Our goal was to provide an early glimpse of plan revision efforts in order to highlight important lessons learned and create a foundation for future research. For example, do planners find that the required topics provide useful guidance for developing their assessments? How can planners become more confident in knowing what BASI is, and how to identify and use it? Is additional guidance needed for incorporation of traditional knowledge and other information? Of particular interest is whether the "science synthesis" information is useful to forest planners in addressing their forest assessment needs, given the significant agency resources devoted to developing science syntheses. Finally, how is information from the assessment used in forest plan revision (development and selection of management options) and monitoring efforts? While draft environmental impact assessment (EIS) reports are available in various stages, as of this writing only one final Record of Decision (ROD) has been issued for a forest plan undergoing revision under the 2012 rule. Thus, it remains to be seen how scientific information will be incorporated in development of alternatives, impact statements, and final management decisions.

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The Interaction of Fire, Fuels, and Climate across Rocky Mountain Forests

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Understanding the relative influence of fuels and climate on wildfires across the Rocky Mountains is necessary to predict how fires may respond to a changing climate and to define effective fuel management approaches to controlling wildfire in this increasingly populated region. The idea that decades of fire suppression have promoted unnatural fuel accumulation and subsequent unprecedentedly large, severe wildfires across western forests has been developed primarily from studies of dry ponderosa pine forests. However, this model is being applied uncritically across Rocky Mountain forests (e.g., in the Healthy Forests Restoration Act). We synthesize current research and summarize lessons learned from recent large wildfires (the Yellowstone, Rodeo-Chediski, and Hayman fires), which represent case studies of the potential effectiveness of fuel reduction across a range of major forest types. A "one size fits all" approach to reducing wildfire hazards in the Rocky Mountain region is unlikely to be effective and may produce collateral damage in some places.

Keywords: fire ecology, forest management, forest health, Rocky Mountain forests, climate

The interaction between climate, fuels, and the frequency and severity of wildfires across Rocky Mountain forests is complex. A comprehensive understanding of the relative influence of fuels and climate on wildfires across this heterogeneous region is necessary to predict how fires may respond to a changing climate (Dale et al. 2001) and to define effective fuel management for controlling wildfires in this increasingly populated region (USDA 2002). The annual area burned by wildfires has apparently increased during the last few decades across North America, and in the southern Rocky Mountain region in particular, possibly in response to recent climate change and the gradual accumulation of fuels following decades of effective fire suppression (figure 1; Grissino-Mayer and Swetnam 2000). However, more complete modern records, and an increase in land under federal protection since the 1960s, may also have contributed to this apparent trend over the last half-century. Nonetheless, the United States recently experienced a series of big fire years: According to the National Interagency Fire Center (*www.nifc.gov*), wildfires in 1988, 2000, and 2002 burned 3.0 million, 3.4 million, and 2.8 million hectares (ha), respectively, Most of these fires took place in the western United States, which is characterized by fire-prone ecosystems.

In an effort to mitigate the risk to life and property from wildfires and the high cost of fighting fire throughout the western United States, fuel reduction has become an important forest and fire management tool. In 2002, thinning and prescribed-fire projects were carried out across 1 million ha of federal land as part of the US National Fire Plan (*www.fireplan.gov*) to reduce the fire hazard and to restore historical species composition and stand structures. The goals of fire-hazard reduction and ecological restoration may converge in some ecosystems, yet they may be incompatible in others (Veblen 2003).

The idea that decades of fire suppression have promoted unnatural fuel accumulation and subsequent unprecedentedly large, severe wildfires across western forests was developed primarily from experience in dry ponderosa pine (*Pinus ponderosa*) forests in the US Southwest, the interior West, and the Sierra Nevada (Covington and Moore 1994, Caprio and Swetnam 1995, Moore et al. 1999). Historically, short-interval, low-severity surface fires maintained sparse, open stands in most dry ponderosa pine forests (Swetnam and

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Baisan 1996). With fire suppression, young fire-intolerant trees can establish during lengthened fire intervals. Denser stands provide "ladder" fuels at intermediate heights that carry fire up into continuous canopy fuels, promoting unprecedentedly large, catastrophic fires. This system has presented a strong case for thinning to reduce the fire hazard and to restore historical stand structure.

Ecological restoration and fire mitigation are urgently needed in dry ponderosa pine forests, where previous research supports this management action. However, we are concerned that the model of historical fire effects and 20th-century fire suppression in dry ponderosa pine forests is being applied uncritically across all Rocky Mountain forests, including places where it is inappropriate (e.g., USDA 2002, White House 2002). Of particular concern is President Bush's Healthy Forests Initiative, which identifies unnatural fuel buildup as a widespread risk across the West: "Today, the forests and rangelands of the West have become unnaturally dense, and ecosystem health has suffered significantly. When coupled with seasonal droughts, these unhealthy forests, overloaded with fuels, are vulnerable to unnaturally severe wildfires. Currently, 190 million acres [77 million ha] of public land are at increased risk of catastrophic wildfires" (White House 2002, executive summary). This initiative was recently enacted as HR 1904, the Healthy Forests Restoration Act of 2003.

The relative contribution of fuels and climate to recent fire activity across forest types throughout the western United States is hotly debated (e.g., see *Conservation Biology,* vol. 15 [2001]). It is easy to identify either local situations in which fire suppression has allowed unusual fuel accumulations or, by contrast, those in which fuel conditions remain within the historical range and the effects and frequency of fire are controlled primarily by weather conditions, not by fuels. What is lacking is a broad synthesis of the geographical patterns in historical fire regimes, and of 20th-century changes in these regimes, addressing these key questions:

- Where, in what ecosystem types, and to what degree have fuels increased with fire suppression across the Rocky Mountain region (Arizona, New Mexico, Colorado, Utah, Wyoming, Montana, and Idaho)?
- Where are forest restoration treatments appropriate, and how will fire respond to fuel-reduction treatments in different forest types?
- Where and when is the influence of short-term (i.e., seasonal and annual) climatic variation expected to override the effectiveness of fuel treatments?

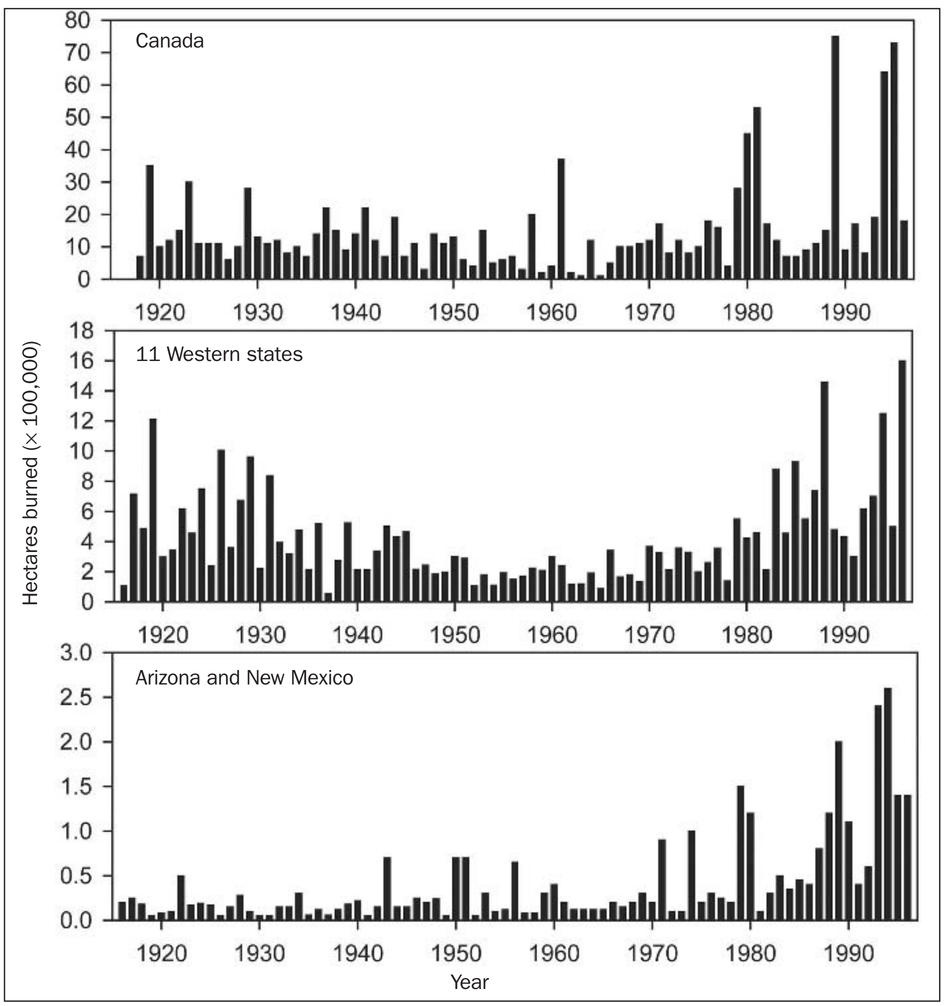

***Figure 1. Area burned by wildfires in different regions under federal protection across North America. The apparent increase in the extent of fires over the last century is most pronounced in the southwestern United States (Arizona and New Mexico), although we urge caution in interpreting these trends. Source: Grissino-Mayer and Swetnam (2000); reprinted with permission from* The Holocene.**

To address these questions, we synthesize current understanding of the different types of fire regimes (defined by the historical range of variability in fire size, severity, and frequency) that occur across the Rocky Mountain region. The fire regime is a central concept in fire ecology and is essential for understanding the character, effect, and variability of disturbance patterns across regions. Our analysis of different fire regimes is based on the classic fire triangle of weather, fuels, and ignition, which identifies the factors controlling combustion. All three factors must be present in a form conducive to combustion, or fire will not occur. However, the inherent variability, and therefore the limiting role, of these three ingredients is dramatically different among forest types and geographic regions. For example, we argue below that fuel types and amounts are less limiting to fire spread in subalpine forests than in low-elevation forests, but suitably dry weather conditions for fire spread in subalpine forests occur infrequently. Hence, variability in seasonal and annual climate is more limiting and has a greater influence on fire extent and severity in these generally cool, moist ecosystems.

In contrast, periods of several months of warm, dry weather occur almost annually in most southwestern ponderosa pine forests, leaving fuels sufficiently desiccated for extensive fires to occur annually. Given the higher frequency of weather conditions that desiccate fuels in this ecosystem, factors that affect fuel type, quantity, and configuration are more limiting than climate in controlling this fire regime. Variations in local site productivity, and in the time elapsed since the last fire event, affect fuel accumulation in the dry, low-elevation ponderosa pine forests. Annual climatic variation affects fuels indirectly in these forests both through short periods of above-average moisture availability, which enhance the production of fine fuels (e.g., leaves, grasses, forest litter), and through fuel-desiccating drought. But overall, climate is more limiting in subalpine forests, where short-term (i.e., months to a few years) variability in climate primarily affects fire severity and spread through fuel desiccation and wind, not fuel abundance. In contrast, the fire regime in dry ponderosa pine woodlands is more limited by annual variability in fine fuel amounts and by ladder-fuels related to the time elapsed since the last fire. Ignition sources also may be important, at least locally, but in this study we do not identify spatial patterns in this component of the fire regime. Assuming instead that ignition sources are always available, we evaluate the relative importance of variability in short-term climatic variation and in fuel quantity and configuration.

We identify three major types of historical fire regimes (Agee 1998): (1) high severity, (2) low severity, and (3) mixed severity. In addition to developing a general theoretical framework for assessing controls on local fire regimes, we summarize the lessons learned from three recent large wildfires (the 1988 Yellowstone fires and the 2002 Rodeo-Chediski and Hayman fires). These case studies reveal the potential effectiveness of fuel reduction under varying climate conditions across a range of major forest types and historical fire regimes. Finally, we develop coarse estimates of the spatial extent of the three major historical fire regimes to broadly quantify heterogeneity in fire regimes and responses to fire suppression across the Rocky Mountain region.

To develop coarse estimates of the proportion and extent of historical fire regimes across the Rockies, we rely on research reported in the peer-reviewed literature to group major forest types that historically experienced each of the three major fire regimes we discuss. Because it is relatively difficult to define the spatial extents of different fire regimes at this scale, we rely on two independent maps of forest cover to highlight general trends and degrees of uncertainty in the relative proportion of major fire types across the Rocky Mountain region. In the first analysis, forest types are based on a map of Küchler's potential natural vegetation (PNV) groups (climax vegetation types that are expected, given the occurrence of natural disturbances such as fire, based on site characteristics such as soils, climate, and topography), modified by Schmidt and colleagues (2002). In our reclassification of these data, we combine eight PNV groups into three main forest types: (1) ponderosa pine (pine forest and Great Basin pine), (2) mixed ponderosa pine (pine–Douglas fir, Douglas fir, grand fir–Douglas fir, and Southwest mixed conifer [Arizona, New Mexico]), and (3) spruce–fir (spruce–fir and spruce–fir–Douglas fir). In the second analysis, forest types are based on a map of current cover types, which Schmidt and colleagues (2002) developed by combining the Forest and Range Resource Planning Act map of US forest type groups with AVHRR (Advanced Very High Resolution Radiometer) satellite imagery. In our reclassification of these data, we combine the current cover types into three main forest types, similar to those obtained by combining the PNV groups: (1) ponderosa pine, (2) Douglas fir, and (3) spruce–fir–lodgepole pine.

In this summary, we assume a one-to-one correspondence between forest types and fire regimes; however, as we emphasize throughout the text, this is a considerable oversimplification. Nonetheless, this summary reveals coarse levels of heterogeneity in fire regimes across the Rocky Mountain region, unaccounted for in current forest policy debates. Other endeavors to define fire regimes at this scale include the work of Schmidt and colleagues (2002), who developed a map of historical fire regimes and departures from historical conditions throughout the continental United States for strategic fire-planning purposes, but who relied primarily on managers' expert knowledge rather than on peer-reviewed empirical studies in defining fire regimes. In addition, McKenzie and colleagues (2000) developed a regional model of fire frequency within the interior Columbia River basin, based on a large fire-history database from the western United States.

Overall, our analysis highlights the heterogeneity of forest types and fire regimes across the Rocky Mountain region. Further, it provides insight into pressing management questions of when and where various fuel treatments are consistent with the goal of ecological restoration, and where such treatments are likely to be successful in reducing the size and severity of wildfires. We focus on the Rocky Mountain region; however, the spatial and geographic heterogeneity in fire regimes across this region is also evident throughout the West (e.g., Agee 1998).

High-severity fire regimes

High-severity or stand-replacing fires are defined by the death of canopy trees, in contrast to low-severity fires, which do not kill overstory trees. High-severity fires typically burn the treetops (crown fires) but may also kill trees through very hot surface fires, which primarily burn the forest floor.

High-elevation subalpine forests in the Rocky Mountains typify ecosystems that experience infrequent, high-severity crown fires (Peet 2000, Veblen 2000). The forest types that occur in the subalpine zone range from mesic spruce–fir forests to drier, dense lodgepole pine stands; and xeric, open woodlands of limber and bristlecone pine. The most extensive subalpine forest types are composed of Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), and lodgepole pine (*Pinus contorta*), all thin-barked trees easily killed by fire.

Extensive stand-replacing fires occurred historically at long intervals (i.e., one to many centuries) in subalpine forests (Romme 1982, Kipfmueller and Baker 2000, Veblen 2000, Schoennagel et al. 2003), typically in association with infrequent high-pressure blocking systems that promote extremely dry regional climate patterns (Romme and Despain 1989, Renkin and Despain 1992, Bessie and Johnson 1995, Nash and Johnson 1996). Persistent high-pressure blocking systems affect regional temperature and precipitation patterns throughout the Rockies and may respond to global climate anomalies (Baker 2003). Regional synchrony of large, high-severity fires across subalpine forests corroborates the idea that high-elevation forest fires respond to broad scale synoptic climate (Nash and Johnson 1996, Kipfmueller and Baker 2000, Veblen 2000, Baker 2003). In moist high-elevation forests, successive seasons of drought can initiate large, stand-replacing fires (Balling et al. 1992, Kipfmueller and Swetnam 2000). In these generally cool subalpine environments, significant drought events are infrequent, which prevents the frequent occurrence of large, high-severity fires. Although they occur infrequently, drought-induced large fire events account for the greatest percentage of the area burned in subalpine forests (figure 2; Bessie and Johnson 1995).

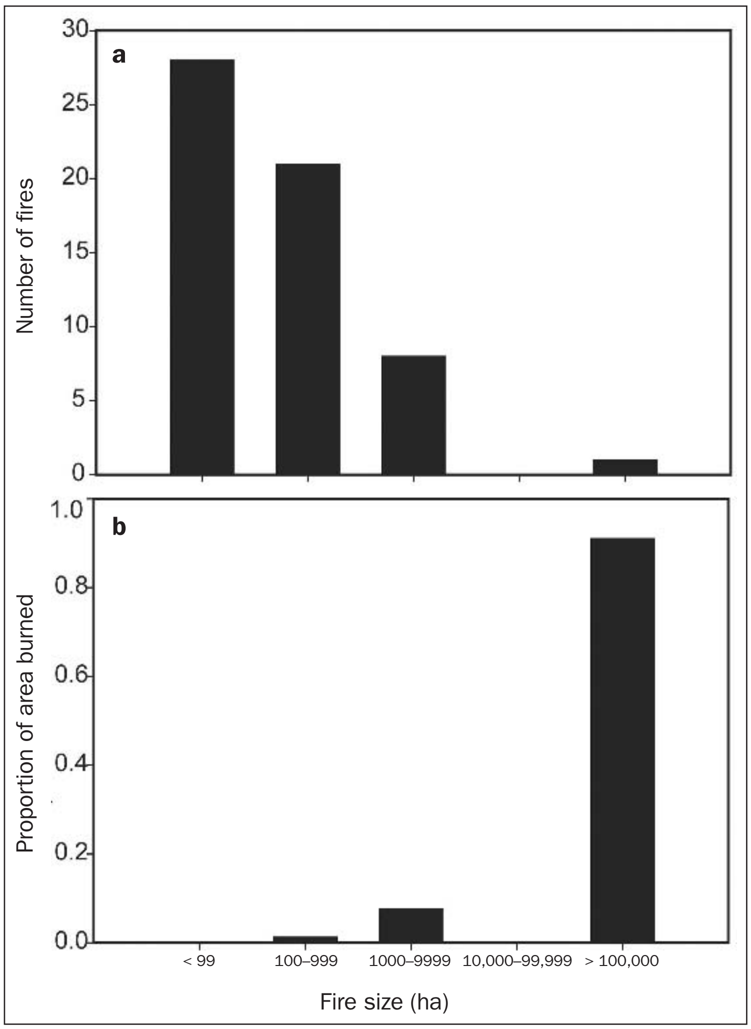

Figure 2. (a) Histogram of the occurrence of different size classes of stand-replacing fires in Yellowstone National Park (1895–1991). (b) Proportion of the total area burned in each size class for the same period (1.0 = 100% of total area). Although large stand-replacing fires (i.e., fires that burn more than 1000 hectares) are infrequent, they are the dominant influence on subalpine forests. Data are from Balling and colleagues (1992).

Subalpine forests typically experience stand-replacing crown fires, rather than low-severity surface fires, because they lack fine fuels on the forest floor but have abundant ladder fuels that carry fire into the treetops. These dense, closed-canopy forests typically support sparse understory vegetation, and the short, stout needles of subalpine trees compact tightly on the forest floor, creating a poor substrate for fire spread (Swetnam and Baisan 1996). This is in stark contrast to the warmer, open-canopied, productive forests at lower elevations, which support abundant, well-aerated fine fuels on the forest floor (Swetnam and Baisan 1996). Although fine surface fuels are sparse in subalpine forests, ladder fuels are abundant. Shade-tolerant fir and spruce trees have abundant lateral branches, which easily carry fire up into the canopy. By contrast, shade-intolerant lodgepole pines have few lateral branches, but these trees tend to grow in very dense stands that thin over time, contributing to abundant dead ladder fuels (figure 3). The abundance of ladder fuels, the proximity of crowns, and the lack of abundant, spatially continuous fine surface fuels all promote high-severity crown fires that dominate subalpine forests.

The low abundance of small fuels, and the relatively high abundance of large dead and live fuels, explains why fires are infrequent but typically large in subalpine forests. Fuel moisture levels respond to ambient environmental conditions and are critical in determining fire potential. Small-diameter dead fuels dry quickly; for example, 1-hour fuels (particles less than 0.6 centimeters [cm] in diameter) approach equilibrium with ambient relative humidity within an hour. By contrast, dead branches, logs, or other large, slow-drying materials (7.6 to 20.3 cm in diameter) are known as 1000-hour fuels because they require 1000 hours to equilibrate (figure 4). Live fuels dry even more slowly than dead fuels and are influenced most strongly by sustained periods of drought. Because of the paucity of small dead fuels such as needles and grasses in subalpine forests, short-duration drying episodes generally do not create sufficiently dry conditions to sustain a fire. However, prolonged dry weather conditions (about 40 days without precipitation) can sufficiently dry live fuels and larger dead fuels to carry large, intense fires once they are ignited (figure 5). Conditions necessary for large fires are infrequent and often coupled with the occurrence of lightning. This suggests that Native Americans probably did not have a major influence on fires in the subalpine forest types, except in some localized areas.

The recent period of consistent, effective fire suppression in remote high-elevation sites, which has lasted 50 years at most, represents only a small portion of typical fire-free intervals in subalpine forests. Studies of fire history show that long fire-free periods (as long as, or longer than, the fire exclusion period during the 20th century) characterized the fire regimes of these forests before Euro-American settlement (Romme 1982, Romme and Despain 1989, Kipfmueller and Baker 2000, Veblen 2000, Schoennagel et al. 2003). Therefore, it is unlikely that the short period of fire exclusion has significantly altered the long fire intervals in subalpine forests (Romme and Despain 1989, Johnson et al. 2001, Veblen 2003). Furthermore, large, intense fires burning under dry conditions are very difficult, if not impossible, to suppress (Wakimoto 1989), and such fires account for the majority of area burned in subalpine forests (figure 2; Romme and Despain 1989, Bessie and Johnson 1995). At lower elevations within its range, lodgepole pine may also experience occasional small surface fires (Kipfmueller and Baker 2000), but their spatial extent and frequency are not well quantified. Suppression of smaller, less intense fires under moderate climate conditions probably has had little influence on the dominant fire regime in subalpine forests (Johnson et al. 2001, Veblen 2003). Our understanding of the dominant fire regime in these high-elevation, cool forests leads us to conclude that any recent increases in area burned in subalpine forests are probably not attributable to fire suppression. Evidence from the subalpine forests of Yellowstone indicates that fires of comparable size to the 1988 fires occurred in the early 1700s (Romme and Despain 1989).

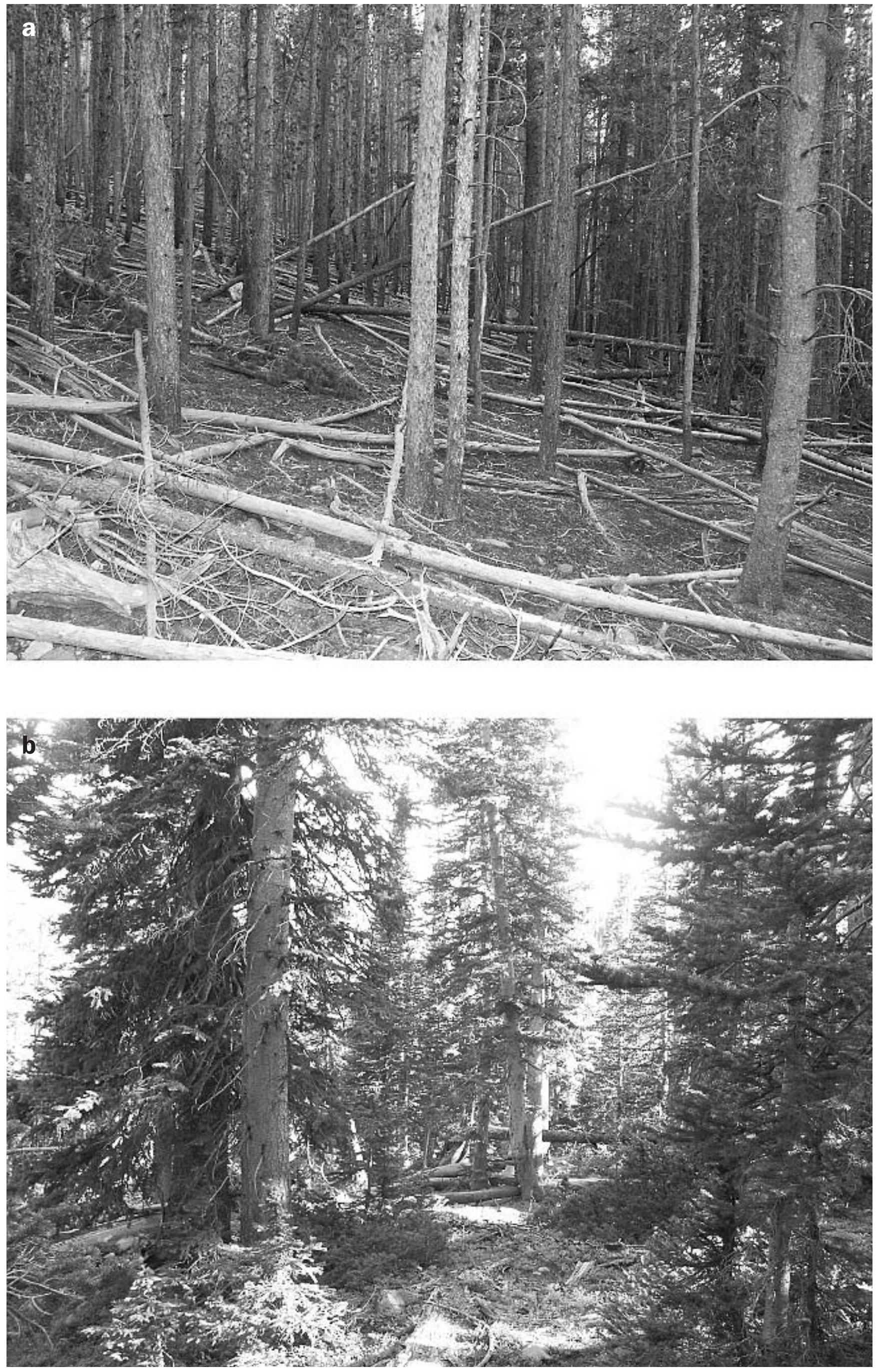

Figure 3. Typical subalpine forest stand structure, which easily carries fire into the canopy, promoting high-severity crown fires. (a) Lodgepole pine stand with sparse understory fuels and high tree densities. (b) Spruce–fir stand with abundant live ladder fuels throughout the vertical profile. Photographs: Tania Schoennagel.

Moreover, there is no consistent relationship between time elapsed since the last fire and fuel abundance in subalpine forests (Brown and Bevins 1986), further undermining the idea that years of fire suppression have caused unnatural fuel buildup in this forest zone. For example, lodgepole pine stands experience high rates of self-thinning that contribute large dead fuels as stands mature (Kashian 2003). However,

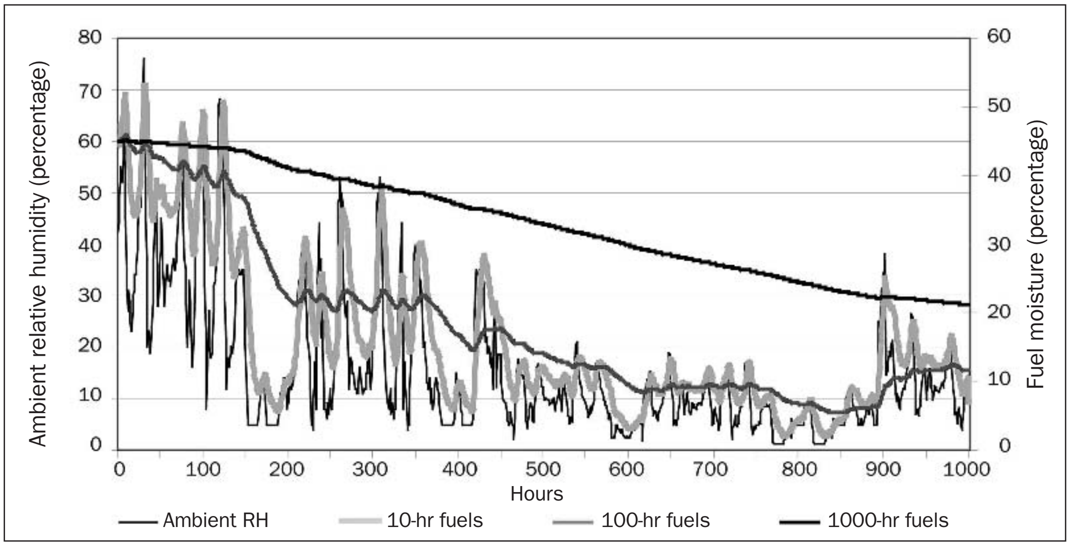

Figure 4. A theoretical example illustrating differences in fuel-moisture time lags for small (10-hour), intermediate (100-hour), and large (1000-hour) fuels. Small fuels dry out rapidly and respond more quickly to short-term variability in ambient relative humidity, while large fuels exhibit a more lagged response, requiring much longer dry periods to reach similar dryness.

the legacy of wood from the prefire stand contributes abundant loads of large fuel to young postfire stands (Romme 1982). Bessie and Johnson (1995) report little variation in total fuel loads, relative to variation in weather, in subalpine forests of different ages. No evidence suggests that spruce–fir or lodgepole pine forests have experienced substantial shifts in stand structure over recent decades as a result of fire suppression. Overall, variation in climate rather than in fuels appears to exert the largest influence on the size, timing, and severity of fires in subalpine forests (Romme and Despain 1989, Bessie and Johnson 1995, Nash and Johnson 1996, Rollins et al. 2002). We conclude that large, infrequent stand-replacing fires are "business as usual" in this forest type, not an artifact of fire suppression.

Case study: The 1988 Yellowstone fires. In 1988, according to the National Interagency Fire Center, more than 700,000 ha burned in mostly high-elevation subalpine forests throughout Wyoming, Montana, and Idaho. Yellowstone National Park was the focus of public attention during these fires. Some 40% of the park burned, much of it at high severity (Turner et al. 1994). Drought, which had started years earlier, extended beyond its immediate region during the summer of 1988. From 1977 to 1989, a strong Pacific North America pattern developed, creating a blocking ridge over the northwestern United States that reduced winter snowpack across Montana and Wyoming (Baker 2003). Low winter snowpack in 1988, followed by an unusually dry, hot, and windy summer, contributed to extreme burning conditions in the park (Balling et al. 1992). Precipitation in July and August was only 20% of normal levels; relative humidity fell to 6%; and strong, dry, gusty winds (60 to100 kilometers [km] per hour) spread multiple fires ignited by humans and lightning.

Variation in daily area burned was highly correlated with the moisture content of 100-hour (2.5- to 7.6-cm diameter) and 1000-hour dead fuels (Turner et al. 1994). Once fuels reached critical moisture levels later in the season, the spatial pattern of the large, severe stand-replacing fires was controlled by weather (wind direction and velocity), not by fuels, stand age, or fire-fighting activities (Minshall et al. 1989, Wakimoto 1989, Turner et al. 1994). Variation in fuel abundance and topography (including formidable barriers such as the Grand Canyon) had little influence on the severity or direction of the fire when fuel moistures were critically low (Turner et al. 1994). Stand-replacing fire affected stands of all ages, including some as young as 7 years old (Schoennagel et al. 2003).

Contrary to popular opinion, previous fire suppression, which was consistently effective from about 1950 through 1972, had only a minimal effect on the large fire event in 1988 (Turner et al. 1994). Reconstruction of historical fires indicates that similar large, high-severity fires also occurred in the early 1700s (Romme and Despain 1989). Given the historical range of variability of fire regimes in high-elevation subalpine forests, fire behavior in Yellowstone during 1988, although severe, was neither unusual nor surprising.

Summary: High-severity fire regimes in subalpine forests. Subalpine forests that experience infrequent, high-severity fires cover approximately 32% to 46% of the forested area in the Rocky Mountain region, which encompasses the three major forest types discussed in this article (table 1). The following insights are drawn from analyses of historical fire regimes and contemporary fire behavior in subalpine forests.

- Infrequent, high-severity, stand-replacing fires dominate the historical and contemporary fire regime in these forests.
- Climatic variation, through its effects on the moisture content of live fuels and larger dead fuels, is the predominant influence on fire frequency and severity.
- Dense trees and abundant ladder fuels are natural in subalpine forests and do not represent abnormal fuel accumulations.

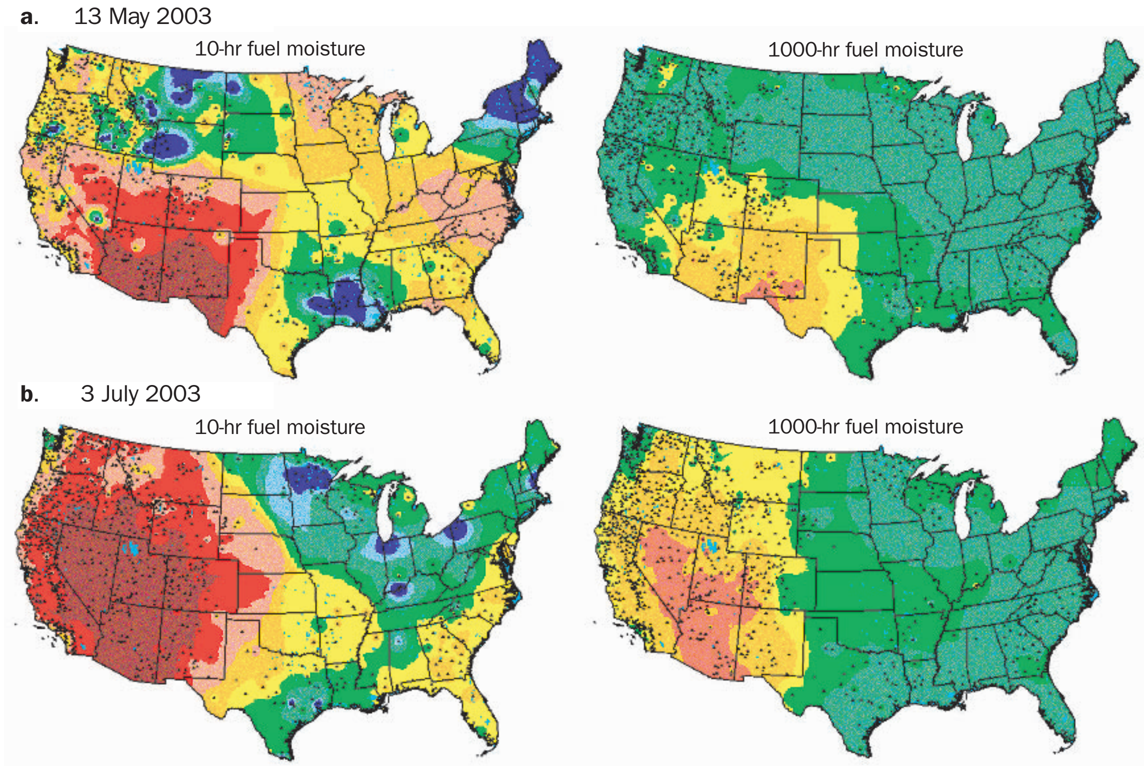

Figure 5. Maps of fuel moisture for small (10-hour) and large (1000-hour) fuels, showing responses to (a) short-term (1- to 2-day) and (b) longer-term (1- to 2-month) drying conditions in the southwestern United States. Large fuels dry sufficiently to carry fire only under longer drying conditions, while smaller fuels may dry sufficiently to carry fire under short-term or moderate drying conditions. The maps were developed by the National Interagency Fire Center (17 June 2004; **www.fs.fed.us/land/wfas/wfas10.html**).

- Fire suppression has had minimal influence on the size, severity, and frequency of high-elevation fires.
- Mechanical fuel reduction in subalpine forests would not represent a restoration treatment but rather a departure from the natural range of variability in stand structure.
- Given the behavior of fire in Yellowstone in 1988, fuel reduction projects probably will not substantially reduce the frequency, size, or severity of wildfires under extreme weather conditions.

Low-severity fire regimes

In marked contrast to the infrequent, high-severity fire regimes characteristic of subalpine forests, many low-elevation ponderosa pine forests historically experienced frequent, low-severity fires. A meta-analysis of 63 fire histories from similar-size southwestern ponderosa pine sites (10 to 100 ha) indicates that surface fires returned at mean intervals of 4 to 36 years (based on fire dates recorded for more than 10% of the sampled trees; Swetnam and Baisan 1996), an order of magnitude shorter than the intervals for subalpine forest stands. Some low-elevation ponderosa pine stands in Colorado, near the Plains grasslands, show evidence of 8- to 10-year intervals for fire returning to the same small stand or tree before the 1900s (Veblen et al. 2000). In the Black Hills of South Dakota, the mean fire interval was 20 to 23 years at each of four low-elevation ponderosa pine sites (about 100 ha each) for the period from 1388 to 1900 (Brown and Sieg 1996). Although detailed comparison of fire-interval statistics across study sites is problematic because of differences in the extent of the study area and the intensity of sampling, these studies clearly indicate a significant difference in fire interval and severity between low-elevation, dry ponderosa pine forests and high-elevation, moist subalpine forests.

Frequent, low-severity fire regimes occurred predominantly in dry, low-elevation ponderosa pine forests that were formerly open woodlands with abundant, contiguous fine fuels in the understory. This surface fuel layer, dominated by grasses and long cast needles, dries easily and thus promotes the spread of frequent surface fires. Historically, climate, fine-fuel abundance, and fire were highly interrelated in dry, low-elevation ponderosa pine forests. El Niño–Southern Oscillation (ENSO) patterns correlate tightly with the incidence of synchronous, low-severity fires in dry, low-elevation forests of the Southwest (Swetnam and Baisan 1996, Grissino-Mayer and Swetnam 2000, Kitzberger et al. 2001). The ENSO cycle alternates between El Niño and La Niña conditions at

Table 1. Two coarse estimates of the extent and proportion of three major forest types across the Rocky Mountain region (Arizona, New Mexico, Colorado, Utah, Wyoming, Montana, and Idaho). The first estimate is based on a map of Küchler's potential natural vegetation groups, modified by Schmidt and colleagues (2002). The second estimate is based on a map of current cover type developed by Schmidt and colleagues (2002). A different historical fire regime is associated with each of the three forest types, although the correspondence is not exact.

Forest type	Area (hectares)	Percentage of total	Associated severity of historical fire regime
Based on PNV groups			
Ponderosa pine (pine forest, Great Basin pine)	8,201,600	17.7	Low
Mixed ponderosa pine (pine–Douglas fir, Douglas fir, grand fir–Douglas fir, Southwest mixed conifer)	23,176,200	49.9	Mixed
Spruce–fir (spruce–fir, spruce–fir–Douglas fir)	15,056,000	32.4	High
Total	46,433,800	100.0	
Based on current cover types			
Ponderosa pine	13,009,100	36.7	Low
Douglas fir	6,176,000	17.4	Mixed
Spruce–fir–lodgepole pine (lodgepole pine, fir–spruce)	16,287,200	45.9	High
Total	35,472,300	100.0	

PNV, potential natural vegetation.

Note: Total is the forested area in the Rocky Mountain region defined by the three major forest types listed. Some other forest types, such as piñon-juniper woodlands, are not included.

2- to 6-year frequencies. In the southern Rockies, El Niño years are characterized by wetter-than-average winter and spring conditions, which enhance the growth of fine fuels (especially grasses). Drier-than-average La Niña years typically follow, desiccating abundant fine surface fuels. Time-lag analysis shows that dry, low-elevation ponderosa pine forests commonly experience more extensive fires when wetter conditions 1 to 3 years before a fire are followed by dry conditions during the year of the fire. Infrequent or anomalous prolonged drought conditions are not the primary factor promoting fires in dry, low-elevation pine forests, as they are in subalpine forests. Summers in the low-elevation forests are typically dry enough to promote low fuel moisture levels that would permit ignition, although the abundance and continuity of fine surface fuel historically were the primary limiting factors (Swetnam and Baisan 1996, Rollins et al. 2002).

Unlike the historical fire regime in subalpine forests, the fire regime in dry, low-elevation ponderosa pine forests has been significantly altered as a result of fire suppression and its effects on historical fuel structure (Arno and Gruell 1983, Swetnam and Baisan 1996, Veblen et al. 2000). Before fire suppression, the frequent, low-severity surface fires in these forests kept dry ponderosa pine stands sparse and open by killing young, newly established trees. With fire suppression and livestock grazing (which reduces the amount of grass fuel), fire intervals have lengthened, and dense stands have developed in which fine grass fuels are less abundant and dense ladder fuels are capable of carrying fire up into the canopy (figure 6). Consequently, high-severity fires potentially can occur in dry ponderosa pine forests, where historically they were rare because of the sparse ladder fuels and the lack of contiguous tree crowns. This pattern has been well documented on the basis of fire scars, repeat photography, and stand age structures, especially for forests in Arizona and New Mexico (Covington and Moore 1994, Allen et al. 1998, Mast et al. 1999, Moore et al. 1999), for some sites in the Colorado Front Range (Veblen and Lorenz 1991, Brown et al. 1999, Kaufmann et al. 2000), and for portions of the Bitterroot Range in Montana (Gruell 1983, Arno et al. 1995). As a consequence of fire suppression, the size and occurrence of high-severity fires has increased in this forest type. Reduction of ladder fuels through mechanical thinning and prescribed fire can effectively reduce the unprecedented occurrence of extensive crown fires and restore the historical surface fire regime in dry, low-elevation ponderosa pine forests (Covington et al. 1997, Allen et al. 2002, Fule et al. 2002).

Case study: The 2002 Rodeo-Chediski fire complex. The Rodeo-Chediski fire, which burned 189,095 ha in northern Arizona from 18 June through 7 July 2002, was the largest Arizona fire in recorded history. The area burned was dominated by ponderosa pine, with isolated pockets of mixed conifers at higher elevations along the Mogollon Rim, where the northern half of the fire burned. Fire-history studies conducted before the fire, in nearby ponderosa pine stands, record frequent surface fires with mean fire intervals of 7 to 10 years (based on fires recorded by more than 10% of sampled trees in 10- to 100-ha study areas; Swetnam and Baisan 1996). In 2002, high-severity crown fire affected 48% of the Rodeo-Chediski fire area, an extent of severe burning that is unprecedented in the low-elevation, dry ponderosa pine forests of this area.

The summer of 2002 marked the fourth year of drought in the Southwest. That May had been the second driest on record across Arizona and New Mexico in 108 years. Levels of fuel moisture before the fire were unusually low: 7% in 1000-hour fuels, as low as 2% in 10-hour (0.6- to 2.4-cm diameter) and 100-hour fuels, and below critical thresholds in live pine and brush fuels (Wilmes et al. 2002). The Haines index is a measure of lower-atmosphere stability and dryness correlated with wildfire growth. Low values (2 or 3) indicate moist, stable conditions; the highest values (5 or 6) represent dry, unstable conditions that favor moderate to high fire activity. The Haines index was 6 on many days during the Rodeo-Chediski fire.

Prescribed fire, salvage logging in previously burned stands, and fuel-reduction treatments (including the removal of slash, or woody debris, from branches and treetops) were effective in reducing fire severity and spread in the Rodeo-Chediski fire, even under extreme weather conditions (figure 7; Wilmes et al. 2002), as predicted by restoration research in Arizona (Fule et al. 2002). High-severity crown fires affected 35% of the stands that had been treated within the last 15 years, compared with 55% of the untreated stands. The average stand density of treated and untreated stands was 387 and 1108 trees per hectare, respectively. All prefire fuel treatments appeared to lower burn severity except for precommercial treatments, which increased it. In precommercial treatments, slash (branches and tree tops) was lopped and scattered throughout the stand, which contributed to higher fuel loads than those in untreated stands. Areas that had high forage production and low tree density experienced less severe burning during the Rodeo-Chediski fire, suggesting that open stands with abundant fine surface fuels were more resistant to high-severity canopy fire (figure 8). Overall, burn severity was positively correlated with overstory tree density (Wilmes et al. 2002). This outcome, in clear contrast with the findings from Yellowstone (where weather rather than fuel type and arrangement influenced fire behavior), highlights the heterogeneity of forest types and fire effects across the Rocky Mountain region.

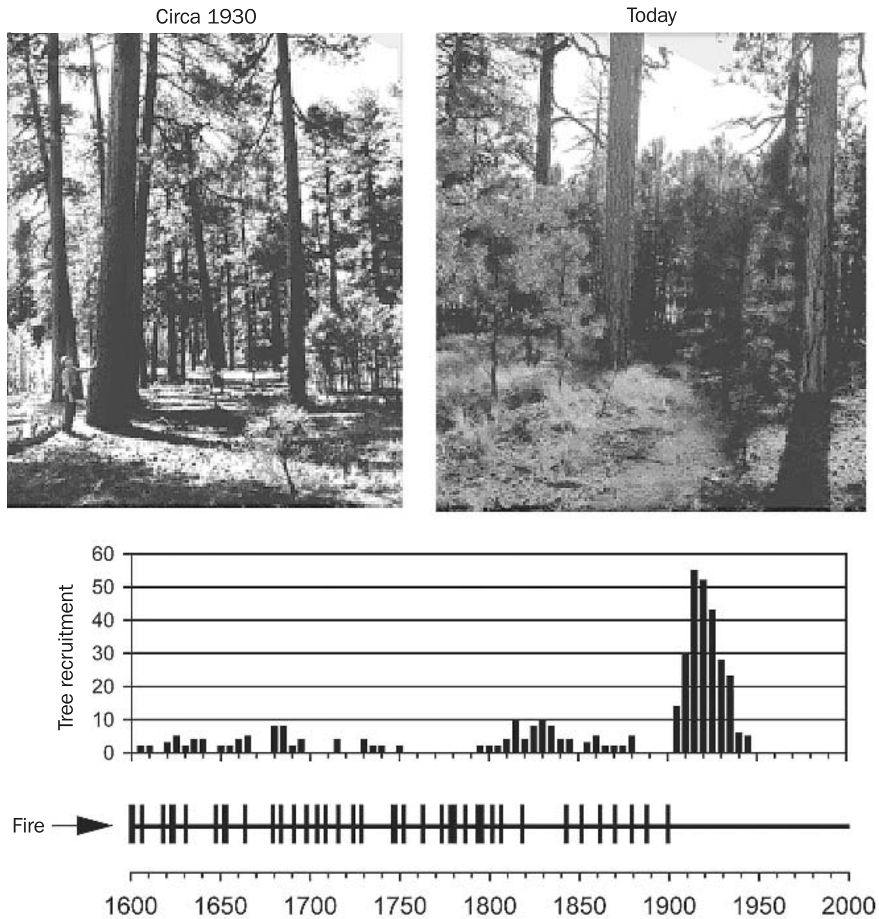

Figure 6. A comparison of historical and contemporary stand structure of dry ponderosa pine stands from the Jemez Mountains of New Mexico, and the relationship of this change to the frequency of low-severity surface fires. Source: Modified from Allen et al. 1998.

Summary: Low-severity fire regimes in low-elevation ponderosa pine forests. Dry, low-elevation ponderosa pine forests in the Rocky Mountain region, which were historically characterized by frequent low-severity fire regimes, make up an estimated 19% to 37% of the forested area that encompasses the three forest types discussed in this article (table 1). Such historically sparse forests, subject to high-frequency fires, comprise much of the ponderosa pine forest in Arizona and New Mexico but only a small fraction of the ponderosa pine forest in the central and northern Rockies. Regional modeling of fire regimes, based on a large fire-history database from the western United States, similarly predicts decreasing fire frequency from southern to northern latitudes (McKenzie et al. 2000). Important lessons about fire regimes in dry, low-elevation ponderosa pine forests are listed below.

- The historical fire regime in these forests was characterized by frequent, low-severity surface fires.
- Historically, the frequency, size, and severity of fires were largely controlled by spatial and temporal variation in fine fuels.
- Fire suppression has significantly increased tree densities and ladder fuels in low-elevation ponderosa pine forests.
- As a consequence of this change in stand structure, unprecedented high-severity fires now occur.

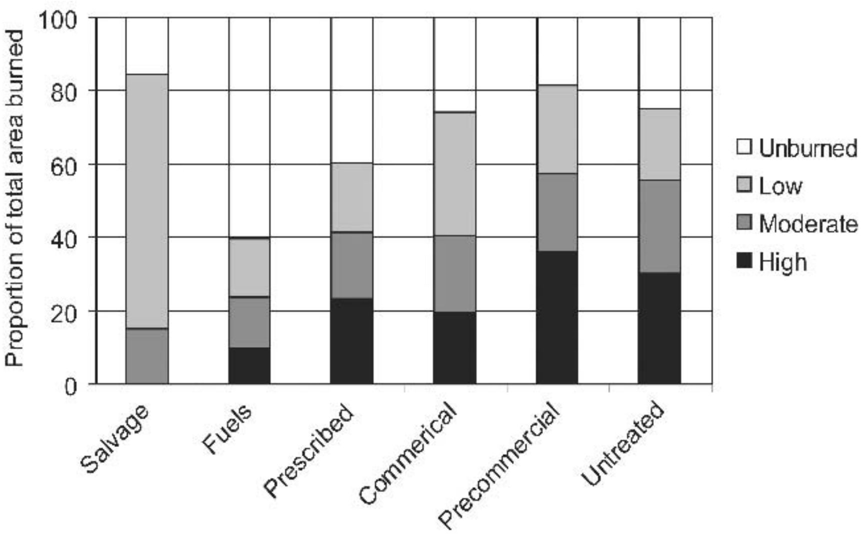

Figure 7. Proportion of different prefire fuel treatments burned at different severities during the Rodeo-Chediski fire in the Apache-Sitgreaves National Forests, Arizona, 2002. Burn severity, defined by the Burned Area Emergency Rehabilitation team **(www.fs.fed.us/r3/asnf/salvage/publications/proj_record/001_rodeo_baer_report_7-29-02.pdf),** ***ranges from unburned (surface fire with little or no canopy damage, tree foliage unscorched) through low severity (some tree crowns scorched but most trees not killed) and moderate severity (variable tree mortality, foliage scorched but not consumed) to high severity (complete tree mortality, foliage completely consumed). Fuel treatments are defined as salvage (removal of trees after a fire), fuels (thinning, chipping, and pile burning), prescribed fire (broadcast burning), commercial (removal, seed cut, regeneration, harvest, partial removal, final cut, or thinning), or precommercial (thinning with chipping, lopping, or both; no slash removal). Data are from Wilmes and colleagues (2002).***

- Fuel-reduction treatments involving mechanical thinning and prescribed fire are likely to be effective in mitigating extreme fire behavior and restoring this forest type to the historical fire regime.

Mixed-severity fire regimes

Mixed-severity fire regimes are intermediate between the infrequent, high-severity fire regimes of high-elevation subalpine forests and the frequent, low-severity fire regimes of dry, low-elevation ponderosa pine forests. Both high- and low-severity fires can occur at varying frequencies in mixed-severity fire regimes. This fire regime occurs predominantly at mid elevations, where topographic variation creates a complex moisture gradient resulting in a mosaic of tree species and densities that is sometimes referred to as mixed conifer forest. There is also evidence of mixed-severity fire regimes that predate fire suppression in some forests dominated by ponderosa pine, and even in pure or nearly pure ponderosa pine stands at low to mid elevation (Veblen and Lorenz 1986, Mast et al. 1998, Kaufmann et al. 2000, Ehle and Baker 2003).

Historically, forests that experienced mixed-severity fire regimes had variable densities of ponderosa pine, Douglas fir (*Pseudotsuga menziesii*), grand fir (*Abies grandis*), and western larch (*Larix occidentalis*), depending on their location. These forests constituted a mosaic of even-aged stands resulting from stand-replacing fire, interspersed with uneven-aged stands that experienced low-severity surface fires and episodic tree regeneration (Arno 1980, Brown et al. 1999, Kaufmann et al. 2000). Pre-1900 stand-replacing fires in these forest types have been documented by historic photographs and by the occurrence of even-age stand structures whose age corresponds to that of fire scars on adjacent trees (Gruell 1983, Veblen and Lorenz 1986, 1991, Arno et al. 1995, Swetnam and Baisan 1996, Shinneman and Baker 1997, Mast et al. 1998, Brown et al. 1999, Kaufmann et al. 2000, Ehle and Baker 2003). Low-severity fires are also well documented by historic photographs, fire scars, and all-age stands that include centuries-old trees, although these surface fires usually occurred less frequently than in the lower-elevation dry ponderosa pine forests described above (Arno 1980, Veblen and Lorenz 1991, Swetnam and Baisan 1996, Brown et al. 1999, Moore et al. 1999, Kaufmann et al. 2000, Veblen et al. 2000). The relative importance of surface versus crown fires and the size of these post-disturbance patches in shaping forests of mixed-severity fire regimes remain uncertain and have probably varied spatially and temporally.

Since the late 19th century, the densities of relatively fire-intolerant and shade-tolerant species, such as Douglas fir and grand fir, have increased in response to the suppression of low-severity fires in areas that historically experienced mixed-severity fire regimes (Arno et al. 1995, Kaufmann et al. 2000). Increases in density probably have occurred more commonly at lower elevations, on drier aspects, and adjacent to grasslands where frequent, low-severity fires were more dominant historically. Sites that previously supported denser stands because of favorable topographic and edaphic conditions have probably changed less as a result of fire suppression; those sites historically experienced stand-replacing fires, and high stand densities are a normal part of the postfire recovery process (Veblen and Lorenz 1986, Arno et al. 1995, Mast et al. 1998, Kaufmann et al. 2000, Ehle and Baker 2003). With fire suppression, forests that historically experienced mixed-severity fire regimes have developed a more homogenous forest structure across the landscape, resulting in larger areas of continuously dense forest and perhaps in larger patches of crown fire than were witnessed historically. In some areas, tree regeneration following logging of these forests in the late

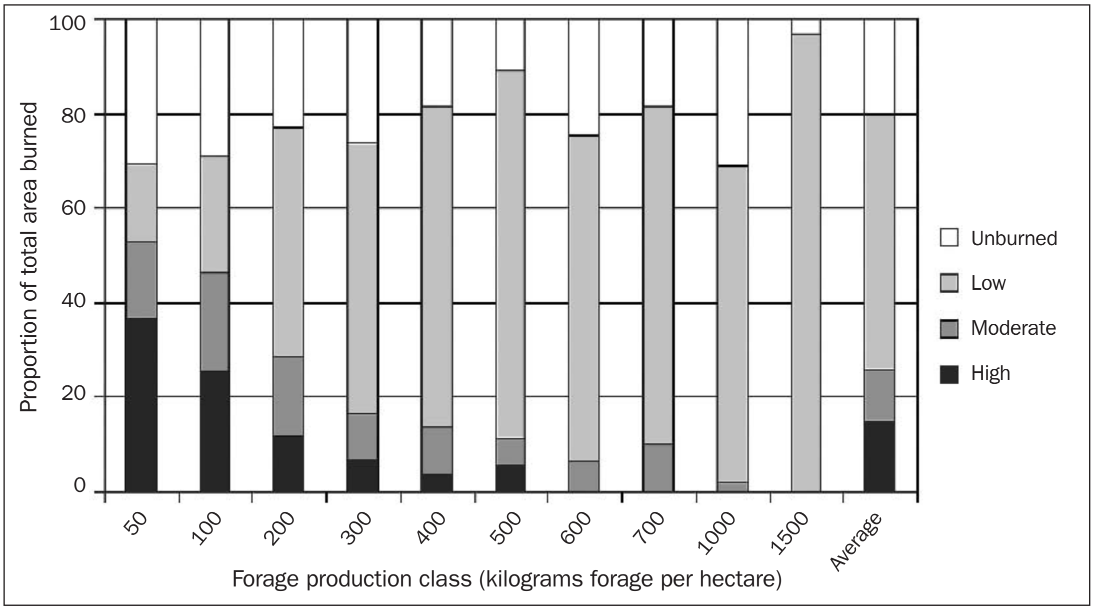

Figure 8. Proportion of different forage production classes burned at different severities during the Rodeo-Chedeski fire in relation to forage production classes for Carlisle and Town Tank allotments on the Lakeside Ranger District, Apache-Sitgreaves National Forest. Data are from Wilmes and colleagues (2002).

19th and early 20th centuries has contributed to high stand densities (Veblen and Lorenz 1986, Kaufmann et al. 2000). Overall, fire suppression has probably significantly affected only sites within the mixed conifer zone at lower elevations, on drier aspects, and adjacent to grasslands where fires historically were more frequent. Therefore, current fire regimes and stand densities in mixed conifer forests are likely to be within the historical range of variability, or at least are not likely to be as far outside this range as those in the dry ponderosa pine forests discussed above (Veblen 2003). However, additional research is needed on the causes of variability in mixed-severity fire regimes and the attendant effects of fire suppression.

In mixed-severity fire regimes, climate and fuels interact in a complex manner to control the frequency and severity of fires. Arno (1980) describes this interaction in mixed-severity fire regimes: "Under severe burning conditions, especially with strong winds, fires sometimes crowned and covered sizeable areas. When conditions moderated, fire would creep along the ground, with occasional flare-ups. Often the major fires burned at several intensities in reaction to changes in stand structure, fuel loadings, topography, and weather. The result was a mosaic of fire effects on the landscape" (p. 463). In mixed-severity regimes, in contrast to the previous two types of fire regime discussed, both climate and fuels (surface and ladder fuels) vary considerably and are important drivers of fire frequency and severity. We look to the example of the Hayman fire to tease apart these interactions in more detail.

Case study: The 2002 Hayman fire. The Hayman fire burned a 55,915-ha area southwest of Denver, Colorado, where previous fire history and forest structure studies (Brown et al. 1999, Kaufmann et al. 2000), mechanical fuel treatments, and burns (wild and prescribed) had occurred. Making use of this unplanned experiment, researchers assessed the relative effect of fuels and climate on fire behavior in the area, which had a historical mixed-severity fire regime (Finney et al. 2003).

Short-term drought during the 5 years before the fire created important antecedent conditions. In particular, below-normal precipitation and unseasonably dry air masses had persisted since 1998, when drier-than-average La Niña conditions began to develop. These conditions persisted intermittently through the spring of 2002. As a consequence, the Colorado Front Range received low snow during the winters of 2001 and 2002, with snowpack recorded in May 2002 at less than 50% of normal levels. By spring 2002, measurements of large-fuel moisture (moisture in 100-hour and 1000-hour fuels) in mid- to low-elevation forests of the southern Rockies were among the driest in the previous few decades, dipping as low as 3% when typically they exceed 12% (Graham 2003).

The size and severity of the Hayman fire can largely be explained by the extreme fire activity during two separate periods associated with sustained, exceptionally dry, forceful winds. First, on 9 June, the fire grew from 485 to 24,700 ha (43% of the total fire size); later, on 18 June, it traveled 5 miles along its southeastern flank (figure 9). During these two periods, mean relative humidity dipped below 8%, maxi-

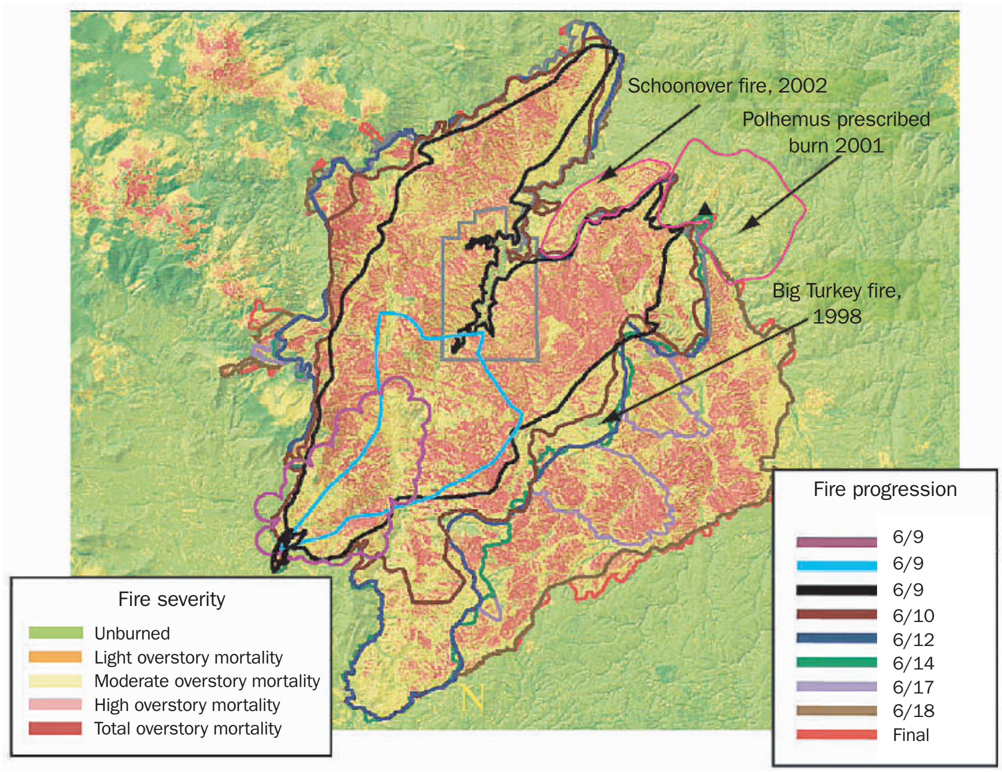

Figure 9. Map of the Hayman fire progression during the period 9–18 June 2002. Note the significant progress of the fire on 9 June (black line) and 18 June (brown line). Not all days are shown, because fire perimeters on slow-growth days overlapped previous days. Burn severity classes are based on the difference-normalized burn ratio from the US Geological Survey's National Burn Severity Mapping Project. Gray line represents the Cheesman Reservoir boundary, pink lines represent the perimeter of recent burns. Source: Modified from Finney and colleagues (2003).

mum wind gusts reached 84 miles (135 km) per hour, and the Haines index was 6, marking very dry, unstable conditions conducive to high fire spread. Both periods produced extensive torching, crown fire, and spotting (firebrands thrown in advance of the fire). These high-activity periods terminated with the passage of fronts followed by upslope winds that substantially increased ambient relative humidity (Finney et al. 2003).

During the substantial fire-progression days of 9 and 18 June, most fuel treatments had very little impact on the severity or direction of the fire (Finney et al. 2003). On 9 June, for example, the burned area included more than 2400 ha that had experienced previous prescribed fires or other fuel-reduction treatments. These treatments, which included previous wildfires (in 1963 and 1998), prescribed fires (in 1990, 1992, 1995, and 1998), and numerous stand modifications with and without subsequent slash removal (table 2), had virtually no effect on the Hayman fire. This is in marked contrast to the behavior of the Rodeo-Chediski fire, whose severity was affected by previous fuel-reduction treatments even under extreme climate and weather conditions. In the Hayman fire, extreme weather conditions overwhelmed the effectiveness of most fuel treatments. However, the fire stopped abruptly at the edge of the area that had been burned by two fires months to weeks before, in fall 2001 (Schoonover fire) and May 2002 (Polhemus prescribed burn), where very little fuel had accumulated during a spring of extreme drought (figure 9; Finney et al. 2003). Overall, the direction, severity, and size of the fire on extreme days were mostly explained by high wind and low relative humidity (table 3), with little effect of past fire or thinning activity. The Hayman review team concluded that "fuel modifications generally had little influence on the severity of the Hayman Fire during its most significant run on June 9th" (Finney et al. 2003) but acknowledged that the small size of these treatments contributed to their lack of effectiveness. On days of moderate fire growth, however, fuel modifications did influence fire spread and severity; of these modifications, recent wild or prescribed fires and thinning with slash removal were most effective. In an example of the interactions between fuels and climate, on 17 June the Hayman fire split into two runs on either side of the area burned by the Big Turkey fire in 1998 (figure 9); however, when the weather became more extreme the following day, this effect on fire shape and extent was obliterated (figure 9; compare 17 June and 18 June perimeters).

Table 2. Distribution of fire severity classes among fuel-modified areas on moderate slopes (defined as slopes of less than 30%) that burned in the Hayman fire on 9 June 2002.

Level of prefire fuel modification	Area (ha)	Fire severity class (percentage) Unburned	Low	Moderate	High
Unmodified	9128	4	18	8	70
		Recent modifications (after 1990)			
Wildfires	5	0	0	25	75
Prescribed fires	291	6	20	11	63
Fuel treatment	0	NA	NA	NA	NA
Improvements and treatment	160	0	19	7	74
Improvements, no treatment	253	3	12	9	76
Harvest and treatment	657	5	14	10	71
Harvest, no treatment	236	0	1	33	66
Plantation	55	0	8	5	87
		Older modifications (before 1990)			
Wildfires	Unknown	NA	NA	NA	NA
Prescribed fires	34	17	50	8	25
Fuel treatment	2	0	86	14	0
Improvements and treatment	0	NA	NA	NA	NA
Improvements, no treatment	592	1	14	8	77
Harvest and treatment	1	0	16	9	75
Harvest, no treatment	384	3	27	2	68
Plantation	127	0	27	10	63

Source: Finney et al. 2003.

Summary: Mixed-severity fire regimes in the Rocky Mountain region. Mixed-severity fire regimes account for an estimated 17% to 50% of the forested area in the Rocky Mountain region that encompasses the three major forest types discussed in this article (table 1). These forests experience the most complex type of fire regime and the least understood. Nonetheless, we have learned several important lessons about mixed-severity fire regimes in Rocky Mountain forests.

- The historical fire regime in these forests is complex, including both low-severity surface fires and infrequent high-severity crown fires.
- Both fuels and climate have major influences on the frequency, severity, and size of fires.
- Fire suppression has had variable effects on fuel densities in mixed-severity fire regimes, with the greatest impacts on sites that formerly supported open woodlands.
- The occurrence of high-severity crown fires is not outside the historical range of variability, although their size and frequency may be increasing.
- Extreme climate and weather conditions can override the influence of stand structure and fuels on fire behavior.
- Fuel-reduction treatments (mechanical thinning and prescribed burning) may effectively reduce fire severity under moderate weather conditions, but these treatments may not effectively mitigate fire behavior under extreme weather conditions and may not restore the natural complexity of historical stand and landscape structure.

Implications for fire mitigation and restoration

What does an understanding of the spatial variation in dominant controls on wildfire frequency and severity mean for ecological restoration and for effective fuel treatments to reduce the threat of large, severe wildfires? The Yellowstone fires in 1988 revealed that variation in fuel conditions, as measured by stand age and density, had only minimal influence on fire behavior. Therefore, we expect fuel-reduction treatments in high-elevation forests to be generally unsuccessful in reducing fire frequency, severity, and size, given the overriding importance of extreme climate in controlling fire regimes in this zone. Thinning also will not restore subalpine forests, because they were dense historically and have not changed significantly in response to fire suppression. Thus, fuel-reduction efforts in most Rocky Mountain subalpine forests probably would not effectively mitigate the fire hazard, and these efforts may create new ecological problems by moving the forest structure outside the historic range of variability (Veblen 2003, Romme et al. 2004).

In contrast, for many low-elevation, dry ponderosa pine forests, it is both ecologically appropriate and operationally possible to restore a low-severity fire regime through thinning and prescribed burning (Covington et al. 1997, Allen et al. 1998, 2002). Fuels rather than climate appear to be the most significant factor affecting fire spread and severity in these forests. Fire suppression in dry ponderosa pine forests appears

Table 3. Comparison of the mean and range of weather indices associated with the type (high, moderate, or low) of fire-growth days during the Hayman fire, 9 June to 18 June 2002.

		Mean (range)			
Fire-growth days	*n*	Relative humidity (percentage)	10-minute average wind (kph)	Maximum wind gust (kph)	Haines index
Low	4	36.6 (8–68)	11.2 (0–30.4)	22.4 (3.2–57.6)	3.7 (2–6)
Moderate	4	27.6 (5–76)	11.2 (0–28.8)	24 (1.6–54.4)	4.2 (2–6)
High	2	7.8 (5–15)	16 (1.6–48)	38.4 (3.2–134.4)	5.7 (5–6)

kph, kilometers per hour.

Note: The Haines index, ranging from 2 to 6, measures the moisture and stability of the lower atmosphere; low values indicate moist, stable conditions, and high values indicate dry, unstable conditions conducive to fire. The two high fire-growth days occurred on 9 and 18 June. High- and moderate fire-growth days are identified on the Hayman fire progression map (figure 9); low fire-growth days are those omitted from the map because fire perimeters were not significantly different from previous days. Data are summarized from Finney and colleagues (2003).

to have contributed to an unprecedented buildup of fuels and to the occurrence of high-severity fires. Indeed, the objectives of fire mitigation and forest restoration generally converge in forests of this type.

Perhaps the most difficult forests to assess are the mid-elevation forests that historically were characterized by mixed-severity fire regimes. Because mixed-severity fire regimes are most complex and least well understood, we must exert caution in developing simple prescriptions for wildfire mitigation that may not bring predictable results under extreme climate conditions. Our analysis reveals that fire regimes, climate, fuel type and abundance, and stand structure vary significantly across the Rocky Mountain region. As a consequence, the heterogeneous forests in this region require very different approaches to restoration and wildfire management (Gutsell et al. 2001). Clearly, policymakers need to incorporate ecological heterogeneity into their decisions in order to implement sound forest management policy.

In addition to the fuel-management operations described above, we need new research to clarify the geographic variation in fire regimes across different forest types in this large, heterogeneous region. There is great geographical variation in the distribution of the three broad fire regimes defined here. In Montana, for example, subalpine forests cover roughly 40% of the forested area, while in Arizona the extent of these forests is significantly smaller and they are more isolated on scattered mountaintops. At a regionwide scale, it is difficult to define the precise extent of these different fire regimes and their spatial location (and especially to distinguish between the low-severity and mixed-severity fire regimes), as illustrated by the variation between the estimates based on PNV groups and those based on current cover type (table 1). There is also significant variation in fire regimes within each of the three broad fire-regime classes in response to local topography and landscape position, and there are other important vegetation types not covered in this brief article (e.g., piñon-juniper woodlands; Romme et al. 2003).

A "one size fits all" approach to reducing wildfire hazards in the Rocky Mountain region is unlikely to be effective and may even produce collateral damage in some places. We do not advocate delaying action until all of the ecological questions have been answered; in many places, there is an urgent need and a solid ecological basis for restoration and fire-mitigation efforts. In other areas, however, where the ecological basis for aggressive fuel reduction is inadequate or lacking, uncritical extrapolation of models from other systems may cause more harm than good.

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PERSPECTIVE

Adapt to more wildfire in western North American forests as climate changes

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Wildfires across western North America have increased in number and size over the past three decades, and this trend will continue in response to further warming. As a consequence, the wildland–urban interface is projected to experience substantially higher risk of climate-driven fires in the coming decades. Although many plants, animals, and ecosystem services benefit from fire, it is unknown how ecosystems will respond to increased burning and warming. Policy and management have focused primarily on specified resilience approaches aimed at resistance to wildfire and restoration of areas burned by wildfire through fire suppression and fuels management. These strategies are inadequate to address a new era of western wildfires. In contrast, policies that promote adaptive resilience to wildfire, by which people and ecosystems adjust and reorganize in response to changing fire regimes to reduce future vulnerability, are needed. Key aspects of an adaptive resilience approach are (*i*) recognizing that fuels reduction cannot alter regional wildfire trends; (*ii*) targeting fuels reduction to increase adaptation by some ecosystems and residential communities to more frequent fire; (*iii*) actively managing more wild and prescribed fires with a range of severities; and (*iv*) incentivizing and planning residential development to withstand inevitable wildfire. These strategies represent a shift in policy and management from restoring ecosystems based on historical baselines to adapting to changing fire regimes and from unsustainable defense of the wildland–urban interface to developing fire-adapted communities. We propose an approach that accepts wildfire as an inevitable catalyst of change and that promotes adaptive responses by ecosystems and residential communities to more warming and wildfire.

wildfire | resilience | forests | wildland–urban interface | policy

Wildfire is a key driver of ecosystem change that increasingly poses a significant threat and cost to society. In western North America (hereafter, the West), warming, frequent droughts, and legacies of past management combined with expansion of residential development have made social–ecological systems (SESs) more vulnerable to wildfire. As the annual area burned has increased over the past three decades, we are confronting longer fire seasons (1, 2), more large fires (3, 4), a tripling of homes burned (5), and more frequent large evacuations. In 2016, the Fort McMurray Fire in Alberta, Canada and the Blue Cut Fire in southern California prompted evacuation orders for a combined total of more than 160,000 people. The costs of wildfire have also risen substantially since the 1990s. The US Congress appropriated $13 billion for fire suppression and $5 billion for fuels management in fiscal years 2006–2015 (6). Other societal costs, including real estate devaluation, emergency services, and postfire rehabilitation, total up to 30 times the direct cost of firefighting (7).

Notwithstanding these costs, many plants, animals, and ecosystem services benefit from fire, and those dependent on frequent fire have been negatively affected by the significantly reduced burning resulting from fire suppression, as compared with the period before European settlement

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(8). However the response of ecosystems to increases in wildfire activity and warming in the coming decades is not well understood. Broad heterogeneity among western forest landscapes in terms of biophysical environment, past management, human footprint, and the role of fire and future warming creates a complicated playing field. Managing ecosystems, people, and wildfire in a changing climate is a complex but critical challenge that requires effective and innovative policy strategies (9, 10).

Our key message is that wildfire policy and management require a new paradigm that hinges on the critical need to adapt to inevitably more fire in the West in the coming decades. Policy and management approaches to wildfire have focused primarily on resisting wildfire through fire suppression and on protecting forests through fuels reduction on federal lands. However, these approaches alone are inadequate to rectify past management practices or to address a new era of heightened wildfire activity in the West (11–14).

In delivering this message, we focus specifically on the distinction between specified, adaptive, and transformative resilience (15, 16). Rigorous definition and critical assessment of resilience to wildfire are needed to develop effective policy and management approaches in the context of climate change. We suggest an approach based on the concept of adaptive resilience, or adjusting to changing fire regimes (e.g., shifts in prevailing fire frequency, severity, and size) to reduce vulnerability and build resilience into SESs. Adaptive resilience to wildfire means recognizing the limited impact of past fuels management, acknowledging the important role of wildfire in maintaining many ecosystems and ecosystem services, and embracing new strategies to help human communities live with fire. Our discussion focuses on western North American forests but is relevant to fire-influenced ecosystems across the globe. We emphasize that long-term solutions must integrate relevant natural and social science into policies that successfully foster adaptation to future wildfire.

Why Has Coping with Wildfire Become Such a Challenge?

Three primary factors have produced gradual but significant change across western North American landscapes in recent decades: the warming and drying climate, the build-up of fuels, and the expansion of the wildland–urban interface (WUI; the zone where houses meet or intermingle with undeveloped wildland vegetation).

In terms of climate, wildfire activity is closely tied to temperature and drought over time scales of years to millennia (2, 17–19). Globally, the length of the fire season increased by 19% from 1979 to 2013, with significantly longer seasons in the western United States (1). Since 1985, more than 50% of the increase in the area burned by wildfire in the forests of the western United States has been attributed to anthropogenic climate change (20). Increases in the number of wildfires and area burned in most forested ecoregions of the West are a result of rising temperatures, increased drought, longer fire seasons, and earlier snowmelt (1–4, 21). Specifically, since the 1970s the frequency of large fires has increased most dramatically in the forests of the Northwest (1,000%) and Northern Rocky Mountains (889%), followed by forests in the Southwest (462%), Southern Rockies (274%), and Sierra Nevada (256%), in response to earlier snowmelt and a longer fire season (21). Based on spatial overlays in western United States forests of large wildfires since 1984 (Monitoring Trends in Burn Severity, available at www.mtbs.gov/dataaccess.html and Existing Vegetation Types, available at https://www.landfire.gov/vegetation.php), we found that in northern regions with dramatic increases in fire activity (the Canadian Rockies, Middle Rockies, and Idaho Batholith ecoregions) cold/wet subalpine forests predominantly burned. These forests characteristically burn at high severity and have not experienced a significant build-up of fuels. Overall, cold/wet forests account for about a quarter of total forest burning in the US West since 1984.

Fire suppression, in addition to past logging and grazing and invasive species, has led to a build-up of fuels in some ecosystems, increasing their vulnerability to wildfire. For example, drier, historically open coniferous forests in the West ("dry forests") have experienced gradual fuels build-up in response to decades of fire suppression and other land-use practices (8, 22, 23). Historically, predominantly frequent, low-severity fires killed smaller, less fire-resistant trees and maintained low-density dry forests of larger, fire-resistant trees. Large, high-severity fires now threaten to convert denser, more structurally homogeneous dry forests to nonforest ecosystems, with attendant loss of ecosystem services (24). However, only forests in the Southwest show a clear trend of increasing fire severity over the last three decades, and only a quarter to a third of the area burned in the western United States experienced high severity during that time (25, 26). Although fuels build-up in dry forests can increase the area burned because of higher contagion, the 462% increase in the frequency of large fires in southwestern forests since the 1970s is also a result of an extension of the fire season by 3.6 mo [the average for the western United States is 2.8 mo (21)]. Overall, dry forests account for about half of the total forest burning in the western United States since 1984.

Alongside these increases in warming and fuels, the WUI has expanded tremendously in the past few decades, augmenting wildfire threats to people, homes, and infrastructure. Between 1990 and 2010, almost 2 million homes were added in the 11 states of the western United States, increasing the WUI area by 24% (27). Currently, most homes in the WUI are in California (4.5 million), Arizona (1.4 million), and Washington (1 million) (27). Since 1990, the average annual number of structures lost to wildfire has increased by 300%, with a significant step-up since 2000 (28). About 15% of the area burned in the western United States since 2000 was within the WUI, including a 2.4-km community protection zone, with the largest proportion of wildfires burning in the WUI zone in California (35%), Colorado (30%), and Washington (24%) (Fig. 1) (27). Additionally, almost 900,000 residential properties in the western United States, representing a total property value more than $237 billion, are currently at high risk of wildfire damage (29). Because of the people and property values at risk, WUI fires fundamentally change the tactics and cost of fire suppression as compared with fighting remote fires and account for as much as 95% of suppression costs (28). Together, these gradually changing variables—climate change, fuels build-up, and residential development—interact with rapid combustion to increase wildfire risks and costs to society and some ecosystems substantially.

Potential Consequences of Future Wildfire

Wildfire activity is predicted to increase in the West over the next century (20, 30, 31). This anticipated ramp-up in burning and possible directional changes in fire regimes (e.g., increases in fire frequency, severity, and/or size) could transform the composition, structure, and function of many forest (8, 32, 33), shrubland, and grassland ecosystems (34). Changes in temperature and precipitation in semiarid shrublands and grasslands may reduce fuel availability subsequently, to the extent that fire occurrence, size, and severity in such areas will eventually decline (35). Thus, although fire activity is projected to increase in the West in the near term (i.e., in the next few decades), longer regional trends will depend on feedbacks between vegetation and fire as well as on anthropogenic alterations in vegetation and land use (36, 37).

Increased exposure of communities to wildfire is also expected with additional warming. More than 3.6 million ha, or almost 40% of the current WUI in the western United States, is predicted to experience moderate to large increases in the probability of wildfire in the next 20 y (Fig. 2). This increase is in addition to the growing wildfire risk to developed nonurban areas (e.g., energy production) and infrastructure (e.g., power lines, pipelines) that define a broader wildland–development

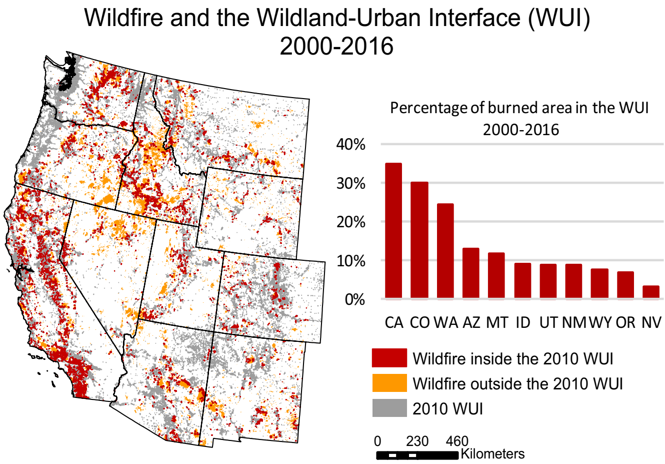

Fig. 1. (*Left*) **Area burned by wildfires between 2000 and 2016 across the western United States inside and outside the 2010 WUI including a 2.5-km community protection zone (27). (*Right*) About 15% of the WUI burned during this period, with largest proportions of the WUI burning in California, Colorado, and Washington.**

interface. Continued WUI growth will further increase human exposure to wildfires (38) and anthropogenic ignitions (37, 39). By midcentury, 82 million people in the western United States are likely to experience more and longer "smoke waves," defined as consecutive days of high, unhealthy particulate levels from wildfires (40). Climate change and increasing exposure of existing and future development to wildfire and smoke present a dangerous and vexing problem for residents, local officials, fire fighters, and managers.

Gradual but significant changes in climate, fuels, and the WUI affect wildfire impacts on ecosystems and society but are difficult to recognize and are challenging to alter meaningfully. There often is a lack of political will to implement policies that incur short-term costs despite their long-term value or to change long-standing policies that are ineffective. For example, few jurisdictions have the will or means to restrict further residential development in the WUI, although modifying and curtailing residential growth in fire-prone lands now would reduce the costs and risks from wildfire in the long term. Furthermore, although the impacts of fire suppression on fuels build-up are now well understood, fire-suppression policies still dominate current fire management (13). Projected global warming of at least 1.1–3.1 °C in the coming century offers a unique opportunity to change policy and the course of our response to wildfires (41). A paradigm shift now in approaches to WUI development and management of fire and fuels can yield tremendous benefits to society later.

Specified, Adaptive, and Transformative Resilience to Wildfire

Resilience is increasingly invoked as a guiding principle in strategies that address the social and ecological dimensions of wildfire. The US Forest Service's National Cohesive Wildland Fire Management Strategy (42) specifically addresses the need to bolster social and ecological resilience to increasing wildfires. Although often invoked in wildfire management and policy, resilience is defined inconsistently or neglects social or ecological contexts, despite the need for uniformity and specification in setting goals and evaluating progress (43, 44).

Defining resilience to wildfire in an SES is especially challenging in the WUI, where people, ecosystems, and wildfire interact over multiple spatial and temporal scales (12). An SES is the intersection and interdependence of biophysical units and associated people and institutions. Resilience in an SES generally has been defined as the capacity to absorb disturbance so as to retain essential structures, processes, and feedbacks and to adapt to and reorganize following disturbance (45). These perspectives of resilience, absorbing versus adapting to disturbance, offer different guiding principles for policy and management in responding to wildfire and measuring success over different planning timelines (44). Here we outline a consistent framework that defines resilience to wildfire in coupled SESs based on the concepts of specified resilience and general resilience, the latter of which includes adaptive and transformative approaches (Table S1) (15, 16, 44).

When climate trends or disturbance regimes are relatively stable and well-characterized and planning horizons are short (years), specified resilience or restoration is an appropriate guiding principle. "Specified resilience" refers to the buffer capacity of a system to retain its identity after a well-specified disturbance (16). Specified resilience reflects the concept of ecological resilience, which refers to the capacity of a system to absorb or tolerate disturbance without shifting to a qualitatively different state controlled by a different set of processes (46). In terms of wildfire, specified resilience applies when fire characteristics are within the bounds of historical range of variability (HRV) of disturbance regimes and a burned forest recovers without converting to another state, e.g., to a nonforest state such as a persistent grassland. In a social context, specified resilience is evident when a community recovers economically and rebuilds similar structures in similar locations following a wildfire (44, 47). Management guided by specified resilience often values recent ecological and social dynamics, particularly when the goal is the conservation of particular species or landscapes. Such management is often informed by short temporal windows of HRV, or "recent HRV" (rHRV) (Fig. 3). This approach can be useful for responding to fires in the short term. However, when social and environmental conditions change rapidly, this approach may foster management goals that are unrealistic or unsustainable in the long run (48, 49).

When climate and wildfire trends are changing and planning horizons are intermediate (decades), general resilience is a more appropriate and desirable guiding principle. "General resilience" refers to the capacity of an SES to adapt or transform in response to unknown shocks or disturbances outside the rHRV (16). Adaptive resilience incorporates aspects of change, reorganization, learning, and adaptability in response to changing climate and disturbance regimes and is an on-going process achieved by harnessing adaptive capacity. In an ecological context, adaptive resilience refers to actively or passively supporting species compositions and fuel structures that are better adapted to a warming, drying climate with more wildfire. Management of specified resilience maintains ecosystems within the rHRV,

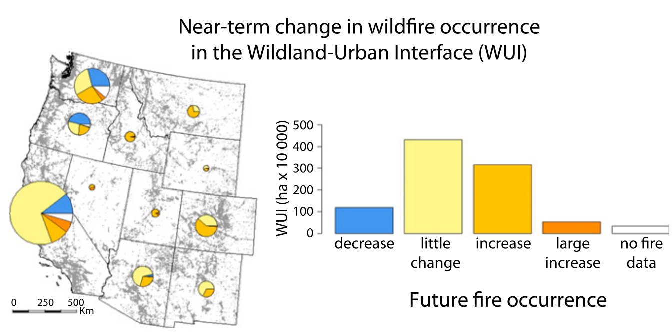

Fig. 2. (*Left*) **Area of the WUI in the conterminous western United States, classified according to projected near-term changes in fire occurrence. The size of each pie is scaled relative to the area of the WUI (both intermix and interface) in each state, based on data from Martinuzzi, et al. (27). Within each pie, slices represent the proportion of WUI area overlapping the five categories of projected fire occurrence for the period 2010–2039, based on data from Moritz, et al. (30). (*Right*) The bar chart summarizes the area of the WUI projected to experience each level of change in fire occurrence in the western United States.**

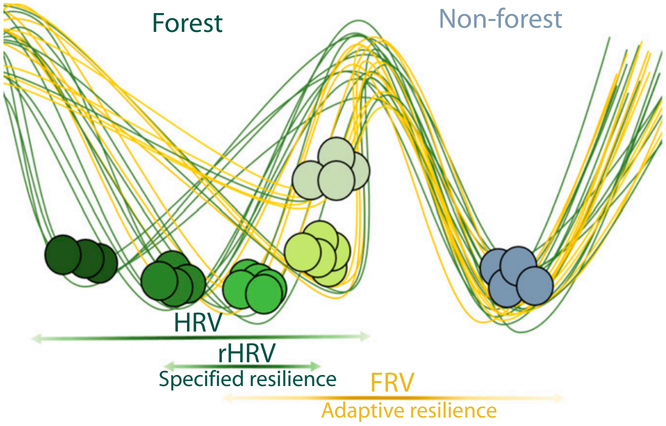

Fig. 3. **Conceptual ball-and-basin representation of specified and adaptive resilience across a forested landscape. Lines defining basins depict the ranges of variation in fire regimes across forest types. Sets of green balls reflect the variation in abundance and composition within different forest types, and the set of blue balls represents nonforest ecosystems. Specified resilience of forests to wildfire is maintained within basins that fall within an rHRV of fire regimes over recent decades to centuries, typically derived from historical documents, remotely sensed data, and tree-ring data. Longer definitions of HRV reflect variation in fire regimes over the last 4,000–5,000 y, when present-day forest types were established in most regions; these data are derived from paleoecological reconstructions. Adaptive resilience to changing fire regimes is reflected within basins that fall within the FRV (yellow). Under the FRV, shifts to nonforest ecosystems remain unlikely in some cases (lower green balls) and more likely in other cases with easier transition to nonforest basin (higher green balls). Changes in the severity, frequency, and size of fire regimes and long-term regeneration following fire events reflect adaptive responses to changing fire regimes and climate conditions across broad scales.**

whereas managing for adaptive resilience considers how changing disturbance regimes may favor suites of traits that are better adapted to a future range of variability (FRV) (Fig. 3) (22). Alignment of fire regimes with adaptive regeneration traits of native vegetation defines a safe operating space (50). The HRV can still play a role by providing insight into how adaptive traits align with changing disturbance regimes to confer adaptive resilience, but under the FRV the safe operating space is shifting (Fig. 3) (50, 51, 52). In a social context, communities exhibiting adaptive resilience engage in ecological, psychological, social, and policy processes that set the community on a trajectory of change to reduce future vulnerability (Fig. 4) (53). Strategies may include changing building codes to make structures more fire-resistant, planning communities to avoid or withstand future wildfire, or providing incentives, education, and resources to reduce vulnerability to future wildfire (47). Adaptive resilience also involves institutional learning, where past management approaches to wildfire evolve.

When climate and wildfire trends are significantly altered from historical trends and/or variability, and planning horizons are long (century), transformative resilience may be necessary. "Transformative resilience" refers to planned fundamental change in response to drastically altered disturbances that have the potential to create broad-scale, systemic shifts in ecological states or radical shifts in values, beliefs, social behavior, and multilevel governance. Examples might include significant regional changes in ecosystem states and associated loss of ecosystem services and/or the relocation of communities of people away from wildfire-prone areas (44, 54). Rapid, planned social–ecological transformation is rare and difficult to implement because of uncertainties about future risk, inflexible institutions and behaviors, and the high cost of transformative action (55).

Although distinct, these approaches to resilience may be nested. Promoting specified resilience may make some forests better poised for adaptive resilience as climate changes, but in some forests or conditions specified resilience may not be effective as climate changes (e.g., refs. 56, 57). Allowing postfire shifts from forest to grassland or shrubland may increase adaptive resilience to changing wildfire and climate conditions. Approaches to adaptive resilience could reduce the need for transformation if efforts keep pace with climate and wildfire trends or may help pave the way toward inevitable social–ecological change. Embracing specified resilience may be the easiest, most familiar path with the least uncertainty, but this approach is short-sighted and could come at the cost of adaptation to future wildfire as climate change continues.

Taking an adaptive resilience approach now is critical, because specified resilience, although useful in some contexts, will become a less useful guiding principle as we exceed HRVs. Adaptive resilience means adjusting to changing fire regimes and climate—in both social and ecological systems—by taking advantage of opportunities to moderate potential impacts and cope better with the consequences. Adapting to wildfire sooner rather than later provides the widest benefits to society at the least cost. If we do not adapt to wildfire now, disruptive and unintended transformations of SESs in the West may ensue.

How Policy and Management Can Promote Adaptive Resilience to Wildfire

Current approaches to managing wildfire focus primarily on controlling fire through suppression and secondarily focusing on managing fuels build-up in forests. Within the context of current and future trends in wildfire, we evaluate the following three approaches in terms of their promise for fostering adaptive resilience in ecosystems and residential communities living with more wildfire: (*i*) managing fire, (*ii*), managing fuels, and (*iii*) promoting adaptive capacity (Fig. 5).

Fig. 4. **Wildfires are catalysts of change that promote adaptive resilience by communities and ecosystems to future wildfires. (*A* and *B*) Example of adaptation in communities. (*A*) A home burned in the 2010 Fourmile fire, Boulder County, CO, which at the time was the most destructive fire in Colorado history in terms of home loss. (*B*) A home that survived the 2016 Cold Springs fire, where many residents managed structural and vegetative fuels around their home to reduce fire hazard after the Fourmile fire through Boulder County's Wildfire Partners program. (*C* and *D*) Heterogeneity in wildfire severity promotes diversity in postfire regeneration and fuels in the 2002 Rodeo-Chediski fire, Coconino and Navajo counties, AZ (*C*) and the 2016 Canyon Creek fire, Grant County, OR (*D*). Photographs courtesy of REUTERS/Alamy Stock Photo (*A*), Wildfire Partners (*B*), Tom Bean/Alamy Stock Photo (*C*), and M.A.K. (*D*).**

Managing Wildfire

Suppressing Fewer Fires and Prescribing More Burning. Increasing the use of prescribed fires and managing rather than aggressively suppressing wildland fires can promote adaptive resilience as the climate continues to warm. Many dry forests currently experience significantly less burning than in the period just before European settlement (8, 35, 58). In recognition of the fire-dependence of many ecosystems, the 1995 Federal Wildland Fire Management policy ushered in the first federal policy aimed at reintroducing more wildfire on public lands; that policy remains in effect today. US federal agencies actively managed an average of 75,000 ha of lightning-caused fires per year under the Wildland Fire Use policy from 1998–2008 and currently burn about 1 million hectares per year with prescribed fires (58). However, prescribed fires still constitute only about 10% of the treatments implemented by the US Forest Service in the West and burn about one-third of the area burned by wildfires (National Interagency Fire Center, https://www.nifc.gov/). In the United States and Canada, suppression remains the primary approach to wildfire, with more than 95% of all wildfires suppressed (28). Continued aggressive fire suppression is counterproductive to building adaptive resilience to increasing wildfire in the long term (13, 14).

Using Fire to Foster Adaptive Resilience to Climate Change. In some systems, fire today attenuates future fire effects, because flames that burn dead and live fuel limit where and how severely subsequent fires burn, at least for a time (59–61). Fires often create complex patterns of burn severity that create variation in postfire regeneration and fuels (62–67). As fire regimes shift over time, individual fire events filter for species adapted to changing fire and climate conditions (68). Strategic planning for more managed and uncontrolled wild fires on the landscape today (69) may help decrease the proportion of large and severe wildfires in the coming decades and may enhance adaptive resilience to changing climate. Prescribed fires, ignited under cooler and moister conditions than are typical of most wildfires, can reduce fuels and minimize the risk of uncontrolled forest wildfire near communities. In contrast to wildfires, prescribed fire risks are relatively low, and more than 99% of prescribed fires are held within planned perimeters successfully (58).

Challenges to increasing use of managed and prescribed fires vary from the public's limited experience with smoke and wildfire to significant direct health impacts of smoke on vulnerable populations, including children, the elderly, and low-income communities (40, 70, 71). Some smoke hazards can be reduced through careful planning and management of fire, public health monitoring, and provisioning of health services for vulnerable populations. Public perceptions of fire are also an important hurdle, given the success of Smokey Bear's fire-prevention campaign and because most urban and suburban residents have very limited experience with wildfire compared with rural residents of the early 20th century. Therefore, public education programs that demonstrate the inevitability of wildfire will be a key aspect of living with increasing fire in the West. We need to develop a new fire culture. Despite these and various legal and operational challenges (58), the benefits of prescribed fire and managed wildfires to ecosystems and communities are high (72). Promoting more wildfire away from people and prescribed fires near people and the WUI are important steps toward augmenting the adaptive resilience of ecosystems and society to increasing wildfire.

Managing Fuels

Limiting Reliance on Fuels Treatments to Alter Regional Fire Trends. Managing forest fuels is often invoked in policy discussions as a means of minimizing the growing threat of wildfire to ecosystems and WUI communities across the West. However, the effectiveness of this approach at broad scales is limited. Mechanical fuels treatments on US federal lands over the last 15 y (2001–2015) totaled almost 7 million ha (Forests and Rangelands, https://www.forestsandrangelands.gov/), but the annual area burned has continued to set records. Regionally, the area treated has little relationship to trends in the area burned, which is influenced primarily by patterns of drought and warming (2, 3, 20). Forested areas considerably exceed the area treated, so it is relatively rare that treatments encounter wildfire (73). For example, in agreement with other analyses (74), 10% of the total number of US Forest Service forest fuels treatments completed 2004–2013 in the western United States subsequently burned in the 2005–2014 period (Fig. 6). Therefore, roughly 1% of US Forest Service forest treatments experience wildfire each year, on average. The effectiveness of forest treatments lasts about 10–20 y (75), suggesting that most treatments have little influence on wildfire. Implementing fuels treatments is challenging and costly (7, 13, 76, 77); funding for US Forest Service hazardous fuels treatments totaled \$3.2 billion over the 2006–2015 period (6). Furthermore, forests account for only 40% of the area burned since 1984, with the majority of burning in grasslands and shrublands. As a consequence of these factors, the prospects for forest fuels treatments to promote adaptive resilience to wildfire at broad scales, by regionally reducing trends in area burned or burn severity, are fairly limited.

Targeting Fuels Treatments in Ecosystems with Fuel Build-Up and on Private Lands. Strategically targeting treatments in areas where fuels build-up has increased the expected burn severity may augment the adaptive resilience of those ecosystems to increasing wildfire. For example, treating drier forests, where the likelihood of fire is

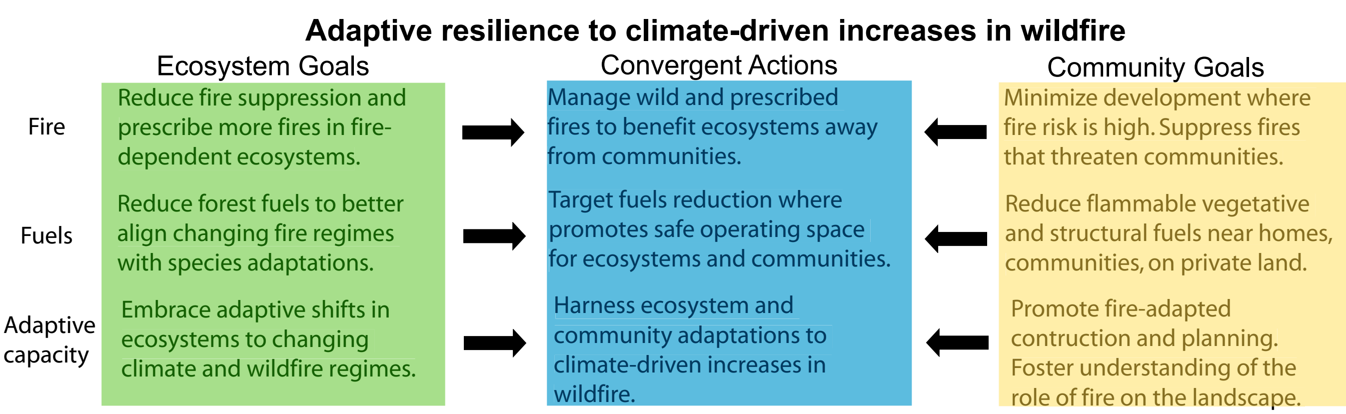

Fig. 5. Convergent actions that promote adaptive resilience to climate-driven increases in wildfire in the West by ecosystems and communities, based on goals related to management of fire, fuels, and adaptive capacity.

high, may also increase opportunities to modify wildfire behavior and postfire recovery. Burn severity has increased because of past fire suppression and fuels build-up in low-elevation dry forests adapted to predominantly frequent, low-severity surface fires (8, 11, 22, 25, 78, 79). In these forests, fuels treatments that remove midstory and understory fuels through thinning and prescribed fire can reduce fire intensity, severity, and rate of spread and may promote adaptive resilience to more frequent fire. Such forests were preferentially treated under the National Fire Plan in 2004–2008 (80). Thinning may effectively restore more frequent, low-severity fire in some dry forests, but when thinning is combined with the expected warming, unintended consequences may ensue, whereby regeneration is compromised and forested areas convert to nonforest (56, 57, 81). Strategic placement of treatments to promote low-severity fire at ecotones between dry and mesic forest distributions may help facilitate postfire migration of species better adapted to warmer, drier conditions.

Midelevation mixed conifer forests, or mesic forests, which typically experienced broad variance in fire frequency and severity, may also benefit from fuels treatments that reduce the likelihood of large patches of high-severity fire and facilitate the migration of species adapted to drier, warmer conditions (77). In contrast, cold/wet forests, such as high-elevation subalpine forests, are adapted to high-severity fire that historically recurred at relatively long (∼100–300 y) intervals (19, 82, 83) and have not experienced unprecedented fuels build-up in recent decades. Severe wildfires have occurred for millennia across a broad range of forests and shrublands, and in many ecosystems species are adapted to severe fire (17, 19, 84, 85), although postfire regeneration may be comprised by drier, warmer conditions (86).

Fuel-reduction treatments also hold promise for locally reducing wildfire hazard around WUI communities if treatments are strategically located to protect homes and the surrounding vegetation. Fuel reduction on federal lands and in municipal watersheds is a primary management tool that has limited application in the WUI, where the majority of land is privately owned (87). Home loss to wildfire is a local event, dependent on structural fuels (e.g., building material) and nearby vegetative fuels (88, 89). Therefore, fuels management for home and community protection will be most effective closest to homes, which usually are on private land in the WUI where ignition probabilities are likely to be high (37). Programs that facilitate the targeted removal of fuels from private land, such as community chipping programs, have been highly successful in some areas, at relatively low cost. The Wyden and Good Neighbor authorities and federal programs, such as the US Forest Service Collaborative Forest Landscape Restoration Program, take an "all-lands" approach to forest management through collaboration with landowners and communities. These policies and programs are roadmaps for augmenting fuel-management efforts across land ownerships. These and other more ambitious policies that facilitate significant fuels management on private land, on a par with fuel-reduction efforts on federal lands, are needed. New policies that facilitate private-land fuels management are critical to augment significantly the adaptive resilience of communities to increasing wildfire.

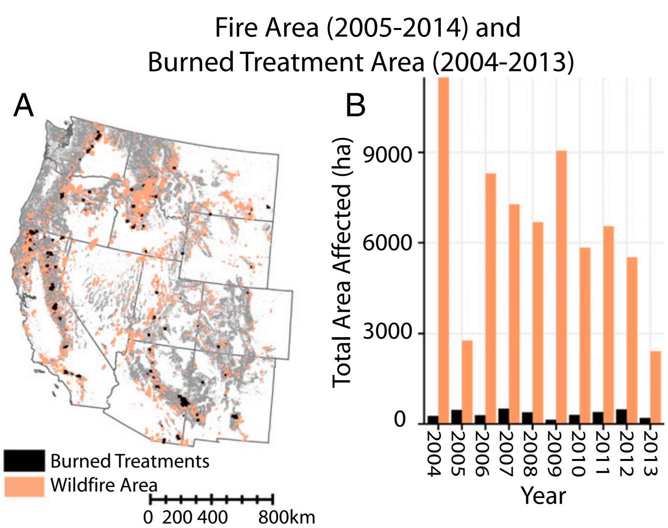

Fig. 6. (*A*) Spatial distribution and area of US Forest Service fuels treatments from 2004–2013 and wildfire from 2005–2014 across forests and woodlands in the western United States. About 3% of the total treated area and 10% of the total number of treatments burned in the period 2005–2014. (*B*) Annual total wildfire area and total burned treatment area. Data are from the following: (1) US Forest Service fuels treatments: Hazardous Fuel Treatment Reduction Polygon (https://data.fs.usda.gov/geodata/edw/datasets.php), (2) Wildfires >1000 ac: Monitoring Trends in Burn Severity Burned Areas Boundaries (www.mtbs.gov/dataaccess.html), (3) Wildfires ≤1000 ac: GeoMAC Historic Fire Perimeters (https://rmgsc.cr.usgs.gov/outgoing/GeoMAC/historic_fire_data/).

Promoting Adaptive Capacity

Fostering and Embracing Adaptive Shifts in Ecosystems. Management of fire and fuels will help some ecosystems withstand more frequent fires and possibly may reduce the risk of larger, more severe fires that may compromise forest recovery. Such efforts will be significant in high-value ecosystems or locations, in helping slow the pace of change and providing a chance for ecosystems and species to adapt to changing fire regimes. The HRV concept can guide management in identifying ecological vulnerabilities and adaptation strategies to changing disturbance regimes (Fig. 3) (50, 51, 52). However, quantifying ecological objectives outside the HRV will be increasingly important in guiding management as fire regimes and climate continue to change (90, 91). Given such uncertainties, management must be adaptive and iterative, and monitoring will be critical to assessing progress. Given the vast area of fire-prone forests in the West, management can directly affect only a small portion of forests. In the majority of forested ecosystems beyond our effective reach, we will have to accept and even embrace changing ecological conditions. While some forests may be entering decades of significant change with high tree mortality in response to drought, wildfire, insect outbreaks, and legacies of past management (86, 92), they also are in the process of adjusting to new conditions to which they will be better adapted and that may challenge our existing philosophies of and approaches to conservation.

Creating Fire-Adapted Communities. The majority of home building on fire-prone lands occurs in large part because incentives are misaligned, where risks are taken by homeowners and communities but others bear much of the cost if things go wrong. Therefore, getting incentives right is essential, with negative financial consequences for land-management decisions that increase risk and positive financial rewards for decisions that reduce risk. For example, shifting more of the wildfire protection cost and responsibility from federal to state, local, and private jurisdictions would better align wildfire risk with responsibility and provide meaningful incentives to reduce fire hazards and vulnerability before wildfires occur. Currently, much of the responsibility and financial burden for community protection from wildfire falls on public land-management agencies. This arrangement developed at a time when few residential communities were embedded in fire-prone areas. Land-management agencies cannot continue to protect vulnerable residential communities in a densifying and expanding WUI that faces more wildfire (12). The US Government Accountability Office questioned the US Forest Service's prioritizing protection of WUI communities that lie under private and state jurisdictions and has argued for increased financial responsibility for WUI wildfire risk by state and local governments (93). This shift in obligation would enhance adaptive governance and could increase the motivation to pursue adaptive resilience of WUI communities to increasing wildfire (94).

Another promising approach for increasing adaptive resilience of WUI residents to wildfire is the promotion of fire-adapted planning in communities. Providing incentives for counties, communities, and homeowners to plan fire-safe residential development for both existing and new homes and discouraging new development on fire-prone lands will make communities safer (89, 94–96). Communities can use land-use and development codes that encourage developers to set aside open space and recreational trails as fuel breaks and require ignition-resistant construction materials in fire-prone settings. For example, San Diego, California enforces strict brush management regulations; the Flagstaff, Arizona fire department uses a WUI development code to protect properties; and Santa Fe, New Mexico applies stringent fire-safe regulations on new developments to protect its watershed (97). Programs such as the Community Planning Assistance for Wildfire (CPAW; planningforwildfire.org), funded by the US Forest Service and private foundations, offer assistance to communities in the form of advice on land-use planning and detailed mapping of wildfire risk. Another example is California, which employs a statewide Fire Hazard Severity Zone map to guide development plans and building codes that reduce wildfire risk. With 84% of potential WUI lands in the West still undeveloped (98), land-use planning now has high potential to reduce the vulnerability of communities to future wildfire. Furthermore, fire-adapted planning may increase management options in terms of how, where, and when fire can be used as a tool for reducing the spread of wildfires into communities and rejuvenating fire-dependent ecosystems, thus increasing the adaptive resilience of communities and ecosystems to more wildfire.

Strengthening and expanding programs such as Fire Adapted Communities, Fire Adapted Communities Learning Network, Firewise Communities USA, and FireSmart Canada will also help communities become more fire-adapted. Capacities to assume these responsibilities will vary significantly among homeowners, communities, and local jurisdictions with markedly different risks and resources (99–101). For example, home hazard mitigation programs and community planning tools are more successful in communities at the fringe of urban areas that have more financial resources and often have a greater trust in government than in more isolated, resource-dependent WUI communities, immigrant non–English-speaking communities, or tribal and First Nations communities (101). Although some tax incentives and rebates are available for wildfire risk mitigation on and around homes, more comprehensive programs that include broader incentives and support are needed for meaningful and widespread impacts. Efforts that combine wildfire-specific efforts with other community capacity-building efforts may leverage the networks that enable communities to act on shared notions of risk (102).

Overall, a shift in resources from the defense of the WUI from wildfire to the mitigation of wildfire hazards and risks in advance of events will build a safe operating space for fire-prone communities that increases adaptive resilience to wildfire. Encouraging development away from fire-prone areas, reducing fuels on private lands in and near communities, and retrofitting and building homes to withstand ignition will increase the adaptive capacity for managing more wildfire (89), similar to adaptive approaches for other natural hazards such as flooding and earthquakes (12). Communities and institutions are long-lived, and disruptive events such as wildfires create windows of opportunity that can shift rules, norms, and expectations to increase adaptive resilience to future wildfires.

Conclusions

Policies that foster adaptive resilience enable WUI communities and fire-prone ecosystems to adjust to increased wildfire risk and reduce future vulnerability. Adaptive resilience provides a realistic framework as the climate warms and wildfires increase, but how will we know if we are achieving adaptive resilience to future fires? On the societal front, minimizing the costs of suppression in the WUI, the number of homes lost to wildfire, the area burned in the WUI, and the number of smoke-related health problems are some metrics. Developing state- or county-wide maps of fire hazard, home survivability rating, and the adaptive capacity of communities would be useful tools in developing this framework.

Some ecosystems will survive and thrive as they adapt to novel future conditions, but not all will. Embracing rather than resisting ecological change will require a significant paradigm shift by individuals, communities, and institutions and will challenge our conservation philosophies. Wildfire is an important catalyst of responses to climate change by communities and ecosystems. Patterns of wildfire are changing with rising global temperatures, and will accelerate in the future. What we can do now is focus management efforts on the places where intervention is needed to slow the pace of change and thereby give particular species and ecosystems a chance to adapt. We also can change how we build, live, and work in fire-prone landscapes to keep our communities safe, healthy, and vibrant.

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Invited Paper

Wildlife Conservation Planning Under the United States Forest Service's 2012 Planning Rule

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ABSTRACT In 2012, the United States Forest Service (USFS) promulgated new planning regulations in accordance with the National Forest Management Act (NFMA). These regulations represent the most significant change in federal forest policy in decades and have sweeping implications for wildlife populations. We provide a brief overview of the history of the NFMA planning regulations and their wildlife provisions and review the current science on planning for effective wildlife conservation at the landscape scale. We then discuss the approach to wildlife conservation planning in the 2012 rule and compare it to alternatives that were not selected and previous iterations of the planning rule. The new planning rule is of concern because of its highly discretionary nature and the inconsistency between its intent on the one hand and operational requirements on the other. Therefore, we recommend that the USFS include in the Directives for implementing the rule commitments to directly monitor populations of selected species of conservation concern and focal species and to maintain the viability of both categories of species. Additional guidance must be included to ensure the effective selection of species of conservation concern and focal species, and these categories should overlap when possible. If the USFS determines that the planning unit is not inherently capable of maintaining viable populations of a species, this finding should be made available for scientific review and public comment, and in such cases the USFS should commit to doing nothing that would further impair the viability of such species. In cases where extrinsic factors decrease the viability of species, the USFS has an increased, not lessened, responsibility to protect those species. Monitoring plans must include trigger points that will initiate a review of management actions, and plans must include provisions to ensure monitoring takes place as planned. If wildlife provisions in forest plans are implemented so that they are enforceable and ensure consistency between intent and operational requirements, this will help to prevent the need for additional listings under the Endangered Species Act and facilitate delisting. Although the discretionary nature of the wildlife provisions in the planning rule gives cause for concern, forward-thinking USFS officials have the opportunity under the 2012 rule to create a robust and effective framework for wildlife conservation planning.

KEY WORDS at-risk species, coarse-filter, fine-filter, focal species, forest planning, monitoring, viability.

In April 2012, the United States Forest Service (USFS) issued its final planning rule in accordance with requirements of the National Forest Management Act of 1976 (NFMA; 77 FR 21162). The 2012 rule took over 2 years to complete and included extensive public involvement, consultation through forums with scientists and policy experts, and environmental analysis conducted in accordance with the National Environmental Policy Act of 1969 (NEPA; USFS 2012). The new rule represents the most substantive change in federal forest policy in 30 years, with sweeping implications for wildlife. We review the administrative history of the planning rule, explore the provisions that affect the conservation of wildlife and biodiversity, and discuss how careful implementation could lead to more efficient and effective wildlife management. To provide a context for interpreting the changes that will come with implementation of the new rule, we begin with a short administrative history, and then provide a conceptual framework for interpreting the management implications of the rule. We also consider the intersection of the NFMA and the Endangered Species Act (ESA) and look at the implications of this rule change for ESA decision making. We conclude with a series of observations and recommendations for how the wildlife profession might help ensure that sound science and practical policy are effectively wed as the planning rule is implemented across the nation's public forest lands over the years to come.

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A BRIEF HISTORY OF THE 2012 PLANNING RULE

The NFMA created a 3-tiered, regulatory approach to planning. At the highest tier, national-level regulations govern the development and revision of second-tier forest plans. Site-specific plans for projects and other activities make up the third tier, and they must be consistent with both sets of higher-level regulations. Forest plans typically make zoning and suitability decisions and regulate various activities within a forest area, therefore acting as a gateway through which subsequent project-level proposals must pass. They do not, however, authorize or mandate site-specific projects. Instead, plans address issues such as the prioritization of various multiple-use goals, requirements for managing resources such as wildlife, watersheds, or soils, and the determination of which land is suitable for timber cutting, along with allowable volume and the choice of harvesting and regeneration methods.

Efforts to revise the rules governing Forest Service planning have been many, and debate has been intense, resulting in considerable confusion regarding the requirements, process, and legal provisions underlying recent forest planning and management. During development of the 2012 rule, the USFS operated under the 1982 planning rule (47 FR 43026), despite the issuance of more recent rules in 2000 (65 FR 67514), 2005 (70 FR 1023), and 2008 (73 FR 21468). The 2000 rule, developed by the Clinton administration with guidance from a Committee of Scientists (Committee of Scientists 1999), was deemed by the subsequent administration too "costly, complex, and procedurally burdensome" (77 FR 21162: 21164) to implement, and the USFS reverted to planning under the terms of the 1982 rule. Both the 2005 and the 2008 rules were enjoined by the courts because of a failure to meet legal requirements. The agency had argued that planning regulations did not have environmental impacts and thus did not require analysis under the NEPA and the ESA, but this argument failed to survive judicial review (*Citizens for Better Forestry v. USDA* 2007, 2009). A desire to address these persistent weaknesses and to avoid a similar judicial outcome is evident in the language of and justification for the 2012 rule.

One of the most controversial and highly litigated aspects of previous USFS planning rules has been the regulations written in accordance with the NFMA's requirement to "provide for a diversity of plant and animal communities based on the suitability and capability of the specific land area in order to meet overall multiple-use objectives" (16 USC 1604[g][3][B]). To interpret the diversity provision and other requirements of the NFMA, a Committee of Scientists was convened in 1977, in accordance with requirements of the NFMA, to assist with the development of the first planning rule (issued in 1979 and revised in 1982). The diversity regulations in the 1982 rule required that "fish and wildlife habitat shall be managed to maintain viable populations of existing native and desired non-native vertebrate species in the planning area" (36 CFR 219.19). The reference to "viable populations," drawn directly from fundamental principles of population biology, embedded specific, scientific intent into the Forest Service's planning and management responsibilities.

Subsequently, this provision caused significant controversy and drove change in forest management (Corbin 1999, Duncan and Thompson 2006). For example, compliance with the viability provision initiated litigation over the northern spotted owl (*Strix occidentalis caurina*), and led the courts to reject forest plans in the Pacific Northwest for failure to protect the viability, not only of the owl, but also of other species associated with late-successional forests (Duncan and Thompson 2006). Implementation of the 1982 rule relied primarily on the selection of management indicator species, like the northern spotted owl, meant to serve as surrogates to indicate management impacts on a broader suite of unmeasured species. Most forests indirectly assessed the status and trends of management indicator species by measuring habitat amount, a controversial practice that has been the subject of numerous court cases (Corbin 1999). Nonetheless, the use of habitat as a proxy for population status was established in court as not necessarily prohibited by the 1982 regulations (*Inland Empire Public Lands Council v. USFS* 1996).

In the 1990s, the USFS made several attempts to revise the planning rule, and in 1997 a second Committee of Scientists was convened. Its recommendations served as the basis for the 2000 rule, which maintained the viability requirement and extended it to all plant and animal species. The Committee of Scientists suggested a combination of coarse-filter approaches, which focus on the maintenance of ecosystems defined in terms of dominant vegetation types and their successional stages (see Hunter 1990), and fine-filter approaches, which involve direct species-specific measurements of population status and trends (Haufler et al. 1996, Committee of Scientists 1999). Specifically, the 2000 rule required that focal (see below) and at-risk species be monitored using fine-filter approaches. Diversity provisions of the 2000 rule were never implemented, because in 2001 the USFS reverted to the 1982 rule, using a transitional provision in the 2000 rule, while the Bush administration initiated revisions to the planning rule. Both the 2005 and 2008 rules relied entirely upon a coarse-filter approach to wildlife conservation. Contrary to assertions from the scientific community (Noon et al. 2003, 2005), the USFS argued that maintenance of broad ecosystem diversity (as represented by coarse-filter approaches) would adequately protect species and address their diversity obligations under the NFMA. These rules did not require any fine-filter, species-specific planning or monitoring. When the 2005 and 2008 rules were enjoined, the court gave the USFS the option of using the 2000 or the 1982 rule. The USFS chose to use the provisions of the 1982 rule, including the viability provision, through the transitional language in the 2000 rule. In its justification of the most recent planning effort, the USFS claims that the 1982 rule is out-of-date in its scientific foundations, planning procedures, and social values, and is too complex, expensive, and procedurally burdensome to implement (77 FR 21162).

CONCEPTUAL BASIS FOR WILDLIFE CONSERVATION PLANNING

In addressing asserted shortcomings of the 1982 rule, the Forest Service adopts an approach to wildlife conservation that hinges primarily on the assessment, analysis, management, and monitoring of habitat. The 2012 Programmatic Environmental Impact Statement for the planning rule states, "The best opportunity for maintaining species and ecological integrity is to maintain or restore the composition, structure, ecological functions, and habitat connectivity characteristics of the ecosystem. These ecosystem components, in essence, define the coarse-filter approach to conserving biological diversity" (USFS 2012:126). This contrasts with the 1982 and 2000 rules that emphasized population viability.

A Combined Coarse-Filter/Fine-Filter Approach

Most wildlife ecologists believe that effective biodiversity conservation planning requires an appropriate balance between habitat-based, coarse-filter approaches and insights from fine-filter, species-level assessment and monitoring (Noon et al. 2009). The 2012 Programmatic Environmental Impact Statement for the planning rule recognizes the limits of the coarse-filter approach stating, "initially at least, some amount of direct species measurement may be needed to assess the effectiveness of the ecological conditions provided under the coarse-filter approach in achieving the goal of conserving the biological diversity of the area" (USFS 2012:124). The impact statement goes on to propose that fine-filter strategies "can be focused on the few species of special concern whose habitat requirements are not fully captured by coarse-filter attributes." However, this language understates the importance of a complementary fine-filter approach. Research indicates that the coarse-filter approach is unlikely to provide a reliable basis for multi-species conservation planning (Cushman et al. 2008), only limited testing of the approach's validity has occurred (Noon et al. 2009), and the monitoring of a select group of species using a fine-filter approach is necessary to determine the efficacy of coarse-filter approaches (Committee of Scientists 1999, Flather et al. 2009). A recent review of the degree to which coarse-filter models can be used to infer animal occurrence concluded that ". . . the observed error rates were high enough to call into question any management decisions based on these models" (Schlossberg and King 2009:609). These authors went on to state, ". . . [coarse-filter] models oversimplify how animals use habitats, and the dynamic nature of animal populations" (Schlossberg and King 2009:609).

Defaulting to vegetation type as a descriptor of a species' habitat has a long history in ecology. It has been driven largely by pragmatism—vegetation is much easier to measure and characterize than prey resources or nest sites, for example. The practice continues because detailed vegetation maps exist for most of the United States based on either extensive ground-surveys or remotely sensed imagery (e.g., USFS LandFire Program). However, vegetation is often a poor proxy for more influential, but difficult to measure resources, and the frequent failure of vegetation-based habitat models to predict a species' distribution and abundance may be because of limitations of this assumed relationship (Van Horne 2002, Cushman et al. 2008). Even with more detailed data on habitat characteristics, unmeasured and unknown factors will still affect populations. For these reasons, population status of focal and at-risk species must be directly assessed. Therefore, a coarse-filter approach based primarily on dominant vegetation communities will have limited ability to predict the distribution and abundance of many wildlife species and effectively address the diversity provisions of the NFMA; this requires both coarse- and fine-filter approaches.

Selecting Species for Fine-Filter Assessment

Striking a balance between coarse-and fine-filter assessments of biological diversity has challenged forest managers for decades. Even if the fine-filter approach was restricted to vertebrates, monitoring the status of all species is not feasible, thus previous planning rules have restricted USFS requirements to an assessment of a small subset of species occurring across the planning area. This pragmatic constraint was recognized in the 1982 planning rule with the designation of management indicator species, species assumed to reflect the effects of management on their populations as well as the populations of many unmeasured species. However, the notion that a single species can serve as an indicator for a suite of species is an untested premise and generally not supported by research studies or ecological theory (Noon et al. 2009, Cushman et al. 2010). The concept that some species act as direct surrogates of others is untenable unless those species share similar population drivers (Cushman et al. 2010).

Instead of management indicator species, the second Committee of Scientists recommended the use of "focal species" (Committee of Scientists 1999) to evaluate status and trends of plant and animal diversity, generally. The Committee of Scientists proposed that focal species would commonly be selected on the basis of their functional role in ecosystems (e.g., they serve keystone functions [Mills et al. 1993], they are indicators of exposure to key stressors [Caro and O'Doherty 1999], they have a role as engineers of ecological processes [Jones et al. 1994], or play an important role in food web dynamics [Soule et al. 2005]). For federal public lands, Noon et al. (2009) suggest a combined coarse-filter and fine-filter approach, with the latter focusing on monitoring threatened, at-risk, and rare species, along with a modest number of focal species selected with complementary and comprehensive functional roles as described above. Systematic approaches exist for identifying and prioritizing an informative subset of species for fine-filter assessment and monitoring. For example, Regan et al. (2008) suggest selecting species based on existing schemes, such as The World Conservation Union (IUCN) Red List, Nature Serve, Partners in Flight databases, and federal or state listings, combined with an assessment of the degree and spatial and temporal characteristics of known threats. Nevertheless, uncertainties regarding the ability to generalize inferences drawn from any subset of species make the selection process of fundamental importance to the successful implementation of the fine-filter approach.

Improved Techniques for Fine-Filter Monitoring

One argument against direct assessment of wildlife populations is that it is not financially feasible. Traditional monitoring programs and viability analyses have been based on estimates of demographic parameters such as abundance, density, survival, and reproductive rates (Beissinger and McCullough 2002). Estimates of these parameters are expensive, require extensive field surveys, often involve capture and marking of individual animals, and are available for only a small number of species. However, indirect estimates of a species' status and trend based on their spatial distribution can provide defensible surrogate measures (MacKenzie and Nichols 2004, Manley et al. 2004). Focusing on distribution, rather than traditional measures of population size and growth rate, greatly increases the efficiency of broad-scale monitoring programs (Noon et al. 2012). Advancements in wildlife monitoring, based on detection/non-detection data, including the use of sign surveys, genetic evaluation, and historical presence–absence survey data decrease the cost of monitoring changes in distribution, which can be inferred from the proportion of sample units at which the species is detected (MacKenzie et al. 2006). One of the most significant advances in detection/non-detection monitoring is the ability to confirm the presence of a species at a survey site based on its genetic signature (e.g., in hair or scat; Waits 2004, Schwartz et al. 2006). The July 2005 issue of the *Journal of Wildlife Management* devoted a special section to the application of presence–absence sampling in wildlife monitoring (Vojta 2005), including an application to National Forest System lands (Manley et al. 2005). One variable estimated by these models is the area occupied by a species, a measure of a species' spatial distribution. Temporal and spatial patterns in detection/non-detection monitoring data allow inference to changes in animal abundance (MacKenzie and Nichols 2004), the single most influential parameter that provides insights into likelihood of species persistence (Lande 1993). Thus, previous arguments citing the practical limitations of the fine-filter approach are blunted by recent technical and statistical research, much of it inspired by the difficulty and expense of implementing earlier approaches to fine-filter assessments under the 1982 planning rule.

Political and Administrative Barriers to Effective Biodiversity Conservation Planning

In the past, very few if any management indicator species have been monitored in a manner that would allow a reliable assessment of their response to management (Noon et al. 2009). Managers cite the lack of monitoring data as a critical limitation in understanding cumulative impacts to species (Schultz 2012). Aside from cost and the technical challenges discussed above, funding and implementation of reliable, species-specific monitoring has been a significant challenge on National Forest System lands because of political reasons. Maintaining the political and fiscal will to support long-term monitoring programs is difficult (Doremus 2008, Biber 2011). In addition to the challenges of chronic under-funding, management agencies face disincentives to implementing robust species-level monitoring plans because monitoring data may reveal the negative impacts of management. For example, documenting the impacts of timber harvest or fuels reduction activities on sensitive wildlife species often highlights conflicts between different agency mandates, each of which enjoys strong political and social support. In addition, funds allocated to monitoring may draw funds away from projects that result in immediate job creation, the provision of marketable goods such as timber, the attainment of fuels reduction and restoration goals, or other accomplishments that can be reported to Congress in a timely manner. Furthermore, an agency could face legal challenges if it makes enforceable monitoring commitments that it does not have the funding to implement. However, at least as they are typically drafted, monitoring plans are difficult to enforce in court, obviating the need to fully implement intended programs. The judiciary usually finds commitments to monitor land-use plans not subject to review under the parameters of administrative law, and even when reviewed in court, determinations regarding the adequacy of monitoring data are traditionally left to the expertise of administrative agencies (Biber 2011).

Several other issues make understanding management effects on wildlife populations problematic. For example, the USFS has often monitored impacts to species at the project level (Schultz 2010), a spatial scale with generally small population-level effects. Small effect sizes require high statistical power for their detection. The disparity between the scale at which population responses can be detected and the scale of individual management actions leads to persistent problems in assessing impacts to species viability (Ruggiero et al. 1994). Monitoring impacts to habitat must be done cumulatively and at multiple spatial scales to assess whether small-scale habitat changes affect individual organisms, interrupt landscape connectivity affecting multiple populations, or synergistically interact with other small-scale disturbances, resulting in broad-scale effects.

Finally, the integrity of any monitoring plan, coarse- or fine-filter, depends on the articulation of clearly stated objectives and triggers to management actions. A trigger point is a threshold value for a monitoring state variable (e.g., percent area occupied by a given focal species within a national forest planning area) that, when exceeded, triggers a particular management response. A monitoring program without triggers selected a priori to call attention to trends provides little more than a retrospective time series of data with no feedback—and therefore little value—to the management decision-making process (Noon 2003). Furthermore, the efficacy of a monitoring program cannot be assessed at adoption without pre-defined trigger points. Trigger points can be most objectively set up-front, before the difficult management changes that might result from crossing such points are proximate. This is especially true if effects are analyzed exclusively at project scales, masking broader trends. In such cases, declines in population size or habitat quality, for example, may occur incrementally with no recognition of impact until a decline in species status is clearly established via listing under the ESA (Schultz 2010). To provide value to the forest planning process, a monitoring program must establish, a priori, the magnitude of change in the monitoring state variable that would trigger a review of management practices.

In summary, a comprehensive wildlife assessment framework would include a combination of both coarse- and fine-filter approaches. It would commit to monitoring at-risk and focal species using recent advances in monitoring approaches that make species-specific monitoring more financially feasible and efficient than it has been in the past (Noon et al. 2012). As required for effective and meaningful adaptive management, monitoring would occur at multiple spatial scales and use pre-defined triggers to meaningfully evaluate the consequences of management actions and to inform future management decisions.

AN OVERVIEW OF THE 2012 PLANNING RULE'S DIVERSITY PROVISIONS

The planning framework for the 2012 final rule involves a 3-step process: assessment; plan development, amendment, and revision; and monitoring (36 CFR §219.5 [2012]). It requires the use of the "best available scientific information to inform the planning process" (36 CFR §219.3 [2012]) and identifies restoration and watershed protection as agency priorities, while emphasizing the contributions of sound forest management to ecological, social, and economic sustainability (36 CFR §219.8 [2012]). Because restoration requires: 1) an assessment of the current system state relative to desired future conditions; 2) measurement of the system state subsequent to management activities; and 3) a comparison of the observed to desired state, restoration is critically dependent on monitoring. In this section, we discuss the approach in the 2012 rule and the alternatives that were considered but not selected in the agency's decision process.

Assessment and Planning

Section 219.9 outlines the approach for providing for diversity of plant and animal communities. It explains that the USFS is adopting "a complimentary ecosystem and species-specific approach," or a combined coarse- and fine-filter approach. Paragraph (a) outlines the coarse-filter requirements to maintain ecosystem integrity and diversity: plans "must include plan components . . . to maintain or restore the ecological integrity of terrestrial and aquatic ecosystems and watersheds in the plan area" and "maintain or restore the diversity of ecosystems or habitat types throughout the plan area" (ecological integrity and diversity are defined in §219.19 of the 2012 rule). Plan components must function to maintain or restore ecosystem structure, function, composition, connectivity, key ecosystem characteristics, rare species communities, and native tree diversity. A commitment to restore or maintain landscape connectivity to facilitate movement, migration, and dispersal is a significant addition to the planning rule. Paragraph (b) outlines the fine-filter approach. It begins by explaining that the responsible official must determine whether the plan components under part (a), the coarse-filter requirements, will provide the necessary conditions to contribute to the recovery of species listed as threatened or endangered under the ESA, or species that are proposed or candidate species for listing. Additionally, the responsible official must determine whether the coarse-filter approach is sufficient for maintaining viable populations of "species of conservation concern." These are species known to occur in the plan area, other than those listed, proposed, or identified as candidate species under the ESA, that are selected by the Regional Forester based on "substantial concern about the species' capability to persist over the long-term in the plan area" (36 CFR §219.9[c] [2012]). If the coarse-filter is deemed to be insufficient, the responsible official must include species-specific plan components (e.g., buffer areas around nest sites), that will contribute to the recovery of populations of species of conservation concern, as well as federally listed, proposed, and candidate species. If the coarse-filter is assumed adequate, no further species-level consideration is employed in planning. Yet how responsible officials will be held accountable for such decisions is unclear. The burden of proof for determining the effectiveness of the coarse-filter approach is not addressed. These species-specific requirements represent the USFS commitment to the fine-filter approach in section 219.9.

Notably, the new rule eliminates the requirement for maintaining viable wildlife populations, in contrast to the 1982 rule's viability provision for vertebrates and the provisions of the 2000 rule that would have extended the requirement to other species. Since the agency only commits to maintaining the viability of species of conservation concern, under the 2012 rule the USFS has no obligation to address the decline of any species not listed, proposed, or a candidate under the ESA, unless the responsible official, in this case the Regional Forester, expresses substantial concern about its persistence. Thus, any number of species could pass from secure to endangered status before any federal intervention would be required. However, in contrast to the 1982 rule, the agency can commit to maintaining viable populations of non-vertebrates by identifying them as species of conservation concern.

Historically, the diversity provisions of the NFMA have been one of the most controversial aspects of the planning rule, and the issue of how the USFS should address the clearly established public values associated with wildlife conservation often has been overshadowed by legal and technical arguments about the practicality of specific approaches to viability assessment. For example, over the course of the drafting and judicial review of multiple rules, considerable disagreement existed as to whether a requirement to maintain viable populations of all species, or just vertebrate species, or just at-risk species was an attainable goal. Understandably, the USFS has been reluctant to commit the agency to a species viability standard with which demonstrating compliance is difficult. At any point in time, all species have some non-zero probability of extinction; thus, viability can never be guaranteed. Viability is a probabilistic concept that invokes a specific level of risk over a stated time horizon, and proponents of the viability standard have had difficulty explaining to the public—and sometimes to their colleagues in wildlife management—how probabilistic events can be addressed in legally enforceable standards.

Nonetheless, in its 2012 record of decision, the agency commits to maintaining the viability of species of conservation concern, arguing that the combination of coarse- and fine-filter approaches it proposes are scientifically defensible, will adequately protect biodiversity on its lands, and will not be too costly to implement (77 FR 21162). However, the planning rule does not specify how viability will be assessed or what information will be used to assess a species' viability. Additionally, species identified as being of conservation concern could experience sharp range restrictions, since the regulations no longer require viable populations to be well-distributed, as was the case under the 1982 rule. Instead, the new rule defines of a viable population as one that "continues to persist over the long term with sufficient distribution to be resilient and adaptable to stressors and likely future events" (36 CFR §219.19 [2012]).

Finally, the USFS may absolve itself of responsibility for species-level conservation if the agency determines that maintaining a viable population of a species of conservation concern is beyond the capability of the plan area. In this case, which might result from stressors extrinsic to the planning area, such as climate change or the loss of habitat in other regions, the responsible official is required to document the basis for that decision and include plan components that contribute to the maintenance of a viable population across multiple land ownerships, in coordination with other managers and private parties working across jurisdictional boundaries, to the extent practicable.

Monitoring

Monitoring requirements are outlined in section 219.12. The planning rule requires a monitoring program for each National Forest, which can be developed jointly across forests and must be developed in coordination with the Regional Forester and the Research and State & Private branches of the agency. Plan monitoring programs must include questions and indicators; for diversity, these include indicators addressing the status of ecological conditions and the status of focal species, defined in the rule as "a small subset of species whose status permits inference to the integrity of the larger ecological system to which it belongs and provides meaningful information regarding the effectiveness of the plan in maintaining or restoring the ecological conditions to maintain the diversity of plant and animal communities in the plan area. Focal species would be commonly selected on the basis of their functional role in ecosystems" (36 CFR § 219.19 [2012]). Regional Foresters are to develop "broader-scale monitoring" for questions that are relevant at scales larger than the planning area. In all cases, monitoring information is to be compiled, evaluated, made available to the public, and used to inform adaptive management of the plan area. Thus, the new rule adopts, for the first time, a multi-scaled approach for monitoring and codifies the intent, although not the process, for implementing a transparent and data-driven approach to adaptive management. Although the adoption of a focal species approach based on functional roles in sustaining ecosystem processes reflects the logic of the 2000 rule, the 2012 rule draws no connection between the monitoring of focal species and the conservation of their roles in the ecosystem. The new rule does not include a requirement to maintain the viability of focal species, despite the fact that it is the status of these species that is meant to indicate whether the USFS is successfully maintaining and restoring ecosystem diversity and integrity. Additionally, the 2012 rule does not provide a requirement to monitor species of conservation concern, despite their established vulnerability to local extirpation. Consequently, the fine-filter approach to monitoring is explicitly separated from the fine-filter approach for biodiversity conservation.

Alternatives Not Selected

Although a review of the key provisions of the planning rule provides direct insight into the place of wildlife conservation in the future of forest planning and management, examination of the alternatives not selected reveals the underlying logic, pivotal choices, and philosophical foundations of the Forest Service's interpretation of the NFMA and reconceptualization of its institutional role and responsibilities to the public. The USFS considered several other alternatives in its Programmatic Environmental Impact Statement, in addition to the selected alternative (i.e., the final rule), which was a modified version of Alternative A. Alternative B closely followed the 1982 rule, notably in regards to the viability provision (". . . fish and wildlife habitat shall be managed to maintain viable populations of existing native and desired non-native vertebrate species in the planning area . . ." [36 CFR 219.19]). The agency provides a lengthy rationale for not selecting Alternative B, focusing on the defects of the 1982 viability provision (see 77 FR 21162:21168). This rationale also pertains to the selection of the final rule (modified Alternative A), which dropped the 1982 viability provision with the exception of "species of conservation concern" (see below). The agency states the 1982 rule "included planning procedures that do not reflect current science or result in unrealistic or unattainable expectations because of circumstances outside of the Agency's control, particularly for maintaining the diversity of plant and animal species" (77 FR 21162:21169). The USFS further justifies dropping the requirement to maintain species viability by stating, "[T]here are limitations on the Agency's authority and the inherent capability of the land" (77 FR 21162:21169). It notes that forest clearing in South America and habitat fragmentation in the Rocky Mountains on private land affect the agency's ability to maintain viable populations on National Forest System lands. For reasons such as these, the agency notes, the USFS cannot ensure a species' existence in the planning area when circumstances outside of its control may be contributing to population declines. It also notes that managing for the habitat of a single species sometimes impinges on management requirements for a species listed under the

ESA, or on other necessary activities the agency must undertake to comply with statutory requirements. Furthermore, the agency writes, some forests simply cannot support viable populations of species that are rare and far-ranging, like wolverines (*Gulo gulo*), and require more habitat than is available on a single National Forest unit.

Alternative C included no specific provisions for biodiversity conservation beyond the minimum requirements of the NFMA. This alternative was highly discretionary, leaving decisions about the requirements for assessment, planning, and monitoring to the USFS Directives' System (i.e., the agency's handbook and manual), whose provisions are not legally binding. The high degree of discretion in this alternative, according to the agency, would have resulted in too much variation in implementation: "There would be no certainty with regard to the inclusion of any plan components beyond the minimum required by this Alternative, and a potential lack of consistency across the National Forest System" (77 FR 21162:21170).

Alternative D "was designed to evaluate additional protections for watersheds and an alternative approach to addressing the diversity of plant and animal communities" (77 FR 21162:21170). This alternative required watershed-scale assessments of climate change vulnerability and designation of key watersheds to anchor the assessment and maintenance of the ecological status of aquatic, riparian, and terrestrial components of watersheds (USFS 2012). Establishing connectivity between habitats and discrete populations of species would also have been required. The alternative maintained and extended the 1982 viability requirement, stating the National Forests would provide for viable populations of native and desired non-native species in each planning area. The USFS was required to use the best available science to determine ecological conditions necessary to support viable populations, as informed by the "current and likely future viability of focal species within the planning area" (USFS 2012:F-9). To address the agency's concern that it cannot ensure the viability of populations on its lands, Alternative D included language that required the Secretary of Agriculture to provide notice to the public and allow for public comment if the agency determined it could not provide for viable populations of native or desired non-native species in a plan area. Furthermore, the agency was required to provide for viability of such a population to the maximum extent practicable and to take no actions that would increase the likelihood of extirpation of a population in the planning area. As with the selected alternative, Alternative D required monitoring of the status and trends of focal species, but with the additional requirement that triggers be identified for focal species' monitoring that would initiate a review of planning and management decisions to achieve compliance with the viability standard. This alternative explicitly stated that population surveys of focal species would be conducted using presence–absence data, occupancy modeling, genetic monitoring, or count-based methods. Alternative D was not selected because of the high anticipated planning and monitoring costs (77 FR 21162). The record of decision states that many plans already incorporate elements of this alternative, but that it is too prescriptive to allow for efficient, effective, and flexible management of all National Forests (77 FR 21162).

Finally, Alternative E was highly prescriptive in terms of requirements for public notification, assessment, and monitoring. It would have required specific monitoring questions, indicators, and triggers for changes in management action. The diversity requirements would have been similar to those in the selected alternative, but with more emphasis on monitoring of species' status and trends. The alternative was rejected for the same reasons as Alternative D.

MANAGEMENT IMPLICATIONS AND RECOMMENDATIONS FOR IMPLEMENTING THE 2012 PLANNING RULE

In theory, the new planning rule could be implemented in a robust way, drawing on the best available science to protect plant and animal diversity on National Forest System lands. However, the primary change introduced by the 2012 rule is the considerable discretion afforded centralized authorities, particularly at the regional level, in how general provisions will be implemented. Based on the management history of the USFS, numerous aspects of the 2012 planning rule are of concern, primarily because they defer many fundamental details to the interpretation of officials who may lack scientific background and disciplinary depth in wildlife biology and may have disincentives to prioritize wildlife. A number of scientists and scientific societies (including The Wildlife Society) commented on the draft rule and noted that it leaves more decisions about diversity conservation to agency discretion than did the 1982 rule. Forest Service officials must strike a fine balance between prescriptive standards and discretion or flexibility in a rule that is meant to guide planning years into the future on the entire National Forest System. Although some discretion is necessary, a rule must be sufficiently prescriptive to ensure that the National Forests do not implement a loosely written and unenforceable standard with so much variability across management units as to compromise the conservation of biological diversity.

Discretion, Authority, and Responsibility in Wildlife Conservation

Highly discretionary mandates are especially problematic for protecting resources such as wildlife that, without clear substantive requirements, have historically received less attention in land management. The 1897 Organic Act gives the USFS wide discretion by providing an open-ended mandate to secure water flows and provide timber. The Multiple Use Sustained Yield Act (MUSYA), passed in 1960, expanded the factors that the USFS must consider in planning, including wildlife conservation. However, the language in the MUSYA does not require the USFS to conserve wildlife in any specific fashion, only to consider the wildlife resource when planning for multiple-use. The concept of multiple-use, according to the courts, "breathes discretion at every pore" (*Perkins v. Bergland* 1979). Wildlife never gained serious consideration in forest management under the MUSYA, in part because of the agency's deference to state wildlife agencies, which have generally focused on game species and sport fisheries.

We have consistently heard many USFS personnel argue that their primary responsibility is to manage the habitat on USFS lands, whereas actual populations are the domain of the states. However, the USFS clearly has the power to manage wildlife on its lands. The United States Constitution's Property Clause (Art IV, section 3) gives Congress proprietary and sovereign powers over its property, and it may delegate decisions regarding federal lands to executive agencies. The Supreme Court has repeatedly observed that this power over federal land is "without limitations" (*United States v. San Francisco* 1940). The Court's expansive reading of the Property Clause also extends to managing wildlife on federal lands. The dispositive case is *Kleppe v. New Mexico* (1976), where the Court states, "the 'complete power' that Congress has over public lands necessarily includes the power to regulate and protect the wildlife living there" (426 U.S. 529: 541). Of course, the states also manage wildlife on federal lands, but as made clear in *Kleppe*, "those powers exist only in so far as [their] exercise may be not incompatible with, or restrained by, the rights conveyed to the Federal government by the Constitution." (426 U.S. 529: 545). Though the USFS seldom chooses to assert its full wildlife management powers, the Courts continue to emphasize the Property Clause's application to wildlife (see, e.g., *Wyoming v. United States* 2002).

Concerns about wildlife were one of the central factors precipitating the passage of the NFMA in 1976, and the USFS has a clear responsibility under the Act to manage for biodiversity. The Act's legislative history shows that its diversity provision was meant to require "Forest Service planners to treat the wildlife resource as a controlling, coequal factor in forest management and, in particular, as a substantive limitation on timber production" (Wilkinson and Anderson 1987:296). When the NFMA was passed, it included language stating that the USFS has a responsibility to be "a leader in assuring that the Nation maintains a natural resource conservation posture that will meet the requirements of our people in perpetuity" (16 U.S.C. §1600[6]) and an explicit requirement to protect plant and animal diversity. To ensure that the agency's new requirements were effectively translated into administrative regulations, Congress required the agency to convene a Committee of Scientists to inform the writing of these regulations, which were finalized in 1982 (16 U.S.C. §1604[h][1]).

Timber harvest on the National Forests, nonetheless, continued to increase steadily, until the late 1980s. At that time, citizen enforcement, frequently manifest through appeals and litigation based on substantive provisions like the 1982 rule's viability standard and the ESA, was a major factor that led to significant declines in timber production (from >13 million board feet/year in the late 1980s to <2 million in the early 2000s). Legal exposure created by the suite of substantive requirements to protect biological diversity under the NFMA and ESA forced the agency to address wildlife conservation, something that had not come to pass under the MUSYA. However, even in the 1990s, pressure to prioritize timber production over the protection of wildlife remained strong because of internal biases, financial incentives, and Congressional intervention (Wilkinson 1992, Government Accountability Office 1997, Corbin 1999).

Although agency culture and priorities have shifted over time, biodiversity conservation still may conflict with activities like timber harvest, fuels reduction, recreation, or energy development, all of which the USFS has strong economic and political incentives to promote. Literature in political science and economics predicts that when given conflicting tasks by Congress, such as the multiple use mandate, agencies systematically prioritize high incentive and measurable goals over those that are lower incentive and more difficult to measure (Biber 2009). A highly discretionary NFMA diversity regulation could lead the USFS to prioritize higher incentive and measurable goals that are supported by political interests.

Given this reality, even when regulations for protecting plant and animal diversity are well meaning and scientifically sound, if they are not specific, measurable, binding, and enforceable, history suggests that effective wildlife conservation planning will end up as a secondary objective (Houck 1997). Specific, mandatory language is needed to protect wildlife on the National Forests, a point not lost on the first Committee of Scientists, who wrote the following in 1979, "It is simply not possible to carry out the planning requirements of NFMA in accordance with a set of regulations that contain nothing but generalities" (44 FR 53967: 53968). Such specificity, said the Committee, is what the NFMA requires. Historically, the NFMA's diversity provision and its associated regulations have provided an effective counterbalance to competing agency demands and political pressures. However, without more specific requirements, the administrative discretion in the 2012 rule's diversity provisions will lead to varied implementation across management units, give managers who are not committed to wildlife conservation the leeway to pursue other management goals without concern for biodiversity, and leave managers who are committed to protecting biodiversity without a solid, legal framework to help them withstand internal and external pressures to prioritize other factors.

Although the diversity provisions in the 2012 planning rule itself are highly discretionary, the agency, through the Directives system, could adopt standards and practices for wildlife conservation that are more prescriptive and would help to ensure that the rule is implemented in a more robust fashion and informed by the best available science. We urge the agency to implement the rule in a manner that closes the gap between the stated purpose of maintaining ecological integrity and diversity, and the highly general and discretionary operational provisions in the rule that are meant to achieve these purposes. The Wildlife Society and other professional organizations can play an important role in guiding this process, and for this purpose, we offer a series of recommendations that would strengthen the key wildlife provisions in the 2012 rule.

Coarse-Filter Contributions

Coarse-filter approaches, typically focused at broader spatial scales than fine-filter strategies, are aimed at communities, ecosystems, or landscapes (Schwartz 1999). Their central role in the 2012 rule complements fine-filter provisions and commits the USFS to multi-scaled assessment and monitoring efforts. Coarse-filter conservation strategies often rely on habitat predictors (e.g., dominant vegetation and landform) derived from satellite imagery (e.g., the California Wildlife Habitat Relationships System, http://www.dfa.ca.-gov/biogeodata/cwhr). Under this approach, all appropriate habitats within a planning unit that intersect the species' geographic range are typically assumed to support the species. This assumption is often based on anecdotal occurrence data because the spatial extent of coarse-filter strategies often constrains the agency's ability to implement probability-based survey designs. The consequence is that commission errors are likely, which can lead to the erroneous conclusion that animal diversity is being maintained when it is not. Although these concerns limit the ability the coarse-filter approach to serve as a substitute for fine-filter assessments, a management objective to sustain dominant vegetation communities and their successional stages at broad spatial scales is an essential aspect of a comprehensive approach for sustaining biological diversity. In the context of the diversity requirements of the 2012 rule, measures of the effectiveness of the coarse-filter are presented in terms of species' metrics (e.g., number of rare and imperiled species conserved, presence of apex consumers, species richness, etc.). Therefore, verifying the efficacy of the coarse-filter approach requires some level of direct species-level assessment, and a comprehensive diversity policy requires a carefully balanced coarse-filter/fine-filter strategy.

Implementing the Fine-Filter Approach

We are concerned with the limited commitment to conduct fine-filter (species-level) assessments in the new rule. We found little scientific evidence to suggest that maintaining the diversity and integrity of a combination of habitat types "will provide the ecological conditions for the long-term persistence of most species within the plan area" (36 CFR §219.9). The Committee of Scientists stated, "Habitat alone cannot be used to predict wildlife populations" and "diversity is sustained only when individual species persist; the goals of ensuring viability and providing for diversity are inseparable" (Committee of Scientists 1999, Chapter 3:19,38). For this reason, the fine-filter species assessment is critical.

The rule is inaccurate in the way it portrays its coarse- and fine-filter approaches. It claims that the coarse-filter approach, along with the inclusion of fine-scale habitat management requirements for species that are not adequately protected, constitutes a combined coarse-filter/fine-filter approach. This discussion misconstrues fine-filter species conservation approaches, which entail direct assessment at the species level, including monitoring state variables such as a species' abundance, density, survival, birth rate, or occupancy. Managing fine-scale habitat components for a given species is not the same as fine-filter assessment.

The USFS defines focal species, in part, based on their functional significance to ecosystem processes (36 CFR §219.19[2012]). The planning rule requires the selection and monitoring of focal species "to assess the ecological conditions required under §219.9 . . ." (§219.12[a][5][iii]), and it is this aspect of the rule that holds the most promise as a genuine, complimentary fine-filter approach to wildlife conservation planning. The USFS defines ecological conditions as "the biological and physical environment that can affect the diversity of plant and animal communities, the persistence of native species, and the productive capacity of ecological systems" (36 CFR §219.19[2012]). An emphasis on monitoring species with known or suspected functional significance to ecosystems process and sustainability is appropriate. Ecosystem resilience is strongly related to native species diversity and functional redundancy (the degree to which multiple species perform similar ecosystem functions [Naeem et al. 2009]). In general, ecosystems with greater native species diversity are more resistant to disturbance, recover more quickly following disturbance, and are less likely to experience irreversible changes than species-poor communities (Cottingham et al. 2001, Hooper et al., 2005, Naeem et al. 2009). Furthermore, species loss ranks among the most severe global change stressors, with effects comparable to those of climate warming, acidification, and elevated carbon dioxide (Hooper et al. 2012). Therefore, it is inconsistent with the stated intent of §219.9 to maintain or restore ecological conditions not to include a commensurate requirement to maintain viable populations of focal species.

Another central requirement of the 2012 rule is the mandate to contribute to the recovery of proposed, candidate, and listed ESA species and to protect viable populations of species of conservation concern. Section 219.9 requires that species-specific habitat management components be built into plans if the responsible official determines that coarse-filter approaches are insufficient for maintaining viable populations of species of conservation concern, and ESA species, within the plan area. We are concerned that, as presently construed, the rule does not require the monitoring of these species. Thus, it is unclear what information will be used to determine if a species maintains a viable population within the plan area, or if it requires additional species-specific conservation actions. Because the coarse-filter approach may be insufficient to provide insights into the status and trend of species (Cushman et al. 2008), some direct species-level monitoring is necessary. Without such monitoring, the USFS's approach is problematic; by the time evidence of further decline for these already at-risk species is found, threats may have significantly increased.

Ideally, the rule would have committed to population-level monitoring and viability for both focal species and species of conservation concern. Extending the viability requirement to focal species, selected in part because of their known or suspected functional significance, is a logical way to address the ecosystem integrity goals of the rule. Further, monitoring species of conservation concern will provide essential information to assess their viability. These changes, incorporated into the Directives, would connect the commitment to species-level conservation with the mandate for adaptive management and bring greater cohesion to the disjointed diversity provisions in the 2012 rule. In addition, all species-level monitoring should include trigger points so that significant declines in either focal species or species of conservation concern would initiate reviews of management policies.

Selecting Species of Conservation Concern and Focal Species

The process for selecting focal species and identifying species of conservation concern, separately or in concert, is not detailed in the rule. The rule simply states that the selection of species of conservation concern will be based on the best available science and evidence of substantial concern about their long-term persistence in the plan area. The Record of Decision indicates that further guidance will be provided in the Directives, but that the Department of Agriculture expects species to be identified based on existing classifications of risk, such as NatureServe conservation status or those listed as threatened or endangered under state law (77 FR 21162:21218). In addition to referencing NatureServe and state law, we recommend the agency also consider IUCN red-list species that are not already listed under the ESA, and high priority species identified in State Wildlife Action Plans; if such species are not selected, a rationale for failing to designate them as species of conservation concern should be required.

Criteria for focal species selection include the species' functional roles in the ecosystem and sensitivity to changing conditions, management activities, particular threats, or desired ecological conditions (77 FR 21162). This is consistent with recommendations of the most recent Committee of Scientists' Report (Committee of Scientists 1999). Additional guidance in the Directives will be necessary to establish and maintain consistency and efficacy across management units in the selection of focal species. Noon et al. (2009) provide useful guidance on focal species selection for fine-filter assessments on federal public lands. Furthermore, we see no reason that species identified as species of conservation concern cannot also be identified as focal species, providing a ready avenue for conceptual integration of the fine-filter approaches under the new planning rule.

Establishing a step-down process to identify and prioritize species for fine-filter monitoring that reflects the reality of Forest Service monitoring budgets remains a major challenge. This topic goes beyond the scope of our paper, but to initiate discussion, we suggest that identifying the core species (Magurran and Henderson 2003) that are 1) persistent members of a given management unit; 2) functionally significant; and 3) at risk in that unit may be a first step in developing a manageable species set.

Developing Informative Monitoring Programs

The planning rule requires forests to develop monitoring programs that will include a set of questions and indicators to track change, measure management effectiveness, and assess progress towards desired future conditions. The rule only commits to monitoring focal species, which as mentioned above, may include species of conservation concern (the fine-filter approach). It also requires monitoring a select set of ecological conditions in accordance with the objectives of §219.9 (the coarse-filter approach). The Regional Forester is required to develop a broad-scale monitoring plan to address issues relevant at a scale larger than a single National Forest. The content of the broad-scale monitoring plan is at the discretion of the Regional Forester, and s/he is required to coordinate with other jurisdictions, other branches of the USFS, and the public. Additionally, monitoring plans may be coordinated across units. The responsible officials are to conduct biennial evaluations of monitoring information and adjust management activities as necessary.

At the outset, the discussion of species monitoring in the Record of Decision (77 FR 21162:21232–21233) is confusing and suggests a critical misunderstanding by the USFS of environmental monitoring. The Record of Decision (77 FR 21162:21233) states, "The final rule does not require monitoring species population trends. Species population trend monitoring is costly, time intensive, and may not provide conclusive or relevant information." This perspective is at odds with the general understanding in the scientific literature of environmental monitoring. For example, Suter (1993:505) states that monitoring is the "measurement of environmental characteristics over an extended period of time to determine status or trends in some aspect of environmental quality." Monitoring of an appropriate state variable (e.g., occupancy) is conducted at regular intervals to assess both the current state and time trend in some ecological resource (e.g., a species' population [Noon 2003, Nichols and Williams 2006])—that is, the stated purpose of monitoring is to estimate temporal trends.

Provisions in the rule encourage the development of robust monitoring strategies. However, our primary concern is whether these strategies will be developed, funded, implemented, and designed in such a way that they inform adaptive planning. As noted previously, monitoring has been chronically underfunded by federal agencies. The rule requires development of a monitoring plan but does not specify a particular standard of quality or utility of monitoring data. Since Congress annually sets the agency's budget, the USFS cannot commit to funding monitoring at a particular dollar amount. However, committing a certain percentage of planning dollars to monitoring may be possible so that the USFS can address its commitment to adaptive management.

Following the United States Supreme Court's decision in *Norton v. SUWA* (2004), enforcing monitoring requirements of federal land use plans is difficult. In language easily extendible to NFMA plans, that case held that commitments to monitor in Bureau of Land Management land use plans are not generally binding or reviewable under the parameters of administrative law. The Court noted that monitoring requirements could perhaps be written in such a way as to make them enforceable, if they were written as clear and binding commitments. In some cases, when monitoring activities are clearly required before undertaking certain activities, monitoring can be enforceable in court (Blumm and Bosse 2007). However, because requiring or enforcing funding levels or data quality standards for monitoring programs is generally difficult, oversight will be necessary to ensure that monitoring occurs in a way that it clearly assesses management and restoration actions.

We recommend that multi-party oversight boards be established to aid in the design of monitoring programs, contribute to the selection and prioritization of monitoring state variables, provide science consistency checks, provide interpretations of the monitoring data, suggest when changes to management practices are needed, and advocate for consistent funding. Because monitoring data will unlikely be subject to judicial review, oversight from a multi-party stakeholder monitoring board could increase the likelihood that monitoring will provide reliable information and useful insights into future decision making (Nie and Schultz 2012). Such boards must consider how monitoring data will inform decision making and the level of statistical certainty required to trigger a change in management actions.

All species-level monitoring should include trigger points so that significant declines in either focal species or species of conservation concern will initiate reviews of management policies. If trigger points are not identified, monitoring data may not feed back into adaptive planning and decision making (Noon 2003). Triggers will be critical for species-level monitoring and for any evaluation of species viability. Monitoring enforceability also would be substantially increased if forest plans included requirements that before approving any major projects, such as those requiring an Environmental Impact Statement, the responsible official find that monitoring programs are being implemented and that no trigger points have been exceeded without corrective action.

Maintaining Current Populations and Adequate Distribution of Species

Whether the planning rule intentionally allows for local extirpation of species or range reductions in cases where this might be avoided is unclear, but the decline and loss of species from the planning area is an allowable outcome of USFS management under the new rule. Aside from the loss of a broader viability requirement, this is the most significant change from the 1982 rule: the replacement of language requiring that viable populations be well-distributed, with the definition of a viable population as one that "continues to persist over the long term with sufficient distribution to be resilient and adaptable to stressors and likely future events" (36 CFR §219.19 [2012]). The impact of the change stems from the fact that what constitutes a "sufficient distribution" is not defined in the rule, providing broad discretion to the responsible official and obfuscating the well-established relationship between geographic distribution and persistence likelihood (e.g., Harris and Pimm 2008).

Furthermore, the rule establishes that the USFS does not need to protect viable populations, as required in the 1982 rule, if this is not within the "inherent capability of the plan area," a vague concept that is never defined in measurable terms. In this case, the USFS is held to a much lower conservation standard: documenting the rationale for such a determination and working across land ownerships to create management standards and guidelines to maintain or restore conditions that will contribute to maintaining a viable population of the species within its range (36 C.F.R. §219.9(b)(2)(i) [2012]). The USFS also states, "the individuals of a species of conservation concern that exist in the plan area will be considered to be members of one population of that species" (77 FR 21162:21217). In light of this, whether the agency is committing to maintaining a viable population of a species of conservation concern when it is not within the inherent capability of a single plan area to protect a viable population is not entirely clear. Depending on how the agency interprets these standards, it might never have to commit to maintaining a viable population of a low-density, wide-ranging species, but it might have to commit to maintaining multiple viable populations of species with more constricted ranges.

To address ambiguities in the 2012 viability requirements, we recommend that the USFS explicitly recognize the importance of maintaining a wide geographic distribution for species viability. Species that are widely distributed across the landscape are much less likely to experience spatially correlated disturbance events (den Boer 1981). Maintaining the distribution and viability of rare or widely distributed species and populations will require close coordination among administrative units. Guidance should be included in the Directives indicating that the agency should assess viability (perhaps employing more efficient distributional analyses based on occupancy [Noon et al. 2012]) across ownerships and plan units, when this will enhance the likelihood of persistence for individual species. When the USFS determines that maintaining a viable population of a species is not within the inherent capability of the plan area, the agency should solicit scientific comment and review. This review will help ensure that the agency is aware of all relevant scientific information that may conflict with their determination and will better prepare the agency to defend its decisions against possible legal challenge. In cases where the USFS determines that providing for a viable population of a species that relies upon National Forest System lands for its habitat is not within the capability of the plan area, we recommend that the agency task itself with restoring populations, to the maximum extent practicable. At the least, a standard should be included in the Directives that directs the agency not to authorize or permit activities that reduce the viability of any species of conservation concern.

Development on private land and other activities external to National Forest System lands may affect species such that the USFS cannot alone ensure their viability. A critical question is to what extent should this compel the USFS to compensate for declines in species status due to factors outside of their control. Recall that the NFMA emphasizes the National Forests' role in conserving resources for the American people, in perpetuity. It does not imply that this objective is restricted to National Forest System lands. There is ample historical precedent for the USFS to consider what is happening outside of its jurisdiction and proactively respond on the National Forests (Nie and Miller 2010). In the view of the first chief of the USFS, Gifford Pinchot, 1 rationale for establishment of the National Forests was to compensate for unsustainable management of resources on private lands (Wilkinson 1992). Pinchot was focused on unsustainable timber harvest at the time, but the reasoning applies widely to other natural resources on USFS lands based on changing public values and priorities. The USFS, in its 2012 rule, emphasizes its responsibility to maintain and restore ecosystem diversity and integrity, and diverse plant and animal communities are fundamental to ecosystem integrity (Naeem et al. 2009). If development on private land is adversely affecting biodiversity, the USFS has a greater, not lesser, responsibility to protect species on its lands. This compensation principle will become even more significant given predictions of private land development in the future, with much of this development projected in the wildland urban interface (Nie and Miller 2010). The National Forests, and federal lands in general, will become more important to wildlife in increasingly developed landscapes. Therefore, the "inherent capacity" clause of the 2012 rule should be used rarely, if at all, and if used, be subject to scientific and public review. The USFS must recognize its increasingly important mission to conserve the nation's forest and grassland ecosystems during the current period of rapid global change and species loss, when unpredictable transformations of ecosystems may be the "new normal" (Barnosky et al. 2012).

Considerations Regarding the Relationship Between the NFMA and the ESA

Important intersections exist between biodiversity conservation requirements under the NFMA and the ESA, which work together as part of this nation's biodiversity conservation policy. Wildlife provisions in forest plans are a significant factor in ESA decision making (see below), and ESA decisions have profound and far-reaching implications for forest management. Ideally, viability protection on National Forests would serve as an early warning signal that a species may be heading towards local extirpation or extinction. A proactive approach to address risks to a species' viability could avoid costly and polarizing ESA decisions that might limit management flexibility for the USFS.

On the National Forests, currently 430 species are listed under the ESA as threatened or endangered, and an additional 60 species are candidates for listing (USFS 2011:14). More than 647,000 ha of terrestrial habitat and 35,000 km of stream habitat on USFS lands are designated as critical habitat under the ESA (USFS 2011:14). For these and other reasons, the 2012 planning rule emphasizes the connections between forest planning and the ESA more than previous regulations:

> The [Department of Agriculture] anticipates that plan components, including standards or guidelines, for the plan area would address conservation measures and actions identified in recovery plans relevant to T&E [threatened and endangered] species. When implemented over time, these requirements would be expected to result in plans that will be proactive in the recovery and conservation of the threatened, endangered, proposed, and candidate species in the plan areas. These requirements will further the purposes of section 7(a)(1) of the ESA, by actively contributing to threatened and endangered species recovery and maintaining or restoring the ecosystems upon which they depend (77 FR 21162:21215).

One way in which the USFS can actively contribute to species conservation and recovery is by providing wildlife and habitat-based standards in individual National Forest plans. The NFMA requires the incorporation of standards and guidelines in land and resource management plans (16 U.S.C. 1694). Standards are mandatory constraints on USFS projects and activities and are used to achieve or maintain desired conditions and planning objectives, to avoid or mitigate undesirable environmental impacts, and to meet applicable legal requirements (76 FR 8480). Guidelines, as commonly applied, also constrain decision making but allow for some deviation from rules as long as the intent of the guideline is achieved (76 FR 8480).

The types of wildlife and habitat-based standards used in forest planning differ in scale, specificity, and complexity. Some standards cover multiple National Forests, such as the Northwest Forest Plan's Aquatic Conservation Strategy (discussed below) and the Inland Native Fish Strategy. The latter, covering at one point 22 National Forests, is used to protect native fish and their habitats in eastern Oregon and Washington, Idaho, western Montana, and portions of Nevada. It does so by using several riparian management objectives, standards, guidelines, and monitoring requirements (USFS 1995). The Inland Native Fish Strategy's standards and guidelines replaced conflicting direction in multiple National Forest plans, except when those forests provided for more protection for inland native fish habitat. Standards can also be applied forest-wide, such as requiring that all snags over a certain size be retained or that a specified percentage of old growth be maintained on a National Forest. Other standards apply to particular management areas or zones as delineated in a land use plan; they often permit or prohibit various uses, such as grazing or the application of herbicides in a municipal watershed zone.

An enduring debate continues over the appropriate role of standards in forest planning. The 2012 rule requires every plan to include standards as 1 of 5 plan components (36 C.F.R. §219.7), but it leaves their application to the discretion of the responsible official, with the expectation that further direction will be provided in the Directives system (77 FR 21162:21206). Regarding the diversity of plant and animal communities, the rule requires standards or guidelines be used "to maintain or restore ecological conditions within the plan area to contribute to maintaining a viable population of the species within its range" (36 C.F.R. §219.9). Standards for wildlife protections should play a significant role in the new forest plans that will be written under the 2012 regulations. Legally binding and enforceable standards promote accountability and provide increased certainty about future management actions. Without them, there is an increased risk that wildlife protections will give way to other agency pressures and priorities.

Forest plan standards can play significant roles in decisions to list or delist a species under the ESA. One of the 5 factors to be considered by the wildlife regulatory agencies that enforce the ESA (the National Oceanic and Atmospheric Agency [NOAA] Fisheries and the U.S. Fish and Wildlife Service [USFWS]) in making ESA listing decisions is "the inadequacy of existing regulatory mechanism[s]" (16 U.S.C. §1533). Vague, voluntary, speculative, and unenforceable measures found in plans are generally not considered a sufficient regulatory mechanism (*Oregon Natural Resources Council v. Daley* 1998). Instead, federal wildlife agencies and the courts typically assess whether a plan contains specific and legally enforceable standards having regulatory force. Forest plan standards also can be relevant for determinations made by the wildlife regulatory agencies under section 7 of the ESA, which requires federal agencies to undergo consultation with the wildlife agencies to ensure their projects will not cause jeopardy to a listed species.

Several cases have been decided in which NOAA Fisheries and the USFWS made a no-jeopardy determination under section 7 of the ESA or decided not to list a particular species because a forest plan contained binding standards and other regulatory mechanisms to protect the petitioned species. One example is the decision not to list the Queen Charlotte goshawk (*Accipiter gentilis laingi*) in southeast Alaska. Roughly 80% of this region is managed by the Tongass National Forest, and petitioners argued that old-growth logging in the region posed a threat to goshawks. Standards and other regulatory mechanisms specified in the 2007 Tongass Land Management Plan were significant factors in the decision by the USFWS to not list the goshawk (72 FR 63133). The USFWS also emphasized the legally binding and enforceable nature of Tongass forest planning standards in its 1997 status review of the species (USFWS 2007), and the Department of the Interior asked the USFS to retain the Conservation Strategy in the 2008 Tongass Forest Plan Amendment. The USFS also recognizes the significance of these wildlife standards. Possible changes to the Strategy, according to Undersecretary of Agriculture Harris Sherman, "could hamper the plan's ability to maintain viable populations of plant and wildlife species [and] this could lead to the need for USFWS to reconsider its previous determinations regarding the goshawk . . ." (Sherman 2011:8).

The Aquatic Conservation Strategy, part of the Northwest Forest Plan, provides another example of the interactions between binding standards and the ESA (USFS and Bureau of Land Management 1994). The purpose of the Aquatic Conservation Strategy is to maintain and restore the ecological health of watersheds in the northwestern National Forests. The Strategy includes several binding standards and guidelines that apply to key watersheds, riparian reserves, required watershed analyses, and watershed restoration. In biological opinions written in accordance with section 7 of the ESA, NOAA Fisheries equates Aquatic Conservation Strategy consistency with no-jeopardy findings, a practice that has satisfied the courts (*Pacific Coast Federation of Fishermen's Associations v. National Marine Fisheries Service* 2001). Standards such as these can be used to protect wildlife while also achieving the restoration and watershed protection purposes of the 2012 rule.

The lack of enforceable standards and clear conservation commitments made in forest plans also has been a factor influencing decisions to list a species. In these cases, NOAA Fisheries and the USFWS determine that a forest plan fails to provide sufficiently certain, binding, and detailed protection to a species to count as an adequate regulatory mechanism. One of the most significant decisions in this regard is provided by the listing of Canada lynx (*Lynx canadensis*) as threatened in 2000 (65 FR 16052). The species was classified as a sensitive species by the USFS before listing, but most National Forests with lynx did not have population viability objectives or management standards and guidelines in place at the time (63 FR 37005). The fact that forest plans in effect at the time did not provide enough protection and guidance for the conservation of the lynx is a primary reason why the species was listed. The USFWS determined that these forest plans permitted several actions that cumulatively could cause a significant threat to lynx persistence across its range (63 FR 37005). The USFS responded to the listing by amending multiple national forest plans to incorporate various lynx standards and guidelines (USFS 2007). Currently, the USFS does not have to engage in ESA consultation with the USFWS on a project-by-project basis if these projects comply with these binding and enforceable lynx standards. Another prominent example is the 2010 decision to list the greater sage-grouse (*Centrocercus urophasianus*) as warranted-but-precluded, meaning the species is warranted for listing but precluded from actually being listed because of funding limitations (75 FR 13910). The USFS manages roughly 8% of the sagebrush habitat significant to the species. Greater sage-grouse were designated by the USFS as a sensitive species on USFS lands across the range of the species, and 14 of these forests designated the bird as a management indicator species (75 FR 13910:13979). But of the 33 National Forests managing greater sage-grouse habitat, "16 do not specifically address sage-grouse management or conservation in their Forest Plans, and only 6 provide a high level of detail specific to sage-grouse management" (75 FR 13910:13980). The lack of detailed protections and the variation among National Forest plans in the greater sage-grouse area was an important factor in making the warranted-but-precluded determination (75 FR 13910).

Enforceable wildlife standards and protections on the National Forests also play a role in delisting species from the ESA. One of the few species to be delisted under the ESA is the Robbin's cinquefoil (*Potentilla robbinsiana*), an endemic plant found in the White Mountains of New Hampshire, in areas managed exclusively by the White Mountain National Forest (67 FR 54968). The USFS was able to assist in the recovery of this species by restricting entry to particular areas of the National Forest, relocating trails, and entering into a Memorandum of Understanding with the USFWS. This Memorandum of Understanding included provisions related to habitat protection and monitoring, and it served as a long-term commitment by the USFS to conserve this plant, irrespective of its status and potential delisting under the ESA (USFS and USFWS 1994). The USFS regulations also prohibited removing, destroying or damaging plants that are classified as threatened, endangered, rare, or unique (36 C.F.R 261.9). All of these specific actions and commitments—the protective actions taken by the White Mountain National Forest, the plant regulations, and the Memorandum of Understanding—served as an adequate regulatory mechanism for delisting the species by the USFWS.

A more controversial example is the proposed delisting of the Yellowstone distinct population segment of grizzly bears (*Ursus arctos horribilis*). The lack of regulatory mechanisms to protect grizzly bear habitat on National Forest System lands was 1 reason why the species was listed in 1975 (40 FR 31734). A conservation strategy for the bear was written pursuant to its recovery plan to provide adequate regulatory mechanisms after the bear's delisting. The USFS amended 6 forest plans to incorporate the habitat standards and other provisions in the conservation strategy. The USFWS considers these standards to be adequate regulatory mechanisms for the purpose of delisting grizzly bears, but much of the debate and litigation over the delisting decision centers on the sufficiency of these standards. A district court found the delisting impermissible, partly because the amended forest plans contained discretionary and legally unenforceable guidelines, rather than binding standards, in the bear's primary conservation area (*Greater Yellowstone Coalition v. Servheen* 2009). The Ninth Circuit disagreed with the lower court on this matter and found the standards, as applied by the USFS within the primary conservation area, to be sufficient under the ESA because they are a legally enforceable part of National Forest plans, and management of these forests must be consistent with their governing forest plans (*Greater Yellowstone Coalition v. Servheen* 2011).

The 2012 rule also requires that forest plans provide the ecological conditions to "contribute to the recovery" of listed threatened and endangered (T&E) species (77 FR 21162:21215, 36 C.F.R. §219.9). The USFS has an expectation that forest plans would use standards or guidelines "to address conservation measures and actions identified in recovery plans relevant to T&E species" (77 FR 21162:21215). Better use of ESA recovery objectives could lead to more proactive, integrated, and strategically coordinated forest plans.

We recommend that more guidance be provided as to how synergies might be developed between forest and ESA recovery planning. Scott et al. (2005:386) show that "most listed species will require continuous management action in order to maintain their recovered status." These "conservation-reliant species" can only be maintained as a self-sustaining population in the wild "if ongoing management actions of proven effectiveness are implemented" (Scott et al. 2005:386). The Memorandum of Understanding and revised forest plan for Robbin's cinquefoil provide this sort of ongoing protection to a conservation-reliant species, and similar standards in forest plans could do the same for other T&E species on the National Forests.

The number of ESA listing decisions will only increase in the future, given the September 2011 settlement between the USFWS and environmental groups requiring the agency to make listing decisions on over 800 species, including 262 candidate species, for which such decisions have been delayed (Center for Biological Diversity 2012). Altogether, another 1,000 listing decisions will possibly have to be made by 2020 (Rylander 2012:10018). Furthermore, conservation scientists, the IUCN, and the Intergovernmental Panel of Climate Change all predict increases in the number of species threatened with extinction (Scott et al. 2010). For these reasons, the impact of ESA listing decisions on National Forest management is likely to increase over time. The use of binding standards in forest plans would likely serve to decrease the number of species listed as threatened and endangered and promote delisting decisions in the future.

If implemented in a robust fashion, the NFMA's diversity mandate will serve as a precautionary and proactive approach to wildlife conservation. In contrast, the ESA provides a more reactive and crisis-based approach to decision making, since the law's protective measures are usually not initiated until a jeopardized species is listed, and by that time, it is already in the proverbial emergency room. Federal wildlife agencies take an average of 11 years to list species (Greenwald et al. 2006), frequently after their long-term viability is in doubt (Wilcove et al., 1993, Neel et al. 2012, Rylander 2012). Waiting until a species is on the brink of extinction before taking protective measures creates unnecessary risks to a species and increases the controversies, costs, and restrictions associated with their recovery. Furthermore, funding is inadequate to meet the needs of species that are already listed, are candidates for listing, or have been petitioned for listing (Scott et al. 2010). Strong wildlife provisions under the NFMA could provide an earlier, proactive response to species declines, lessening the trend for more listings under the ESA. Allowing populations to decline towards listing is not good policy ecologically, politically, or economically. It will only reduce management flexibility for states, private citizens, and federal agencies and will further burden managers implementing the already underfunded ESA.

CONCLUSIONS

Given clear guidance in the Directives and sufficient funding, the 2012 planning rule has the potential to be a highly effective framework for wildlife conservation on National Forest System lands. It commits the Forest Service to a formal adaptive management process, adopts a landscape perspective as the primary context for forest planning, strives to find an appropriate balance between coarse- and fine-filter approaches to the assessment of biological diversity, and codifies the need to monitor focal species at multiple spatial scales. These are all significant advances that signal the Forest Service's commitment to a new planning rule that is responsive to the status and trends of ecological systems, as well as the expectations of the nation for the wise stewardship and conservation of public lands and resources.

Although we are confident that the rule can be implemented so as to effectively conserve wildlife populations, we are concerned about the ambiguity of the plan's diversity provisions and the level of discretion permitted when interpreting and implementing the plan's most fundamental actions: identifying focal species, monitoring status and trends, establishing triggers for adaptive management, and taking action to sustain viable populations. Effective implementation of the rule will require a commitment to direct monitoring of focal species, species of conservation concern, and ESA species, as well as a commitment to maintaining their viability. Without this commitment, the provision to sustain biological diversity is incoherent and unlikely to be effective. Triggers will have to be established for monitoring of species to signal when a review of management approaches is necessary. Without an assessment of the effects of management actions via monitoring, the agency cannot fulfill its obligation to manage adaptively. When private land development or other more distant factors affect the viability of species, the USFS should place more, not less, emphasis on providing for viable populations to the extent practicable. The design of monitoring programs, determinations about the inherent capability of the land, and selection of focal and species of conservation concern should be based on the best available scientific information.

The language of the new rule is more discretionary than the 1982 rule, and it removes the requirement to maintain viable populations of all vertebrate species. Although this is unquestionably a significant change in regulatory language, some might argue the 2012 rule merely codifies the way the USFS has managed for diversity since 1982. In practice, management indicator species seldom have been monitored directly in a way that allowed for a clear understanding of their response to management actions, and the USFS has been managing for Regional Forester Sensitive Species by relying primarily on habitat measurements as proxies for the species' current status. In effect, the 2012 rule simply makes it more explicit that this relaxation of the standards established in the 1982 rule will be the USFS's accepted standard for managing for diversity—to focus primarily on coarse-filter approaches, with the expectation that currently abundant species will remain abundant, and that sensitive but stable wildlife populations will, by and large, persist. The problem with this approach is that the NFMA includes clear requirements to provide for a diversity of plant and animal species, not just a range of ecological conditions that may or may not support diversity. In the end, habitat is a meaningless concept if it is never occupied by actual individuals of the species in question.

With the new rule, the USFS faces a new set of decisions that it can address from a position of power, with greater discretion over its approach to wildlife, and forest management in general. It has the opportunity to improve upon past efforts to conserve wildlife and biological diversity, or it could retreat from the responsibilities established in the NFMA and the 1982 rule. At this juncture, the USFS and the broader community of foresters and wildlife managers should pause to consider whether a relaxation of standards—most notably with respect to population viability—and the consequent lessening of agency responsibility and authority is in the best interest of the nation or the agency itself. We respectfully argue that conservation of the nation's biological wealth, including the persistence of viable populations of wildlife species, is an important service that a strong and professional USFS can and should provide to the American public. To the extent that the agency uses its new discretion to lessen its responsibility to wildlife and its exposure to controversy and criticism, the 2012 rule is likely to represent a retreat from an essential public responsibility and a blow to the wildlife profession. But to the extent that the agency signals its leadership on these issues by voluntary committing itself to a nationwide, science-based, and outcome-oriented program of adaptive management of both forest ecosystems and their full complement of species, the 2012 rule will signal a new era of leadership, where increased discretion is used to elevate intent and expectations, accept greater responsibility, and provide energetic leadership in the conservation and management of the nation's public lands and wildlife.

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Review and assessment of LANDFIRE canopy fuel mapping procedures

Joe Scott

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Executive Summary

The LANDFIRE procedures for quantifying and mapping canopy fuel characteristics follow generally accepted scientific procedures in the fields of fuel science and remote sensing. Accuracy of LANDFIRE canopy fuel products is low, but consistent with constraints imposed by the very large (national) extent of the effort and the high inherent variability of the characteristics being mapped. Other canopy fuel mapping efforts have achieved greater accuracy than LANDFIRE's products, but at greater cost per acre mapped, and by employing methods that can't be applied at LANDFIRE's extent. The problem of low map accuracy of LANDFIRE canopy fuel products is a greater problem for project-level geospatial fire analyses than for the national-level analyses which LANDFIRE was designed to support. Insufficient accuracy can be resolved by end users through a routine process of critique and calibration (refinement using local information) and refreshing (to account for changes in the landscape since the effective date of LANDFIRE products). Work is now underway to develop a standard procedure for critiquing and calibrating LANDFIRE data layers and to refresh the LANDFIRE data to the present time. These efforts will improve accuracy for both project- and national-level analyses.

Artificial seams in LANDFIRE data products may exist both within and between map zones. The problem of data seams is very difficult to resolve once the data have been published by LANDFIRE, but are unavoidable given the scale and constraints of the project. The utility of LANDFIRE data for national-level analyses is not significantly compromised by these seamlines, but regional- and project-level analyses may suffer from the difficult-to-remove seams.

This report is organized around seven potential shortcomings or problems with canopy fuel related LANDFIRE data products:

- canopy cover values are too high,
- data discontinuities exist within and between map zones,
- canopy bulk density values are too low for use in FARSITE,
- canopy base height is too high to generate crown fire,
- treelist data sources may not be best for canopy fuel calculations
- alternative canopy fuel calculation programs may produce different results
- Refreshing and calibrating LANDFIRE data is needed

Canopy Cover values too high

The canopy cover values used in the LANDFIRE process were obtained from the National Land Cover Dataset (NLCD). The NLCD dataset was produced using a Classification and Regression Tree (CART) analysis relying on a method combining satellite remote sensing and field data. Unfortunately, there are several cover-related quantities measured by ecologists and used by fire modelers; the different quantities are frequently interchanged, erroneously.

As used in fire modeling software and envisioned by fire behavior specialists, canopy cover is the proportion of the forest floor covered by the vertical projection of tree crowns. Some field methods estimate this quantity without bias, but the most common field measurement technique uses a spherical densitometer that actually measures a quantity sometimes called canopy *closure*—the proportion of the sky hemisphere obscured by vegetation when viewed from a single point. Canopy closure is usually a higher value than canopy cover; canopy cover rarely exceeds about 70 percent, whereas canopy closure often approaches 100 percent. Refer to the FireWords glossary of fire science terminology (Scott and Reinhardt 2007) for more details (available at www.fs.fed.us/fmi/downloads/firewords.html). It is not clear if this is the reason for the discrepancy between the NLCD canopy cover values and on-the-ground experience. Nonetheless the canopy cover values produced by NLCD are acknowledged by the LANDFIRE developers to be too high relative to the quantity used by existing fire models.

Canopy Cover is a key LANDFIRE variable because it is used as an independent variable for estimating a wide range of dependent variables like fuel model and canopy bulk density. As directly used in fire modeling programs, canopy cover is used to estimate wind adjustment factor and fine dead fuel moisture. The wind adjustment factor sub-model in fire modeling systems is relatively insensitive to the magnitude of apparent errors in the canopy cover maps. The dead fuel moisture model, however, is more sensitive to errors in canopy cover. In an unpublished analysis, LANDFIRE's Matt Reeves[1] found that correcting the apparent canopy cover error using an alternative approach resulted in a dead fuel moisture decline of roughly 2 percentage points across example landscapes. This change in fuel moisture led to modest changes in potential fire behavior as simulated with FlamMap[2], but a factor-of-two increase in fire growth using FARSITE[3], a significant increase.

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[1] Matt Reeves is a GIS Specialist and leads the LANDFIRE Fuels Team, stationed at the Missoula Fire Sciences Laboratory.

[2] FlamMap is software that maps potential fire behavior across a landscape for a single specified weather condition, and has features that allow simple fire growth simulation, identification of fire travel paths, and locating fuel treatments. Available at www.firemodels.org

[3] FARSITE is software that simulates the growth of one fire for one projected weather scenario. Available at www.firemodels.org.

Moreover, canopy cover mapping errors may lead to significant indirect fire modeling effects. Because canopy cover is a keystone variable, these indirect effects are difficult to quantify. If canopy cover is overestimated, LANDFIRE may subsequently map the incorrect fuel model, incorrect CBD, incorrect CBH, etc., all of which can strongly affect fire modeling outputs in a geospatial fire analysis. Using the current LANDFIRE fuel mapping procedure, Tobin Smail[4] believes these indirect effects may be small, because they are so heavily calibrated by end users before publication of the data.

Unfortunately, most of the direct and indirect effects of overestimating canopy cover tend to under-predict fire behavior; the effects are not necessarily compensating. For example, overestimating canopy cover in forested areas can lead to slight underestimation of midflame wind speed, slight-to-moderate overestimation of dead fuel moisture content, choosing a too-benign fuel model (one with little or no live fuel, for example). Together, these factors conspire to underestimate surface fire behavior. Overestimating canopy cover can potentially lead to overestimating canopy bulk density in the LANDFIRE process, which in some cases can partially balance the underestimation.

Because it is used as an independent variable, the importance of an accurate canopy cover layer in the LANDFIRE process should not be underestimated. Matt Reeves reports that a newer type of FIA plot allows independent calculation of canopy cover for FIA plots installed since 2005. This new method appears to agree well with the unbiased (but infrequently used) line-intercept field method of estimating canopy cover, whose values correlate very well with what is expected in the fire behavior models, without manipulation. If enough of such plot data is available, it may be possible for LANDFIRE to generate canopy cover maps using this new approach, with significant improvement in fire modeling. Such improved canopy cover maps may also affect dependent LANDFIRE maps such as CBD.

[4] Tobin Smail is a LANDFIRE fuel specialist based at the Missoula Fire Sciences Lab.

Seamlines within and between map zones

LANDFIRE data is “gapless” because it maps fuel and vegetation characteristics across all ownerships across the U.S. That is a critical feature because important aspects of geospatial fire analysis (fire growth modeling and mapping potential fire behavior and effects) require gapless coverage of not only the analysis area but of a large buffer around the area as well. However, despite using a consistent methodology across the U.S., LANDFIRE data is not “seamless” in the sense that obvious artifacts of the mapping process are evident in surface and canopy fuel layers. Seams in LANDFIRE maps can arise from two sources. First, a seam can exist along map zone boundaries, even if the satellite imagery were the same in both map zones, because different protocols and different fuel and fire experts can be used in each map zone. Second, a seam may exist *within* a map zone due to the developers’ need to stitch satellite scenes into a composite image for a whole map zone. This procedure is similar in nature to stitching together digital photos to make a panorama—if the exposure is not the same for each photo, then the boundary between photos becomes obvious in the final panorama. In the LANDFIRE process, if those separate satellite images are of similar quality (captured during times of similar atmospheric conditions, for example) then the compositing process works well and a seam may not be created. However, the separate images may differ in many respects (primarily atmospheric conditions) such that the information contained in one image may differ from another image for the same pixel. The CART analysis assumes that all variation in the images is due to on-the-ground differences, not atmospheric differences unrelated to actual differences on the ground. When used in subsequent CART analyses, the boundaries where the two images were merged can become a noticeable data seam where the map indicates a strong change in value that is not actually present on the landscape. This is a difficult problem to reconcile; there is no easy way to remove such a seam—it’s in the base imagery that the data layers are built upon, and it runs along an artificial (satellite image) boundary. Such data seams can also exist in the inherited NLCD canopy cover data used in the LANDFIRE fuel mapping process, but those seams are generally “hidden” along natural terrain features such as rivers and major ridgelines where changes in vegetation structure are not uncommon. Despite being hidden, such seams can produce disconcerting data discontinuities in the final map.

For example, Charley Martin[5] provided this LANDFIRE CBD data for southern Oregon, which shows the distribution of CBD values in a small watershed that crosses a map zone boundary (figure 1).

[5] Charley Martin is a Fire Ecologist with the Bureau of Land Management’s Medford District, Oregon. Charley has been closely involved in LANDFIRE’s calibration workshops and participated in a separate project to assess accuracy of LANDFIRE fuel maps.

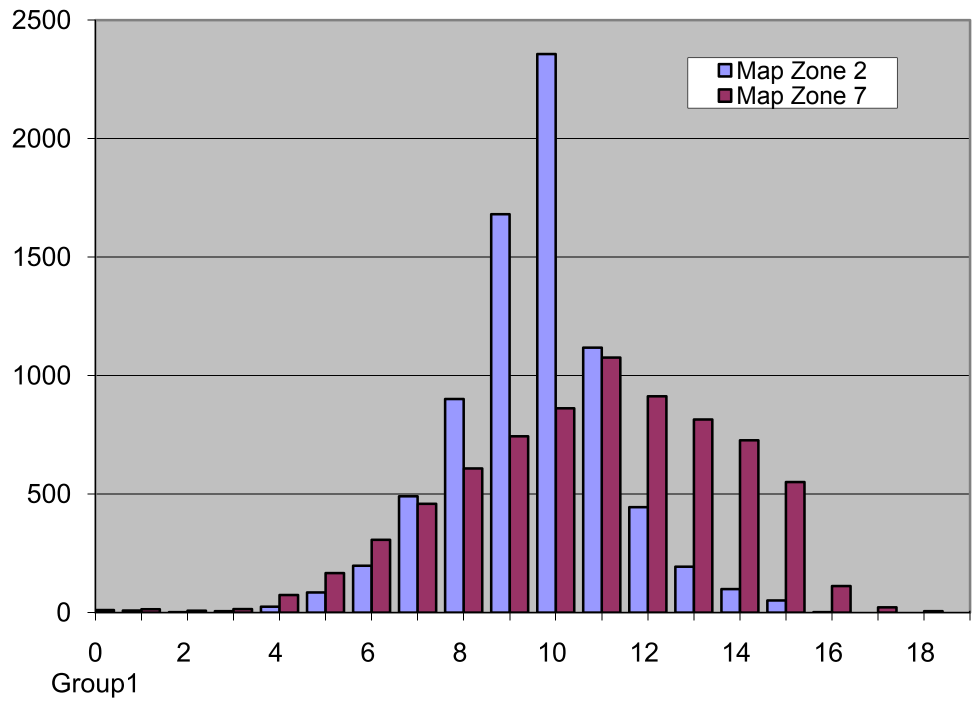

Figure 1 -- Distribution of CBD values (kg/m3 * 100) for a southern Oregon watershed that crosses two map zones .

The expectation, based on field experience in the watershed, is that the distributions should have the same shape. Information such as this can help in a calibration exercise designed to force the map zones into similar distibutions, but there is no way to know which distibution is "correct". The following map (figure 2) shows the nature of the data discontinuity on the CBD map. Similar data seams are evident in nearly all LANDFIRE maps for this watershed.

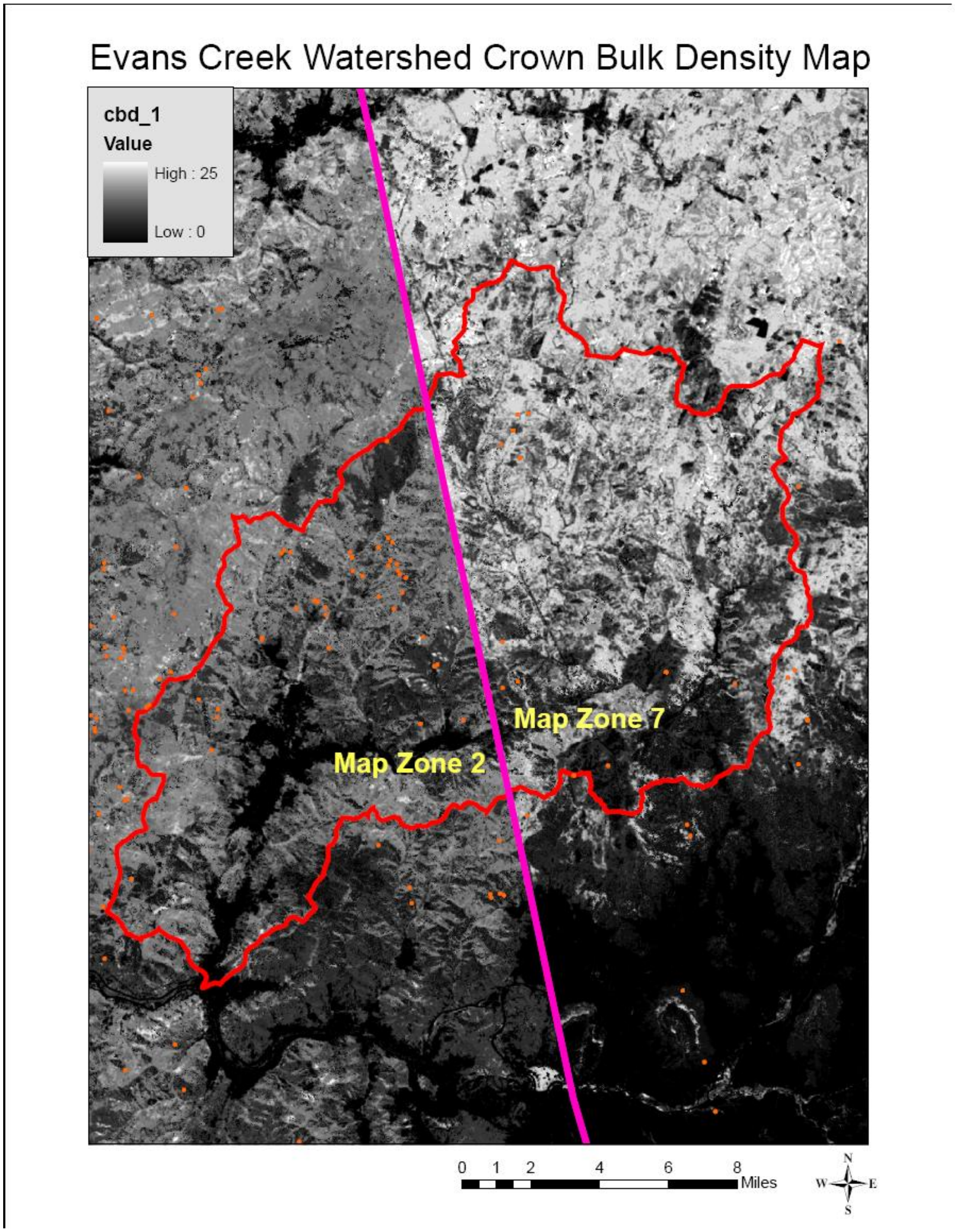

Figure 2 -- LANDFIRE map of CBD in a watershed crossing the map zone 2/7 boundary. CBD is shown as higher in the Map Zone 7 portion of the watershed; on-the-ground experience does not support that result.

Rick Stratton[6] of Systems for Environmental Management[7] is currently working on a procedure for calibrating and updating LANDFIRE data for use in some fire modeling

[6] Rick Stratton is a Fire Modeling Specialist with Systems for Environmental Management, based at the Missoula Fire Sciences Lab.

systems. Fire Program Analysis (FPA) and the USDI National Park Service are co-funding that work. The current in-preparation version of Stratton's work does not yet suggest a method for mitigating seams. Charley Martin, with the BLM in Oregon, is trying an approach that smoothes the data on both sides of the seam to reduce its effect. Such an approach is visually appealing on a map, but does not effectively deal with the problem.

Alternative approaches to using remote sensing imagery in creating the LANDFIRE data layers may reduce the intensity or extent of seams. An alternative approach, which avoids seamlines by conducting the mapping and analysis one strip (satellite image) at a time, rather than one map zone at a time, is being used to generate LANDFIRE maps in Alaska. This approach requires that field data be well-distributed across the area, because sufficient field data must exist within each image, not just the map zone. Assuming such data exist, this approach may work well to avoid seamlines and improve accuracy.

Seamlines in LANDFIRE data primarily affect project-level analyses, but regional- and national-level analyses may also be affected. Even a national analysis like FPA is broken into smaller units (FPUs and FMUs) for analysis and comparison. If the analysis unit is small compared to the map zone or satellite imagery, then the potential exists for the data discontinuities to affect results. The larger the analysis area, the smaller the effect seamlines will have on the results.

[7] Systems for Environmental Management (SEM) is a private, nonprofit research and education foundation based in Missoula Montana. In conjunction with federal partners, SEM has developed a host of fuel, weather and fire behavior modeling software and procedures, which are available at www.fire.org.

CBD too low for crown fire in the FARSITE family

Users of the Mark Finney's[8] family of geospatial fire analysis programs (FARSITE, FlamMap, FSPro[9], FSIM[10]) have long noted that values of canopy bulk density (CBD) produced by treelist methods are too low to generate the expected amount of crown fire in their simulations. LANDFIRE has used a prototype of FuelCalc, which applies a treelist method for estimating CBD, to generate its CBD map, so the complaint has been extended to LANDFIRE CBD maps. A general rule-of-thumb was developed to cope with this apparent disconnect: double the LANDFIRE treelist-generated values for use in Finney's geospatial programs to achieve the expected results.

Early CBD mapping procedures (Selway-Bitterroot, Gila wilderness areas) were developed before any plot-level methods of estimating CBD had been developed. Instead of relying on observed CBD at plots, the early efforts instead populated CBD by working backward from expected fire behavior to determine the CBD values that produce that behavior using a given fire model. Given the lack of plot-level CBD observations available at the time, the approach was reasonable. Nonetheless, that procedure produces a value that is good only for the particular fire model used. In this case, the fire model used, FARISTE, produces fire behavior quite different than all others developed since then (except FARSITE's geospatial relatives).

Since those initial mapping efforts, our ability to estimate CBD has improved considerably, and is codified in FuelCalc[11], Fuels Management Analyst Plus (FMAplus[12]), and the Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS), all of which use a treelist approach. Such methods are based on decades of biomass research. The CBD algorithm in FuelCalc was conservatively designed to over-estimate rather than under-estimate CBD (by using the highest CBD found in any 11-ft layer of a canopy as the value for the whole plot, which is commonly more than twice the average bulk density). Comparison of predicted CBD with meticulously observed CBD (Scott and Reinhardt 2005) has generally verified the utility of the approach for estimating CBD in various stand structures. The values the treelist method produces fall squarely in the

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[8] Mark Finney is a research forester at the Missoula Fire Sciences Lab. Mark is the developer of a suite of geospatial fire modeling software tools, including FARSITE, FlamMap, FSPro, and FSim.
[9] FSPro is online software that simulates the likelihood of fire spread across a landscape by simulating fire growth under a large sample of possible future weather scenarios. FSPro is an integral component of the Wildland Fire Decision Support System (WFDSS).
[10] FSIM is prototype software that simulates the likelihood of fire growth and behavior across a landscape for a sample of possible weather conditions and for a sample of possible escaped-fire frequencies and locations. FSIM simulations are being considered for use in FPA, and are also used in prototype quantitative wildland fire hazard and risk assessments.
[11] A prototype version of FuelCalc designed by Elizabeth Reinhardt at the Missoula Fire Sciences Lab was used by the LANDFIRE fuel staff. A more complete version of FuelCalc is currently under development.
[12] FMAplus is commercially available software produced by Don Carlton of Fire Program Solutions LLC, available at www.fireps.com.

range of values that Agee's (1996) analysis found would lead to crown fire. The treelist methods generate CBD values work well in all fire modeling software programs except Finney's geospatial family, including BehavePlus[13], FMAplus and NEXUS[14]. FlamMap and FARSITE offer users a choice of crown fire modeling methods to use: the original "Finney (1998)" method, and a method similar to that used in NEXUS, which is labeled "Scott and Reinhardt (2001)" in those programs (figure 3).

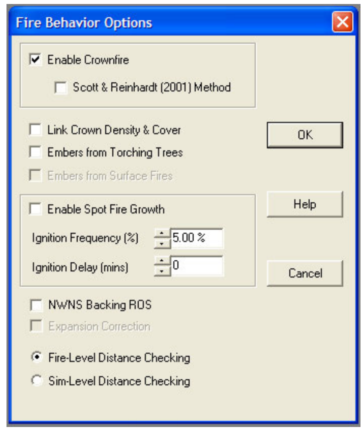

Figure 3--The *Model | Fire Behavior Options* dialog box in FARSITE, showing the checkbox that allows calculation of crown fire similar to the method described in Scott and Reinhardt (2001).

Users have generally found that using LANDFIRE or other treelist-generated CBD data with the crown fire option set to "Scott and Reinhardt" produces very reasonable results for crown fire occurrence, but not when using the "Finney 1998" default setting.

Scott (2006) suggests that the significant difference in fire model outputs (fire type, crowning index, etc.) between Finney's geospatial fire models and the others can be attributed to an error in modeling logic made initially in the Canadian Forest Fire Behavior System and subsequently used in Finney's programs. The error in modeling logic had little practical effect as implemented in the Canadian prediction system, so it went unnoticed; the same logic error when implemented in the U. S. system, however, has led to great differences in predicted fire behavior. See Scott (2006) for a detailed discussion of this topic.

The problem that LANDFIRE-generated CBD may be too low for use in Finney's geospatial fire models is best addressed by the fuel and fire modeling community, not by LANDFIRE. For users who wish to use those programs in their default setting (or those using FSPRO and FSIM, which do not yet have an option to use the Scott and Reinhardt 2001 method), the current rule of thumb may be appropriate. Otherwise,

[13] BehavePlus is software that allows simulation of fire behavior and effects for a specific point in space and time. Available at www.firemodels.org
[14] NEXUS is software that allows simulation of crown fire potential for a specific point in space and time. Available at www.fire.org

many users report that using the “Scott and Reinhardt” switch with LANDFIRE CBD maps produces acceptable results.

CBH too high for crown fire

At first glance this issue appears similar to the above issue with CBD, but in reality it is much more difficult to address. Unlike CBD, CBH is difficult to define in such a way that it can be measured in the field or estimated from a treelist. Moreover, CBH is not strongly correlated with other stand characteristics, making it difficult to produce reliable maps using the LANDFIRE approach (or any mapping approach, for that matter). For example, within any given forest type, CBH can be low in areas with low canopy cover, because there may have been little self-pruning in such a low-density stand, or CBH can be high if the stand has low cover because it was thinned. The LANDFIRE procedure can only broadly distinguish those cases.

Such difficulties led to the development of an alternative method of estimating CBH based on expert opinion. (Note that this is a similar approach taken for the Selway-Bitterroot and Gila mapping projects when faced with a lack of available CBD data.) The fuel and fire behavior experts did not offer their opinion of CBH directly, but instead were asked to identify the weather conditions that typically lead to torching (because CBH is used to predict when torching will occur). From that information, along with the fuel model and canopy cover already assigned, the CBH that leads to torching is then identified by working backward through the crown fire initiation model. This expert opinion CBH therefore depends on the fuel model and canopy cover for the area, as well as the weather conditions identified by the experts. Any errors in mapping of those layers, and any changes or adjustments made by users to those layers invalidate this CBH estimate—transition to torching would no longer take place at the identified threshold.

Fortunately, unlike with CBD, all point-based and geospatial fire models, regardless of developer, use CBH in the same way, so estimates of CBH made this way are valid in all U.S. fire modeling programs.

The difficulties with estimating CBH to simulate transition to crown fire cannot be resolved by LANDFIRE. The fire modeling community may need to find a different approach that is more amenable to mapping and less dependent on surface fire behavior (see Cruz and others 2004).

In most fire modeling systems, especially in Finney's geospatial models, the downside of conservatively estimating a low CBH is small compared to the downside of estimating a CBH that is too high. Until fire modeling uses a different approach, a stop-gap measure that LANDFIRE could employ is to modify the FuelCalc procedure for estimating CBH to identify the height of the lowest biomass of any density. Responsibility for this task lies not with LANDFIRE but with the fuel and fire behavior modeling community.

Treelist data sources

LANDFIRE has gathered treelist data from a variety of sources that use a variety of inventory methods. Two tree inventory methods are generally used: fixed-area plots and variable-radius plots (some tree inventories combine both methods). The treelist-based calculation methods used by LANDFIRE in FuelCalc are designed to be used with fixed-area plots of approximately 0.1 ac in size. The developers of that method felt that variable-radius plot may not adequately represent stand structure of a plot because it emphasizes sampling of large trees at the expense of small trees. For canopy fuel estimation, the contribution of a large number of small trees can be much more important than a small number of large trees, so it is important to have as much information as possible for those trees. Moreover, the trees sampled at a variable radius plot can be very far apart from each other, so their individual crown characteristics may not necessarily reflect growing conditions near the plot center.

Nonetheless, a large amount of treelist data available to LANDFIRE is of the variable-radius or hybrid plot type. The magnitude of potential problems with using variable-radius plots is unknown. In theory, the CBD predicted for a variable radius plot is probably slightly lower than if a fixed-area plot had been established at the same location, but this is impossible to know without research comparing the two approaches at the same plot.

For this report, a comparison of fixed-radius and variable-radius plot types was conducted using a dataset for a single even-aged ponderosa pine/Douglas-fir stand in western Montana. The dataset consisted of a complete list of tree attributes, including (X,Y) coordinates, of every tree on a square, 100 x 100 m (1-ha) plot. (The plot was established in 2006 by Elizabeth Reinhardt[15] to eventually test the use of upward-looking LIDAR for estimating canopy fuel characteristics.) From this complete dataset we established four virtual sample points within the megaplot, each located 25 meters from the edge. At each of these sample points we identified which trees would be counted in fixed- and variable-radius plots of different sizes. We then computed the average canopy fuel characteristic across the four sample points for each plot size. The results are summarized below. The results for a one-tenth-acre fixed-radius plot are shown in bold for emphasis. Plot sizes are listed in descending order of “size”; plots at the top sample a larger number of trees than plots at the bottom.

[15] Elizabeth Reinhardt is a research Forester at the Missoula Fire Sciences Lab. Elizabeth has led or participated in the development of several fuel, fire behavior and fire effects modeling systems, including FOFEM, FFE-FVS, NEXUS, and FuelCalc.

Table 1 -- Mean canopy characteristics (n = 4) for various plot types and plot sizes. The highlighted row indicates the plot type and size recommended by the developers of FuelCalc.

Fixed-radius Plots							
Plot Id	CBD (kg/m3)	CFL (t/ac)	CBH (ft)	SH (ft)	CC (percent)	Basal Area (ft2/ac)	Trees Per Acre
0.50 ac	0.054	3.0	22	86	38	102	278
0.25 ac	0.058	2.9	23	85	38	110	239
0.20 ac	0.058	2.8	23	85	38	105	253
0.10 ac	**0.065**	**3.1**	**22**	**83**	**39**	**109**	**298**
0.05 ac	0.086	4.0	29	85	46	143	310
0.02 ac	0.099	4.2	42	85	53	173	183
0.01 ac	0.148	6.2	38	72	58	239	233
Variable-radius Plots							
	CBD (kg/m3)	CFL (t/ac)	CBH (ft)	SH (ft)	CC (percent)	Basal Area (ft2/ac)	Trees Per Acre
BAF10	0.065	2.9	39	87	37	115	110
BAF20	0.088	3.7	38	84	45	135	151
BAF30	0.104	4.3	37	85	49	157	150
BAF40	0.110	5.0	37	86	56	190	207
BAF50	0.109	4.8	38	85	51	175	170
BAF60	0.130	5.7	38	86	57	210	204

Plot size appears to matter significantly for the fixed-radius plots—CBD ranged from 0.054 kg/m3 for the half-acre plots to 0.148 kg/m3 for the hundredth-acre plots, a factor of three difference (for the very same plot centers). These averages mask the increasing variability as plot size decreased—CBD at the four half-acre plots ranged from 0.046 to 0.065 kg/m3, whereas the hundredth-acre plots ranged from 0.000 to 0.366 kg/m3. This situation resulted in increasing CBD values with decreasing plot size, but that is unlikely to be a universal truth. In fact, one of the four plots was located such that many trees were found on the hundredth-acre plot, whereas the others had few or none.

The variable radius plots did not tend to underestimate CBD compared to the fixed-radius plot, an unexpected result. In fact, the BAF10 (variable-radius plot with 10-factor prism), a common BAF used in vegetation sampling, produced an estimate of CBD similar to the tenth-acre plot. In fact, larger BAFs, which sample fewer trees but puts more weight on each, tended to increase the estimated CBD. While very encouraging, this result applies to this one even-aged stand; a similar result may not be found for more complex fuel structures.

Canopy fuel load, canopy cover, and stand height estimated from variable-radius plots was also similar to the fixed-radius plots.

Canopy base height estimates differed significantly between the fixed-radius plots and the variable-radius plots, which predicted much greater CBH. This is likely due to the fact that the variable-radius plots do not adequately sample small trees, so tend to under-predict biomass in the lower part of the canopy. This effect would be even greater in more complex stand structures than present in this analysis. A hybrid plot with both variable- and fixed-radius plot elements could mitigate this effect.

Finally, the variable-radius plots underestimated tree stem density relative to the fixed-radius plots, again due to the under-sampling of small trees.

In summary, this analysis supports the conclusion that variable-radius plots under-sample small trees. That is, in fact, the purpose of that plot design. For this even-aged stand, the under-sampling of small trees led to underestimation of CBH, but not CBD, CFL, or SH. Only a more exhaustive analysis with other stand structures will confirm or refute this result.

Canopy fuel calculation programs

LANDFIRE used a customized-prototype version of FuelCalc coded by Larry Gangi[16] to estimate CBD and CBH. Other fuel analysts have used the Fire and Fuels Extension to FVS or Fuels Management Analyst Plus for making the same estimates. The canopy fuel calculations in FuelCalc and FFE-FVS[17] were designed by Elizabeth Reinhardt, and FMAplus was also patterned after those programs. All three programs use the same general approach to estimating CBH and CBD, but there are slight differences in the equations used in each tool, and slight differences in certain parameters and internal models. For example, the user has control over whether any of the biomass of broadleaf tree species is factored into the CBD and CBH estimates. In theory, differences in output generated from these three programs should be small.

As a quick test of this assumption, Charley Martin's dataset of 700 FIREMON[18] plots was run through both FuelCalc and FMAplus. The resulting differences between the programs were larger than expected—FMAplus consistently over-predicted relative to FuelCalc (Figure 4).

[16] Larry Gangi is a computer programmer with Systems for Environmental Management. Larry has also served as software developer for the FOFEM and FireMon software tools.
[17] FFE-FVS is software to simulate vegetation growth and quantify fuel characteristics over time. Available at www.fs.fed.us/fmsc/fvs/description/ffe-fvs.shtml.
[18] FIREMON is software to catalog and monitor fuel and vegetation characteristics. Available at www.fire.org

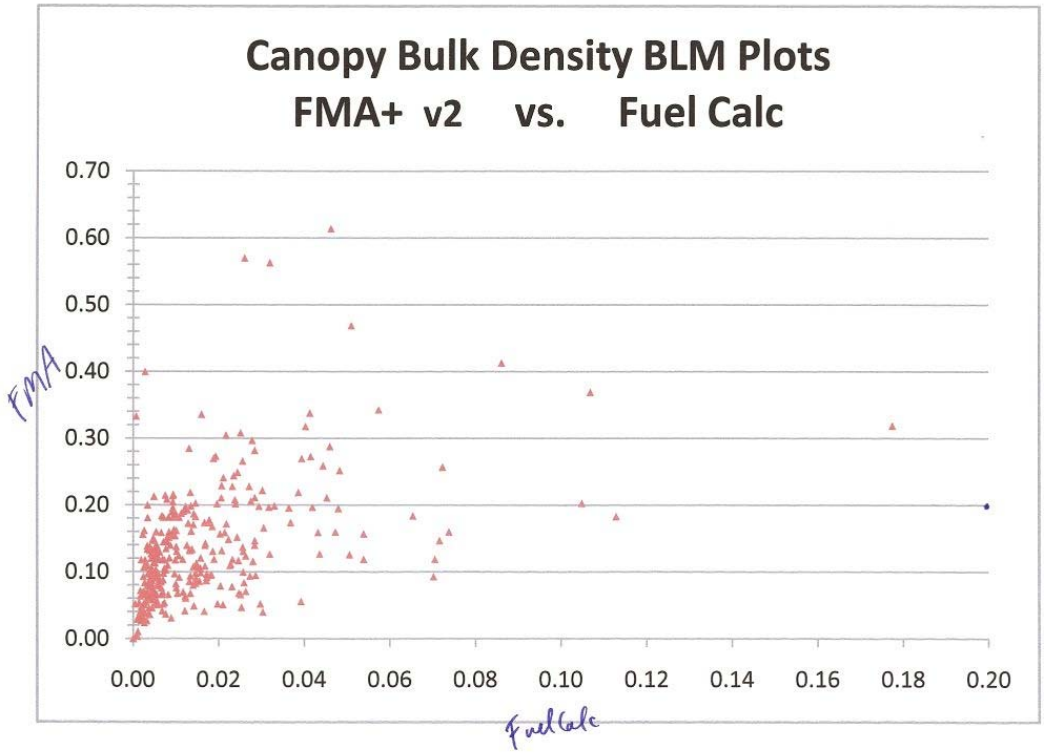

Figure 4 -- Predicted CBD (kg/m3) for FMAplus (Y-axis) and FuelCalc (X-axis). FMAplus overpredicts relative to FuelCalc, but it is not possible to know which is closest to "observed".

It is impossible at this point to know which is more accurate or reliable, but the FMAplus CBD values did not seem unreasonably high. (I don't have enough experience with the vegetation and fire behavior in the study area to confirm that conclusion, though.) It is possible that FMAplus is over-emphasizing the contribution of broadleaf species to canopy bulk density, or that FuelCalc is under-emphasizing those species.

I have forwarded this finding to Elizabeth Reinhardt, lead developer of FuelCalc, for further investigation. At this point I surmise that the FuelCalc-FMAplus comparisons were not apples-to-apples; user settings controlling different aspects of the calculation may not have been equal. FuelCalc remains the standard government application for quantifying canopy bulk density; LANDFIRE can rely on its output in mapping efforts.

The fuel and fire behavior modeling community should investigate this issue by thoroughly analyzing the outputs from a common set of treelist inputs for a variety of calculation tools. Any differences in output should be explained, and recommendations for resolving differences among the various programs should be provided.

Refreshing and calibrating LANDFIRE data

Two important limitations result from LANDFIRE's national extent: early date of validity (*ca*. 1999), and poor project-level accuracy for some fire planning applications. Refreshing data to the current year is a critical task before applying LANDFIRE's spatial data for any analysis, whether national-, regional- or project-level in extent. Improving local-level accuracy is important for project-level planning, but is not required (and may in fact hinder) regional- and national-level analyses by mixing adjusted and unadjusted data. LANDFIRE and others are addressing these issues by publishing procedures for calibrating and adjusting LANDFIRE data using a variety of *ad hoc* software tools.

To address the first limitation, LANDFIRE has developed a data-refresh plan to reflect landscape changes due to fire biennially. To jump-start the process, NIFTT[19] has conducted and nearly completed a Rapid Refresh of LANDFIRE data—a first-cut refreshing of LANDFIRE data to reflect landscape changes between 1999 and 2007. The products of this effort are expected to be replaced by a more thorough refreshing on a two-year cycle. In addition, the entire LANDFIRE mapping process will be repeated on a 10-year cycle. This procedure should ensure that high quality, up-to-date landscape data is always available. See the LANDFIRE Operations and Maintenance Business Case and Plan at http://www.landfire.gov/documents_updatedprod.php for more information.

Two separate efforts are underway to address the adjustment of LANDFIRE data to meet the needs of project-level analysis. One effort, co-funded by FPA and the National Park Service, is being carried out by Rick Stratton of Systems for Environmental Management. The product of that effort will be a document describing a process for critiquing and adjusting LANDFIRE data. A draft of this document will be available soon.

Second, NIFTT is continuing development and training of software tools and developing a training package designed to help users to download and prepare LANDFIRE. Two tutorials are available, and a course is being developed.

The DataPrep tutorial shows users how to prepare LANDFIRE data for use in NIFTT tools. This tutorial does not address adjustment or calibration of spatial data; it simply instructs users on how to download, clip, and re-project LANDFIRE data for use in a project-level analysis.

The LANDFIRE Data Access Tool tutorial describes the use of this tool for obtaining LANDFIRE data. This tutorial also does not address calibration and adjustment of the data itself.

[19] NIFTT is the National Interagency Fuel Technology Transfer team, co-funded by LANDFIRE and the National Interagency Fuel Coordination Group.

Finally, a course titled “GIS Tools for Wildland Fire and Fuels Planning” is under development. The course will teach students to download and edit LANDFIRE data for use in NIFTT’s GIS tools.

The combination of the NIFTT courses and tutorials and Stratton’s NPS/FPA-funded process for critiquing and editing LANDFIRE data should be enough guidance for most users.

Discussion

At times during their development process, LANDFIRE faced the choice of producing data that was consistent with biological science (for example, producing CBD values based on methods derived from the biomass literature) or producing data specifically adjusted so that could be consumed by a fire behavior modeling tool (CBD values manipulated so they work better in FARSITE). The LANDFIRE philosophy for the current effort was to base all data maps on the best available biological science, knowing that adjustment would be required for certain models. This is the only scientifically supportable approach. Should LANDFIRE's best biological estimate of a certain quantity end up not working well in a fire model, a quick investigation would indicate whether the problem was with the data, with the model, or with the fire modeling science. LANDFIRE should take steps to adjust any data layers it produces that are not consistent with scientifically valid field data, as they did for canopy cover values. In other cases, the fuel and fire modeling community may need to make accommodations in their fire models for the biologically estimated data.

Although users may need to critique and calibrate LANDFIRE data for use in project-level analysis, the goal of producing a nationally-consistent dataset is met without such effort. The scope of a critique and editing effort should be tied to the extent of analysis to be conducted. A national-level analysis would require a nationally consistent critique and calibration effort – LANDFIRE has already accomplished this task. A mid-scale analysis (state or region, for example) should have a critique and calibrate effort at the same scale, or none at all – mixing base LANDFIRE data for some areas with critiqued and calibrated data for others may lead to spurious results.

LANDFIRE's success at producing biologically based fuel and vegetation maps has created a situation where fire modeling difficulties can be addressed by the fire modeling community. Before LANDFIRE, without consistently created maps, fire modeling errors were always attributed to problems with the data, with no consideration for problems with the model. Geospatial fire modeling systems have been developed with a very rigid fire behavior model—no way to accommodate model error. (FARSITE has model-side spread rate adjustment factors, but other geospatial fire modeling tools do not). As a result, calibration of the fire model has always focused on changing the underlying data. When based on reliable fuel maps and weather data, many FARSITE simulations under-predict fire growth and behavior. The approach to improve simulation accuracy has been to adjust the data: reduce canopy base height, increase canopy bulk density, increase wind speed, etc. Unless there is specific evidence of a data accuracy problem, adjusting the data to suit the model is not the best approach to calibration. Instead, the fire modeling community should focus on adjusting parameters in the fire model itself. Few such adjustment factors currently exist in geospatial fire models, especially the emerging FSPro and FSIM.

Conclusion

LANDFIRE has done an admirable job integrating emerging fuel and fire modeling technologies into their mapping efforts. Given the large extent of the project, high inherent spatial variability of the characteristics being mapped, emerging (and sometimes contradictory) nature of the fuel and fire modeling technologies involved, and time constraints, no better map products could have been produced. More accurate, seamless maps can be produced at greater cost and smaller scale than required by LANDFIRE's mission. Most remaining problems with LANDFIRE data for local-level projects can be addressed through a process of calibration and adjustment. Both FPA and LANDFIRE are funding the development of procedures for accomplishing that task.

Two significant problems can potentially be addressed by LANDFIRE. First, LANDFIRE can explore whether the new canopy cover estimation techniques developed for recently placed FIA plots can be used to generate a LANDFIRE-produced forest canopy cover map to replace the inherited NLCD maps. The adjustment of this map will significantly improve fire modeling by facilitating better estimates of wind adjustment and dead fuel moisture, both of which depend on forest canopy cover. Second, in an effort to reduce data discontinuities caused by seamlines, LANDFIRE can consider strip-based mapping for any future efforts (as opposed to the present zone-base). LANDFIRE mapping for Alaska is already planning to use strip-based approach, and will serve as a good test of that approach.

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Review

Management for Mountain Pine Beetle Outbreak Suppression: Does Relevant Science Support Current Policy?

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Abstract: While the use of timber harvests is generally accepted as an effective approach to controlling bark beetles during outbreaks, in reality there has been a dearth of monitoring to assess outcomes, and failures are often not reported. Additionally, few studies have focused on how these treatments affect forest structure and function over the long term, or our forests' ability to adapt to climate change. Despite this, there is a widespread belief in the policy arena that timber harvesting is an effective and necessary tool to address beetle infestations. That belief has led to numerous proposals for, and enactment of, significant changes in federal environmental laws to encourage more timber harvests for beetle control. In this review, we use mountain pine beetle as an exemplar to critically evaluate the state of science behind the use of timber harvest treatments for bark beetle suppression during outbreaks. It is our hope that this review will stimulate research to fill important gaps and to help guide the development of policy and management firmly based in science, and thus, more likely to aid in forest conservation, reduce financial waste, and bolster public trust in public agency decision-making and practice.

Keywords: bark beetle; clearcut; climate change; climate change adaptation; daylighting; *Dendroctonus ponderosae*; forest pest management; monitoring; sanitation; thinning

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1. Introduction

Insect outbreaks are increasing in size and severity on a global scale [1]. In North America alone, three massive insect outbreaks occurred within the last two decades, all involving native bark beetles in conifers [2]. Of these, the mountain pine beetle (*Dendroctonus ponderosae*) outbreak is an order of magnitude larger than any previously recorded. A variety of factors, natural and anthropogenic, converged to result in these dramatic events [2]. Each outbreak has not only had severe ecological effects, but each has also triggered human responses that, for better or for worse, have resulted in additional impacts along with massive expense [3]. Predictions are that outbreaks of bark beetles will become more frequent and severe in the future [4,5] indicating an imperative need to critically assess the efficacy and impacts of our approaches to their management.

Outbreaks of bark beetles are not new. They have been occurring for millennia and have played a major role in shaping coniferous forest ecosystems of the world. While considerable research has been conducted on controlling bark beetles, massive gaps in knowledge remain. In particular, there is a disturbing dearth of rigorous replicated empirical studies assessing the effects of various management strategies, particularly timber harvest treatments, for bark beetle outbreak suppression. Even fewer studies have focused on how such treatments meet explicit goals or affect forest structure, function and future outbreak dynamics [6]. Particularly pertinent at this time, there is a lack of information to address forest adaptation to climate change in light of increasingly "out of historic norm" behavior of bark beetles. Despite this, there is a widespread belief in the policy arena that timber harvesting is an effective and necessary tool to address beetle infestations. That belief has led to proposals for, and enactment of, significant changes in federal environmental laws to encourage more timber harvests. Our question is, does that belief have a sound grounding in current science?

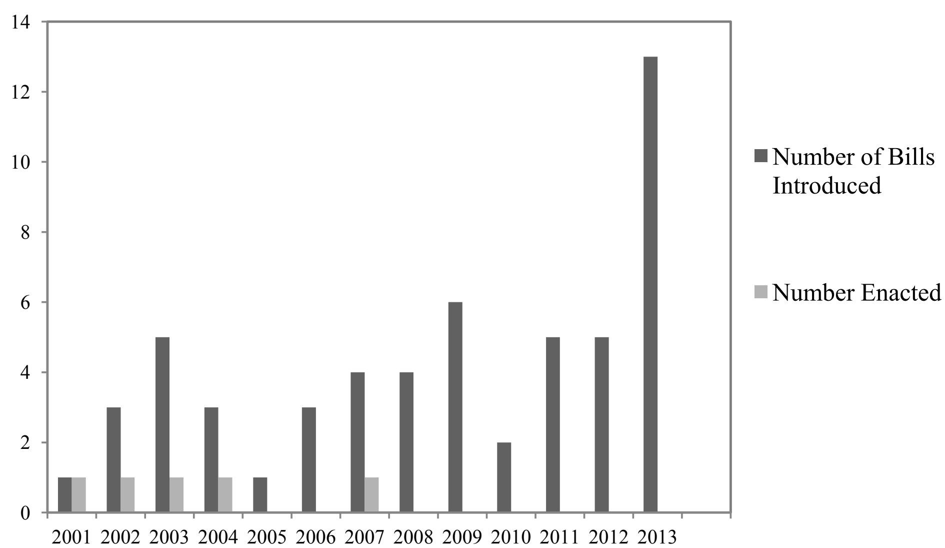
In this review, we focus on mountain pine beetle as an exemplar to critically evaluate the state of science behind the use of timber harvest treatments for bark beetle suppression during outbreaks. The mountain pine beetle was chosen because it is the most studied, most intensively managed, and most aggressive of the irruptive bark beetles. It has also responded strongly to climate change, resulting in a recent massive outbreak of unprecedented size that, in turn, has initiated numerous human responses, mostly involving implementation of timber harvests. It has also initiated many policy changes with many more currently in the pipeline.

We begin with an overview of the current policy situation. We then briefly review the biology of mountain pine beetle to form a foundation for understanding the factors that initiate and maintain outbreaks and how anthropogenic factors are contributing to current problems. We then describe the primary timber harvest treatments used to suppress bark beetle outbreaks and examine how well relevant science and ecological principles support their use. We conclude with a discussion on how well policy reflects the actual state of current science and identify where significant gaps between science and practice occur particularly in light of climate change. We also discuss the need to use advanced tools, including genetics and remote sensing, to adapt old practices to new situations-particularly in the realm of climate change adaptation. It is our hope that this review will stimulate research to fill important gaps and to help guide the development of policy and management firmly based in science, and thus, more likely to aid in forest conservation, reduce financial waste, and bolster public trust in public agency decision-making and practice.

2. The Current Policy Situation

There have been many recent proposals to streamline, reduce, or eliminate perceived legal obstacles to implementing timber harvests to address beetle epidemics on federal public lands (Figure 1). Between the 107th Congress (January 2001) and the 113th Congress (present), we found 55 bills that were introduced where at least one goal of the legislation was to increase timber harvests in order to respond to beetle infestations (Figure 1). Most of these proposals focused on the US Forest Service, which manages the majority of forests on federal public lands.

Figure 1. Number of bills involving timber sales that included bark beetle control that were introduced and/or enacted from 2001 to 10 July 2013.



Some of these proposals have been enacted. By far, the most important legal change has been the Healthy Forest Restoration Act of 2003 (HFRA). HFRA reduced the level of environmental analysis required for certain timber projects under the National Environmental Policy Act (NEPA), specifically by limiting the number of alternatives that the Forest Service was required to analyze. It also significantly restricted the ability of members of the public to challenge certain timber projects in court (by making participation in the agency's administrative process a precondition for filing suit). Further, it sought to streamline the Forest Service's internal administrative process for considering citizen challenges to certain timber projects. HFRA applies nationally to all National Forest System and Bureau of Land Management lands, and has resulted in forest treatment projects on an average of 220,000 acres of federal land per year since its enactment [7]

HFRA authorizes this streamlined process for timber projects on "Federal land on which…the existence of an epidemic of disease or insects, or the presence of such an epidemic on immediately adjacent land and the imminent risk it will spread, poses a significant threat to an ecosystem component, or forest or rangeland resource, on the Federal land or adjacent non-Federal land" [8,9].Moreover, while other types of HFRA projects in old growth forests are subject to limitations intended to protect old growth structure and large trees, timber projects to address insect epidemics can occur in old growth forests without those limitations [10,11].

HFRA also sets up a special experimental management process to develop better management methods for beetle infestations. After a long list of findings by Congress about the risks of beetle infestations in US forests, Congress authorized up to 250,000 acres of "applied silvicultural assessment and research treatments" on National Forests that would be categorically excluded from NEPA; these treatments could include timber harvesting [12,13]. HFRA section 401(b)(3) [14] requires that these applied silvicultural assessments and treatments must be peer reviewed by non-agency scientists.

HFRA is not alone. Another enacted bill created exemptions from environmental laws to allow timber harvest projects in a geographically limited area. As part of a massive supplemental appropriations act to address recovery from the September 11, 2001 terrorist attacks, Congress exempted a series of timber harvest projects in the Black Hills of South Dakota from any and all environmental laws; the law specifically stated that the projects were intended to reduce both fire risk and beetle infestations [15].

Other recent enactments create additional incentives for timber harvests intended to address beetle infestations. Congress permitted state forestry agencies to perform beetle control timber harvest projects on federal lands in Colorado and Utah under what is called "Good Neighbor Authority" [16]. These state forestry agencies must also implement "similar and complementary" services on state land adjacent to federal land in order to use the authority. Additionally, in the 2008 Farm Bill, Congress expanded subsidies for the production of "renewable biomass" energy to include timber produced from projects intended to reduce or contain disease or insect infestation [17].

There have been many more recent proposals for additional changes. Congress has considered multiple bills to expand the scope of HFRA. One proposal would require the Forest Service to implement at least one insect and disease control pilot project in at least one subwatershed in every national forest in a state that is "subject" to an insect or disease epidemic [18–24].Congress has also considered many other changes to encourage timber harvesting to control beetle infestations besides expanding HFRA. Some proposals would expand the exemptions to the Forest Service's Roadless Rule (which prohibits commercial timber projects and road construction in unroaded areas of National Forests) in order to allow more timber projects that are intended to address beetle infestations; some of these projects would be exempt from judicial review [25–27].

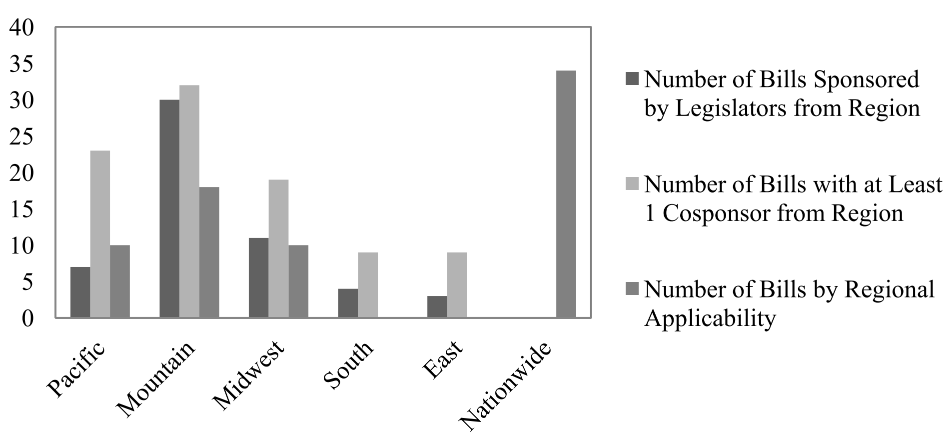
Congress has considered giving additional benefits under the Clean Air Act for "renewable biomass" produced from timber projects on federal lands, including projects intended to control beetle infestations [28,29], giving grants and other subsidies for beetle control timber projects [30], extending the Good Neighbor Authority to more states [31–33], and reducing or eliminating the fee that private timber contractors pay for timber contracts in exchange for agreements to implement restoration work, such as culvert removals, road improvements, or invasive weed removal, if the project provides insect control and other forest management benefits [26]. Finally, two bills have proposed that designation of additional federal lands as protected wilderness be paired with exemptions of beetle-related timber projects from environmental laws [34,35].

Throughout this policy debate, members of Congress and major stakeholders have regularly stated that timber harvest on federal lands is a necessary component of efforts to fight beetle infestations and control outbreaks and that additional flexibility under environmental laws is necessary for agencies to pursue these timber harvest projects [36–41].

Likewise, the U.S. Forest Service and other U.S. federal land management agencies have prescribed timber harvests as a necessary component of beetle control. For example, the Forest Service's Western Bark Beetle Strategy calls for the agency to "reduce the number of trees per acre and create more diverse stand structures to minimize extensive epidemic bark beetle areas" by using thinning and other harvest treatments [42]. While the Forest Service has applauded HFRA as "very helpful" in addressing beetle outbreaks (U.S. Forest Service, Review of the Forest Service Response: The Bark Beetle Outbreak in Northern Colorado and Southern Wyoming, September 2011), available at [43], agency leaders do not look favorably upon all legislative proposals to weaken environmental laws to facilitate timber harvest for beetle control. For example, Tom Tidwell, Chief of the Forest Service, criticized recent bipartisan legislation [25] because it would "shortchange the environmental review process, cut out public engagement and collaboration…and override roadless protections." (Testimony from House Subcommittee on Public Lands and Environmental Regulation Legislative Hearing on H.R. __, H.R. 1294, H.R. 818, H.R. 1345, H.R. __, and H.R. 1442 available at [44].

Given the geographic concentration of federal public lands in the West, most of the bills have a specific focus on western states, and were introduced or supported by westerners (Figure 2). But that is not universally the case. Two of the proposals to expand the scope of HFRA were sponsored by Representative Markey, a Democrat from Massachusetts [19,23]. Moreover, support for these bills is bipartisan, showing that the belief that timber harvest can address beetle infestations crosses the political spectrum. Of the 55 total bills, 17 were sponsored by Democrats alone, 21 sponsored by Republicans alone, and 17 had bipartisan sponsors. Markey himself has received very high ratings from the League of Conservation Voters, with a 94% lifetime score from the group.

Figure 2. Bill sponsorship, co-sponsorship, and applicability by region. (Pacific = CA, OR, W, AK, HI; mountain states = MT, ID, NV, WY, UT, CO, AZ, NM; Midwest = ND, SD, NE, KS, MN, IA, MO, WI, IL, IN, MI, OH; SOUTH = TX, OK, AR, LA, KY, TN, MS, AL, GA, FL, SC, NC, VA, WV; east = ME, NH, VT, MA, NY, RI, CT, NJ, DE, MD, PA).



The 55 bills introduced since 2001 show that many legislators, particularly those from western states, believe that timber harvests are a necessary tool to address beetle infestations. This belief has led to the enactment of laws that reduce compliance burdens under NEPA and other federal environmental laws. There are many more proposals for additional significant changes to federal environmental laws to encourage more timber harvests for beetle control. While "there is certainly a tremendous amount of social and political pressure to 'do something' about beetles," there is also growing concern by many that timber harvests for beetle control are expensive and ineffective and that long-term impacts on forests are unknown [42 citing Ann Merwin, director of policy and government affairs for the Wilderness Society]. The policy debate demonstrates the need to critically examine how well these treatments work and place policy in the context of the best available science.

3. A Mountain Pine Beetle Primer

The mountain pine beetle is native to pine forests in western North America [45]. During outbreaks, it can kill millions of trees across extensive areas. The ability to cause such widespread mortality has led it to be described as the most destructive forest pest on the continent [46]. Indeed, economic and aesthetic impacts of outbreaks can be severe. From a manager's perspective, outbreaks are often perceived as a symptom of poor "forest health", while ecologists more often view outbreaks as natural ecological processes integral to the maintenance and resilience of the forest. These differing human perceptions have led to conflicting and ambiguous management goals as well as scientific, social, and political conflict.

The mountain pine beetle is polyphagous on pines (*Pinus*) [45]. It attacks not only native pines but also exotic pines used in ornamental landscaping. Within the natural range of the beetle, only *P. jeffreyi* appears to be avoided, likely due to its unusual chemistry [45]. Pines are well defended and are not easy targets for the beetle. They produce constitutive defenses consisting of resin that can flush the tiny beetles from trees, often drowning them [47–49]. Pines also produce induced defenses in the phloem comprised of resin containing elevated concentrations of toxic monoterpenes [49,50]. Induced defenses develop in response to attack, and thus, involve a lag time of one or more days to develop and can last for a month or more even when trees are killed [51].

To contend with a defensive host, the mountain pine beetle has evolved a complex chemical communication system it uses to coordinate a mass attack on a tree [52]. A female beetle will land, begin to tunnel, and release an aggregation pheromone that attracts conspecifics of both sexes to the tree. Subsequent arrivals release additional pheromone increasing attraction to the tree [53]. If enough beetles respond, the tree can be overwhelmed in just a few days. As defenses are depleted, the beetles release an anti-aggregation pheromone which repels late arriving beetles and acts to reduce intra-specific competition among brood [53]. At this point, the tree has reached "a point of no return" [54]. It will not recover and will slowly die, although it may remain green for nine months or more due to translocation of water to needles by capillary action in the xylem.

The number of beetles needed to kill a tree varies and depends, in part, on the strength of its defenses [55]. In general, as the strength of defenses increase so does the number of beetles needed. Several factors influence the strength of tree defenses. Trees weakened by drought, disease or damage can be overwhelmed by only a few hundred beetles while very vigorous trees may require many hundreds or even thousands [56]. Genetics of the host tree also play an important role. Within a tree species, different genotypes result in differing levels of resistance and susceptibility [57,58]. Genetic differences are even more pronounced when considering differences in defenses among *Pinus* species [59,60].

The ability of tree defenses to affect mountain pine beetle success varies by whether the beetle is in endemic (non-outbreak), incipient (building) and eruptive (outbreak) phases. During the endemic phase, when beetle populations are low, host tree defenses are the major constraint in the ability of beetles to kill trees. However, tree defenses become inconsequential once the threshold to the incipient stage has been surpassed [61]. When numbers are low, beetles attack smaller diameter trees with low defenses. However, once populations rise to the incipient stage, beetles choose larger, healthier, resource-rich trees, despite their superior defenses [61]. Because larger trees have thicker phloem resources to support larval development, they support greater beetle productivity which results in positive feedback that helps fuel the expansion of the outbreak. Thus, host tree traits (primarily host defenses and diameter class) that determine which trees are killed when populations are low, may be unimportant or even have an opposing effect on beetle success when populations are high [61].

It is often reported in the press that mountain pine beetle populations are cyclical. This is not the case. The population dynamics of insects that develop cyclical outbreaks are typically dominated by *delayed* negative density dependent feedback involving regulation by natural enemies and induced resistance mechanisms [62]. This type of feedback results in predictable intervals (cycles) between outbreaks although the amplitude of population peaks can vary due to spatiotemporal variation in abiotic conditions. Bark beetle dynamics, instead, are driven by alternations of negative density dependent and positive density dependent feedbacks resulting in sporadic unpredictable population eruptions primarily driven by threshold effects and typically triggered by abiotic factors, particularly climate [61–63]. It is critical to distinguish between cyclical and eruptive population dynamics as insects exhibiting these two types of dynamics demand different management and monitoring approaches. In particular, eruptive dynamics are triggered by abiotic factors typically outside the realm of human manipulation.

Mountain pine beetle can remain in non-outbreak phase for very long periods of time, even when forests are composed of suitable age classes of host trees and in a condition often considered to be highly susceptible and "unhealthy". Outbreaks occur *only* when multiple thresholds involving temperature, tree defenses, and brood productivity are surpassed that allow positive feedbacks to amplify across several scales [2,64]. While outbreak development is complex, the primary elements that must exist are an abundance of suitable hosts *and* a trigger [63]. Triggers for mountain pine beetle that allow population amplification and subsequent widespread outbreak initiation are warm temperatures and drought, conditions that often co-occur [65]. There can also be a substantial lag period, even several years, from the initiation of the abiotic factors that trigger an outbreak to when populations actually amplify [65,66]. However, once a threshold number of beetles is surpassed, the outbreak becomes self-perpetuating.

While forest conditions alone do not cause outbreaks, certain forest conditions can support larger and more severe outbreaks once they are initiated. Mountain pine beetle attacks only pines (except in rare instances where it "bleeds over" into spruce) [67], and typically only those larger than ca. 15 cm in diameter [68]. Therefore, forests comprised mainly of large diameter pine can be at higher risk of widespread mortality when a trigger occurs than are forests comprised of young, small diameter pine or composed of a mix of tree species including non-pines [68]. Processes that homogenize forest structure and composition such as abnormally widespread stand replacement events (e.g., fires of 1910, Yellowstone 1988) or particular types of forest management (e.g., some timber harvest practices, fire suppression) that alter forest composition and structure over large areas, can contribute substantially to the extent and severity of an outbreak once it is initiated. Processes that result in heterogeneity, such as "normative" wildfires and bark beetle outbreaks, and some land management practices (e.g., restoration treatments focused on restoring a mosaic structure of forest stands of different age classes) tend to reduce outbreak severity and extent by reducing the amount of contiguous susceptible hosts [68].

Climate acts as a trigger for mountain pine beetle outbreaks for a very good reason. Like all insects, mountain pine beetle is poikilothermic-it cannot regulate its body temperature, and thus, all its metabolic rates and vital functions are dependent upon the temperature of its environment [69]. As temperatures rise, feeding, activity, development and reproductive rates increase. Importantly, this also means that the length of the mountain pine beetle life cycle is determined by temperature [69]. Under optimal thermal conditions, development is univoltine (one year). A univoltine cycle allows synchronized emergence of brood adults in mid-late summer, supporting not only mass attacks, but also attacks at a time that allows subsequent offspring to enter winter as cold-hardened larvae [70,71]. Cold hardening is a gradual process that occurs as temperatures fall in autumn. Once larvae are cold hardy it can take temperatures as low as −40 °C to kill significant numbers [72]. However, cold air incursions in fall when beetles are not yet cold hardened or in spring when larvae have lost cold hardening in preparation for transitioning to the adult stage can result in widespread mortality. This can halt an outbreak if subsequent conditions are no longer favorable for the beetle. However, if favorable conditions return, beetle populations rebuild. Importantly, outbreaks require a univoltine life cycle combined with moderate winter temperatures [73].

In areas where temperatures are too cool to support a univoltine life cycle, a semivoltine (longer than one year) life cycle occurs [73]. A semivoltine life cycle is maladaptive for the beetle in several ways. First, adaptive seasonality is disrupted, increasing the percentage of brood that enter winter in stages vulnerable to freezing (eggs, pupae and adults). Additionally, mortality increases when beetles must pass through two winters and feed on a food source increasingly depleted in moisture, nutrients, and symbiotic fungi [74]. Warm periods support not only greater brood production and survival in areas typically suitable for the beetle, but also allow a transition from a semivoltine to a univoltine life cycle in areas otherwise too cool. This increases the spatial extent of suitable habitat and tree mortality. Thus, abnormally warm periods can vastly increase the total area suitable for the beetle and play a major contribution to the synchronicity and coalescence of outbreaks across regions [2,65].

Drought can also play an important role in outbreak initiation. Host tree defense mechanisms are compromised during drought allowing beetles to more easily attack trees [2,75]. Tree defenses are major constraints when beetles are in non-outbreak phase. However, drought-weakened trees can support population amplification until a point where stand level densities surpass a critical threshold. Once this threshold is passed, tree defenses lose their importance in regulating beetle populations [61]. Very importantly, drought stresses large numbers of trees at a regional scale. This results in large numbers of trees that are easier for the beetles to kill, further supporting outbreak intensification [65,76].

Recent studies have found that drought occurring years or even decades before the outbreak can influence outbreak initiation. Furthermore, prolonged drought stress appears to pre-condition trees to be more susceptible, an effect that can continue for years after normal precipitation has returned [58,65,77]. There also appears to be a genetic component to tree sensitivity to drought, and subsequently, susceptibility to beetles. In two studies, one conducted in whitebark pine and the other in ponderosa pine, differences in growth of surviving trees and trees killed by beetles over the last century suggest that adaptive differences to changes in climate exist. In the whitebark pine study, the trees studied were co-dominants and not significantly different in diameter age or mean growth over their lifetimes [58]. However, trees that were killed exhibited faster rates of growth in the first half of the century suggesting they were better adapted to the cooler wetter conditions of that period. The surviving trees had greater growth in the latter half of the century when conditions were warmer and drier. Millar *et al.* [58]) suggested that the beetle-caused tree mortality in the stands they studied resulted in a strong natural selection event that removed trees less fit under our current climate while leaving those more well-suited.

Likewise, Knapp *et al.* [77] found genotypes of ponderosa pine that were slow-growing in the two to three decades prior to the outbreak were much more vulnerable to beetle infestation than those that were fast-growing, again suggesting the beetle may act as a selective agent shifting genetic structures in stands over time to those most suited to prevailing climatic conditions. In lodgepole pine, trees of similar age and diameter growing intermixed in the same stand and under the same conditions exhibited different levels of sapwood moisture that were highly correlated with susceptibility to beetle attack [74] hinting at genetic differences in water efficiency. Those with lower sapwood moisture were attacked and killed by the beetle while those with higher sapwood moisture were not [74].

While mountain pine beetle has developed outbreaks for millennia, the current outbreak is far outside the historic norm [2,78]. The unprecedented size and severity of this outbreak is due to a combination of increasingly favorable climate for the beetle and forest conditions. Warming trends have supported the development of a univoltine cycle in many areas that previously were too cool and have resulted in greater beetle productivity and survival [79]. This has led to massive tree mortality, not only in areas previously favorable for the beetle, but also in areas previously suboptimal or unusable. Warmer temperatures and high population levels have also supported expansions of the beetle's range hundreds of kilometers further north in British Columbia and eastward across Alberta [80–82]. In these new locations, the beetle is infesting naïve hosts including (in the eastern expansion) a novel species, jack pine [80,82]. These naïve hosts exhibit lower defenses to beetle attack [83] as well as similar chemical compositions to natural hosts [84] promoting establishment. Predictions are that the beetle will continue to move across the continent through the boreal forest and finally into eastern pine forests [78].

Warming has also allowed the beetle to move higher in elevation where it is devastating whitebark pine, a tree that is foundational to the western North American subalpine ecosystem and that was previously protected from the beetle by cold [73,85]. Movement into the subalpine has been supported by overall warmer temperatures and milder winters allowing the beetle to switch from a semivoltine to a univoltine life cycle while simultaneously reducing winter mortality [85–87]. The resulting mortality to whitebark pine in many areas, particularly the greater Yellowstone Ecosystem, has been so severe the tree is now proposed for listing as an endangered species [88]. The tree is already listed as an endangered species in Canada due to the combined effects of mountain pine beetle and white pine blister rust [89].

4. Mountain Pine Beetle Outbreak Suppression

Treatments used to mitigate the effects of mountain pine beetle are grouped into three broad categories. Treatments that strive to reduce or eliminate beetle populations are termed direct controls [90]. Treatments aimed at increasing tree vigor and altering stand conditions to be less favorable for beetles are called indirect controls [90,91]. Prophylactic treatments aim to protect high value individual trees or stands of trees from infestation. Salvage, while often included in beetle management programs does not actually reduce or impact beetle populations-it is the removal of dead trees for economic or other reasons and often involves removal of trees that are already ‘empty’ of beetles and thus has no impact on beetle population size. Because our focus is on how well science supports the use of timber harvests (including tree felling and destruction of trees in place) to reduce or suppress bark beetle outbreaks, we will focus primarily on direct and indirect controls concentrating on these treatments.

Direct control includes sanitation treatments such as removing single trees or small patches of trees that are infested with the insect, clearcutting (also called block harvesting) and prescribed burning of infested trees, as well as fell and burn, trap trees, debarking, and application of insecticides or toxins such as MSMA (monosodium methanearsonate). Sanitation cuts attempt to remove most or all beetles in an area by removing infested trees before the beetles developing within them can emerge and disperse [90,92]. Prescribed burns, fell and burn, debarking, and toxin applications attempt to destroy beetles in infested trees on-site. Trap trees are trees that are baited with attractant pheromone baits in an attempt to draw beetles into specific areas where they are concentrated into the baited trees which are subsequently taken to the mill or destroyed. Each of these methods relies on killing as many beetles as possible in order to lower beetle population thresholds below which they can maintain outbreak dynamics.

Indirect controls are primarily silvicultural in nature. The main treatment used for mountain pine beetle is thinning. Thinning is thought to act by reducing inter-tree competition for water, nutrients, and light, enhancing greater tree vigor, and thus defenses against the beetle [93]. Thinning treatments are also thought to reduce successful beetle attacks by altering microsite conditions by increasing temperatures on bark surfaces on bark in summer and decreasing them in winter, as well as disrupting beetle communication by increasing wind flow [94,95]. A new treatment recommended for reducing bark beetle infestation is “daylighting” which involves removing trees and shrubs from around trees that are to be protected to increase light on the tree’s stems to disrupt beetle colonization. Other silvicultural treatments include removal of beetle-suitable hosts (mature trees and old growth) and conversion of stands from species preferred by beetles (pines) to species that are not hosts or converting stands that are primarily pine to a mixed species composition [91,92]. Most of these approaches involve, completely or partially, the use of timber harvests.

4.1. Efficacy of Direct Controls

Direct control treatments are extremely expensive in time, effort and resources. They address only one aspect of an outbreak which is the amount of beetles present in a stand or area. Because they do not address the underlying conditions that support an outbreak (climate, tree condition/stress) their effects are considered a holding action until conditions shift to being less favorable for the beetle [92].

Direct control efforts must be maintained at a high level on an annual basis until the outbreak ceases [3,90,96]. It is highly controversial whether direct controls are effective in reducing tree mortality in the short-term, and if they can be effective in halting or suppressing outbreaks in the long-term.

One of the biggest problems in assessing the utility of direct controls is a general lack of monitoring or *post hoc* assessments of the outcomes of implementing these practices. Despite decades of direct control and large-scale implementation of these practices, few rigorous studies on its efficacy have been done and there remains no agreement among scientists or foresters regarding its ability to reduce beetle populations or losses of trees. Studies conducted prior to the current outbreak have variously concluded that direct treatments may merely act to delay infestation of susceptible stands [97], or that if used correctly, can be effective [98,99]. Many studies found that while some treatments slowed the rate of infestation, overall, they had little to no impact on mountain pine beetle populations [97,100–104].

The US and Canadian governments have spent hundreds of millions of dollars in direct control efforts to address the current outbreak. However, assessments of the efficacy of these efforts are nearly non-existent and only a few studies on assessments have been published. The few that have been published are reviewed here. Although much of our review addresses how well science supports US policy, we use primarily studies conducted in Canada as few studies have been published on direct control measures during the current outbreak in the US.

Nelson *et al.* [3] evaluated the efficacy of five direct control treatments in British Columbia roughly midpoint in the portion of the current outbreak as it progressed in that province. The assessment was extremely short-term and looked only at the response of beetles in the year immediately post-treatment. However, it provides one of the very few broadscale assessments ever conducted of the efficacy of direct controls during an outbreak. The treatments assessed were applications of MSMA, trap trees, fell and burn, and clearcutting. The study was split into three geographic regions to account for potential sources of variability due to location and different background levels of beetles. The northern-most region was at the margin of the beetles range (expansion zone) and possessed relatively low beetle populations, while the central and southern regions had higher beetle populations and were known to have supported high beetle populations historically. The study found that, overall, sites receiving MSMA treatments exhibited higher infestation intensities (a metric based on kernel density estimators) than randomly selected untreated sites with similar characteristics. This was particularly pronounced in the southern region. Results for trap tree treatments showed substantial variability within and among regions. A reduced infestation rate in response to treatment was observed more often than not in the northern area where beetle pressure was low. However, in the central and southern regions where beetle pressure was higher, the range of infestation intensities was similar for treated and untreated sites although a larger number of comparisons found higher infestation intensities in the treated sites. The overall conclusion was that MSMA and trap tree treatments may be effective, but not reliably, and only when beetle pressure is low and environmental conditions are not highly favorable for the beetle.

Results for fell and burn were also variable. In the northern region, intensities were lower overall in treated *vs.* untreated sites. However, in the central area, treated areas tended to have greater infestation intensities. In the southern area, no discernible effect of treatment was seen. Therefore, like with trap trees, fell and burn appeared to sometimes be effective, but only when populations of beetles were low, and became increasingly unreliable as beetle pressure increased and the infestation moved into outbreak phase.

Removal of trees in patches was studied only in the central region. No significant effect of treatment was detected. Clearcuts were assessed in the central and southern areas and were found to lead to a significant reduction in infestation intensity. In almost all cases, infestation intensities were lower in treated *vs.* untreated areas. However, this was likely due to the removal of all living trees (potential subsequent hosts) that survived the beetle as well as the infested trees. The overall conclusion of the study was that mitigation treatments are effective when populations are low to moderate and if infested trees can be kept to 2.5 or fewer per hectare. Efficacy was also recognized to be contingent upon a high level of accuracy in detecting infested trees and wide-scale and continuous implementation of treatments. However, with only one year of data, the authors could not predict how long treatments would need to be sustained to remain effective, nor what effect beetle pressure from surrounding areas might have on the subsequent fate of treated stands. No follow up study has been published to report how these treatments fared as the outbreak progressed.

Fell and burn has been a stalwart component of the direct control efforts against mountain pine beetle in Canada during the current outbreak, particularly on the advancing front as the beetle expands its range eastward. Coggins *et al.* [105] examined the efficacy of fell and burn treatments to "stabilize" such infestations (*i.e.*, prevent expansion) using field plot data from sites at the expanding edge of the mountain pine beetle infestation in 2008 in eastern British Columbia and western Alberta. The authors used multiple modeling scenarios along with ground data to demonstrate how infestations may develop with and without mitigation, and to predict how long mitigation may need to be maintained to be effective given different levels of infestation and detection accuracy. They found non-mitigated plots experienced more tree mortality due to the beetle and that infestations in these plots expanded more rapidly. The higher the expansion factor (means rate of increase, e.g., 2 would indicate a doubling of the population each year) the greater the detection accuracy that was required to maintain a static population. When a beetle population had an expansion factor of 5.1 (high), an 80% detection rate was required, whereas with a population with an expansion factor of 1.1 (very low), the minimum detection rate could be as low as 10% and still be effective. The authors also modeled how long it would take to achieve population stability given different levels of infestation. On average, across their stands, with a 70% detection accuracy rate, mitigation would take 11 years, at 80% 6 years, and at 90% 3 years. The actual mean mitigation efficiency at their sites was found to be 43%, a level at which no control could occur. They concluded that the stabilization of mountain pine beetle populations is possible, but only with a much higher detection accuracy than commonly occurs coupled with an intense level of mitigation maintained potentially over a very long timeframe.

Wulder *et al.* [96] looked at the effectiveness of sustained mitigation on slowing the beetle's expansion in western Canada. The results were difficult to assess because of the unevenness of application of mitigation treatments (for example, in one year only 68% of sites slated for mitigation were treated) and differences in background beetle populations. However, such a situation is typical and thus may represent the reality of many on-the-ground direct control efforts. One site where little mitigation was conducted early on, did exhibit a strong increase in tree mortality due to the beetle that declined once extensive mitigation efforts were implemented. However, overall, the conclusion was that mitigation must be extensive and continuous to work and may only be effective when populations are low to moderate.

Trzcinski and Reid [104] studied the trajectory of beetle populations in treated and untreated zones in Banff National Park from 1997–2004. The Park used a combination of pheromone-baited trees and fell and burn to remove as many beetles as possible from treatment zones—they also conducted prescribed burns to reduce beetle numbers and lodgepole pine hosts. The area colonized by the beetle increased rapidly over this time period in both the untreated and treated zones. After four years of treatment, control measures did not reduce the area affected by beetles and infestations continued to expand at a similar rate in both zones. The authors estimated that between 45% and 79% of infested trees had failed to be detected in the treated areas. This equated to *only* 0.7–3.7 infested trees remaining per thousand ha yet still was sufficient to support subsequent rapid beetle population growth.

A general consensus of these studies is that suppression of a beetle outbreak would require massive sustained efforts with extremely high detection rates to succeed. It has been estimated that 97.5% of beetles in an area must be killed to merely stabilize a mountain pine beetle population [90]. Even a small increase in survival above this value can allow a substantial increase in population size. For example, if mortality drops to 95%, this would allow a population to *double* in size annually. If the goal is not just to stabilize a population, but to reduce it, mortality of beetles would need to be higher than 97.5%, a goal that is highly unlikely given the vast areas that would need to be treated on a continual basis when conditions are favorable for outbreak development. Even if 100% removal of infested trees from an area was feasible, the migration of beetles into treated stands from surrounding areas allows reestablishment and subsequent tree mortality further decreasing the potential for effective direct control.

The on-the-ground reality is that direct control efforts typically fall far below the levels needed to stabilize, let alone control, mountain pine beetle populations. In the above cited studies, rates of detection in mitigated stands ranged from 45%–79%. These situations are not unusual. Direct control treatments are laborious, extremely costly and time consuming, and require high levels of training. Logistical difficulties, including proper seasonal timing, access, inclement weather, and lack of trained personnel, increase the odds that they will not be effective. The high financial cost of such efforts coupled with a volatile market for sawtimber, pulp and pellets further complicates the use of direct controls. Importantly, outbreak development is extremely swift and the amount of mitigation required can rapidly outstrip the ability of managers to respond.

During an outbreak the number of trees killed annually is often in the millions and infestations may cover hundreds of thousands of hectares [90]. Carroll *et al.* [90] presents an example of the degree of mitigation that would be required for an outbreak that covers 300,000 hectares with a rate of increase of 2 (the population doubles in one year-a conservative rate for an outbreak). In this case, 150,000 ha of infested trees would need to be removed each year just to maintain a *static* beetle population–this would still allow tree mortality to occur for many years, potentially until most or all mature trees were killed. In reality, such a high level of detection and mitigation is impossible. Given that the goal of direct management is to reduce populations and protect trees, the effort that would be needed to actually reduce such a high beetle population would require an even more unlikely effort.

Studies in other bark beetle systems also have found that a high degree of detection accuracy and intensity of mitigation is required to reduce beetle numbers. Fahse and Heurich [106] found that control of *Ips typographus*, a less aggressive European bark beetle, requires a detection and removal level of around 80% to be effective. They concluded that direct control efforts are useless and should be dropped if survival probabilities of the beetle after treatment are above 20%–30%. This estimate is in line with those developed in studies on mountain pine beetle in North America and highlights the challenge the high reproductive capacity of bark beetles poses when conditions are favorable for outbreak development.

It is not just the difficulty of dealing with the extreme spatial extent of outbreaks and the challenge of detection and treatment that makes the efficacy of direct control measures unlikely, but also the time frame over which direct controls must be maintained. Carroll *et al.* [90] estimated that to control a population involving 10,000 infested trees with expansion factor of 2 (conservative) and with a detection and removal rate of 80% (difficult), it would take at least 10 years of annual treatment to reduce the population to a single tree. If the population was tripling or quadrupling, a more likely scenario during an outbreak, it would take 18 or 41 years, respectively. A costly, intensive detection and treatment program lasting that long, assuming sufficient trees even remained to be infested, would be unlikely [90].

Carroll *et al.* [90] emphasized three requirements for direct controls to be effective in treating *individual* infestations: infestations must be detected early, efforts must be applied quickly and intensively, and control programs must be maintained continuously until the desired population level is achieved. Because of the cost and intensity of treating individual infestations, the US Forest Service recommends that direct control measures only be applied to higher value stands [92]. However, treating individual infestations or stands during outbreaks can fail because of the regional nature of outbreaks. Outbreaks are driven by abiotic factors that affect entire regions (warm temperatures and drought). Thus, they consist of many infestations that occur synchronously across a very large area. These infestations often coalesce to form vast expanses where beetle populations are extremely high. These characteristics mean that many stand level efforts are prone to failure due to high beetle pressure and migration into treated areas by beetles from surrounding areas. Given that treating entire regions is impossible, and that many treatments are not in line with other land use objectives, direct control efforts may in some cases, not be worth their costs. The consensus of studies and retrospectives over the course of several outbreaks is that even after millions of dollars and massive efforts, suppression using direct controls has never been effectively achieved, and at best, the rate of mortality to trees was reduced only marginally [90,101,102,105]

4.2. Efficacy of Indirect Controls

Thinning is the primary indirect control measure used to manage the mountain pine beetle. It is generally considered a preemptive measure to be implemented prior to the initiation of a mountain pine beetle outbreak, although it is increasingly employed to reduce damage by the insect during outbreaks. It is often touted as a global panacea for problems with pest bark beetles. One type of thinning is even termed “beetle-proofing” [107], further reinforcing the view among managers, the public, and policy makers, that this approach is failsafe. While overall, evidence suggests that thinning can reduce mortality of trees due to mountain pine beetle, the outcome is frequently more variable than is often recognized or reported. This is particularly true when outbreak populations are involved.

So how exactly does thinning work, and how well does thinning hold up under outbreak conditions? Surprisingly, the mechanism(s) by which thinning affects beetle activity in forest stands is still not well understood. Two, non-mutually exclusive, lines of thought exist. One hypothesis is that thinning increases tree vigor, and thus tree defenses, by reducing competition among trees for light, nutrients and water [93,108]. Intuitively, this makes sense, and indeed, immediate impacts of thinning on reducing water stress have been seen [109]. Likewise, increases in growth and photosynthetic rates also have been observed post-thinning, albeit after a lag period of one or more years [107,109,110]. Increases in growth and vigor are predicted to increase the amount of energy that trees allocate to defense, leading to greater resistance to beetle attack through increased resin and monoterpene production. In fact, the initial impetus for the use of thinning to manage mountain pine beetle came from an early study that found that ponderosa pines in thinned stands produced more defensive resin [93]. However, subsequent studies have reported a variety of responses in resin production as well as growth in response to thinning. For example, Zausen *et al.* [111] found that ponderosa pines in the thinned stands exhibited lower water stress but also produced less resin. This, along with the thicker phloem (greater food resources) found in trees in thinned stands, indicates they might be not only more susceptible to attack but also a more productive resource for beetles. In contrast, McDowell *et al.* [112] found greater resin flow in thinned stands. Both studies were conducted in southwestern US ponderosa pine forests indicating that the variable responses observed were not due to major regional differences in hosts. Six and Skov [113], in a study conducted in ponderosa pine in the northern Rocky Mountains looking at effects of thinning and burning treatments, found that resin flow was highest in trees in burn treatments, intermediate in controls, and lowest in thinned treatments. Raffa and Berryman [114] tracked the fate of trees over time during an outbreak and found no significant difference between resin flow for lodgepole pines that survived attack vs those killed by the beetle.

A number of studies have noted a reduction in beetle caused-mortality of trees immediately after thinning treatments were applied and before trees had time to respond physiologically to lower stocking densities. This timing suggests that the effects of thinning may have more to do with microsite conditions than to changes in tree vigor or defense. These observations led to the second line of reasoning that thinning affects beetle activity through changes in microsite conditions.

Thinning alters temperature, light intensity and wind speed within a forest stand; factors that can have major effects on insect behavior and success. A number of studies have tried to describe how shifts in microsite conditions due to thinning may influence mountain pine beetle activity. Bartos and Amman [94] investigated how incident solar radiation, wind speed, wind direction and temperature were altered by thinning and whether changes affected beetle responses to stands. They did not conduct statistical analyses on their data; however, there was a trend for south sides of trees in thinned stands to be warmer, and ambient temperatures in thinned stands to be overall warmer during parts of the day. Incident solar radiation was higher in the thinned stand. It is not known if bark temperature affects beetle attack behavior, although higher temperatures on south sides of trees in thinned stands have been suggested to be deleterious to beetle development [94]. However, this speculation does not account for differences in local environmental conditions. For example, at cool sites, increased temperatures and insolation could ostensibly support better beetle development by increasing thermal units sufficiently to support a univoltine life cycle.

Light intensity affects the flight behavior of mountain pine beetles [115]. However, if and how different levels of light in treated and untreated stands affect beetle attack behavior is unclear. It has been hypothesized that a reduced propensity for flight in darker stands might concentrate beetles for mass attack, while beetles may be more likely to disperse in open stands [116].

The hypothesis that light has a strong effect on mountain pine beetle behavior, particularly in reducing attacks, has led to a new treatment called daylighting. This approach is currently being implemented on a broad scale by federal and western state agencies. Daylighting involves removing trees and vegetation from around trees that are targeted for retention and is believed to work by repelling beetles from the boles of trees by increasing light and solar radiation [117]. While widely recommended, the efficacy of this treatment is unknown; there are no published studies on its effects on bark beetles.

Changes in wind speed and direction due to thinning have also been suggested to alter beetle behavior by disrupting beetle communication via disruption of pheromone communication. Schmid *et al.* [118] found no statistically significant differences in horizontal and vertical wind patterns in thinned and unthinned stands. However, disruption of pheromone plumes by greater wind speeds may affect communication and thus the potential for successful attacks [95]. Ultimately, we need to look at actual population dynamics of beetles in treated and untreated stands to understand if microsite effects hold under epidemic conditions. MacQuarrie and Cooke [119] found that, under outbreak conditions, mountain pine beetle populations exhibited density-dependent dynamics and that thinning did not change the epidemic equilibrium. In this study, population growth curves did not exhibit responses that would be expected if microsite conditions played a role in beetle behavior. It is evident that more research is needed to understand how these effects ultimately influence tree mortality due to beetle attack.

While we may not have a complete understanding of how thinning works, it is clear that this practice can have a significant effect on mountain pine beetle infestations. Several studies have reported striking differences in mortality to trees caused by beetles in thinned *vs.* un-thinned forests (reviewed in [120,121]). In contrast, only a small number of studies have reported failures. However, the disparity in numbers of successes and failures must be placed within a broader context. Many studies assessing the efficacy of thinning have been conducted under non-outbreak conditions. Their results do not reflect how stands perform during an outbreak. Additionally, failures are often not reported, dismissed as a result of poor management 'next door' or targeted for management without evaluation. This is unfortunate because thinned stands that fail may have particular characteristics that could inform a better understanding and application of this approach.

Studies conducted during outbreaks indicate that thinning can fail to protect stands. In Colorado, thinning treatments in lodgepole pine implemented in response to the outbreak that began in the 90s often only slowed the spread. Klenner and Arsenault [122] reported high levels of mortality due to the mountain pine beetle across a wide range of stands densities in lodgepole pine in British Columbia during the same outbreak. They noted that silvicultural treatments were largely ineffective in reducing damage to the beetle. Preisler and Mitchell [123] found that once beetles invaded a thinned stand the probability of trees being killed there can be greater than in unthinned stands and that larger spacings between trees in thinned stands did not reduce the likelihood of more trees being attacked. Whitehead and Russo [107] reported on the performance of 'beetle-proofed' (stands thinned to an even spacing of about 4–5 m between mature trees) and un-thinned stands in five areas in western Canada during approximately the same time period. These treatments were successful in protecting stands when they were combined with intensive direct control measures (removal of infested trees) in the areas surrounding the thinned units, but failed if units were exposed to beetle pressure from the neighboring area—a situation most thinned stands experience during an outbreak.

Unfortunately, long-term replicated studies monitoring beetle responses to thinned forests from non-outbreak to outbreak to post-outbreak phase are virtually non-existent. One large fully-replicated long-term study was initiated in 1999 under non-outbreak conditions and continues to track beetle activity [113]. In this study, mountain pine beetle was low in all treatments in the period leading up to the outbreak, but increased in some controls and burn treatment replicates as the outbreak developed. Although more trees were killed overall in control units during the outbreak, all controls still retained a greater number of residual mature trees than did thinned stands as they entered the post-outbreak phase [124].

Two factors contribute substantially to our inability to assess how well thinning performs under outbreak conditions. One, very few thinning treatments are monitored after implementation over either the short- or the long-term. Thus, for the vast majority of stands that have been treated, we have no data on how well they perform once an outbreak of the insect initiates (or for that matter, even under non-outbreak conditions). Second, stands that become infested, thinned or otherwise, are often targeted for intensive suppressive management and are cut without assessment or data collection. This even includes studies and sites that are intended to inform management. For example, at the sites studied by Whitehead and Russo [107], infested trees were being removed from the study sites even before data collection for their study could be completed. The long-term study discussed previously [113,124] is under continual pressure to be logged to remove beetle kill even though the site lies within an experimental forest designated specifically for studies assessing the outcomes of forest management.

5. What are the Goals?

When we manage forests, we do so in an attempt to achieve one or more outcomes, preferably with minimal negative effects on non-target resources. To be effective, management must have explicit and appropriate goals as well as clear metrics for success. Ideally, management is monitored to assess how well it meets its goals, where it falls short, and whether and how it can be improved. This approach is called adaptive management and implies an iterative process through time whereby we learn from the outcomes of our actions and base future actions on improving performance [125].

Not only outcomes, but the costs of management must be factored into decision making. These include direct financial costs as well as the less tangible (at least in dollar values) effects on ecosystem services and functions. By considering the full cost of management along with benefits as verified through monitoring and evaluation, we lessen the risk of failure, financial waste, and unnecessary negative environmental impacts.

In assessing how well we meet goals when managing for mountain pine beetle, we must ask several questions. Do our management practices actually control the beetle during outbreaks? Do the outcomes justify the financial and ecological costs? And, what long-term impacts do these treatments have on forests and their ability to adapt to climate change? These questions are difficult to answer. Only limited data are available on the short-term efficacy of direct and indirect controls, and information on long-term effects is virtually nonexistent. The results of short-term assessments can be difficult to interpret. For example, often only the proportion or numbers of trees killed by beetles post-treatment are reported. This does not allow a complete evaluation of outcomes. A study may report that 75% of trees in controls are killed by the beetle, whereas only 10% are killed in thinned stands. At first glance, this appears to be a resounding success in saving trees. However, if we approach this situation from a pretreatment perspective, our interpretation of success may change. In this example, 400 mature trees existed in each plot prior to treatment. After treatment, 100 mature trees remain in the thinned plots (300 trees have been removed by thinning). Doing the math, we find that once the beetles have run their course, more residual living trees (100) actually remain in the control plot than in the thinned plot (90) and, in fact, humans have contributed more to tree mortality than have the beetles. In the case of silvicultural intervention, humans typically must expend considerable effort and expense. They also choose the trees that remain, and thus the structure and composition of the remaining forest. This may result in very different trajectories for residual forests as discussed below.

When we include pre-treatment conditions as well as post-treatment responses we can assess the management efficacy from a more informed position. For instance, in a retrospective study investigating the effects of management on spruce beetle, researchers found that post-infestation, untreated stands had more live spruce trees and greater basal areas. When comparing only residual large spruce, final densities in both stand types were similar [126]. Six [124] found higher numbers of mature living trees remained in control stands of ponderosa pine than in thinned stands post-mountain pine beetle outbreak. In a study in Canada focusing on stocking density of living lodgepole pine post-outbreak, the authors found that, even in hard hit stands, stocking density in post-outbreak unmanaged stands was sufficient to maintain desired levels of productivity [127]. Klutsch *et al.* [128] in a study conducted in lodgepole pine forests in Colorado, found greater mortality of trees due to the beetle in more densely stocked stands. However, while the density and basal area of lodgepole pine in infested plots declined 62% and 71%, respectively, the number of trees that remained and their size distribution post-outbreak indicated that lodgepole pine would remain the dominant overstory tree. In another study in Colorado, the beetle killed 60%–92% of overstory lodgepole pine. However, these stands retained residual overstory trees as well as advance regeneration. Furthermore, untreated stands were predicted to return to pre-outbreak stocking levels approximately 25 years sooner than treated stands [129]. Other studies have found similar results for both lodgepole and ponderosa pine [130–134]. These studies highlight a seldom considered impact of mountain pine beetle- that it can act as a natural thinning agent and seldom removes all mature trees during outbreaks. These effects are an important part of the ecological role that the beetle plays in western pine forests [135].

It is also important to recognize there can be significant differences in long-term forest trajectories for stands thinned by beetles *vs.* those thinned by humans. When humans thin, they select for particular size classes, often favoring the retention of larger, older trees, selecting toward one desired tree species, and often ‘thinning from below’ which removes advanced regeneration (small trees) [123,136]. Thinning prescriptions also typically call for relatively even spacing between residual trees [92,107,121]. Mountain pine beetle, on the other hand, often selects the largest trees during outbreaks (with exceptions; [121,123,131]) which can lower the mean diameter of the stand [128]. However, beetles often leave sufficient numbers of large diameter trees to maintain a dominant overstory of pine. Beetles also leave substantial amounts of advanced regeneration to replace the mature trees that arekilled [121,129]. Spacing among trees after an outbreak is uneven, resulting in a clumpy network of living trees [129]. Patches where all trees are killed are seldom extensive and add to a mosaic structure as forests recover post-outbreak. Heterogeneous stand and mosaic forest structures are more typical of natural conditions and can support greater biodiversity and resilience against fire and subsequent beetle outbreaks [137–139]. In contrast, intensive thinning treatments by humans typically favors the retention of mature pines. Over time, these pine-dominated stands grow, they are predicted to have increased susceptibility and potential for tree mortality from future mountain pine beetle outbreaks [123,136].

Very importantly, the beetle exercises selectivity in the trees it kills. While extremely high numbers may override this selectivity, evidence is accumulating that, even under outbreak conditions, beetles choose trees that have particular qualities. Beetles commonly select trees for attack that exhibit lower growth rates, defenses, and higher water stress [58,74,77]. While these factors can be influenced both locally and regionally by site conditions and climate, much of the variation in these properties within individual stands that affect bark beetle choice likely has a genetic basis. Outbreaks can result in strong natural selection against trees with phenotypes (and likely genotypes) favorable for the beetle and for those that possess unfavorable qualities [58,77]. However, when humans thin forests, trees are removed according to size, species, and density, without consideration of genetics. Thus, trees best adapted to surviving beetle outbreaks are as likely to be removed as those that are not.

When humans thin forests, they typically manage for resistance and resilience, rather than adaptation which involves genetic change. It is very important to distinguish between resistance, resilience, and adaptation, as each have different goals and operate on different temporal scales [140]. Resistance is a short-term holding action where we try to maintain an existing state. Approaches focusing on resistance often require massive interventions and increasing physical and financial investments over time. Such approaches may set forests up for future outbreaks [136] and even catastrophic failure as they surpass thresholds in a warming climate [140]. In contrast, practices that promote resilience attempt to allow forests the ability to adjust to gradual changes related to climate change and to recover after disturbance. However, like resistance, resilience is not a long-term solution. In the long term, forests must be able to adapt to change. Adaptation involves genetic change driven by natural selection. Currently, much of forest management, including bark beetle management, focuses on resistance and resilience, mainly through direct and indirect management, respectively. However, neither approach allows for true adaptation. For long term continuity of our forests, it will be imperative to begin to incorporate this aspect of management into our approaches.

We also need to reassess the ecological role of bark beetles, including the mountain pine beetle, in our forest ecosystems. As has been well demonstrated by a century of fire suppression, the dampening or suppression of natural disturbance can alter forest trajectories in undesirable ways, many of which can be irreversible. Although beetle outbreaks, like fire, can have negative impacts on timber values and aesthetics, their natural role in many forest ecosystems is seldom considered and beetle suppression is often perceived as something that must be conducted at all costs. However, as with fire, suppression of beetles over the long term may alter forests in ways that are not desirable or sustainable. While intensive management for bark beetle suppression is called for in some situations such as in the wildland urban interface, it may not be appropriate in many other areas where natural processes including natural selection are needed to maintain a dynamic and functional forest.

6. What are the Needs in Research and Monitoring?

There is clearly a need to better understand how well management programs aimed at reducing mountain pine beetle work, particularly under outbreak conditions, and what impacts these treatments have on forests in both the short and long term.

Perhaps the biggest area of need is in monitoring. Monitoring is essential to understanding whether mountain pine beetle treatments work, and in which contexts, but as noted above there has been all too little long-term monitoring of the effectiveness of various treatment efforts. This is a failing among both agencies and researchers. Agencies often do not have strong incentives to conduct long-term monitoring: Monitoring is costly; external and internal political pressures focus on short time frames; and monitoring may produce information that conflicts with agency goals or missions. It is also difficult to get strong public pressure to force agencies to conduct the necessary monitoring, particularly when the public has been led to believe that outbreaks are strictly the result of a lack of management. Even for scientists, long-term monitoring projects are not encouraged by short-term funding time frames and professional incentives or norms; monitoring is often not viewed as "real" science, and the long-time frames required for monitoring to result in significant gains in information are often longer than the time frames used for professional advancement (e.g., completion of a dissertation, tenure review) [141].

Addressing the shortage of monitoring for beetle treatments may, therefore, require far more than simply trying to provide additional funds (even assuming additional funding is politically feasible). Scientists can help by encouraging and rewarding projects that involve long-term monitoring. Agencies might try to establish units that are focused specifically on monitoring forest health, insulating monitoring projects from adverse political or bureaucratic pressure [141]. Finally, tools that might reduce the cost of monitoring significantly, such as retrospective studies and remote sensing, should be used to complement traditional monitoring and decrease its costs.

Monitoring is all the more essential if forest health management in general, and beetle treatments in particular, are truly to be guided by adaptive management. The high levels of uncertainty and dynamism associated with beetle infestations and the effectiveness of beetle treatments make adaptive management a very appealing tool to reduce uncertainty and allow us to respond to changes in global climate and forest ecosystems. But adaptive management requires monitoring to be successful [141], monitoring that is currently not occurring even as agencies conduct massive beetle treatments and propose to pursue even more.

There is also a real need to increase research on management efficacy and, in particular, how our approaches affect forest adaptation including genetic responses of trees to climate and the role in bark beetle selectivity and fitness. With a changing climate we will need to develop new approaches rather than trying to force old methods of questionable efficacy onto new conditions.

Unfortunately, most funding for research on bark beetles is very short-term, sometimes even as short as on an annual cycle, and thus cannot hope to address the complexities of beetle responses to treatments. Funding cuts to research personnel, particularly in agencies like the US Forest Service, have exacerbated this problem exactly at the time when the need for rigorous research is increasing at a rapid pace. The US Forest Service has recognized that long-term planning must include explicit goals to increase forest resilience and adaptation to disturbance, including outbreaks of the mountain pine beetle. However, with extreme cuts to budgets and personnel, they are highly constrained to meet these needs at this time. Likewise, cuts in federal funding to agencies such as United States Department of Agriculture and the National Science Foundation concurrently reduce the ability of academic researchers to address these problems.

7. Aligning Policy to Science

Our survey of the relevant literature finds that there is significant uncertainty about whether the most commonly used beetle timber harvest treatments are, indeed, effective. Yet there has been little discussion of this uncertainty in the relevant policy debates. Politicians have instead latched on to beetle timber treatments as a cure-all for beetle infestations and have pushed to weaken or eliminate environmental laws that are perceived to be obstructing these treatments. Agencies such as the US Forest Service, to their credit, have been more nuanced in their support for bills that package beetle timber harvest treatments with weakened environmental laws; they have opposed several proposals to alter environmental laws to allow more treatments, but on the other hand, the agencies have at times also aggressively pushed for the implementation of treatments.

It seems clear that the policy debates–both in the agencies and in Congress–need to be better informed by science. Researchers should be more proactive in communicating their understandings of the current science to policymakers. This does not mean that researchers need to take a position pro or con vis-à-vis beetle treatments, or even vis-à-vis specific legal proposals. In the face of uncertainty, aggressive beetle timber harvest treatments may be warranted in some instances. However, policymakers should be aware of uncertainty when they are making the relevant decisions and should also be more willing to include the voices of scientists in the development of policy.

Given the uncertainty about the effectiveness of many beetle timber harvest treatments, the high financial costs of those treatments, the impacts on other environmental resources and values, and the possibility that in the long-run those treatments may interfere with the ability of North American forests to adapt to climate change, our position is that weakening or eliminating environmental laws to allow more beetle timber harvest treatments is the wrong choice for advancing forest health in the United States. Indeed, given the uncertainty, the costs, and the possibilities of both short-term harm to other resources and long-term ineffectiveness, we believe that the current structure of thoughtful, detailed environmental review for these projects is, in general, appropriate. If agencies believe that they need to be able to react quickly to specific infestations with treatments, and that this quick reaction is incompatible with existing legal procedures, we encourage the agencies to adopt overall programmatic environmental reviews based on the principles of adaptive management. Agencies should be able to build (or tier) on these programmatic reviews to respond quickly to individual events as needed. However, the programmatic reviews should allow the agency to build in the monitoring, replication, and variance of treatments that are essential for successful adaptive management [142].

8. Conclusions

The manner in which policy makers have accepted beetle timber harvest treatments as a panacea for responding to bark beetle outbreaks in North American forests raises a number of red flags. As ecosystems and places that have economic, social, and cultural value to human communities are altered by climate change, there is a risk that people will overreact because of a need to "do something" to respond to change, and to give themselves some sense of control over broader forces that appear to be out of control. That pressure, to "do something", might also interact with the uncertainty about which choices are effective and appropriate (as with beetle timber harvest treatments) to create an opportunity for political pressures to force the adoption of particular choices that benefit specific interest groups [143]. It is perhaps no accident that the beetle treatments that have been most aggressively pushed for in the political landscape allow for logging activities that might provide revenue and jobs for the commercial timber industry. The result is that the push to "do something," uncertainty, and political pressures might lead us to act to respond to climate change before we understand the consequences of what we are doing, in the end producing more harm than good.

Our argument here is not to forgo management, but rather that management should be led by science and informed by monitoring. Both direct and indirect management for bark beetles have their place. However, to manage our forests in a way that best ensures their long-term function while wisely using limited financial resources, policy makers and the public need a clearer understanding of current science and gaps.

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Conflicts of Interest

The authors declare no conflict of interest.

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Chapter 10

c0050 Carbon Dynamics of Mixed- and High-Severity Wildfires: Pyrogenic CO_2 Emissions, Postfire Carbon Balance, and Succession

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s0010 10.1 MIXED-SEVERITY FIRES: A DIVERSITY OF FUELS, ENVIRONMENTS, AND FIRE BEHAVIORS

p0010 Recent increases in global temperatures are projected by some research to increase the frequency and severity of wildfires in certain regions, particularly those experiencing warmer, drier summers (McKenzie et al., 2004; Flannigan et al., 2006). While the annual area burned in most forests of western North America remains well below historical levels (see Chapter 9), many areas have experienced significant increases in annual burning, particularly from 1970 to 1986 (Westerling et al., 2006), prompting concerns about the additional release of carbon, primarily in the form of carbon dioxide. However, concerns over a positive feedback between wildfire-caused carbon emissions and temperature increase must be considered in the context of the physical magnitudes of pyrogenic carbon emissions and the respective constituents of forest carbon storage from which they are derived. Here I discuss the factors influencing the combustion of different constituents of forest carbon storage and how rates of fuel combustion vary among fires of low, medium, and high severity. This chapter also addresses the relationship of fuel reduction treatments with regard to reducing fire severity and carbon emissions at the potential expense of forest carbon storage. Finally, I discuss postfire carbon emissions from the decomposition of fire-killed biomass, postfire forest succession, and the eventual recovery of forest carbon storage.

p0015 Rates of pyrogenic carbon emission from wildfires can be highly variable among mixed-severity wildfires. The consumption of each respective component of forest fuel is strongly determined by individual particle geometry, often expressed as the surface area-to-volume ratio for the purposes of quantifying the amount of fuel that is likely to be consumed. Combustion generally occurs at the surface of the fuel particle, and the size of each particle and its surface area-to-volume ratio control the amount of heat required for ignition and consumption. Fuels with large surface area-to-volume ratios, such as grasses and pine needles, require less heat for ignition and combustion. Conversely, large fuels with low surface area-to-volume ratios, such as standing trees, as well as snags, downed logs, and other forms of coarse woody debris, require considerably more energy for ignition and combustion. Fuel particle size also influences the rates of moisture absorption and release, as smaller fuel particles release moisture more rapidly than larger particles in response to increasing atmospheric vapor pressure deficits, as well as in response to the thermal energy brought about by an approaching flaming front. Consequently, large fuels are much more likely to burn during the smoldering stage, in which the emissions of combustible gases and vapors are too low to support flaming combustion (Lobert and Warnatz, 1993).

p0020 Fuel consumption also is influenced by the compactness of the fuel bed, in part because of the two-stage process of consumption through pyrolysis and combustion. While these processes are nearly simultaneous, pyrolysis occurs first and is the heat-absorbing reaction that converts fuel elements such as cellulose into char, carbon dioxide, carbon monoxide, water vapor, highly combustible vapors and gases, and particulate matter (DeBano et al., 1998; Ward, 2001; Ottmar, 2014). Pyrolysis is followed by combustion, in which escaping hydrocarbon vapors are released from the surface of the fuels and are oxidized. Thus fuel compaction presents a tradeoff between heat transfer and oxygen diffusion. Highly compacted fuels facilitate a more efficient transfer of heat between fuel particles while limiting the diffusion of oxygen and, by extension, limiting consumption. Conversely, low fuel compaction allows for high diffusion of oxygen, albeit with a low diffusion of heat between fuel particles (Hardy et al., 2001). Fuel consumption also is influenced by the spacing, or continuity, of fuels across the forest floor (Finney et al., 2010) (Figure 10.1).

p0025 While the amount of consumption that is to be expected can be strongly determined by the fuel's physical and chemical characteristics, it is also a function of climate and topography. Regional climate exerts a top-down influence on fire frequency through seasonal patterns of temperature and precipitation (Littell et al., 2010), whereas local factors such as topography, vegetative composition, and fuel loads exert a bottom-up influence on fire behavior (Perry et al., 2011; Miller et al., 2012). Topography can influence the species composition of a forest, the composition and accumulation of fuels from a forest, and the topographically mediated content of fuel moisture. Among landscapes at elevations dominated by ponderosa pine (*Pinus ponderosa*) in eastern Oregon and Washington, white fir (*Abies concolor*) and grand fir (*Abies grandis*) are more common on north-facing slopes

f0010 **FIGURE 10.1** Aerial view of a smoke plume. *(Photo courtesy of M. Welling, Max Planck Institute for Chemistry.)*

because of the cooler and moist conditions that result from less incoming solar radiation (Cowlin et al., 1942). Stand composition and structure interact with the edaphic (pertaining to soils) moisture gradients to determine patterns of fire severity (Hessburg et al., 2000; Miller, 2003; Hessburg et al., 2004). In areas north of the Klamath Mountains in northwestern California, north-facing slopes may burn with mixed severity, whereas south-facing slopes can burn with mixed or low fire severity. However, the opposite occurs in the more xeric (dry) forests of the Klamath Mountains, wherein mixed-severity fires have historically dominated on south- and west-facing aspects, whereas low-severity fires were dominant on north- and east-facing aspects (Taylor and Skinner, 1998). Extreme weather conditions can override these effects, however, as was the case in the Biscuit Fire of 2002 in southwest Oregon; hot, dry winds from the northeast drove the fire, thereby eclipsing much of the influence of topographic positions (Thompson and Spies, 2010). Other fires with severe conditions have shown a stronger response to topographic controls, such as the Megram Fire in northern California (Jimerson and Jones, 2000).

p0030 The expected fuel consumption for a given level of fire severity is often expressed as a combustion factor (CF). A CF is the proportion of a biomass constituent that is expected to be consumed in a wildfire. CFs vary with respect to different biomass components such as live foliage, litter, stem, branches, shrubs, and soil. CFs can also vary as a function of fire severity: Lower levels of fire severity typically result in lower levels of combustion for each respective constituent of forest carbon storage. Note, however, that the use here of the term "fire severity," expressed as the proportion of mortality observed in overstory trees, can be misleading when used as a determinant of fuel combustion. Fuel combustion often ~~is~~ determined by fire intensity, a measure of energy output

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from a fire (Keeley, 2009). A fire of relatively low intensity could conceivably result in a fire of medium or even high severity if it occurred among trees with relatively low tolerance to fire. Because this is a book concerned about forest ecosystems with mixed- and high-severity fire regimes, however, we are largely dealing with ecosystems that have evolved at least some adaptations to moderate- or high-severity fire.

p0035 An improper use of a CF in estimating the carbon emissions of a given fire can produce vastly different estimates of pyrogenic carbon emissions. Worldwide, forests store about 45% of terrestrial carbon (861 ± 66 pg carbon) in soils, ~42% in above- and belowground live biomass, ~8% in dead wood, and ~5% in litter (Bonan, 2008). Given the magnitude of carbon stored in, say, dead wood, a poorly derived CF for dead wood can have a considerable impact on the resulting estimates of carbon dioxide emissions. Estimates of average pyrogenic carbon emissions for a given time period can produce a considerable range of values, some of which can be over four times higher than those of others (Wiedinmyer and Neff, 2007; Ghimire et al., 2012), in part because of methodological differences in the approaches used to estimate biomass accumulation and area burned, as well as different approaches used by different studies to obtain CFs.

p0040 Here I discuss factors controlling the combustion of different constituents of carbon storage in forest ecosystems and how these constituents can influence, and can be influenced by, different levels of fire severity in forested landscapes with mixed- and high-severity fire regimes. I also discuss the indirect impacts of wildfire through the long-term carbon emissions of fire-killed biomass and how emissions after wildfire can influence the source-sink dynamics throughout a postfire landscape.

s0015 10.2 DUFF, LITTER, AND WOODY DEBRIS COMBUSTION

p0045 Duff carbon comprises the dead organic matter found in the O_a (almost complete decomposition) through the O_e (moderate composition) horizons, whereas litter comprises the dead materials found in the O_i horizon (undecomposed plant parts) and includes small, woody fragments <0.51 cm in diameter, also known as 1 h fuels. Small, woody debris consists of particles 0.51-2.54 cm in diameter, also known as 10-h fuels. While only a small fraction of total forest carbon storage, these components of carbon storage on the forest floor often constitute the majority of combusted fuel for fires of all severities. Campbell et al. (2007) estimated that duff, litter, and small, downed, woody debris consumption constituted about 60% of direct carbon emissions in the Biscuit Fire of 2002. High rates of combustion among these components are consistent with the principle that fuels with large surface area-to-volume ratios have higher CFs than fuels with lower surface area-to-volume ratios, much of which can be attributed to the short time periods required for woody materials (1- to 10 h fuels) to dry out. Seasonal variation in fuel moisture can thus have a considerable impact on carbon emissions. Knapp et al. (2005) found that early season burns, in which fuel

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moisture was higher, left approximately five times more litter and duff unconsumed in areas where fire passed over the forest floor than late season burns.

p0050 Noting that this pool of carbon storage is destined for biogenic emission to the atmosphere in the absence of wildfires is important. Pools of litter, foliage, and small, downed wood are thought to have a mean residence time of 10-20 years (Law et al., 2001), and while a portion of this eventually transitions into more stable forms of soil carbon storage, much of it is lost through decay. Furthermore, much of the carbon stored in a pool with such high turnover should equate to a subsequent reduction in heterotrophic (requiring organic matter for food) respiration until these pools become recharged by the addition of leaf litter and small, woody debris (Campbell et al., 2007).

p0055 Because additional energy is necessary to remove water before combustion is possible, more energy is required to propagate flaming combustion in moist fuels than dry fuels (Nelson, 2001). In theory (Finney et al., 2013), as well as in some modeling studies (Hargrove et al., 2000; Miller and Urban, 2000), the probability that fire will propagate to neighboring fuels is reduced at higher fuel moisture levels. Knapp et al. (2005) found that the amount of area within the fire perimeter burned, and greater patchiness of early season burns conducted under higher fuel moisture conditions, are consistent with these model predictions. Thus the combustion of large, woody debris (1000-h fuels) can be particularly sensitive to fuel moisture. Estimates of combustion of downed, coarse, woody debris suggest that the majority of carbon contained therein will remain after the fire, with CFs of 0.04 for low- and very-low-severity fires and up to 0.08 and 0.24 for medium- and high-severity fires (Table 10.1). CFs are even lower f standing coarse, woody debris, ranging from 0.02 for low- and very-lo severity fires to 0.04 and 0.12 for medium- and high-severity fires (Table 10.1).

p0060 Interestingly, levels of fuel consumption for woody debris, duff, and litter exhibit a surprisingly high level of similarity at different levels of fire severity, even among different forest types (Table 10.2). CFs for woody debris (including all diameter classes) averaged 0.56, 0.63, and 0.79 for low-, medium-, and high-severity fires, respectively (Table 10.2). Average duff combustion (0.46) was lower than average woody debris combustion among stands burned by low-severity fires, but it was higher in stands burned by medium- and high-severity fires, with average CFs of 0.70 and 0.90, respectively (Table 10.2). The highest rates of combustion were observed in litter biomass, which had CFs of 0.68, 0.7 0.95 for low-, medium-, and high-severity fires, respectively (Table 10.2).

s0020 10.3 LIVE FOLIAGE COMBUSTION

p0065 Estimates of live, crown foliage combustion are difficult because few studies have attempted to distinguish between crown consumption and noncombustive mortality (Wyant et al., 1986; McHugh et al., 2003; Hull Sieg et al., 2006; Campbell et al., 2007; Keyser et al., 2008). While live foliage can be consumed

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t0010 **TABLE 10.1** Constituents of Biomass Storage and Combustion Factors[a] for the 2002 Biscuit Fire in the Rogue River-Siskiyou National Forest in Southwestern Oregon

C Storage Constituent	C Storage (kg C ha^{-1})	High CF (%)	Medium CF (%)	Low CF (%)	Very Low CF (%)
Foliage					
Large conifers	3242	0.69	0.27	0.08	0.02
Large hardwoods	1698	0.58	0.29	0.12	0.03
Small conifers	1863	0.89	0.76	0.44	0.01
Small hardwoods	417	1.00	0.80	0.50	0.00
Grass and forbs	2	1.00	0.76	0.75	0.70
Branch					
Large conifers	9858	0.05	0.02	0.00	0.00
Large hardwoods	4350	0.05	0.02	0.01	0.00
Small conifers	609	0.64	0.69	0.41	0.00
Small hardwoods	579	0.79	0.63	0.40	0.00
Bark					
Large conifers	11,199	0.20	0.06	0.03	0.01
Large hardwoods	4523	0.22	0.11	0.03	0.01
Small conifers	597	0.70	0.70	0.42	0.01
Small hardwoods	69	0.79	0.63	0.40	0.00
Bole					
Large conifers	57,419	0.00	0.00	0.00	0.00
Large hardwoods	30,748	0.00	0.00	0.00	0.00
Small conifers	288	0.61	0.68	0.40	0.00
Small hardwoods	700	0.79	0.63	0.40	0.00
Dead wood					
Large standing	5927	0.12	0.04	0.02	0.02
Small standing	1642	0.61	0.68	0.40	0.00
Large downed	9324	0.24	0.08	0.04	0.04

Continued

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TABLE 10.1 Constituents of Biomass Storage and Combustion Factors for the 2002 Biscuit Fire in the Rogue River-Siskiyou National Forest in Southwestern Oregon—Cont'd

C Storage Constituent	C Storage (kg C ha^{-1})	High CF (%)	Medium CF (%)	Low CF (%)	Very Low CF (%)
Medium downed	1798	0.79	0.73	0.67	0.62
Small downed	1543	0.78	0.58	0.61	0.62
Forest floor and soil					
Litter	9499	1.00	0.76	0.75	0.70
Duff	6335	0.99	0.51	0.54	0.44
Soil to 10 cm	45,500	0.08	0.04	0.04	0.02

Litter consists of materials in the O_i horizon, and duff is in the O_e and O_a horizon. Soil is all mineral soil to a depth of 10 cm, including fine roots. For live trees, small is a <7.62 cm diameter at breast height (DBH); large is a >7.62 cm DBH. For dead wood, small is 0.51-2.54 cm, medium is 2.54-7.62 cm, and large is a >7.62 cm diameter.

np0010 [a]*Data from Campbell et al. (2007).*

by wildfires, foliage can also be scorched and damaged by direct contact with or indirect convective heating from flames, leading a yellowing or browning of foliage. Once scorched, the foliage is usually killed and subsequently falls to the ground.

p0070 Understory and shrub-layer vegetation can have a significant impact on foliage consumption, but these effects depend on species composition. In the 2002 Biscuit Fire, open conifer forests with a predominantly sclerophyllous (trees and shrubs with hard, thick leaves) shrub understory experienced the most crown mortality (Thompson and Spies, 2009). Conversely, an assessment of the foliar moisture content of several grass and nonsclerophyllous shrub species suggested the possibility that the presence of a grass and/or shrub in the understory could reduce flame height throughout most of the fire season (Agee et al., 2002). If true under field conditions of fire ignition and development, such a finding would suggest a possible caveat to the common assumption that fuels with high surface area-to-volume ratios are among the most combustible and efficiently burning fuel types. The abundance of foliage fuel found throughout densely stocked, uniform forests, however, clearly has a high probability of combustion capable of propagating fires with high subsequent mortality. In a mixed conifer system in the Sierra Nevada range, North and Hurteau (2011) examined the effects of "thin from below" treatments, in which trees of a given diameter are removed to minimize the presence of ladder fuels that could t0015 **TABLE 10.2** ~~Mortality Factors (MFs) and~~ Combustion Factors (CFs) for Woody Debris (WD), Litter, and Duff Fuels for Different Forest Species Groups and Levels of Fire Severity[a]

	Low Severity			Medium Severity			High Severity		
Dominant Vegetation	*WD CF*	*Litter CF*	*Duff CF*	*WD CF*	*Litter CF*	*Duff CF*	*WD CF*	*Litter CF*	*Duff CF*
Pinyon/juniper	0.56	0.63	0.48	0.62	0.77	0.77	0.81	0.97	0.97
Douglas-fir	0.53	0.70	0.47	0.60	0.73	0.81	0.81	0.97	0.97
Ponderosa pine	0.52	0.65	0.54	0.65	0.75	0.84	0.82	0.96	0.97
Fir/spruce/mountain hemlock	0.53	0.60	0.44	0.63	0.76	0.69	0.77	0.92	0.83
Lodgepole pine	0.68	0.50	0.21	0.77	0.56	0.33	0.96	0.72	0.42
Hemlock/sitka spruce	0.59	0.75	0.54	0.58	0.76	0.51	0.77	1.00	0.99
California mixed conifer	0.56	0.64	0.48	0.62	0.77	0.77	0.80	0.97	0.97
Elm/ash/cottonwood	0.58	0.75	0.51	0.66	0.74	0.77	0.77	1.00	0.99
Aspen/birch	0.43	0.77	0.40	0.48	0.74	0.64	0.60	1.00	0.81
Western oak	0.56	0.76	0.50	0.67	0.80	0.79	0.81	0.98	0.95
Tanoak/laurel	0.59	0.75	0.54	0.68	0.66	0.79	0.77	1.00	0.99

np0015 [a] *Data from Ghimire et al. (2012).*

propagate a crown fire. Following wildfire, differences in fire mortality between treated (53%) and untreated (97%) forest suggest that fuel reduction treatments can allow for a considerable reduction in the presence of foliage and ladder fuels throughout the stand, though this did not include the effects of direct mortality from the mechanical thinning itself, which would substantially increase overall mortality in the thinned areas.

p0075 The potential for fire to spread vertically to the forest canopy is highly dependent upon the successional stage of the forest stand. As densely stocked stands of shade-intolerant species mature, self-thinning raises the crown height, and the resulting shading discourages the development of ladder fuels, thereby reducing the probability of fire propagation from the ground fuels into the canopy (Odion et al., 2004; Perry et al., 2011). Collins and Stephens (2010) found that stands were most susceptible to high-severity reburn when they were between 17 and 30 years old (also see Chapter 1). Consequently, mature, closed conifer stands can be more resistant to foliage combustion and tree morality than their younger counterparts (Thompson and Spies, 2009, 2010). These findings bear relevance to the commonly held assumption that the probability of high severity fires tends to increase with stand age. Such assumption is often made on the premise that forests accumulate more biomass through time, and thus have more total fuel that could be burned, thereby resulting in fires of higher severity. However, the infrequent occurrence of high-severity wildfires is not necessarily the result of infrequently high amounts of forest fuel availability. For many ecosystems, it is the infrequent occurrence of extreme weather conditions that may lead to a high-severity, foliage-consuming crown fire (Perry et al., 2011).

p0080 Foliage combustion rates may thus be best thought of as a function of fire severity and the vertical strata of the foliage. CFs for grass and forbs range from 0.70 to 0.75 in very-low-/low-severity fires to 1.00 in high-severity fires, whereas the combustion of fuels of small (<7.62 cm diameter at breast height [dbh]) trees and shrubs at a slightly higher vertical strata is slightly less: CFs for low-, medium-, and high-severity fires are 0.44, 0.76, and 0.89 for conifers and 0.50, 0.80, and 1.00 for hardwoods, respectively. Estimated CFs for the foliage of large trees are, as expected, lower than the others because of the vertical distance between foliage and surface fuels, where the majority of combustion takes place. CFs for large (>7.62 cm dbh) foliage in low-, moderate- and high-severity fires are 0.09, 0.27, and 0.69 for conifers and 0.12, 0.29, and 0.58 for hardwoods, respectively (Table 10.1).

s0025 10.4 SOIL COMBUSTION

p0085 Soil represents a considerable fraction of forest carbon, comprising approximately 44% of total forest carbon storage worldwide (Bonan, 2008). Soil carbon storage is usually low among ecosystems with frequent, low-severity fire regimes, such as those found in semiarid ponderosa pine forests. Conversely, soil carbon storage can be very high in ecosystems with infrequent (i.e., a mean fire return interval of >200 years) fires. Fires of high intensity and severity typify many forests with infrequent fire regimes. Because of the high magnitude of soil carbon storage in stands with infrequent, high-severity fires, estimates of carbon emissions from wildland fires are highly sensitive to the CF used to estimate the proportion of soil carbon that is consumed. However, estimates of soil carbon combustion are difficult to obtain, particularly in high-severity wildland fires, because of the lack of prefire estimates of soil carbon content.

p0090 The process of soil carbon consumption is dominated by smoldering, as opposed to flaming, combustion. Smoldering combustion is a result of insufficient amounts of oxygen required to support flaming combustion and is most prevalent in organic soils and rotting logs. The combustion of forest soils is highly dependent on the magnitude of the temperatures they are exposed to and the duration of exposure. Agee (1993) suggested that soils can be combusted at temperatures as low as 100 °C, but laboratory-based experiments suggest that significant amounts of soil carbon volatilization require temperatures between 200 °C and 315 °C (Lide, 2004), with peak smoldering temperatures ranging from 300 °C to 600 °C (Rein et al., 2008). Work by Fernández et al. (1997) heated the top 10 cm of soil taken from a *Pinus sylvestris* stand to 150° at a gradually increasing rate (+3 °C min^{-1}), at which point the soil was heated for 30 min thereafter, yet no significant amount of soil carbon combustion was observed. Upon applying the same heating regime at temperatures of 220°, 350°, and 490°C, however, there were significant changes in the content of soil organic matter (i.e., soil carbon). Temperatures of 220°, 350°, and 490°C resulted in losses of 37%, 90%, and nearly 100%, respectively. Others have noted that shorter heating times at 350 °C resulted in a 50% weight loss after only 180 s (Almendros et al., 2003), compared with 90% at 350 °C observed by Fernández et al. (1997). Consequently, exposure to increased temperatures is highly dependent on combustion times and rates of fire spread; the relatively high rates at which fire moves across western North American landscapes, combined with the relatively limited diffusion of oxygen into the relatively nonporous soil profile, limit soil carbon emissions. CFs for soils described by Campbell et al. (2007) were 0.04 for low- and medium-severity fires and 0.08 for high-severity fires (Table 10.1).

p0095 The combustion of soils in boreal forests represents an important exception to the relatively low rates of soil carbon emissions observed in most western US forests. Turetsky et al. (2011) and Kasischke and Hoy (2012) found that the combustion of soil carbon in Alaskan boreal forests can actually constitute the majority of carbon emissions during fires, representing 54-70% of total carbon emissions. Turetsky et al. found that three factors explained most of the variation in the depth of burning/carbon consumption in the surface organic layers of black spruce forests. First, topography was a significant control: Higher fractions of consumption were observed in upland sites compared with lowland sites. Second, season of the fire was also a factor: Seasonal thawing of permafrost resulted in drier ground layers as the growing season progressed. Finally, in upland sites, fires that exhibited higher consumption occurred in the early season in years where fires had a large spatial extent compared to those in years where fires had a smaller spatial extent because of drier conditions and more extreme fire behavior.

p0100 Large amounts of biomass with long-term smoldering potential also are found in pocosin shrublands (a type of wetland with deep, sandy, and acidic soils) in the southeastern United States. While pocosin systems can have substantial amounts of combustible fuel contained in deep peat layers, they differ most notably from boreal forests in their lack of both a freeze-thaw cycle and a strong, seasonally sensitive decline in moisture as the growing season progresses. Consumption of fuel beds in these systems is poorly understood, and additional research on moisture dynamics, biogeochemical processes, and combustion is needed (Reardon et al., 2007, 2009).

s0030 10.5 BOLE BIOMASS CONSUMPTION

p0105 While many studies report tree mortality rates, relatively little on the fraction of fire-killed trees that were combusted during wildfire has been reported. In estimates of pyrogenic carbon emissions taken from the Biscuit Fire in 2002, Campbell et al. (2007) found no combustion of bole biomass among large (>7.62 cm dbh) trees, regardless of fire severity (Table 10.1). The lack of combustion for the boles of large trees seems to have been effectively mediated by the combustion of bark, which had CFs of 0.03, 0.06, and 0.20 for conifers and 0.03, 0.11, and 0.22 for hardwoods in low-, medium-, and high-severity fires, respectively. Such a finding is consistent with what is expected of fuels with low surface area-to-volume ratios (Table 10.1).

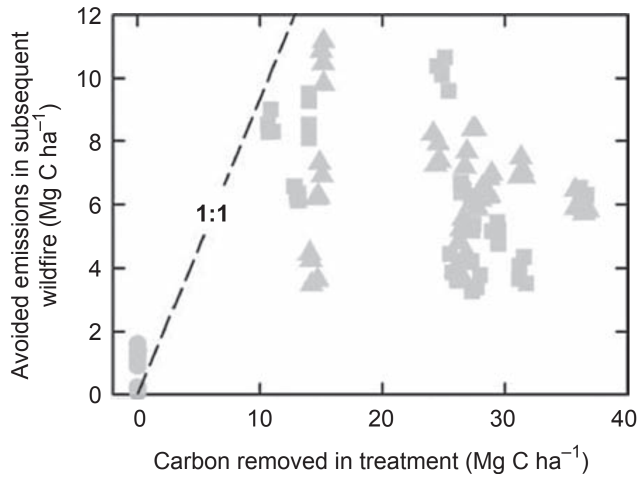
p0110 Bark CFs were much higher for small trees; for low-, medium-, and high-severity fires there were CFs of 0.42, 0.70, and 0.70 for conifers and 0.40, 0.63, and 0.79 for hardwoods, respectively (Table 10.1). As expected, the thinner bark of smaller trees, much of which was combusted, was not effective in protecting the bole biomass from combustion. Estimates of the combustion of the boles of small trees for low-, medium-, and high-severity fires were 0.40, 0.68, and 0.61 for conifers and from 0.40, 0.63, and 0.79 for hardwoods, respectively (Table 10.1). Weighted CFs for all trees, adjusted for the abundance of small tree biomass versus large tree biomass, would be approximately 0.03, 0.07, and 0.08 for low-, medium-, and high-severity fires, respectively (Campbell et al., 2007). Others have used far higher CFs for high-severity fires in modeling studies. An estimated high-severity CF of 0.30 has been used for Siberian forests (Soja et al., 2004), which may be realistic, given the small diameters prevalent in boreal forest stands. Estimates of bole CFs, however, some of which are as high as 0.30 for North American forests (Wiedinmyer et al., 2006), seem to be at odds with those estimated by Campbell et al.

(2007), given the majority of biomass is stored in boles of large trees, none of which is combusted by high-severity fires. Such estimates, if inaccurate, can result in substantial overestimates of pyrogenic carbon emissions because of the considerable stocks of carbon in bole biomass of large trees. Overall, the CFs for total forest biomass (i.e., trees, snags, shrubs, woody fuels, litter, duff, and soil), weighted according to their respective prefire biomass, were 0.13, 0.15, and 0.21 for low-, medium-, and high-severity fires, respectively, in the Biscuit Fire (Campbell et al., 2007) (Table 10.1).

s0035 10.6 FUEL REDUCTION TREATMENTS, CARBON EMISSIONS, AND LONG-TERM CARBON STORAGE

p0115 The application of fuel reduction treatments have become common in many fire-adapted forests throughout the western North America. Such treatments are intended to reduce the severity of fires, primarily out of concern over public safety in fire-prone regions, as well as ~~because of land management agencies that want~~ to minimize ~~widespread mortality in the forests within their jurisdiction~~. ~~Common fuel~~ reduction treatments ~~include~~ understory removal, whereby midstory and understory vegetation ~~is~~ removed through pruning or harvesting. ~~Anothe~~[illegible] ~~reduction treatment is~~ prescribed fire, which reduces surface fuels in orde[illegible]mit the flame height of a wildfire that might enter the stand. ~~In the field, this~~ is done by removing fuel through prescribed fire or pile burning, both of which reduce the potential magnitude of a wildfire by making it more difficult for a surface fire to ignite the canopy. The timing of prescribed fire can be central to its effectiveness. If performed *after* an understory removal treatment, it may burn any additional residue created by the treatment. ~~Performing~~ prescribed fire under cooler and moisture conditions than those experienced during the fire season is also ideal to avoid the propagation of an unplanned fire. Other fuel reduction treatments involve a partial harvest of overstory trees to limit the potential of fire to spread from crown to crown.

p0120 While such treatments can sometimes be effective in reducing fire severity, if and when fires occur in thinned areas (Rhodes and Baker, 2008), they can come at the expense of carbon storage. The majority of carbon stored in leaves, leaf litter, and duff is typically consumed by high-severity wildfire and often constitutes the majority of the carbon emissions during the a given fire, yet most of the carbon stored in forest biomass (stem wood, branches, and coarse, woody debris) remains unconsumed even by high-severity wildfires. Consequently, fuel removal via forest thinning almost always reduces carbon storage more than the additional carbon that a stand is able to store when made more resistant to wildfire. For this reason, removing large amounts of biomass to reduce the fraction by which other biomass components are consumed via combustion is inefficient (Mitchell et al., 2009). Fuel reduction treatments that involve the removal of overstory biomass (i.e., intermediate-sized and large trees) are, perhaps unsurprisingly, the most inefficient methods of reducing wildfire-related



f0015 **FIGURE 10.2** Simulated effectiveness of various fuel-reduction treatments in reducing future wildfire combustion in a ponderosa pine ~~(*P. ponderosa*)~~ forest. In general, protecting one unit of carbon (C) from wildfire combustion came at the cost of removing approximately three units of carbon in treatment. At the very lowest (least biomass removed) treatment levels, more carbon was protected from combustion than removed during treatment; however, the absolute gains were extremely low. Circles show understory removal, squares show prescribed fire, and triangles show understory removal and prescribed fire. Simulations were run for 800 years with a treatment-return interval of 10 years and a mean fire-return interval of 16 years. Forest structure and growth were modeled to represent mature, semiarid ponderosa pine forest growing in Deschutes, Oregon. Further descriptions of these simulations are given by Mitchell et al. (2009).

carbon losses because they remove large amounts of carbon for only a marginal reduction in expected fire severity (Figure 10.2).

s0040 10.7 INDIRECT SOURCES OF CARBON EMISSIONS

p0125 Our discussion thus far has focused on the *direct* effects of wildfire on carbon emissions as a result of the combustion of live vegetation, dead biomass, and soil organic matter. *Indirect* effects, by contrast, are not the result of the active combustion of biomass or soil organic matter; instead, they result from the long-term decomposition of vegetation killed in wildfire. The magnitude of indirect emissions, and the temporal scales at which they affect the net ecosystem carbon balance, vary with different fire behaviors. Most of the mortality resulting from low-severity fires is limited to understory plants, shrubs, and small trees, which do not typically constitute a significant portion of total stand carbon storage and, by extension, do not represent a significant source of carbon emissions upon decomposition. High-severity fires, by contrast, result in the near-total death of all trees within a stand, including overstory dominants. While the addition of any unburned leaf litter and fine, woody debris from fire-killed trees represent pools with relatively high turnover (10-20 years), a large pool of coarse woody debris (e.g., logs, snags) can be a significant source of carbon

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emissions (Bond-Lamberty et al., 2003), one that can ~~continue to release carbon~~ for periods of up to, and even exceeding, 100 years (Kashian et al., 2006).

p0130 Fire severity has a significant impact on postdisturbance rates of net primary production and net ecosystem production (NEP). Net primary production is the difference between photosynthesis and autotrophic (i.e., plant) respiration, whereas NEP is a measure of net ecosystem carbon uptake, defined as the difference between photosynthesis and autotrophic respiration plus heterotrophic (i.e., decomposition) respiration. Following a high-severity disturbance, rates of heterotrophic respiration are, for a period of time, far higher than rates of photosynthesis, resulting in negative NEP (Harmon et al., 2011). While indirect sources of carbon emissions following fire can be substantial, particularly following high-severity fire, the postdisturbance regrowth of a new cohort of trees is also a significant contributor to total ecosystem carbon storage and the net ecosystem carbon balance (Figure 10.3).

p0135 The amount of time required for a recently disturbed forest to shift from a source to a sink depends on fire severity, forest type, and local climate. Following high-severity wildfires, forests with low rates of productivity, such as the ponderosa pine forests of the southwestern United States, take relatively longer to make the postfire transition from carbon source to carbon sink (Ghimire et al., 2012). Dore et al. (2008) examined a ponderosa pine forest in northern Arizona 10 years after a stand-replacing fire and found it to be a moderate source of carbon (109 g carbon m^{-2} $year^{-1}$), but they observed a moderate carbon sink (164 g carbon m^{-2} $year^{-1}$) in an unburned stand nearby. The burned stand remained a source of carbon during all months of the year that were measured, even during the growing season in the summer months. Annual ecosystem respiration was 33% lower in the burned stand. The slow recovery of such stands is largely attributed to the climate, whereby cold winters combine with low springtime precipitation to limit gross primary production (GPP), whereas warm summers with periodic precipitation are conducive to respiration-driven losses of soil carbon (Dore et al., 2008). However, this analysis was based on only five plots with a 25 m radius; therefore, some caution regarding broader inferences is appropriate.

p0140 Differences in the postfire carbon balance of uptake were observed in semiarid, mixed-conifer forests of eastern Oregon. Meigs et al. (2009) found that 4-5 years after a mixed-severity fire, areas that burned at low severity were modest net carbon sinks. By contrast, ponderosa pine forests that also were affected by a low-severity fire were carbon neutral in low-severity fire areas. Differences in the recovery time to being a source of carbon once again may be because of differences in productivity; ponderosa pine forests are typically less productive than mixed-conifer forests (Franklin and Dyrness, 1973). Among areas affected by high severity fires, both ponderosa pine and mixed-conifer stands were sources of carbon emissions 4-5 years following fire. Modeled estimates of the postfire transition from carbon source to carbon sink suggest that ~40 years may necessary for low-productivity ponderosa pine forests to shift

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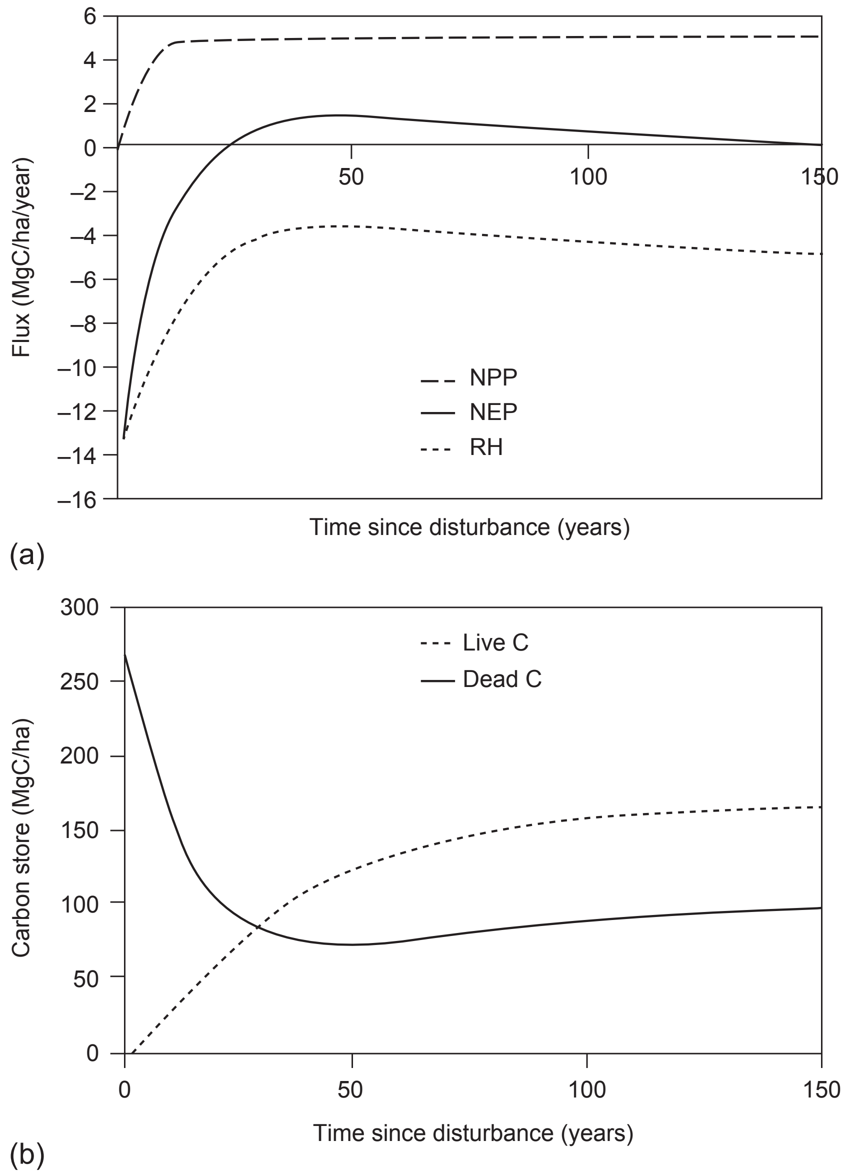

f0020 **FIGURE 10.3** The classic pattern of net primary production (NPP), heterotrophic respiration (RH), and net ecosystem production (NEP) (A) and associated carbon stores (B) following a high-severity disturbance. *(From Harmon et al. (2011).)*

from being a carbon source to a carbon sink (Ghimire et al., 2012), though this analysis did not control for the potentially confounding effect of postfire logging, which is common after high-severity fire in ponderosa pine and mixed-conifer forests (see Chapter 11). Forests with higher rates of productivity, such as coastal range Sitka spruce (*Picea sitchensis*)/western hemlock (*Tsuga* *heterophylla*) forests in the Pacific Northwest, seem to make the postfire transition from carbon source to carbon sink in a shorter amount of time than any other coniferous western forest, potentially in <30 years. Harmon et al. (2011) reviewed the scientific literature on this question for various forest types and concluded that the transition from source to sink following fire sufficiently severe to reset the successional "clock" varied from 14-50 years in forests types characteristic of the Pacific northwestern United States and 5-15 years in boreal forests. High-severity fire rotation intervals are currently several hundred years to more than 1000 years in most mixed-conifer and ponderosa pine forest regions of the western United States, however, and these rates are generally substantially lower than historical rates (see Chapter 1). Thus a long-term spatiotemporal perspective is important to understand more fully the natural disturbance dynamics in these systems (see Chapter 9).

s0045 10.8 CONCLUSIONS

p0145 The majority of carbon stored in montane forest ecosystems of western North America remains unconsumed, even in high-severity wildfires. Large carbon stores in the bole biomass of large forest trees are not consumed, and the substantial proportion of carbon stored in forest soils is only slightly consumed. Most of the carbon emissions in a wildfire are from combustion of litter, duff, and woody debris. In the 2002 Biscuit Fire, CFs for total forest biomass (i.e., trees, snags, shrubs, woody fuels, litter, duff, and soil), weighted according to their respective prefire biomass, were 0.13, 0.15, and 0.21 for low-, medium-, and high-severity fires, respectively. Such factors can be even lower among stands with a higher proportion of carbon storage in bole biomass that likewise remains unconsumed in high-severity wildfires, such as Sitka spruce (*P. sitchensis*)/Western Hemlock (*T. heterophylla*) forests in the coast range of the Pacific Northwest (Smithwick et al., 2002; Mitchell et al., 2009). The application of fuel treatments can be effective in reducing fire severity and carbon emissions, but such treatments come at the cost of a net reduction in carbon storage relative to fire alone (Mitchell et al., 2009).

p0150 Postfire carbon emissions from fire-killed biomass can be substantial for decades following wildfires. Low- or even moderate-severity fires, however, do not necessarily result in a postfire source of carbon released to the atmosphere. High-severity fire temporarily creates a source of postfire carbon emissions as a result of the decomposition of fire-killed biomass, which lessens each year with natural postfire succession of vegetation, transitioning from a carbon source to a carbon sink within 5-50 years, depending on the ecosystem. Rates of postfire recovery are highest among systems with high productivity, whereas high-severity wildfires in forests with low productivity transition from source to sink over a relatively longer timeline, though there are important limitations in the amount and scope of existing studies of these systems. ~~Future~~ research on the relationship between climatic change, disturbance regimes, and postdisturbance successional trajectories may prove to be a crucial step toward projecting the future of pyrogenic carbon emissions in mixed-severity fire regimes.

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Non-Print Items

Abstract

Among the concerns raised by climatic change is the potential for the additional release of carbon dioxide as a result of biomass combustion. Most of the carbon emissions from wildfires are from the combustion of litter, duff, and small woody debris, whereas most, if not all, of the biomass stored in the boles of large trees is not combusted. Consequently, most of the carbon stored in forests remains unconsumed, even by high-severity wildfires. Thus the application of fuel reduction treatments, while sometimes effective in reducing fire severity and carbon emissions, nearly always result in a net reduction in carbon storage. Postfire carbon emissions from the decomposition of fire-killed biomass can continue for decades, but effects of forest regrowth can exceed the losses of carbon from biomass combustion and the decomposition of fire-killed biomass within 5-50 years, depending on the ecosystem.

Keywords: Carbon sequestration; Carbon emissions; Climate change; Fuel reduction treatments; Biscuit Fire; Combustion factor; Carbon dioxide.

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Forest fuel reduction alters fire severity and long-term carbon storage in three Pacific Northwest ecosystems

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Abstract. Two forest management objectives being debated in the context of federally managed landscapes in the U.S. Pacific Northwest involve a perceived trade-off between fire restoration and carbon sequestration. The former strategy would reduce fuel (and therefore C) that has accumulated through a century of fire suppression and exclusion which has led to extreme fire risk in some areas. The latter strategy would manage forests for enhanced C sequestration as a method of reducing atmospheric CO_2 and associated threats from global climate change. We explored the trade-off between these two strategies by employing a forest ecosystem simulation model, STANDCARB, to examine the effects of fuel reduction on fire severity and the resulting long-term C dynamics among three Pacific Northwest ecosystems: the east Cascades ponderosa pine forests, the west Cascades western hemlock–Douglas-fir forests, and the Coast Range western hemlock–Sitka spruce forests. Our simulations indicate that fuel reduction treatments in these ecosystems consistently reduced fire severity. However, reducing the fraction by which C is lost in a wildfire requires the removal of a much greater amount of C, since most of the C stored in forest biomass (stem wood, branches, coarse woody debris) remains unconsumed even by high-severity wildfires. For this reason, all of the fuel reduction treatments simulated for the west Cascades and Coast Range ecosystems as well as most of the treatments simulated for the east Cascades resulted in a reduced mean stand C storage. One suggested method of compensating for such losses in C storage is to utilize C harvested in fuel reduction treatments as biofuels. Our analysis indicates that this will not be an effective strategy in the west Cascades and Coast Range over the next 100 years. We suggest that forest management plans aimed solely at ameliorating increases in atmospheric CO_2 should forgo fuel reduction treatments in these ecosystems, with the possible exception of some east Cascades ponderosa pine stands with uncharacteristic levels of understory fuel accumulation. Balancing a demand for maximal landscape C storage with the demand for reduced wildfire severity will likely require treatments to be applied strategically throughout the landscape rather than indiscriminately treating all stands.

Key words: biofuels; carbon sequestration; fire ecology; fuel reduction treatment; Pacific Northwest, USA; Picea sitchensis; Pinus ponderosa; Pseudotsuga menziesii.

Introduction

Forests of the U.S. Pacific Northwest capture and store large amounts of atmospheric CO_2, and thus help mitigate the continuing climatic changes that result from extensive combustion of fossil fuels. However, wildfire is an integral component to these ecosystems and releases a substantial amount of CO_2 back to the atmosphere via biomass combustion. Some ecosystems have experienced an increase in the amount of CO_2 released due to a century-long policy of fire suppression that has led to increased levels of fuel buildup, resulting in wildfires of uncharacteristic severity. Fuel reduction treatments have been proposed to reduce wildfire severity, but like wildfire, these treatments also reduce the C stored in forests. Our work examines the effects of fuel reduction on wildfire severity and long-term C storage to gauge the strength of the potential trade-off between managing forests for increased C storage and reduced wildfire severity.

Forests have long been referenced as a potential sink for atmospheric CO_2 (Vitousek 1991, Turner et al. 1995, Harmon et al. 1996, Harmon 2001, Smithwick et al. 2002, Pacala and Socolow 2004), and are credited with contributing to much of the current C sink in the coterminous United States (Pacala et al. 2001, Hurtt et al. 2002). This U.S. carbon sink has been estimated to be between 0.30 and 0.58 Pg C/yr for the 1980s, of which between 0.17 Pg C/yr and 0.37 Pg C/yr has been attributed to accumulation by forest ecosystems (Pacala et al. 2001). While the presence of such a large sink has been valuable in mitigating global climate change, a substantial portion of it is due to the development of understory vegetation as a result of a national policy of fire suppression (Pacala et al. 2001, Donovan and Brown 2007). Fire suppression, while capable of incurring

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short-term climate change mitigation benefits by promoting the capture and storage of atmospheric CO_2 by understory vegetation and dead fuels (Houghton et al. 2000, Tilman et al. 2000), has, in part, led to increased and often extreme fire risk in some forests, notably *Pinus ponderosa* forests (Moeur et al. 2005, Donovan and Brown 2007).

Increased C storage usually results in an increased amount of C lost in a wildfire (Fahnestock and Agee 1983, Agee 1993). Many ecosystems show the effects of fire suppression (Schimel et al. 2001, Goodale et al. 2002, Taylor and Skinner 2003), and the potential effects of additional C storage on the severity of future wildfires is substantial. In the *Pinus ponderosa* forests of the east Cascades, for example, understory fuel development is thought to have propagated crown fires that have killed old-growth stands not normally subject to fires of high intensity (Moeur et al. 2005). Various fuel reduction treatments have been recommended for risk-prone forests, particularly a reduction in understory vegetation density, which can reduce the ladder fuels that promote such severe fires (Agee 2002, Brown et al. 2004, Agee and Skinner 2005). While a properly executed reduction in fuels could be successful in reducing forest fire severity and extent, such a treatment may be counterproductive to attempts at utilizing forests for the purpose of long-term C sequestration.

Pacific Northwest forests, particularly those that are on the west side of the Cascade mountain range, are adept at storing large amounts of C. Native long-lived conifers are able to maintain production during the rainy fall and winter months, thereby out-competing shorter-lived deciduous angiosperms with a lower biomass storage capacity (Waring and Franklin 1979). Total C storage potential, or upper bounds, of these ecosystems is estimated to be as high as 829.4 Mg C/ha and 1127.0 Mg C/ha for the western Cascades and Coast Range of Oregon, respectively (Smithwick et al. 2002). Of this high storage capacity for west Cascades and Coast Range forests, 432.8 Mg C/ha and 466.3 Mg C/ha, respectively, are stored in aboveground biomass (Smithwick et al. 2002), a substantial amount of fuel for wildfires.

High amounts of wildfire-caused C loss often reflect high amounts of forest fuel availability prior to the onset of fire. Given the magnitude of such losses, it is clear that the effect of wildfire severity on long-term C dynamics is central to our understanding of the global C cycle. What is not clear is the extent to which repeated fuel removals that are intended to reduce wildfire severity will likewise reduce long-term total ecosystem C storage (TECμ). Fuel reduction treatments require the removal of woody and detrital materials to reduce future wildfire severity. Such treatments can be effective in reducing future wildfire severity, but they likewise involve a reduction in stand-level C storage. If repeated fuel reduction treatments decrease the mean total ecosystem C storage by a quantity that is greater than the difference between the wildfire-caused C loss in an untreated stand and the wildfire-caused C loss in a treated stand, the ecosystem will not have been effectively managed for maximal long-term C storage.

Our goal was to test the extent to which a reduction in forest fuels will affect fire severity and long-term C storage by employing a test of such dynamics at multi-century time scales. Our questions were as follows: (1) To what degree will reductions in fuel load result in decreases in C stores at the stand level? (2) How much C must be removed to make a significant reduction in the amount of C lost in a wildfire? (3) Can forests be managed for both a reduction in fire severity and increased C sequestration, or are these goals mutually exclusive?

Methods

Model description

We conducted our study using an ecosystem simulation model, STANDCARB (Appendix A), that allows for the integration of many forest management practices as well as the ensuing gap dynamics that may result from such practices. STANDCARB is a forest ecosystem simulation model that acts as a hybrid between traditional single-life-form ecosystem models and multi-life-form gap models (Harmon and Marks 2002). The model integrates climate-driven growth and decomposition processes with species-specific rates of senescence and stochastic mortality while incorporating the dynamics of inter- and intraspecific competition that characterize forest gap dynamics. Inter- and intraspecific competition dynamics are accounted for by modeling species-specific responses to solar radiation as a function of each species' light compensation point as well as the amount of solar radiation delineated through the forest canopy to each individual. By incorporating these processes the model can simulate successional changes in population structure and community composition without neglecting the associated changes in ecosystem processes that result from species-specific rates of growth, senescence, mortality, and decomposition.

STANDCARB performs calculations on a monthly time step and can operate at a range of spatial scales by allowing a multi-cell grid to capture multiple spatial extents, as both the size of an individual cell and the number of cells in a given grid can be designated by the user. We used a 20×20 cell matrix for all simulations (400 cells total), with 15×15 m cells for forests of the west Cascades and Coast Range and 12×12 m cells for forests of the east Cascades. Each cell allows for interactions of four distinct vegetation layers, represented as upper canopy trees, lower canopy trees, a species-nonspecific shrub layer, and a species-nonspecific herb layer. Each respective vegetation layer can have up to seven live pools, eight detrital pools, and three stable C pools. For example, the upper and lower tree layers comprise seven live pools: foliage, fine roots, branches, sapwood, heartwood, coarse roots, and heart-rot, all of which are transferred to a detrital pool following mortality. Dead wood is separated into snags and logs to capture the effects of spatial position on microclimate. After detrital materials have undergone significant decomposition, they can contribute material to three increasingly decay-resistant, stable C pools: stable foliage, stable wood, and stable soil. Charcoal is created in both prescribed fires and wildfires and is thereafter placed in a separate pool with high decay resistance. Additional details on the STANDCARB model can be found in Appendix A.

Fire processes

We generated exponential random variables to assign the years of fire occurrence (sensu Van Wagner 1978) based on the literature estimates (see experimental design for citations) of mean fire return intervals (MFRI) for different regions in the U.S. Pacific Northwest. The cumulative distribution for our negative exponential function is given in Eq. 1 where X is a continuous random variable defined for all possible numbers x in the probability function P, and λ represents the inverse of the expected time $E[X]$ for a fire return interval given in Eq. 2:

$$P\{X \leq x\} = \int_0^x \lambda e^{-\lambda x} dx \qquad (1)$$

where

$$E[X] = \frac{1}{\lambda}. \qquad (2)$$

Fire severities in each year generated by this function are cell specific, as each cell is assigned a weighted fuel index calculated from fuel accumulation within that cell and the respective flammability of each fuel component, the latter of which is derived from estimates of wildfire-caused biomass consumption (see Fahnestock and Agee 1983, Covington and Sackett 1984, Agee 1993). Fires can increase (or decrease) in severity depending on how much the weighted fuel index of a given cell exceeds (or falls short of) the fuel level thresholds for each fire severity class (T_{light}, T_{medium}, T_{high}, and T_{max}), and the probability values for the increase or decrease in fire severity (P_{i} and P_{d}). For example, while the natural fire severity of many stands of the west Cascades can be described as high severity, other stands of the west Cascades have a natural fire severity that can be best described as being of medium severity (~60–80% overstory tree mortality) (Cissel et al. 1999). For these stands, medium-severity fires are scheduled to occur throughout the simulated stand and can increase to a high-severity fire depending on the extent to which the weighted fuel index in a cell exceeds the threshold for a high-severity fire, as greater differences between the fuel index and the fire severity threshold will increase the chance of a change in fire severity. Conversely, medium-severity fires may decrease to a low-severity fire if the fuel index is sufficiently below the threshold for a medium-severity fire. High-severity fires are likely to become medium-severity fires if the weighted fuel index within a given cell falls sufficiently short of the threshold for a high-severity fire, and low-severity fires are likely to become medium severity if the weighted fuel index in a given cell is sufficiently greater than the threshold for a medium-severity fire. Fuel level thresholds were set by monitoring fuel levels in a large series of simulation runs where fires were set at very short intervals to see how low fuel levels needed to be to create a significant decrease in expected fire severity. We note that, like fuel accumulation, the role of regional climate exerts significant influence on fire frequency and severity, and that our model does not attempt to directly model these effects. We suspect that an attempt to model the highly complex role of regional climate data on fine-scale fuel moisture, lightning-based fuel ignition, and wind-driven fire spread adds uncertainties into our model that might undermine the precision and applicability of our modeling exercise. For that reason we incorporated data from extensive fire history studies to approximate the dynamics of fire frequency and severity.

Final calculations for the expected stand fire severity $E[F_{\mathrm{s}}]$ at each fire are performed as follows:

$$E[F_{\mathrm{s}}] = \frac{100}{C} \sum_{i=1}^{n} c_{i(\mathrm{L})} m_{i(\mathrm{L})} + c_{i(\mathrm{M})} m_{i(\mathrm{M})} + c_{i(\mathrm{H})} m_{i(\mathrm{H})} \qquad (3)$$

where C is the number of cells in the stand matrix and $c_{i(\mathrm{L})}$, $c_{i(\mathrm{M})}$, and $c_{i(\mathrm{H})}$ are the number of cells with light, medium, and high-severity fires, and $m_{i(\mathrm{L})}$, $m_{i(\mathrm{M})}$, and $m_{i(\mathrm{H})}$represent fixed mortality percentages for canopy tree species for light, medium, and high-severity fires, respectively. This calculation provides an approximation of the number of upper-canopy trees killed in the fire. The resulting expected fire severity calculation $E[F_{\mathrm{s}}]$ is represented on a scale from 0 to 100, where a severity index of 100 indicates that all trees in the simulated stand were killed.

Our approach at modeling the effectiveness of fuel reduction treatments underscores an important trade-off between fuel reduction and long-term ecosystem C storage by incorporating the dynamics of snag creation and decomposition. Repeated fuel reduction treatments may result in a reduction in long-term C storage, but it is possible that if such treatments are effective in reducing tree mortality, they may also offset some of the C losses that would be incurred from the decomposition of snags that would be created in a wildfire of higher severity. STANDCARB accounts for these dynamics by directly linking expected fire severity with a fuel accumulation index that can be altered by fuel reduction treatments while also incorporating the decomposition of snags as well as the time required for each snag to fall following mortality.

Total ecosystem C storage (TEC) is calculated by summing all components of C (live, dead, and stable). For each replicate (i = 1, 2, ... 5) and for each period

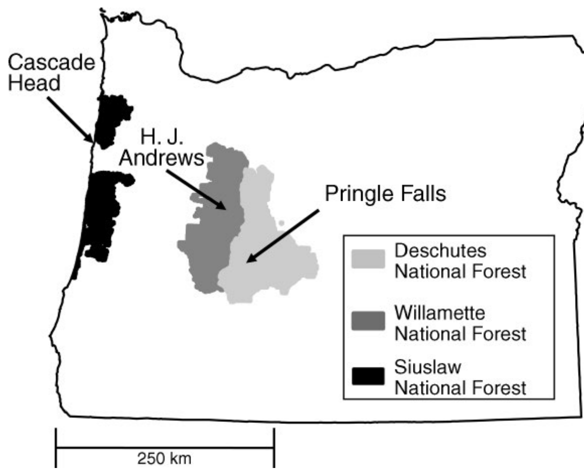

FIG. 1. Site locations in Oregon. Pringle Falls is our representative site for the east Cascades, H. J. Andrews is our representative site for the west Cascades, and Cascade Head is our representative site for the Coast Range.

between fires ($x = 1, 2, \ldots P_i$), the mean total ecosystem C storage (TEC_μ) is calculated by averaging the yearly TEC values ($k = 1, 2, \ldots R_x$).

$$\mathrm{TEC}_{\mu(i,x)} = \frac{1}{R}\sum_{k=1}^{R}\mathrm{TEC}_{(i,x,k)}.$$

Aggregating TEC_μ values in this manner permits the number of TEC_μ values to be the same as the number of $E[F_s]$ values, permitting a PerMANOVA analysis to be performed on $E[F_s]$ and TEC_μ.

Fuel reduction processes

STANDCARB's fire module allows for scheduled prescribed fires of a given severity (light, medium, high) to be simulated in addition to the nonscheduled wildfires generated from the aforementioned exponential random variable function. In addition to simulating the prescribed fire method of fuel reduction, STANDCARB has a harvest module that permits cell-by-cell harvest of trees in either the upper or lower canopy. This module allows the user to simulate understory removal or overstory thinning treatments on a cell-by-cell basis. Harvested materials can be left in the cell as detritus following cutting or can be removed from the forest, allowing the user to incorporate the residual biomass that results from harvesting practices. STANDCARB can also simulate the harvest of dead salvageable materials such as logs or snags that have not decomposed beyond the point of being salvageable.

Site descriptions

We chose the *Pinus ponderosa* stands of the Pringle Falls Experimental Forest as our representative for east Cascades forests (Youngblood et al. 2004). Topography in the east Cascades consists of gentle slopes, with soils derived from aerially deposited dacite pumice. The *Tsuga heterophylla–Pseudotsuga menziesii* stands of the H. J. Andrews Experimental Forest were chosen as our representative of west Cascades forests (Greenland 1994). Topography in the west Cascades consists of slope gradients that range from 20% to 60% with soils that are deep, well-drained dystrochrepts. The *Tsuga heterophylla–Picea sitchensis* stands of the Cascade Head Experimental Forest were chosen as our representative of Coast Range forests. We note that most of the Oregon Coast Range is actually composed of *Tsuga heterophylla–Pseudotsuga menziesii* community types, similar to much of the west Cascades. *Tsuga heterophylla–Picea sitchensis* communities occupy a narrow strip near the coast, due to their higher tolerance for salt spray, higher soil moisture optimum, and lower tolerance for drought compared to forests dominated by *Pseudotsuga menziesii* (Minore 1979), and we incorporate this region in order to gain insight into this highly productive ecosystem. Topography in the Cascade Head Experimental Forest consists of slope gradients of ~10% with soils that are silt loams to silt clay loams derived from marine siltstones. Site locations are shown in Fig. 1 and are located within three of the physiographic regions of Oregon and Washington as designated by Franklin and Dyrness (1988). Additional site data are shown in Table 1.

Experimental design

The effectiveness of forest fuel reduction treatments is often, if not always, inversely related to the time since their implementation. For this reason, our experiment incorporated a factorial blocking design where each ecosystem was subjected to four different frequencies of each fuel reduction treatment. We also recognize the fact that fire return intervals can exhibit substantial variation within a single watershed, particularly those with a high degree of topographic complexity (Agee 1993, Cissel et al. 1999), so we examined two likely fire regimes for each ecosystem. Historic fire return intervals may become unreliable predictors of future fire intervals (Westerling et al. 2006); thus ascertaining the differences in TEC_μ that result from two fire regimes might be a useful metric in gauging C dynamics resulting from fire regimes that may be further altered as a result of continued global climate change.

We based the expected fire return time in Eqs. 1 and 2 on historical fire data for our forests based on the following studies. Bork (1985) estimated a mean fire return interval of 16 years for the east Cascades *Pinus ponderosa* forests, and we also considered a mean fire return interval of 8 years for this system. Cissel et al. (1999) reported mean fire return intervals of 143 and 231 years for forests of medium- and high-severity (stand-replacing) fire regimes, respectively, among the *Tsuga heterophylla–Pseudotsuga menziesii* forests of the west Cascades. Less is known about the fire history of the Coast Range, which consists of *Tsuga heterophylla–Pseudotsuga menziesii* communities in the interior and *Tsuga heterophylla–Picea sitchensis* communities occu-

TABLE 1. Site characteristics (from Smithwick et al. 2002).

Site characteristic	Pringle Falls	H. J. Andrews	Cascade Head
Vegetation	PIPO	TSHE–PSME	TSHE–PISI
Elevation (m)	1359	785	287
Mean annual temperature (°C)	5.5	8.4	8.6
Mean annual precipitation (mm)	544	2001	2536
Soil porosity	sandy loam	loam	loam
Mean C storage potential (Mg C/ha)	183	829	1127

Note: Species codes: PIPO, *Pinus ponderosa*; TSHE, *Tsuga heterophylla*; PSME, *Pseudotsuga menziesii*; PISI, *Picea sitchensis*.

pying a narrow edge of land along the Oregon Coast. Work by Impara (1997) in the interior region of the Coast Range suggested a natural fire return interval (expected fire return time) of 271 years in the *Tsuga heterophylla–Pseudotsuga menziesii* zone, and Long et al. (1998) reported lake-derived charcoal sediment-based estimates of mean fire return interval for the Coast Range forests to be fairly similar, at 230 years. However, the *Tsuga heterophylla–Picea sitchensis* community type dominant in our study area of the Cascade Head Experimental Forest has little resistance to fire, and thus rarely provides a dendrochronological record. We estimated a mean fire return interval of 250 years as one fire return interval for a high-severity fire, derived from interior Coast Range natural fire return interval estimates, and also included another high-severity fire regime with a 500-year mean fire return interval in our analysis.

It is important to note that while the forests of the east Cascades exhibit a significant and visible legacy of effects from a policy of fire suppression, many of the mean fire return intervals for the forests of the west Cascades and Coast Range exceed the period of fire suppression (~100 years), and these forests in the west Cascades and Coast Range will not necessarily exhibit uncharacteristic levels of fuel accumulation (Brown et al. 2004). However, the potential lack of an uncharacteristic amount of fuel accumulation does not necessarily preclude these forests from future fuel reduction treatments or harvesting; thus we have included these possibilities in our analysis. The frequencies at which fuel reduction treatments are applied were designed to be reflective of literature-derived estimates of each ecosystem's mean fire return intervals, since forest management agencies are urged to perform fuel reduction treatments at a frequency reflective of the fire regimes and ecosystem-specific fuel levels (Franklin and Agee 2003, Dellasala et al. 2004). Treatment frequencies for the Coast Range and west Cascades were 100, 50, 25 years, plus an untreated control group, while treatment frequencies in the east Cascades were 25, 10, and 5 years, and an untreated control group.

We incorporated six different types of fuel reduction treatments largely based on those outlined in Agee (2002), Hessburg and Agee (2003), and Agee and Skinner (2005). Treatments 2–5 were taken directly from the authors' recommendations in these publications, treatment 1 was derived from the same principles used to formulate those recommendations, and treatment 6, clear-cutting, was not recommended in these publications but was incorporated into our analysis because it is a common practice in many Pacific Northwest forests. Treatments 1–4 were applied to all ecosystems, while treatments 5 and 6 were applied only to the west Cascades and Coast Range forests, as such treatments would be unrealistic at the treatment intervals necessary to reduce fire severity in the high-frequency fire regimes of the east Cascades *Pinus ponderosa* forests. Note that these treatments and combinations thereof are not necessarily utilized in each and every ecosystem. Managers of forests on the Oregon Coast, for example, would be unlikely to use prescribed fire as a fuel reduction technique. Our experimental design simply represents the range of all possible treatments that can be utilized for fuel reduction and is applied to all ecosystems purely for the sake of consistency.

1. *Salvage logging (SL)*.—The removal of large woody surface fuels limits the flame length of a wildfire that might enter the stand. Our method of ground fuel reduction entailed a removal of 75% of salvageable large woody materials in the stand. Our definition of salvage logging includes both standing and downed salvageable materials (sensu Lindenmayer and Noss 2006).

2. *Understory removal (UR)*.—Increasing the distance from surface fuels to flammable crown fuels will reduce the probability of canopy ignition. This objective can be accomplished through pruning, prescribed fire, or the removal of small trees. We simulated this treatment in STANDCARB by removing lower canopy trees in all cells.

3. *Prescribed fire (PF)*.—The reduction of surface fuels limits the flame length of a wildfire that might enter the stand. In the field, this is done by removing fuel through prescribed fire or pile burning, both of which reduce the potential magnitude of a wildfire by making it more difficult for a surface fire to ignite the canopy (Scott and Reinhardt 2001). We implemented this treatment in STANDCARB by simulating a prescribed fire at low severity for all cells.

4. *Understory removal and prescribed fire (UR + PF)*.—This treatment is a combination of treatments 2 and 3, where lower canopy trees were removed (treatment 2) before a prescribed fire (treatment 3) the following year for all cells.

5. *Understory removal, overstory thinning, and prescribed fire (UR + OT + PF)*.—A reduction in crown density by thinning overstory trees can make crown fire spread less probable (Agee and Skinner 2005) and can reduce potential fuels by decreasing the amount of biomass available for accumulation on the forest floor. Some have suggested that such a treatment will be effective only if used in conjunction with UR and PF (Perry et al. 2004). We simulated this treatment in STANDCARB by removing all lower canopy trees (treatment 2), removing upper canopy trees in 50% of the cells, and then setting a prescribed fire (treatment 3) the following year. This treatment was excluded from the east Cascades forests because it would be unrealistic to apply it at intervals commensurate with the high-frequency fires endemic to that ecosystem.

6. *Understory removal, overstory removal, and prescribed fire (clear-cutting) (UR + OR + PF)*.—Clear-cutting is a common silvicultural practice in the forests of the Pacific Northwest, notably on private lands in the Oregon Coast Range (Hobbs et al. 2002), and we included it in our analysis for two ecosystems (west Cascades and Coast Range) simply to gain insight into the effects of this practice on long-term C storage and wildfire severity. We simulated clear-cutting in STANDCARB by removing all upper and lower canopy trees, followed by a prescribed burn the following year. This treatment was excluded from the east Cascades forests because it would be unrealistic to apply it at intervals commensurate with the high-frequency fires endemic to that ecosystem.

7. *Control group*.—Control groups had no treatments performed on them. The only disturbances in these simulations were the same wildfires that occurred in every other simulation with the same MFRI.

In sum, our east Cascades analysis tested the effects of four fuel reduction treatment types, four treatment frequencies, including one control group, and two site mean fire return intervals (MFRI = 8 years, MFRI = 16 years). Our analysis of west Cascades and Coast Range forests tested the effects of six fuel reduction treatment types, four treatment frequencies, including one control group, and two site mean fire return intervals (MFRI = 143 years, MFRI = 230 years for the west Cascades, MFRI = 250 years, MFRI = 500 years for the Coast Range) on expected fire severity and long-term C dynamics. This design resulted in 32 combinations of treatment types for the east Cascades and 48 combinations of treatment types and frequencies for each fire regime in the west Cascades and Coast Range, with each treatment combination in each ecosystem replicated five times.

Biofuel considerations

Future increases in the efficiency of producing biofuels from woody materials may reduce potential trade-offs between managing forests for increased C storage and reduced wildfire severity. Much research is currently underway in the area of lignocellulase-based (as opposed to sugar- or corn-based) biofuels (Schubert 2006). If this area of research yields efficient methods of utilizing woody materials directly as an energy source or indirectly by converting them into biofuels such as ethanol, fuels removed from the forest could be utilized as an energy source and thus act as a substitute for fossil fuels by adding only atmosphere-derived CO_2 back to the atmosphere. However, the conversion of removed forest biomass into biofuels will only be a useful method of offsetting fossil fuel emissions if the amount of C stored in an unmanaged forest is less than the sum of managed stand TEC_{μ}, and the amount of fossil fuel emissions averted by converting removed forest biomass from a stand of identical size into biofuels over the time period considered. We performed an analysis on the extent to which fossil fuel CO_2 emissions can be avoided if we were to use harvested biomass directly for fuel or indirectly for ethanol production. We recognize that many variables need to be considered when calculating the conversion efficiencies of biomass to biofuels, such as the amount of energy required to harvest the materials, inefficiencies in the industrial conversion process, and the differences in efficiencies of various energy sources that exist even after differences in potential energy are accounted for. Rather than attempt to predict the energy expended to harvest the materials, the future of the efficiency of the industrial conversion process, and differences in energy efficiencies, we simply estimated the maximum possible conversion efficiency that can be achieved, given the energy content of these materials. The following procedure was used to estimate the extent to which fossil fuel CO_2 emissions can be avoided by substituting harvested biofuels as an energy source:

1) Estimate the mean annual biomass removal that results from intensive fuel reduction treatments.

2) Calculate the ratio of the amount of potential energy per unit C emissions for biofuels (both woody and ethanol) to the amount of energy per unit C emissions for fossil fuels.

3) Multiply the potential energy ratios by the mean annual quantity of biomass harvested to calculate the mean annual C offset by each biofuel type for each forest.

4) Calculate the number of years necessary for biofuels production to result in an offset of fossil fuel C emissions. This procedure was performed for two land-use histories: managed second-growth forests, and old-growth forests converted to managed second-growth forests.

Calculations for each ecosystem are shown in Appendix B.

Simulation spin-up

STANDCARB was calibrated to standardized silvicultural volume tables for Pacific Northwest stands. We then calibrated it to permanent study plot data from three experimental forests in the region (Fig. 1) to

TABLE 2. Treatment abbreviations.

Treatment abbreviation	Treatment
SL	salvage logging
UR	understory tree removal
PF	prescribed fire
UR + PF	understory tree removal + prescribed fire
UR + PF + OT	understory removal + prescribed fire + overstory thinning
UR + PF + OR	understory removal + prescribed fire + overstory removal

incorporate fuel legacies, which were taken from a 600-year spin-up simulation with fire occurrences generated from the exponential distribution in Eq. 1, where λ was based on each ecosystem's mean fire return interval. Spin-up simulations were run prior to the initiation of each series of fuel reduction treatments, and simulations were run for a total of 800 years for forests of the east Cascades, and a total of 1500 years for simulations of the west Cascades and Coast Range.

Data analysis

We employed a nonparametric multivariate analysis of variance, PerMANOVA (Anderson 2001), to test group-level differences in the effects of fuel reduction frequency and type on mean total ecosystem C storage and expected fire severity. PerMANOVA employs a test statistic for the F ratio that is similar to that of an ANOVA calculated using sum of squares, but unlike an ANOVA, PerMANOVA calculates sums of squares from distances among data points rather than from differences from the mean. PerMANOVA was used instead of a standard MANOVA because it was highly unlikely that our data would meet the assumptions of a parametric MANOVA. PerMANOVA analysis treated fuel reduction treatment type and treatment frequency as fixed factors within each respective fire regime for each ecosystem simulated. The null hypothesis of no treatment effect for different combinations of these factors on TEC_{μ} and $E[F_s]$ was tested by permuting the data into randomly assigned sample units for each combination of factors so that the number of replicates within each factor combination were fixed. Each of our 12 PerMANOVA tests incorporated 10 000 permutations using a Euclidian distance metric, and multiple pairwise comparison testing for differences among treatment types and treatment frequencies was performed when significant differences were detected (i.e., $P < 0.05$).

RESULTS

Results of the PerMANOVA tests indicate that mean expected fire severity ($E[F_s]$) and mean total ecosystem C storage (TEC_{μ}) were significantly affected by fuel reduction type ($P < 0.0001$), frequency ($P < 0.0001$), and interactions between type and frequency ($P < 0.0001$) in all three ecosystems. These results were significant for type, frequency, and interaction effects even when clear-cutting was excluded from the analysis for the west Cascades and Coast Range simulations, just as it was a priori for simulations of the east Cascades. When the PerMANOVA was performed on only one of our response variables ($E[F_s]$ or TEC_{μ}), groupwise comparisons of effects of treatment type showed that the most significant effects of treatment and frequency were related to TEC_{μ}. TEC_{μ} was strongly affected by treatment frequency for each fire regime in each ecosystem ($P < 0.0001$) and consistently showed an inverse relationship to the quantity of C removed in a given fuel reduction treatment, and was thus highly related to treatment type. $E[F_s]$, similar to TEC_{μ}, showed significant relationships with treatment frequency for all three ecosystems ($P < 0.0001$), with statistically significant differences among most treatment types. Boxplots of TEC_{μ} and $E[F_s]$ for each treatment type in each fire regime for each ecosystem are shown in Appendix C.

Fuel reduction treatments in east Cascades simulations reduced TEC_{μ} with the exception of one treatment type; UR treatments (see Table 2 for acronym descriptions) in these systems occasionally resulted in additional C storage compared to the control group. These differences were very small (0.6–1.2% increase in TEC_{μ}) but statistically significant (Student's paired t test, $P < 0.05$) for the treatment return interval of 10 years in the light fire severity regime No. 1 (MFRI = 8 years) and for all treatment return intervals in light fire severity regime No. 2 (MFRI = 16 years). The fuel reduction treatment that reduced TEC_{μ} the least was SL, which, depending on treatment frequency and fire regime, stored between 93% and 98% of the control group, indicating that there was little salvageable material. UR + PF, depending on treatment frequency and fire regime, resulted in the largest reduction of TEC_{μ} in east Cascades forests, storing between 69% and 93% of the control group.

Simulations of west Cascades and Coast Range forests showed a decrease in C storage for all treatment types and frequencies. Fuel reduction treatments with the smallest effect on TEC_{μ} were either SL or UR, which were nearly the same in effect. The treatment that most reduced TEC_{μ} was UR + OT + PF. Depending on treatment frequency and fire regime, this treatment resulted in C storage of between 50% and 82% of the control group for the west Cascades, and between 65% and 88% of the control group for the Coast Range. Simulations with clear-cutting (UR + OR + PF), depending on application frequency and fire regime, resulted in C storage that was between 22% and 58% of

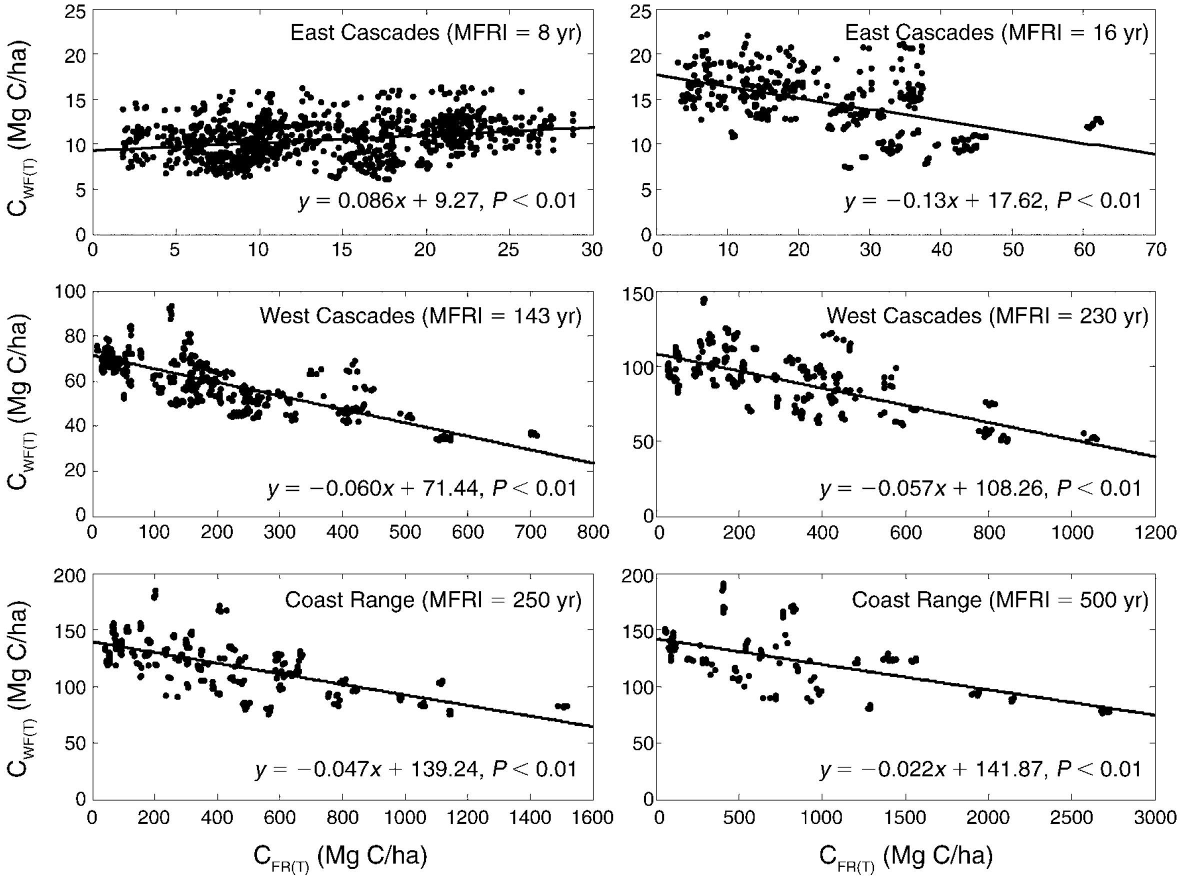

FIG. 2. Scatterplots of C removed in fuel reduction treatments between wildfires $C_{FR(T)}$ (representing fuel reduction [treatment]) and C lost in wildfires $C_{WF(T)}$ for the east Cascades, west Cascades, and Coast Range. Notice the differences in the axes scales. Also note the downward sloping trend for all ecosystems except for the east Cascades where MFRI = 8 years.

the control group for the west Cascades and between 44% and 87% of the control group for the Coast Range.

Similar to TEC_{μ}, $E[F_s]$ was significantly affected by fuel reduction treatments. Fuel reduction treatments were effective in reducing $E[F_s]$ for all simulations. UR treatments had the smallest effect on $E[F_s]$ in the east Cascades simulations and $E[F_s]$ in the east Cascades simulations was most affected by combined UR + PF treatments applied every five years, which reduced $E[F_s]$ by an average of 6.01 units (units range from 0 to 100, see Eq. 3) for stands with an MFRI = 8 years and by 11.08 units for stands with an MFRI = 16 years. In the west Cascades and Coast Range, $E[F_s]$ was least affected by UR treatments, similar to the east Cascades simulations. The most substantial reductions in $E[F_s]$ were exhibited by treatments that removed overstory as well as understory trees, as in treatments UR + OT + PF and UR + OR + PF. In the west Cascades simulations, depending on treatment frequency, $E[F_s]$ was reduced by an average of 11.72–15.68 units where the MFRI = 143 years and by an average of 3.92–26.42 units where the MFRI = 230 years when UR + OT + PF was applied. When UR + OT + PF was applied to the Coast Range, $E[F_s]$ was reduced by an average of 7.06–23.72 units where the MFRI = 250 years and by an average of 1.95–20.62 units where the MFRI = 500 years, depending on treatment frequency. Some UR + OR + PF treatments, when applied at a frequency of 25 years, resulted in $E[F_s]$ that was higher than that seen in UR + OT + PF in spite of lower TEC_{μ} in UR + OT + PF. A result such as this is most likely due to an increased presence of lower canopy tree fuels as a consequence of the increased lower stratum light availability that follows a clear-cut, as lower canopy tree fuels are among the highest weighted fuels in our simulated stands.

Modeled estimates of $E[F_s]$ were reflective of the mean amounts of C lost in a wildfire ($\bar{C}_{WF}$). $\bar{C}_{WF}$ was lower in the stands simulated with fuel reduction treatments compared to the control groups, with the exception of the east Cascades stands subjected to understory removal. Reductions in the amount of C lost in a wildfire, depending on treatment type and frequency, were as much as 50% in the east Cascades, 57% in the west Cascades, and 50% in the Coast Range. In the east Cascades simulations, amounts lost in wildfires were inversely related to the amounts of C removed in an average fire return interval for each ecosystem (Fig. 2), except for the Light Fire Regime No. 1 (MFRI = 8 years). Simulations in this fire regime revealed a slightly increasing amount of C lost in wildfires with increasing amounts removed, though amounts removed were nonetheless larger than the amounts lost in a typical wildfire.

Biofuels

Biofuels cannot offset the reductions in TEC_μ resulting from fuel reduction, at least not over the next 100 years. For example, our simulation results suggest that an undisturbed Coast Range *Tsuga heterophylla–Picea sitchensis* stand (where MFRI = 500 years) has a TEC_μ of 1089 Mg C/ha. By contrast, a Coast Range stand that is subjected to UR + OT + PF every 25 years has a TEC_μ of 757.30 Mg C/ha. Over a typical fire return interval of 450 years (estimated MFRI was 500 years, MFRI generated from the model was 450 years) this stand has 1107 Mg C/ha removed, a forest fuel/biomass production of 2.46 Mg C$\cdot$ha$^{-1}\cdot$yr^{-1}, which amounts to emissions of 1.92 Mg C$\cdot$ha$^{-1}\cdot$yr^{-1}and 0.96 Mg C$\cdot$ha$^{-1}\cdot$yr^{-1} that can be avoided by substituting biomass and ethanol, respectively, for fossil fuels (see calculations in Appendix B). This means that it would take 169 years for C offsets via solid woody biofuels and 339 years for C offsets via ethanol production before ecosystem processes result in net C storage offsets (see Fig. 3). Converting Coast Range old-growth forest to second-growth forest reduces the amount of time required for atmospheric C offsets to 34 years for biomass and 201 years for ethanol, and like all other biofuel calculations in our analysis, these are assuming a perfect conversion of potential energies. West Cascades *Tsuga heterophylla–Pseudotsuga menziesii* ecosystems (where MFRI = 230 years) that are subjected to UR + OT + PF every 25 years would require 228 years for C offsets using biomass as an offset of fossil-fuel-derived C and 459 years using ethanol. Converting west Cascades old-growth forest to second-growth forest reduces the amount of time required for atmospheric C offsets to 107 years for biomass fuels and 338 years for ethanol. Simulations of east Cascades *Pinus ponderosa* ecosystems had cases where stands treated with UR stored more C than control stands, implying that there is little or no trade-off in managing stands of the east Cascades for both fuel reduction and long-term C storage.

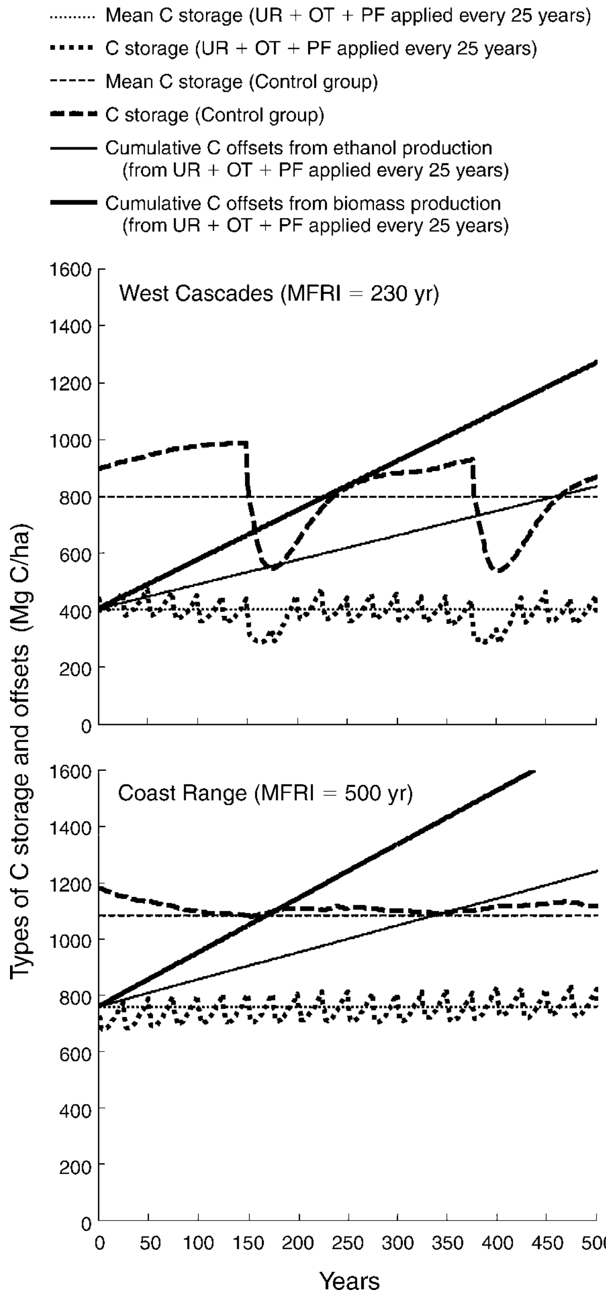

FIG. 3. Time series plots of C storage, mean C storage, and biofuels offsets for control groups and fuel reduction treatment UR + OT + PF (understory removal + overstory thinning + prescribed fire) applied to a second-growth forest every 25 years for the west Cascades and Coast Range. East Cascades simulations were excluded from this plot because there was little or no trade-off incurred in managing these forests for both fuel reduction and C sequestration.

DISCUSSION

We employed an ecosystem simulation model, STANDCARB, to examine the effects of fuel reduction on expected fire severity and long-term C dynamics in three Pacific Northwest ecosystems: the *Pinus ponderosa* forests of the east Cascades, the *Tsuga heterophylla–Pseudotsuga menziesii* forests of the west Cascades, and the *Tsuga heterophylla–Picea sitchensis* forests of the Coast Range. Our fuel reduction treatments for east Cascades forests included salvage logging, understory removal, prescribed fire, and a combination of understory removal and prescribed fire. West Cascades and Coast Range simulations included these treatments as well as a combination of understory removal, overstory thinning, and prescribed fire. We also examined the effects of clear-cutting followed by prescribed fire on expected fire severity and long-term C storage in the west Cascades and Coast Range.

Our results suggest that fuel reduction treatments can be effective in reducing fire severity, a conclusion that is shared by some field studies (Stephens 1998, Pollet and

Omi 2002, Stephens and Moghaddas 2005) and modeling studies (Fulé et al. 2001). However, fuel removal almost always reduces C storage more than the additional C that a stand is able to store when made more resistant to wildfire. Leaves and leaf litter can and do have the majority of their biomass consumed in a high-severity wildfire, but most of the C stored in forest biomass (stem wood, branches, coarse woody debris) remains unconsumed even by high-severity wildfires. For this reason, it is inefficient to remove large amounts of biomass to reduce the fraction by which other biomass components are consumed via combustion. Fuel reduction treatments that involve a removal of overstory biomass are, perhaps unsurprisingly, the most inefficient methods of reducing wildfire-related C losses because they remove large amounts of C for only a marginal reduction in expected fire severity. For example, total biomass removal from fuel reduction treatments over the course of a high-severity fire return interval (MFRI = 230 years) in the west Cascades could exceed 500 Mg C/ha while reducing wildfire-related forest biomass losses by only ~70 Mg C/ha in a given fire (Fig. 2). Coast Range forests could have as much as 2000 Mg C/ha removed over the course of an average fire return interval (MFRI = 500 years), only to reduce wildfire-related biomass combustion by ~80 Mg C/ha (Fig. 2).

East Cascades simulations also showed a trend of decreasing $E[F_s]$ with increasing biomass removal, though a higher TEC_μ was seen in some understory removal treatments compared to control groups. We believe that the removal of highly flammable understory vegetation led to a reduction in overall fire severity that consequently lowered overall biomass combustion, thereby allowing increased overall C storage. Such a result may be indicative of actual behavior under field conditions, but the very low magnitude of the differences between the treated groups and the control group (0.6%–1.2%) suggests caution in assuming that understory removal in this or any ecosystem can be effective in actually increasing long-term C storage. Furthermore, we recognize that the statistically significant differences between the treated and control groups are likely to overestimate the significance of the differences between groups that would occur in the field, as the differences we are detecting are modeled differences rather than differences in field-based estimates. Field-based estimates are more likely to exhibit higher inter- and intrasite variation than modeled estimates, even when modeled estimates incorporate stochastic processes, such as those in STANDCARB. Our general findings, however, are nonetheless consistent with many of the trends revealed by prior field-based research on the effects of fuel reduction on C storage (Tilman et al. 2000), though differences between modeled and field-based estimates are also undoubtedly apparent throughout other comparisons of treated and control stands in our study.

We note an additional difference that may exist between our modeled data and field conditions. Our study was meant to ascertain the long-term average C storage (TEC_μ) and expected fire severities ($E[F_s]$) for different fuel reduction treatment types and application frequencies, a goal not be confused with an assessment of exactly what treatments should be applied at the landscape level in the near future. Such a goal would require site-specific data on the patterns of fuel accumulation that have occurred in lieu of the policies and patterns of fire suppression that have been enacted in the forests of the Coast Range, west Cascades, and east Cascades for over a century. We did not incorporate the highly variable effects of a century-long policy of fire suppression on these ecosystems, as we know of no way to account for such effects in a way that can be usefully extrapolated for all stands in the landscape. *Pinus ponderosa* forests may exhibit the greatest amount of variability in this respect, as they are among the ecosystems that have been most significantly altered as a result of fire suppression (Veblen et al. 2000, Schoennagel et al. 2004, Moeur et al. 2005). Furthermore, additional differences may be present in our estimates of soil C storage for the east Cascades. Our estimates of soil C storage match up very closely with current estimates from the Pringle Falls Experimental Forest, but it is unclear how much our estimates would differ under different fuel reduction treatment types and frequencies. Many understory community types exist in east Cascades *Pinus ponderosa* forests (i.e., *Festuca idahoensis*, *Purshia tridentata*, *Agropyron spicatum*, *Stipa comata*, *Physocarpus malvaceus*, and *Symphoricarpos albus* communities) (Franklin and Dyrness 1988). An alteration of these communities may result from fuel reduction treatments such as understory removal or prescribed fire, leading to a change in the amount and composition of decomposing materials, which can influence long-term belowground C storage (Wardle 2002). Furthermore, there may be an increase in soil C storage resulting from the addition of charcoal to the soil C pool, whether from prescribed fire or wildfire (DeLuca and Aplet 2008).

By contrast, ecosystems with lengthy fire return intervals, such as those of the west Cascades and Coast Range, may not be strongly altered by such a policy, as many stands would not have accumulated uncharacteristic levels of fuel during a time of fire suppression that is substantially less than the mean fire return intervals for these systems. Forests such as these may actually have little or no need for fuel reduction due to their lengthy fire return intervals. Furthermore, fire severity in many forests may be more a function of severe weather events rather than fuel accumulation (Bessie and Johnson 1995, Brown et al. 2004, Schoennagel et al. 2004). Thus, the application of fuel reduction treatments such as understory removal is thought to be unnecessary in such forests and may provide only limited effectiveness (Agee and Huff 1986, Brown et al. 2004). Our results provide additional support for this notion, as they show a minimal effect of understory removal on expected fire severity in these forests, and if in fact climate has far stronger control over fire severity in these forests than fuel abundance, then the small reductions in expected fire severity that we have modeled for these fuel reduction treatments may be even smaller in reality.

We also note that the extent to which fuel reductions in these forests can result in a reduction in fire severity during the extreme climate conditions that lead to broad-scale catastrophic wildfires may be different from the effects shown by our modeling results, and are likely to be an area of significant uncertainty. Fuel reductions, especially overstory thinning treatments, can increase air temperatures near the ground and wind speeds throughout the forest canopy (van Wagtendonk 1996, Agee and Skinner 2005), potentially leading to an increase in fire severity that cannot be accounted for within our particular fire model. In addition to the microclimatic changes that may follow an overstory thinning, logging residues may be present on site following such a procedure, and may potentially nullify the effects of the fuel reduction treatment or may even lead to an increase in fire severity (Stephens 1998). Field-based increases in fire severity that occur in stands subjected to overstory thinning may in fact be an interaction between the fine fuels created by the thinning treatment and the accompanying changes in forest microclimate. These microclimate changes may lead to drier fuels and allow higher wind speeds throughout the stand (Raymond and Peterson 2005). While our model does incorporate the creation of logging residue that follows silvicultural thinning, increases in fire spread and intensity due to interactions between fine fuels and increased wind speed are neglected. However, we note that even if our model is failing to capture these dynamics, our general conclusion that fuel reduction results in a decrease in long-term C storage would then have even stronger support, since the fuel reduction would have caused C loss from the removal of biomass while also *increasing* the amount that is lost in a wildfire.

The amounts of C lost in fuel reduction treatments, whether nearly equal to or greater than our estimates, can be utilized in the production of biofuels. It is clear, however, that an attempt to substitute forest biomass for fossil fuels is not likely to be an effective forest management strategy for the next 100 years. Coast Range *Tsuga heterophylla–Picea sitchensis* ecosystems have some of the highest known amounts of biomass production and storage capacity, yet under the UR + OT + PF treatment a 169-year period is necessary to reach the point at which biomass production will offset C emitted from fossil fuels, and 338 years for ethanol production. Likewise, managed forests in the west Cascades require time scales that are too vast for biofuel alternatives to make a difference over the next 100 years. Even converting old-growth forests in these ecosystems would require at least 33 and 107 years for woody biomass utilization in the Coast Range and west Cascades, respectively, and these figures assume that all possible energy in these fuels can be utilized. Likewise, our ethanol calculations assumed that the maximum theoretical ethanol yield of biomass is realized, which has yet to be done (Schubert 2006); a 70% realization of our maximum yield is a more realistic approximation of contemporary capacities (Galbe and Zacchi 2002).

In addition to these lags, management constraints could preclude any attempt to fully utilize Pacific Northwest forests for their full biofuels production potential. Currently in the Pacific Northwest there are $\sim 3.6 \times 10^6$ ha of forests in need of fuel reduction treatments (Stephens and Ruth 2005), and in 2004 the annual treatment goal for this area was 52 000 ha (1.44%). Unless a significantly larger fuel reduction treatment workforce is employed, it would take 69 years to treat this area once, a period that approximates the effective duration of fire suppression (Stephens and Ruth 2005). The use of SPLATs (strategically placed area treatments) may be necessary to reduce the extent and effects of landscape-level fire (Finney 2001). SPLATs are a system of overlapping area fuel treatments designed to minimize the area burned by high-intensity head fires in diverse terrain. These treatments are costly, and estimates of such treatment costs may be underestimating the expense of fuel reduction in areas with high-density understory tree cohorts that are time consuming to extract and have little monetary value to aid in offsetting removal expenses (Stephens and Ruth 2005). Nevertheless, it is clear that not all of the Pacific Northwest forests that are in need of fuel reduction treatments can be reached, and the use of strategically placed fuel reduction treatments such as SPLATs may represent the best option for a cost-effective reduction in wildfire severity, particularly in areas near the wildland–urban interface. However, the application of strategically placed fuel reduction treatments is unlikely to be a sufficient means in itself toward ecosystem restoration in the forests of the east Cascades. Stand-level ecosystem restoration efforts such as understory removal and prescribed fire may need to be commenced once landscape-level reductions in fire spread risk have been implemented.

Conclusions

Managing forests for the future is a complex issue that necessitates the consideration of multiple spatial and temporal scales and multiple management goals. We explored the trade-offs for managing forests for fuel reduction vs. C storage using an ecosystem simulation model capable of simulating many types of forest management practices. With the possible exception of some xeric ecosystems in the east Cascades, our work suggests that fuel reduction treatments should be forgone if forest ecosystems are to provide maximal amelioration of atmospheric CO_2 over the next 100 years. Much remains to be learned about the effects of forest fuel reduction treatments on fire severity, but our results demonstrate that if fuel reduction treatments are effective in reducing fire severities in the western hemlock–Douglas-fir forests of the west Cascades and the western hemlock–Sitka spruce forests of the Coast Range, it will come at the cost of long-term C storage, even if harvested materials are utilized as biofuels. We agree with the policy recommendations of Stephens and Ruth (2005) that the application of fuel reduction treatments may be essential for ecosystem restoration in forests with uncharacteristic levels of fuel buildup, as is often the case in the xeric forest ecosystems of the east Cascades. However, this is often impractical and may even be counterproductive in ecosystems that do not exhibit uncharacteristic or undesirable levels of fuel accumulation. Ecosystems such as the western hemlock–Douglas-fir forests in the west Cascades and the western hemlock–Sitka spruce forests of the Coast Range may in fact have little sensitivity to forest fuel reduction treatments and may be best utilized for their high C sequestration capacities.

Acknowledgments

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APPENDIX A

STANDCARB model description (*Ecological Archives* A019-028-A1).

APPENDIX B

Biofuels analysis calculations (*Ecological Archives* A019-028-A2).

APPENDIX C

Carbon storage and fire severity results for each treatment type and frequency (*Ecological Archives* A019-028-A3).

A Statement of Common Ground Regarding the Role of Wildfire in Forested Landscapes of the Western United States

Fire Research Consensus Working Group

Final Report

September 2018

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Cover photo caption: Night and day on the Pioneer Fire in central Idaho in 2016. How this fire burned and what will happen next reflects the history of fire and fire suppression in this region, as well as land use and changing climate. The Pioneer Fire burned >188,000 acres in 2016, despite active fire management to limit its spread, at a cost of >$100 million. Photo ID_16-08-30 by Kari Greer, Kari Greer Photography, www.wildland-fires.smugmug.com and used with her permission.

Contents

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The Steering Committee (Craig Allen, Paul Hessburg, Penelope Morgan, Max Moritz, Dennis Odion, Christopher Topik, and Thomas Veblen) is collectively responsible for the work summarized here, and we adhered to a Project Charter that clarified our goals, roles, and responsibilities. The contributions of Ian McCullough were valuable and substantial enough to merit coauthorship. Excellent project facilitation was provided by Julian Griggs of the Dovetail Consulting Group. The substantial suggestions of Rachel White, science writer/editor for the US Forest Service, were very helpful; additional constructive comments were provided in the USGS internal review process.

Many scientists and managers contributed to this report, either through input to a questionnaire or by providing feedback on interim drafts. Please refer to the online supplemental materials associated with the questionnaire and review process, which include a record of contributors and other documents. The project would not have been possible without the time, effort, and trust of our colleagues, and we are grateful for all contributions.

Executive Summary

For millennia, wildfires have markedly influenced forests and non-forested landscapes of the western United States (US), and they are increasingly seen as having substantial impacts on society and nature. There is growing concern over what kinds and amounts of fire will achieve desirable outcomes and limit harmful effects on people and nature. Moreover, the increasing complexity surrounding cost and management of wildfires suggests that science should play a more prominent role in informing decisions about the need for fire in nature, and the need for society to adapt to the inevitable occurrence of different kinds and amounts of fire and smoke.

Scientists widely view the natural wildfire regime as essential to western US forest ecosystem functioning. However, debates continue over how much low-, moderate-, and high-severity fire is “natural” or desirable in these forests. Ongoing disagreement centers on the characteristics and importance of historical proportions and patch size distributions of low-, moderate-, and high-severity fires of dry, moist, and cold forests, and on the ecological consequences of changing fire-patch patterns and relative abundances. Scientists also debate the relative importance of climate and extreme weather versus fuel as drivers of high-severity fire, as well as the effectiveness and value of fuel treatments for reducing risks of undesired fire effects.

Climate research shows that we should expect shifting future climates in all ecoregions. These expected changes make it difficult for scientists, land managers, and decision-makers to know the degree to which future forest management should be informed by historical conditions. There also is disagreement about how to make western forests more resilient to future disruptions in both climatic and fire regimes. To complicate matters, areas of scientific agreement -- the “common ground” shared by those in the research community -- are poorly articulated. Thus, the focus of the Fire Research Consensus (FRC) project has been to identify common ground among scientists, and provide a summary that can inform management. Land and fire managers are one audience for this report, as are stakeholders and the interested public.

Our analysis, which results from extensive scientific literature reviews and questionnaires sent to western fire scientists and land managers, is summarized in nine key topics:

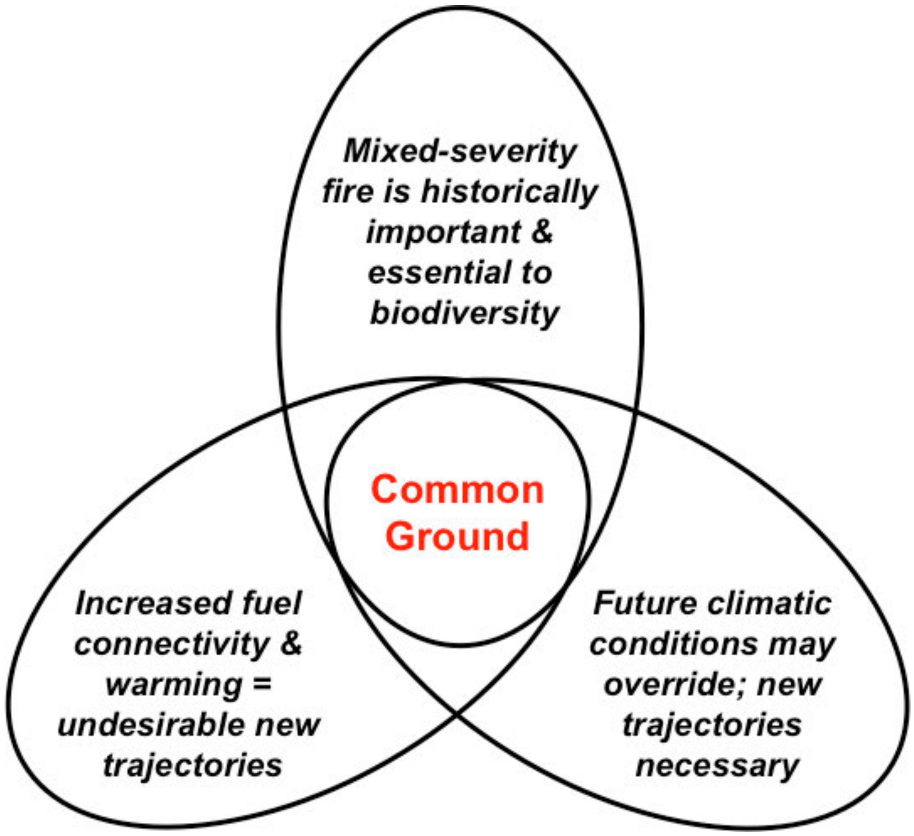
A. Fire history and fire ecology vary with geography.
B. Human impacts and management history vary with geography.
C. Fire is a keystone process, which occurs in almost all western US forest types.
D. Knowledge of historical range of variability (HRV) is useful but does not dictate land management goals.
E. Forest structure, composition, and fuels have changed, affecting burn severity and fire extent.
F. Climate and fuels both influence current fire sizes and their severities.
G. The role of changing climatic conditions is increasingly important.
H. Multiple fire ecology and fire history research approaches can be useful for characterizing fire regimes.
I. Many existing fire management tools and strategies can be useful moving forward.

We found much common ground that will be useful to scientists, managers, citizens, and policy decision-makers. For example, there is wide agreement among scientists that fire is one of the most essential influences on western forests and that more fire is needed on most landscapes, but not all wildfire behavior or extent will do. Fires can produce more positive benefits and fewer negative impacts when they burn with an ecologically appropriate mix and pattern of low, moderate, and high severity. Managers will need assistance and funding to create landscape conditions that favor more desirable fire behavior at broad spatial scales. Note that much societal impact from western wildfires occurs in non-forested landscapes that are not covered in this report, where findings would differ from those reported here for forested landscapes. We summarize additional key points below.

High-severity fire

Respondents disagreed about whether large, high-severity fires have increased to a significant and measurable degree in all forest types *in comparison to historical fire regimes* (i.e., prior to modern fire suppression). There was strong agreement that in dry pine forests at low elevations there has been either an observed increase in high-severity fires or an increase in the potential for fires of elevated severity as the result of increased abundance and connectivity of woody fuels since the late 19th century. There was similar strong agreement about dry mixed-conifer forests in the Inland Northwest, Pacific Southwest, and Inland Southwest (Arizona and New Mexico) that there has been an increase in high-severity fires and an increase in the potential for fires of elevated severity. There was less agreement about the changes in extent, and causes of changes in extent, of high-severity fires in moist mixed-conifer forests. Although there is general agreement that high-severity fires historically played an important role in moist mixed-conifer and cold subalpine forests, there is strong disagreement over the degree of changes in burn severity patch-size distributions and associated successional conditions for these forests between different regions.

Opinions also vary over the consequences of any increases in fire severity. For most dry forests, although there may be some disagreement about trends in burn severity and their causes, there is broad agreement that under current and projected climate, post-fire forest resilience is less than in the past. Some forest habitats, particularly at drier sites, but also in some moist and cold forest sites, show evidence of converting to more flammable non-forest vegetation or less dense forests following recent fires where large patches burn severely, especially if reburned. Reburn potential may depend on the interaction of vegetation, weather, rate of fire spread, time since prior fire, ignitions and fire suppression. Opinions are varied concerning the ecological consequences of departures from historical patterns of fire severity in various mixed conifer and subalpine forests. For example, one viewpoint supports the historical precedence of mixed-severity fire (including relatively large patches of high-severity fire), and the concept that pyrodiversity begets biodiversity. Another viewpoint asserts that increased woody fuel connectivity in combination with a warming climate trend is setting large areas of landscapes on fundamentally new trajectories, with significant undesirable ecological and societal consequences. Still a third viewpoint emphasizes that climatic changes increasingly are of overriding importance, and that new trajectories are unavoidable and thus may be considered desirable in many cases to incrementally foster necessary ecosystem transitions. The figure below characterizes these divergent viewpoints – typical of many areas of disagreement we addressed – and the potential common ground among them.



Uncertainties associated with relative proportions of different burn severities and patch-size distributions combine to cloud key points of consensus that have important management implications. We suggest that resolving many fire science disagreements depends on greater consideration of specific geographical context. This may imply that a narrow range of field experience can limit one's ability to accept findings that depart from that range. A logical way forward is to increase in-depth cross-regional field research experiences of the fire research community. Cross-regional comparisons of top-down and bottom-up determinants of fire activity in similar forest cover types is a fertile area of future research to examine how differences in seasonality, productivity, understory fuels, land use history, and other factors may explain some of the reported geographical differences in historical fire regimes in broadly similar forest types.

There are several reasons for the disagreements about the amount and roles of past higher-severity fire. Both scientists and managers often transfer concepts and findings from one place to another, yet we know that "no one size fits all" for historical fire regimes, even within the same forest type. Likewise, the extent of change in abundance and connectivity of woody fuels varies across forest types and ecoregions. Some of the disagreement derives from use of different scientific approaches. For instance, there is strong debate about the fire regime inferences made from historical and modern tree inventory data, simulation models, and other approaches. We believe that application of diverse research approaches will be useful going forward. Further, multiple approaches will be useful in "triangulating" interpretations for which there is some scientific consensus (see Topic H). We challenge fire scientists who do not share similar perspectives on historical fire regimes in particular ecosystems to engage in civil discourse to better understand the reasons for their disagreement, and to objectively communicate those reasons to managers and other stakeholders. We are heartened by the positive outcomes achieved by some previous attempts when small or large groups work together to find common ground.

The Wildland Urban Interface and Beyond

Respondents strongly agreed on the need for fuel treatments and fire suppression to protect human infrastructure within and adjacent to the wildland urban interface (WUI). There is a strong consensus that preventing undesired human-set fires in the WUI is essential to reducing societal vulnerability. The strategies for managing fire may be different within and adjacent to the WUI than in areas far from the WUI. However, what fire managers do beyond the WUI has implications for fire behavior approaching the WUI, forest resilience, smoke production and its human impacts, water quality, and many other ecosystem services people value.

Fuels management alone, especially if limited to public land, will be insufficient to address the vulnerability of WUI communities to fires. Fuels management will be important for influencing how wildfire behavior will approach the WUI. Thus, policies to make current WUI communities more fire adapted (e.g., implementing current WUI codes) are a critical piece of the puzzle, as are changes in land use policies that influence where and how future WUI areas develop, and the spatial extent and arrangement of managed and wildfire fuel treatments. Controlling human ignitions is important to address fire risk, especially in landscapes where ignitions have the potential to radically increase fire frequency. Communities in fire-prone areas need to learn to live with fire and increase their use of fire and other methods to reduce susceptibility to unacceptable fire damage.

Pattern and Process for Fires in Forest Landscapes

Heterogeneity of fire effects, including the patterns of patches created by fires of all severities, is important to forest resilience to future fires (see Topic E). The scale of the problem is vast, however, so it is likely that the scale of analysis and solutions (e.g., fraction of landscape treated via wildfire use) is also necessarily vast. There are potentially profound implications for forest regeneration, watershed protection, biodiversity, and carbon sequestration if the proportion and spatial pattern of area burned with high-severity fire change. Where wildfires severely burn large areas of forest, local elimination of conifer tree seed sources and reduced tree regeneration under emergent warmer-drier conditions can occur. Large areas of forest are converting to persistent grasslands or shrublands post-fire in some regions. Even relatively small changes in the proportion of large patches can alter system behavior for decades and even centuries. Thus, the patch-size distributions of both forest and non-forest patches are of concern to policy makers, scientists, and managers.

Climate, Fuels, and Implications of Landscape Change

Both fuel and climate are important drivers of fire activity. Increased woody fuel connectivity in combination with a warming climate trend are setting large areas of many landscapes on new trajectories where very large patches burn with high severity. There is agreement that all fire regimes are the product of interactions among varying degrees of top-down climate and weather forcing and bottom-up spatio-temporal controls of local topography and fuels, which reflect legacies of past fires and other agents altering vegetation. In other words, fires respond to interacting influences of climate, weather, fuels, topography, legacies of prior disturbance, and management. The relative importance of these factors varies across landscapes and through time.

While climate is of increasing importance, fuels management is also important. Indeed, fuels are the main landscape characteristic that management can change. But an ecologically and socially appropriate mix of fuel management tools and practices is needed. More flexible management of wildfires and prescribed fires will be useful, depending on local objectives and conditions, to increase the footprint of land areas showing reduced surface and canopy fuel abundance and connectivity. Increased use of prescribed burning combined with thinning will be helpful where forest conditions are not currently manageable via wildfires and prescribed fires alone, and where high certainty about fire perimeter control and fire behavior are key objectives (e.g., adjacent to WUI). Some respondents suggested that accepting a more proactive approach to fire and fuels management on public lands may initially be more expensive, but may reduce overall costs and improve climate change adaptation in the long-term. Other respondents questioned the practicality and effectiveness of fuel treatments under a changing climate. Notably, in their responses, respondents did not integrate the concomitant effects of weather, climate, topography, and fuel abundance.

Decades of research in landscape ecology show that emergent properties have central importance to ecosystems and their pattern and process regulation, whereas many recent studies of climate-driven fire and vegetation change are less focused on local-scale feedbacks and emergent patterns. This difference creates a fundamental problem in linking climate change and landscape ecology research. Climate models assume that top-down climate covariates drive temperature, precipitation, and solar radiation conditions. Landscape ecology research shows that those top-down inputs can be highly modified by meso- and fine-scale bottom-up environmental controls to produce emergent climatic conditions that are strictly speaking neither the top-down or bottom-up inputs, but are influenced by these inputs. Climatic forcing alone poorly explains the shifts in landscape patterns because lagged patterns of historical disturbances continue to influence emergent patterns, under all but the most extreme events. The path forward to more effective projection of future fire and landscape change includes better integration of feedbacks from landscape ecological models into climate-driven models of future fire and landscape change. Broad-scale studies are still needed to tease apart the roles of changing climate and changes in fuels in the observed trends in frequency of large fires.

Effective Management will Depend on Both Science and Trust

Our understanding of historical fire regimes can inform decision-making; indeed, such evidence-based decision-making can build trust. While history does not provide precise prescriptions for managing landscapes, it does offer precautionary principles. Adaptive resilience for the future will require applying what we learn from history to some future range of variability, where fires burn and ecosystems respond in both similar and different ways.

At the same time, fire science points to complex patterns that vary with local conditions. Unique ranges of vegetation and fuel patterns are the result of interactions among regional climate, topography, landforms, geology, and biotic communities of an area, along with associated meso- to fine-scale pattern heterogeneity. Thus, no single solution, such as logging or limiting all logging, will accomplish desired objectives in all forests. Further, any management, including no intervention, has consequences, so all decisions need monitoring to evaluate the assumptions of management. Effective monitoring can improve knowledge, and through collective learning can build common understanding and trust.

Fire management can become more proactive and strategic. Existing tools, such as mechanical fuel treatments, prescribed fire, prevention of accidentally-ignited human fires, and managing wildfires, will all be useful, but adaptation and mitigation responses to climate change and changing fire activity will require using these tools in strategic ways to fit area-specific goals. Some past disagreements about fire and fuel management strategies may be due to lack of clarity about specific goals, such as resident and firefighter safety, cost reduction, biodiversity issues, and ecosystem resilience under a changing climate.

The timing of fires is important, particularly in the context of a changing climate. While recognizing that wildfire seasons are long and getting longer, we must also take advantage of the milder fire weather and associated effects of fires in the “shoulder seasons.” Managers may find that both less-aggressive fire suppression and expanded use of managed wildfire under relatively moderate weather conditions can aid them where reducing the vulnerability of people and natural resources to fires is the objective. Managing wildfires may be one important way to achieve relatively widespread vegetation change at the spatial scales and in the short timeframe needed.

One of the grand challenges of fire management is balancing the reality that wildfires will occur and are needed by western forest ecosystems, yet people, property, and economies need protection from the adverse effects of fire. Another grand and fairly urgent challenge is discovering the tipping points of transformative change for various forest landscapes in their respective geographies, where large, high severity fires (regardless of whether they are considered unprecedented or not) may tip forest ecosystems into persistent non-forest states by constraining tree regeneration opportunities. Particularly as climate changes, we also need a deeper understanding of which landscapes may not be able to sustain forests in the future and how fast such transitions are likely to occur. It is clear that our western history of substantial forest fire activity will continue, one way or another -- many fires will occur in the future and some will be large. Ultimately, we must find ways to both sustainably use and live with fires that are well-adapted to both ecosystem and societal needs of local landscapes.

Introduction

Wildfires have, for millennia, markedly influenced forests and non-forested landscapes of the western United States (US), and they are increasingly seen as having substantial impacts on society and nature, even though less area burns in many forests than burned historically. Informed planning and fire management preparations and responses are thus becoming more important, with lives, property, government expenditures, biodiversity, and ecosystem services at stake. Federal and state firefighting costs now routinely exceed available funds, which are then either borrowed or permanently taken from funds that would ordinarily support resource management activities.

At the same time, climate research shows that we should expect to experience future climates in many ecoregions that will, to varying degrees, differ from those of the recent past. These expected changes make it more difficult for scientists, land managers, and decision makers to know the degree to which future forest management and wildfire policy should be informed by the past. The increasing complexity surrounding cost and management of wildfires suggests that science might play a prominent role in informing decisions about the need for fire in nature, and the need for society to adapt to fire.

Scientists widely view fire as a normal part of ecosystem functioning and one of the most essential influences on forests of the western US. They also recognize that fire directly affects the health and wellbeing of people living near fire-prone landscapes, influencing the water, wildlife, recreation, forest products, and other aesthetic and spiritual benefits these landscapes provide. However, scientific debates continue over several important fire-related topics, including: how much low-, moderate-, and high-severity fire[1] is "natural" or desirable in varied forests across the western US; the relative importance of climatic versus fuel factors as drivers of high-severity fire; and the effectiveness and value of fuel treatments for reducing risks of undesired fire effects.

There is also apparent disagreement about how to make these forests more resilient to future disruptions in both climatic and fire regimes. In many policy and management arenas—from national forest policy to state, county and Tribal-level management—debates about wildfire have sometimes slowed effective integration of research into public policy, and hindered informed planning and management. Much is clearly at stake.

[1] Low, moderate, and high burn severity are usually defined in the western USA by level of mortality of overstory trees or shrubs within individual fires. Low-severity burns are often surface fires with scattered tree torching, where most trees survive (e.g., <20% mortality), while high-severity burns are often stand-replacing fires that kill >70% of the overstory trees (derived from some mix of surface and crown fire behavior). Moderate-severity fires include areas with intermediate levels of overstory mortality (20-70% of basal area or canopy cover of a given patch) from fire. All low-, moderate- and high-severity fire regimes result in intermixed patches of burned and unburned vegetation, but the scale of patchiness differs. Note that we use moderate for intermediate effects of individual fires and mixed for fire regimes. From Agee (1993) Fire ecology of Pacific Northwest forests. Island Press.

Overview: Purpose and Scope of this Report

This report summarizes work of the Fire Research Consensus (FRC) project, which formed to provide insights for scientists, land managers, and human communities with respect to recent controversies over the role of low-, moderate-, and high-severity fires in western US forests. The goal has been to clarify agreements, disagreements, research needs, and possible management implications of scientific common ground. Our hope is that stakeholder groups will avoid the selective use of particular scientific papers to argue for their particular ends. Instead, they will be able to point to key shared assumptions, common understandings considering the entire body of fire science literature, and terminology to support decision-making in constructive ways. This should facilitate better awareness and application of existing and future scientific findings. In particular, land and fire managers are a key audience for this report, as are other stakeholders and the interested public engaged in discussions about land management. Future work is needed to more directly emphasize fire-related research needs and open scientific questions.

We acknowledge that public land management agencies are charged by society to make management decisions, and take associated actions on those lands. Actions are constrained and focused by existing laws, land use plans, policies, and pertinent Acts. We further acknowledge that management agencies are required to accomplish annually-funded land management targets, which can be at odds with some societally-held land-use values, or other landscape and resource management goals. Our focus on areas of broad scientific agreement within the fire science community is intended to make the application of fire science more useful to land management agencies and lawmakers, but it does not, and cannot, resolve diverse social, economic, philosophical, and political debates about preferred land use values across a spectrum of ideologies and management methods. These societal debates play out through broader public conversations and decision-making processes that are only partly informed by fire science. Key roles of fire science are to provide high-quality information to support high-quality societal conversations and decision-making about land (and fire) management, and to assist in monitoring outcomes. In our implications comments at the close of each major section, we provide case examples of how areas of agreement might be considered in the development of management applications.

As a prelude to more in-depth coverage in the report, our analysis can be summarized to nine key topics:

A. Fire history and fire ecology vary with geography.
B. Human impacts and management history vary with geography.
C. Fire is a keystone process, which occurs in almost all western US forest types.
D. Knowledge of historical range of variability (HRV) is useful but does not dictate land management goals.
E. Forest structure, composition, and fuels have changed, affecting burn severity and fire extent.
F. Climate and fuels both influence current fire sizes and their severities.
G. The role of changing climatic conditions is increasingly important.
H. Multiple fire ecology and fire history research approaches can be useful for characterizing fire regimes.
I. Many existing fire management tools and strategies can be useful moving forward.

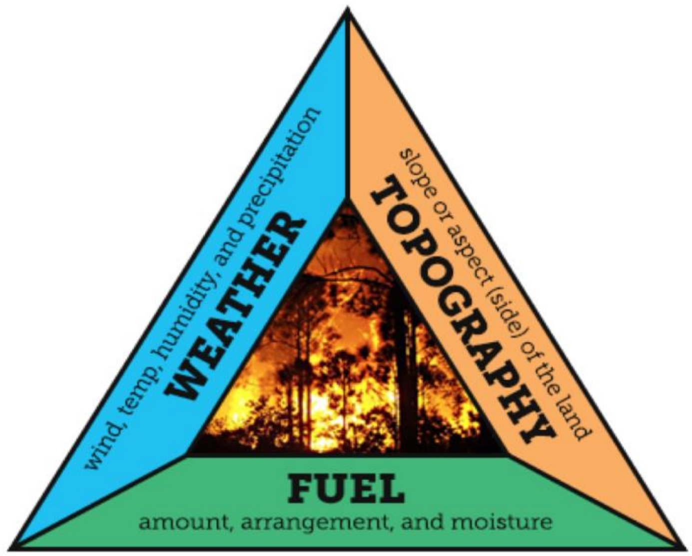
Given the intertwined nature of these topics, there is repetition of themes among some of the material presented. The FRC steering committee believes that the summaries derived from this work are representative of current fire science and can usefully inform fire and land management. It is our intent that in the future, land managers and community leaders will be able to better understand, and more accurately and precisely communicate, the need for fire in the environment and how to better prepare for its impacts. Further, the goals and priorities for fuel and climate change adaptation treatments will be better understood, such that responses to them are less polarized. Scientists will have a clearer picture of the key research questions that underpin current debates. Instead of a focus on disagreements, a deeper appreciation of the research that is agreed upon will allow us all to be more deliberate and proactive when thinking about and managing wildfire environments in the West.

Note that much societal impact from western wildfires occurs in non-forested landscapes that are not covered in this report, where findings would differ from those reported here for forested landscapes.

Fundamental Principles vs. “Common Ground”

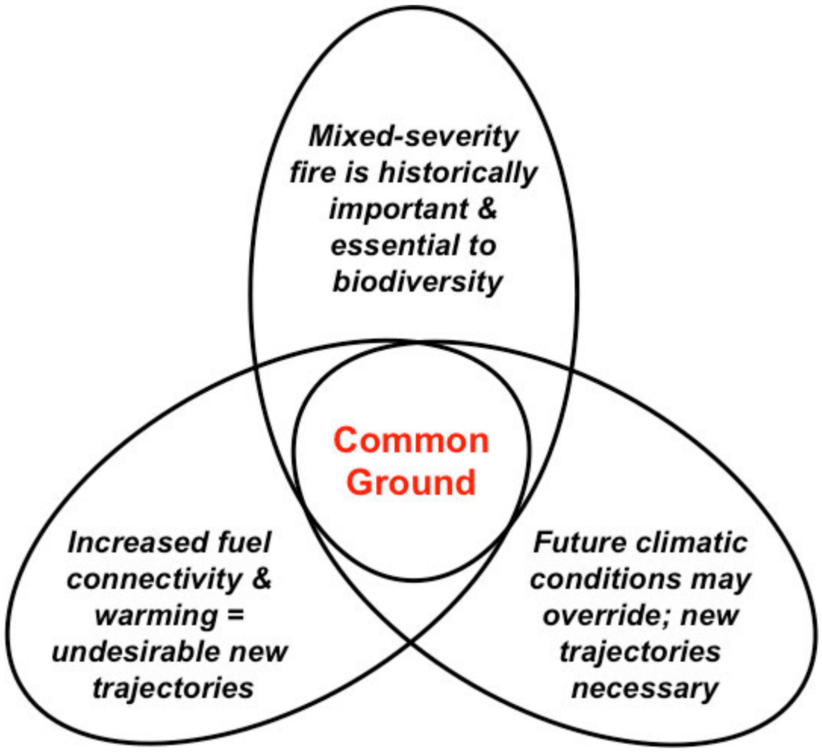
At the outset, we acknowledge some core scientific principles that are widely accepted by those engaged in all sides of these debates. One example is the idea that wildfire is inevitable, and it is a process essential to all western forest ecosystems. Wide agreement therefore exists about the extensive benefits of fire, even if this agreement may not be shared outside of the research community. The notion that fire is an essential ecological process was universally shared and was a guiding principle of most questionnaire respondents.

Another key example is the set of factors making up what is considered the “fire behavior triangle” shown below. This construct was developed by scientists to capture the physical and chemical principles that govern fire behavior, namely characteristics of 1) fuel, 2) weather, and 3) topography in affecting a given fire’s rates of spread, flame lengths, and intensities. There is also extensive agreement about there being trade-offs in the relative importance of these factors, such as the influence of fuel characteristics in some instances diminishing in more extreme topographic settings (e.g., steeper slopes) and weather conditions (e.g., higher wind speeds, lower humidities). There is natural variation in how different factors intersect in space and time, resulting in often complex dynamics and only semi-predictable outcomes. Even so, certain relationships are predictable enough at finer scales to be useful for models of fire spread and crown fire initiation; broad-scale simulation of fire behavior patterns is also possible, although with known limitations.



Fire Behavior Triangle

In the context of our project, the fundamental science that underpins the study of fire is not what we mean by “common ground” shared among disagreeing groups. Here instead we are referring to areas of agreement, or the overlap in perspectives, that emerge when debates over a given issue are deconstructed. As a hypothetical but realistic example, consider the Venn diagram below, which represents three partly overlapping views about possible causes and consequences of increases in high-severity fires. There is evidence that some forest habitats, particularly at drier sites, are converting to non-forest vegetation or less-dense forests following recent fires, where large and severely burned patches are created. Conversely, afforestation has occurred in some forest types as a result of fire suppression, which can reduce fire intensity and spread, compared to some non-forest vegetation. Opinions are varied concerning departures from historical patterns of fire severity in various mixed-conifer and subalpine forests, as well as their ecological consequences. One viewpoint supports the historical precedence of mixed-severity fire (including relatively large patches of high-severity fire), and the concept that pyrodiversity begets biodiversity. Another viewpoint asserts that increased woody fuel connectivity in combination with a warming climate trend is setting large areas of landscapes on fundamentally new trajectories, with significant undesirable ecological and societal consequences. Still a third viewpoint emphasizes that climatic changes increasingly are of overriding importance, and that new trajectories are unavoidable and thus may be considered desirable in many cases to incrementally foster necessary ecosystem transitions.



In the realm of public discourse, these three perspectives might be reduced to simplistic and utterly contrasting sound bites, spanning the following extremes:

- Fuel treatments are urgently needed across nearly all forests.
- Fuel treatments should be focused around communities and plantations; hazard reduction elsewhere is futile.
- There is high uncertainty about where and when fuel treatments are beneficial.

Regardless of public perception, there is still a solid scientific basis for each of the three perspectives shown in the example above, and much can be learned by examining the common ground of their intersection. We explore the common ground of these and other such areas of overlap in divergent scientific perspectives in this document.

Philosophical and Contextual Issues

At times, differences in perspective may be linked to whether one's research emphasizes fire effects on tree survival, residual vegetation structure, or fire effects on overall ecosystem function and biodiversity. Frustrations and value judgements about management activities and their impacts on public lands have also contributed to differing scientific perspectives about possible paths forward. Scientist and public mistrust of past and current management on some public lands is one of the largest impediments to forward progress, and yet most discussions focus on improving fire science rather than improving trust. Fire scientists, ecologists, and land managers need to better understand how science has been used in the past to justify various management actions, and how various breaches of trust have affected adoption of modern scientific findings. Such trust can be rebuilt with monitoring and stakeholder engagement in land management decision making.

There was wide agreement among questionnaire respondents that fire science often gets overly simplified in the media, even when more nuanced views may be held among scientists doing the research. Sometimes simplification links back to early narratives and research findings, which may then be inappropriately applied by others beyond their original context. An example of this is the notion that climate change will universally increase fire frequencies and severities, despite growing evidence of more complex outcomes. In other cases, scientists, journalists, policy makers, land managers, NGOs, or politicians may simplify stories to increase their clarity or impact, or to deliver specific messages to the public, and these stories are then carried forth as "debates."

Many respondents also recognize the need for better terminology and conceptualizations of fire regimes[2], both for communicating with the public, and for use among scientists. To some extent, imprecision or ambiguity of terms and concepts may be partly responsible for certain debates in the fire literature. For example, numerous respondents commented on how fire regimes have been oversimplified as fitting into one of the three broad classes of low-, mixed-, or high-severity. Imprecision or lack of agreement on objective classification of fire regimes conflates with actual disagreements over the interpretation of fire history evidence. Although disagreements are not simply based on semantics, poor semantics contribute to confusion. In addition, scientific interpretations of fire regimes made at specific spatial and temporal scales are sometimes fraught with unspecified assumptions, imprecision, or error in the scope of the inferences made.

Additionally, respondents often had differing priorities for, and definitions of, "restoration" and "resilience," generally reflecting the plurality of these definitions and priorities in modern society. For example, how important is the past to understanding and planning for the future? What exactly is being restored, to what benchmark, and for what purposes? What ecological and social values support any intended restoration? Underpinning each of these questions are differing perspectives about the importance of historical ecology based on differences in human social values, even on the part of scientists.

Restoration[3] and resilience[4] are often identified as goals of forest management, and yet the enabling legislation and funding sources of different land management agencies actually

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[2] A fire regime is the pattern, frequency, fire size, spatial complexity, and severity of fires over space and time. Fire regimes are characterized based on fire frequency (how often fires occur), intensity (amount of heat released at the flaming front), severity (both soil and tree mortality effects), type (ground, surface, crown), size, spatial pattern (including patch size distribution) and seasonality. Ground fires burn organic matter in the soil. Surface fires burn leaf litter, fallen branches, and plants on and near the soil surface. Crown fires burn through to the top layer of trees and shrubs. From Morgan et al. (2001) Mapping fire regimes across time and space: understanding coarse and fine-scale fire patterns. International Journal of Wildland Fire, 10(4): 329-342.

[3] Restoration is "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (Society for Ecological Restoration International Science & Policy Working Group [2004] Primer on Ecological Restoration. www.ser.org). Also see Hessburg et al. (2015) Restoring fire-prone forest landscapes: Seven core principles. Landscape Ecology 30(10): 1805-1835.

[4] Resilience: The capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity, and feedbacks (Society for Ecological Restoration www.ser.org). See also Schoennagel et al. (2017) Adapt to more wildfire in western North American forests as climate changes. Proceedings of the National Academy of Sciences. 114(18): 4582-4590.

dictate how restoration and resilience are defined and implemented. Therefore, even if common ground might exist on the need for fire to play a more natural or culturally central role, there can be widely varying opinions about what to do, and varying options as to how to make that happen, legislatively and administratively.

Not surprisingly, there were differing opinions about tradeoffs between human social values and the ecological benefits of fire. For example, smoke from wildfires or prescribed fires is a great concern that can have important influences on how various fire treatments are applied. Reconciling these varied opinions and the associated trade-offs was not in the purview of this effort. Views on forest restoration and ecosystem resilience are thus embedded in this larger context of other human social values, which greatly adds to the complexity of consensus-building and informed decision-making.

Looking forward, assessments of the effectiveness of fire management under climate warming will provide important results, ideally through science-based monitoring and management actions that are intentionally adapted by lessons learned. Most fire scientists and managers agree that fuel treatments can affect fire behavior, though effectiveness can vary with weather, treatment type, location, and time since treatment. Clearly, wildfire researchers recognize the importance of both extreme weather and fuel conditions on fire behavior. However, some respondents suggested that policy makers are unaware of uncertainties associated with attaining fire mitigation goals in the face of more frequent extreme-fire weather, but the management requirement to address such goals persists. Fire managers look to fire science for clear answers about methods, and their reasonable application, because planning and implementing actions in response to climate change, forest restoration, and other needs are essential to their mission.

A number of respondents lamented the time and energy devoted to disagreements over the interpretation of fire history in forest management debates. They suggested that the real challenge is to face the reality of a changing climate and changing fuels by considering the effectiveness of fire mitigation strategies (both old and new). There was also wide agreement on the need for land-use strategies that reduce societal and resource vulnerability to negative consequences of wildfire and climate change, while providing for the essential role and many benefits of fire in forests. We acknowledge that this is an example of fire scientists pointing to a need for stronger engagement with social scientists.

Numerous western fire scientists, when asked, chose not to participate in this survey, and others reluctantly participated. Several cited previous unproductive and unprofessional interactions in the context of debating fire science and related land management issues. Some questioned the motives of researchers not sharing compatible viewpoints on fire issues. Quite a few, including individuals from all sides of the debate, expressed frustrations with the peer review process of some mainstream ecology and forest science publications, and the resulting contradictory messages conveyed to land managers. The FRC Project Steering Committee is well aware of deep division within a portion of the fire research community that is impeding healthy, productive scientific debate. It is beyond the scope of the FRC to examine the non-scientific bases of these conflicts. Instead, our focus has been to identify the common ground shared among a majority of fire scientists on key issues, and to provide a summary that can support informed management decision-making going forward.

Methods and Data

We considered the entire extent of the scientific literature and views of scientists relating to fire research in western US forests. The Steering Committee is committed to inclusion of the full range of scientific perspectives reflected in the questionnaire responses and in the peer-reviewed literature. To facilitate a broad scope of input, we invited responses to a multi-part questionnaire from scientists from many different geographic areas and scientific perspectives. Invited respondents were those who had "*published significant primary research on fire occurrence and fire effects on ecosystem attributes in forests of the western US prior to intensive management, or in areas with limited active management such as large wilderness areas*."

This invitation criterion filtered out potentially important scientific contributions (e.g., those focusing on Native American use of fire, wildlife, post-European settlement periods, ecosystem resilience, and climate change adaptation) from the initial questionnaire. However, a broader range of scientific perspectives was included when the draft common ground statement was distributed for external review. After several rounds of invitation, 77 researchers were contacted, which yielded 36 respondents, including steering committee members. We believe that the depth and geographical breadth of responses were sufficient to identify key areas of agreement and disagreement among fire scientists.

Individual questions in the questionnaire were often intentionally structured as false dichotomies. Using this mechanism, we intended to generate thoughtful responses that would include details as to why a respondent might agree or disagree with the framing of a given issue. While this approach worked overall, it was clearly frustrating to some, and even appeared to a few as evidence of inherent bias in the process.

Between November 1st and 4th, 2016, our steering committee convened a workshop to summarize and organize responses to the questionnaire. Due to great variation in the nature of the questions and how much respondents tended to use literature citations in their responses, we opted not to incorporate citations throughout this report; doing so in a consistent manner was simply seen as intractable. In addition, our common ground document draws upon our own experience and critical reading of the literature, also without the use of citations. As an archived supplement to this report, however, we list citations that were used by respondents in supplemental online materials; any future refereed publications derived from this work will incorporate citations. Note that an exception to this approach is our inclusion of citations in a relatively small number of definitional footnotes throughout the report.

An external evaluation of the completeness and tone of our common ground statement was undertaken in June of 2017. For scientific perspectives, we invited 100 researchers for feedback on our draft statement; this group was larger than the original 77 invitees, to include a broader range of expertise. We received feedback from 36 individuals, not including the FRC steering committee. We also invited review and comments on the draft and its usefulness from 60 land managers and other stakeholders, 22 of whom provided feedback. To the best of our ability, we then integrated the feedback we received into this final document.

Forest Type Classifications

We based our discussion on three broad forest types in the western US, which we refer to as 1) dry pine and/or dry mixed-conifer (*aka*, dry forests), 2) moist mixed-conifer (moist forests), and 3) cold subalpine (cold forests). These are broad terms that are used colloquially to generalize forest types across the western US. Within particular regions, these terms can be crosswalked to classifications that are commonly used by land managers and in peer-reviewed literature.

To the extent that it may be helpful for cross-regional communication and possible generalizations we provided some examples of forest types covered in the questionnaire based on the US National Vegetation Classification (US NVC).

Dry forests, including for example:

- Central Rocky Mountain Dry Forest Macrogroup M501 (1.B.2.Nb.2 *Pinus ponderosa var. ponderosa - Pseudotsuga menziesii - Pinus flexilis*)
- Southern Rocky Mountain Forest & Woodland Group G228 (1.B.2.Nb.1.b *Pinus ponderosa*)

Moist forests, including for example:

- Central Rocky Mountain Mesic Lower Montane Forest Macrogroup M500 (1.B.2.Nb.3 *Tsuga heterophylla - Abies grandis - Larix occidentalis*)
- Central Rocky Mountain Forest Group (1.B.2.Nb.3.c *Abies grandis - Pseudotsuga menziesii* East Cascades Forest Group)
- Mesic Southern Rocky Mountain Forest Group G225 (*Abies concolor - Picea pungens - Pseudotsuga menziesii)*
- Vancouverian Lowland & Montane Forest Macrogroup M023 (*Calocedrus decurrens – Pinus jeffreyi – Abies concolor* var*. lowiana* Forest Macrogroup (exclude *Pseudotsuga macrocarpa –Quercus chrysolepis*)

Cold subalpine forests, including for example:

- Rocky Mountain Subalpine-High Montane Conifer Forest (1.B.2.Nb.5 *Abies lasiocarpa - Picea engelmannii - Pinus albicaulis)*
- California Red Fir - Mountain Hemlock - Sierra Lodgepole Pine Forest (1.B.2.Nd.4 *Abies magnifica - Tsuga mertensiana - Pinus contorta* var. *murrayana)*

Topic A. Fire history and fire ecology vary across geography

Common Ground

Key points of common ground among respondents to the questionnaire include:

- Generalized models of historical fire regimes vary by ecoregion and forest type.
- Even within the same ecoregion and forest type, there is variation in historical fire regimes among differing environmental gradients.
- There are many different historical fire regimes throughout the western US, and a single model cannot represent this variation (i.e., one size does not fit all).
- Historically, some degree of low-, moderate-, and high-severity fire has occurred in all forest types, but in substantially different proportions and patch size distributions at different locations.
- Classification of historical fire regimes according to forest types can be coarse; thus, failure to recognize variation of historical fire regimes *within* forest types can lead to overgeneralization and oversimplification of landscape conditions.

Respondents strongly emphasized how geographical context is critical in understanding and characterizing past, present, and future fire regimes. Many respondents commented that their responses were dependent on geographical context, or they simply noted that the geography under consideration is important. Respondents described numerous examples of how fire regimes vary at a broad scale across large gradients from warm-dry to cool-wet habitats.[5] Within the fire research community, there is essentially unanimous agreement that historical fire regimes differed fundamentally among strongly contrasting forest types such as low-elevation dry pine forests (mainly involving relatively frequent surface fires) versus cool to cold subalpine forests (mainly involving relatively infrequent high-severity fires), so that a one-size-fits-all approach clearly should not apply to management discussions. The spatial and temporal scales at which generalizations about natural or cultural fire regimes are valid vary, and can be uncertain or as yet poorly researched, which may be an important explanation for some disagreements about fire history among researchers, and appropriate management goals among practitioners. In these latter cases, managers with a need to make progress toward agency goals may inappropriately apply knowledge gained from different but related systems, or from expert panels.

A majority of respondents agreed that any singular characterization of fire regimes and how they have been altered by modern land-use practices—at the scale of the western US—is clearly inappropriate. For example, only at the scale of an ecoregion can we estimate patch size distributions of low-, moderate-, and high-severity fires of any particular forest type. However, individual landscapes within ecoregions do not show the full variability extant within an ecoregion. Neither is it always appropriate to simply assign fire regimes by forest type. Within an ecoregion, gradients of climate and vegetation attributes are well understood as determinants of fire regimes and their variation. Most significantly, broad-scale spatial variability of fire regimes results from broad spatial variability in long-term climate, annual weather,

[5] The existence of major differences in fire regimes in strongly contrasting ecosystems such as low-elevation, dry pine forests and high-elevation cold forests is relatively non-controversial and well documented in the literature (e.g., Schoennagel et al. 2004; Hessburg et al. 2007, Perry et al. 2011).

environmental and topographic conditions suitable for burning, *and* variability in amounts and spatial continuity of fuels, the nature of fuels, (e.g., forest vs. shrub vs. grass vegetation), and in the history of prior fires.

Many respondents emphasized that the commonly applied classification of fire regimes as "low-, mixed- , or high-severity" adequately describes dominance but inadequately describes variation in fire regimes across the western US (see Topic C). Whereas low- and high-severity fires are at least theoretically well understood endpoints of a continuum, a broad, poorly defined "mixed" category is the source of much confusion and misunderstanding. For example, "mixed-severity" is used to describe both the temporal variability in fire effects over multiple fire events at one site, and the spatial variation in burn severity within a single fire. Even in the case of the two extremes of low-and high-severity, respondents noted that there is often some degree of variability, with under-appreciated ecological impacts. However, there is agreement among respondents that all fire regimes are the product of interactions between varying degrees of top-down climate and weather-forcing and of bottom-up spatio-temporal controls of topography and local fuel, that reflect legacies of past fires and other agents altering vegetation, and hence fuel properties (see also Topic E).

Some respondents questioned whether commonly used vegetation classification schemes are a suitable basis for generalizing about fire regimes, and expressed that known geographic variation in fire regimes within forest types argues for improved forest and fire regime classifications. Many noted that broad classifications such as "dry forest" encompass substantial amounts of variability in historical, current, and future fire regimes, making generalizations at the level of an entire forest type suspect. Numerous respondents emphasized that variability in historical fire regimes within a broad forest type often reflects dominant influences of neighboring forest types and their associated fire regimes.

Areas of Divergence

Key areas of divergent opinion among respondents included:

- The relative proportions of different historical fire severities in particular geographical areas.
- The relative importance of extreme weather events to historical burn severity.
- Desirable proportions of low-, moderate-, and high-severity fire in the future.

Respondents disagreed about the relative proportions of different severities of historical fires for some of the *same geographical areas of study*. While this is a key source of debate, it is noteworthy that most studies conducted in the same study areas find qualitative similarities in historical fire regimes. Some studies stress the quantitative differences in the proportions of a study area interpreted as fitting into various classes of burn severity. Most commonly, such disagreement involves potential inferences from different types and scales of evidence of past fire, or past vegetation attributes. For example, tree-ring evidence sometimes supports conclusions that contrast with those derived from landscape-scale inventory and monitoring data using different sampling frames (see Topic H). Yet these different types of evidence of past fire sometimes also yield overlapping or even similar estimates of past fire activity.

In other cases, disagreements about proportions of low-, moderate-, and high-severity fire are based on findings from studies conducted in one area that were applied to another. In other words, some respondents assumed transferability of research findings across ecoregions, based on similarity of forest type. In certain instances, this may be true, but in others it may be inaccurate. Respondents expressed a fairly high degree of consensus about historical fire regimes within particular forest types and ecoregions. Few fire history researchers have significant field experience in more than one ecoregion, but a few of those with cross-regional experiences articulated support for the occurrence of contrasting fire regimes in similar dry, moist, and cold forest types among differing ecoregions. Certainly a narrow range of field experience can limit the ability to interpret and accept findings that differ from one's own experience.

Respondents exhibited a wide range of opinions, explicit or implied, about the potential importance of extreme weather events in overriding historical fire behavior and burn severities (see Topic F). Respondents noted that historical fires in some areas were mostly low-severity, but some high-severity events were also evident in tree-ring records incorporating stand ages, tree growth changes, and tree mortality dates, consistent with other evidence. Most others emphasized the greater frequency and extent of low-severity events and their role in reducing fuel quantities, creating fuel-limited systems or open canopy forests. Some respondents stressed the importance of long-lasting ecological effects from infrequent, moderate- or high-severity fires in the same study areas. Respondents who emphasized the longer time scales of charcoal records noted that most areas of predominantly low-severity fires also showed some incidence of moderate- or high-severity fire over longer time frames. However, the spatial imprecision of those longer charcoal records relative to particular forest types and their location makes these insights difficult to interpret. Some respondents related the occurrence of high-severity fires to extreme climate/weather conditions (both past and present), whereas other literature stresses fuel accumulation or both climate and fuel as the main explanations for high-severity fire.

Determining what proportions and patterns of various burn severities[6] may be desirable in the future is a question that goes far beyond the information available from either fire history research or elicited in our questionnaire. What is desirable will be based on fire's expected influence on ecosystem goods and services that are valued by people, and the social acceptability of those influences. Thus, the predominant viewpoint among land managers and policy makers is that wherever feasible, fire and fuels management should promote the fuel and successional conditions that will support the natural fire regime going forward. In areas such as wilderness, where commodity production is not a management objective, the goals are much the same. Regardless of the management allocation, heterogeneity of fire effects, including the pattern of patches created by fires and other disturbances, is important to forest resilience to future fires (see Topic E).

Respondents exhibited a wide range of opinions about desirable future proportions of burn severity. Some stressed that fire and forest managers often propose treatments designed to reduce future potential for large areas burned with high severity. In contrast, others explicitly stated the benefits of high-severity fire, generally stressing its role in providing habitats for certain wildlife species, forest successional heterogeneity, and biodiversity. Proponents of this latter viewpoint stressed recognition and agreement that allowing high-severity fires in the Wildland-Urban Interface (WUI)[7] was not socially acceptable. Some respondents noted a

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[6] Burn severity is ecological change due to fire, often characterized within the first year or more after fire. In contrast, fire severity refers to effects during the fire. From 1) Morgan et al. (2014). Challenges of assessing fire and burn severity using field measures, remote sensing and modelling. *International Journal of Wildland Fire 23*(8):1045-1060, and 2) Keeley (2009) Fire intensity, fire severity and burn severity: a brief review and suggested usage. *International Journal of Wildland Fire 18*(1): 116-126.

[7] Wildland Urban Interface (WUI): The area where structures and other human development meet or intermingle with undeveloped wildland or vegetative fuels. From NWCG glossary of wildland fire terms (https://www.nwcg.gov/glossary-of-wildland-fire-terminology, accessed 8 May 2017).

paradigm shift from a prevalent view in the 1990s that the only acceptable or “good fire” is a low-severity fire, to a growing viewpoint stressing the benefits of some level of moderate- and high-severity fires, as well as the need for societal adaptation to “managing wildfire” and “living with fire.” Many respondents stressed the importance of different management objectives in different settings (e.g., remote areas versus the WUI, general forest versus wilderness management), and of the clearly different historical fire regimes in low-elevation dry mixed-conifer forests versus cold subalpine forests.

Implications

Managers and scientists alike are challenged with overcoming the tendency to simplify historical fire regimes across and within ecoregions and forest types. While managing for the inherent complexity of fire regimes can be daunting and painstaking work, the resulting patterns and effects on processes provide important and compensating benefits. There is no single model of historical fire regimes applicable to all forest types and ecoregions. Managers should exercise care when applying scientific understanding developed in different landscapes, and recognize that this may result in erroneous scientific underpinnings and failure to meet objectives. Thus, management decisions are generally best-informed by area-specific understanding of fire ecology, which in some cases may require new partnerships between managers and researchers, both in implementation and monitoring. Scientists must clarify the importance of place when characterizing and presenting knowledge about historical fire regimes, and would benefit by sharing methodological approaches and collaborating across ecoregions. Stakeholders—from the general public to land managers to society at large—must wrestle with and decide what future proportion and pattern of burn severity might be desirable in each locality, both for the ecosystem, and for the people who live nearby and depend upon their services. Bearing this in mind, stakeholders will need to discuss the ability of various management prescriptions to achieve their desired changes, the social cost and acceptability of the changes, and alternative approaches to accomplishing them (see Topic I).

A logical way forward is to increase cross-regional and in-depth field research experiences within the fire research community. Improved collaboration across research groups, defined geographically or by previous narratives, can overcome some of the current atmosphere of deep distrust and interpersonal acrimony. Cross-regional comparisons of top-down and bottom-up determinants of fire activity is a fertile area of future research, which can examine how differences in seasonality, productivity, surface and canopy fuels, climatic differences, and other factors may explain some of the reported geographical differences in historical fire regimes in broadly-similar forest types. Likewise, inter-regional comparisons of various land-use practices by Native Americans and EuroAmerican settlers would improve our understanding of how these practices have contributed to past and present geographic differences in fire regimes.

Agencies like the US Forest Service and Bureau of Land Management, by virtue of their enabling legislation and Congressionally appropriated annual budgets, are legally required to manage for improved fuel and fire behavior conditions. Actions that can effectively treat large areas over a short period of time often suffer from an oversimplified understanding of the desired conditions. Because there are strong relationships among spatial patterns of surface and canopy fuels, seral stages, expected burn severity patterns, and onsite climate and fire weather conditions, care must be taken to avoid oversimplifying those patterns for the sake of simply reducing expected wildfire severity. Such oversimplifications can have profound effects on habitat patterns resulting from all burn severities, and their spatial complexity and connectivity. Thus, in each geographic area, managers must seek to obtain a clear understanding of the historical spatial patterns of surface and canopy fuels, and of seral stages through focused study and reconstruction of those conditions. Further, they should use modern climate change evaluation tools to assess how these historical patterns would be altered under the 21st century climate anticipated for that area. This larger understanding would enable managers to then consider conditions in this larger context, and develop landscape prescriptions to make the needed adjustments. Tools to be applied would be those that matched the land allocation and the specific needs for change.

Topic B. Human impacts and management history vary with geography

Common Ground

Key points of common ground among respondents to the questionnaire included:

- The influence of humans directly on fire ignitions and suppressions as well as on landscape drivers of fire activity is ubiquitous and important.
- Impacts of humans vary through time, and are not uniform geographically.
- Human influences are pronounced in dry, moist, and cold forests, but impacts vary.
- The role of human ignitions on wildfire prevalence and severity varies markedly in western forests.
- Climate change is a human impact and a strong driver of fire occurrence and effects.

Respondents to the questionnaire strongly emphasized that fire suppression[8], despite its widespread effects, was not the only human activity profoundly affecting fire regimes and fire-prone ecosystems. Most respondents mentioned other activities as influential in altering fire regimes, such as domestic livestock grazing, logging (selective, post-fire, and clearcut), diverse types of anthropogenic ignitions, mining, overly generalized reforestation practices, invasive plants and animals, road and rail construction, and land-use or development changes. Respondents also spoke to the decimation of Native American communities through the introduction of human diseases, and later marshalling onto reservations, which significantly reduced ignitions and cultural fire uses by native aboriginal people. Human impacts vary with degree of access via roads and trails, but even remote areas have been influenced by people. For example, selective and clear-cutting timber harvests have widely affected dry and moist mixed-conifer forests, where the favored commercial species principally grew.

Many respondents noted that the impacts of these different activities are known to have varied over space and time, posing difficulties for generalized characterizations of human impacts over broad geographical areas or forest types. In other words, there was strong consensus that geographical context matters, and this influences the local assortment of human impacts. Some respondents noted that wilderness areas and actively managed forests often have had different human use histories, including Native American influences, and therefore different trajectories. Wilderness areas, along with some national parks and large roadless areas, offer examples of potentially different human influences, and related opportunities for both research and management. A few respondents elaborated on similarities

[8] Fire suppression is the act of extinguishing or fighting fires. Fire exclusion has partially eliminated fires from the landscape using fire suppression and other land uses, such as grazing, settling in valleys, road and railroad building, and agricultural conversion of most native grasslands.

between the effects of some human activities (e.g., fewer fires may be due to active fire suppression, reduced Native American ignitions, and/or grazing that removed surface fuels) in certain ecosystems, while most noted that dense recruitment of shade-tolerant species has been a direct result of nearly-ubiquitous fire suppression efforts, or logging of large, fire-tolerant trees across many western ecosystems.

A notable point of common ground among many respondents and a chord that was detected throughout the literature was that human impacts have been most detectable and pronounced in dry and many moist mixed-conifer forests, where the most commercially desirable species were logged. This logging, along with fire suppression, has resulted in generally altered and often more-homogenous forest compositions and structures. Such homogenization of forests is often due to harvest of larger and older trees and species (like western white pine, sugar pine, western larch, Douglas-fir, ponderosa and Jeffrey pines) followed by regeneration of higher density, young shade-tolerant forests (of grand fir, white fir, subalpine fir, Douglas-fir, red fir, and incense cedar, or mixes of these species), or due to fire exclusion and a variety of other related mechanisms. Less agreement exists on the degree and causes of homogenization with regard to cold subalpine forests. Regardless, this relative consensus about where human impacts have been most pronounced hopefully provides a stepping-stone for further discussion and common ground.

Many also recognized that climate change at broad scales is a dominant human influence affecting fires and fire effects in all ecosystems. We address it here and in sections F and G because it is the one common denominator affecting all forest types and all fire regimes.

Areas of Divergence

Key areas of divergence of opinion among respondents included:

- The general applicability of “thinning and prescribed burning remedies” to offset human influences.
- The significance of human impacts on forest successional conditions in moist and cold forests.

The questionnaire was intended to elicit a wide variety of responses about the generalized applicability of forest thinning and prescribed burning techniques, in response to changes in fire regimes and forest successional and fuel properties that have occurred across different forest types, and in different geographic locations. These topics might have been better separated, which could have made the areas of agreement and disagreement more distinct. Regardless, there was a general pattern among respondents, based on whether they viewed the fire regime of the forest in question as more driven by fuels versus weather and climate (see also Topic F).

For low-elevation ponderosa pine forests and woodlands and to a lesser extent in dry mixed-conifer forests, respondents generally viewed thinning and prescribed burning to have wide utility, both for ecological and social reasons. However, some asserted that, even where such activities may be useful and justified, their effects may be better accomplished primarily through wildfire.

While a majority of respondents agreed with the statement that cold subalpine forests have been little affected by fire suppression, many studies highlight that human impacts on forest successional conditions have been significant in dry, moist, and cold forests in ecoregions of the northern Rockies, Inland Northwest, Pacific Southwest, and Inland Southwest. In particular, there is evidence in these ecoregions that once-complex cold subalpine forest patchworks composed of early, mid, and late-seral forest conditions have been simplified by extensive timber harvesting, fire exclusion and fire suppression, but also to a lesser degree by livestock grazing of the often widespread wet and dry meadows, and road development.

Implications

There is general consensus that human impacts vary widely across western US forests in terms of type of activity and associated ecosystem effects. Although some human activities had similar influences on many forest ecosystems, failure to recognize the heterogeneity of human impacts can lead to overly generalized prescriptions for forest restoration and management. Thus, there is likely no one-size-fits-all management or restoration approach—available to all conditions—due to the importance of locally-coupled human-natural histories, and current social or political considerations. Most fire scientists assume prehistoric Native American influences on fire and forests to have been relatively widespread, but to varying degrees in different landscapes and habitats. However, more clarity is needed about differences in how Native American and more modern human influences shaped forests of today.

The importance of local context in the management of fire-prone landscapes underscores the need to move away from oversimplified narratives that encourage application of fire research beyond its original scope of inference. Nonetheless, a widespread challenge facing land managers is the need to make forest management decisions in the substantial areas of landscape where fire-vegetation history research has not been conducted; this is a major future research need. General agreement about drier forests being the most impacted by human activities could provide a path forward among those disagreeing about the extent of high-severity fire in these ecosystems. Human impacts have been pronounced but with different effects and implications for moist and cold subalpine forests. Additional studies of landscape changes, and of vegetation response to fires and fuel treatments in these forest types, will inform discussions about forest landscape restoration and management.

To apply knowledge of the relative human impacts on local vegetation conditions, managers need to develop a clear understanding of the specific impacts geographically, their period of influence, and some understanding of their relative strength (also see Topic D). Important human impacts to date include:

- domestic livestock grazing, period of grazing, and density and types of animals grazed;
- introduction of non-native plants or animals, their distribution, and influence on herbivory and the local fire regime;
- wildfire suppression, including the number, locations, and timing of wildfires suppressed;
- timber harvest, type of timber harvest, and frequency of harvesting;
- presence of roads and railroads, their density, and the period of road impacts;
- historical frequency of Native American burning and time since that burning ceased;
- other changes in patterns and trends of anthropogenic (e.g., recent EuroAmerican) and natural (lightning) fire ignitions;
- conversion to cropland, exurban, or urban development, other conditions.

Research shows that the presence or absence of even a single one of these human influences can have profound effects on the resulting vegetation and fire behavior conditions. For example, the absence of timber harvest in some studied wilderness areas reveals significant differences in species composition and tree density in comparison with harvested locations growing in similar climatic conditions and forest types. Knowledge of the local human impacts, their period, and relative intensity can help guide the selection of areas needing and not needing restorative treatments, and it can aid in the selection of appropriate management tools.

Topic C. Fire is a keystone process[9] that occurs in almost all western US forest types

Common Ground

Key points of common ground among respondents to the questionnaire included:

- Low-, moderate-, and high-severity fires historically occurred in nearly all forest types.
- Fires of all severities play important ecological roles.
- Since nearly all western US forests are significantly fire-influenced, fire is a key driver of ecosystem patterns and processes.
- Burn severity patterns and resulting successional and fuel bed conditions have changed due to human activities in most forest types.
- In many western forests, a period of fire exclusion persists, reflecting successful passive and active suppression of the vast majority of ignitions (95-98%) over the past century.

There was consensus among respondents that various combinations of low-, moderate-, and high-severity fire occur in nearly all western US forest types, and associated agreement that fires of all severities play important ecological roles in each forest type. Unsurprisingly, there is also consensus that fire has been, is, and will continue to be an essential ecosystem process across nearly all western US forest types. A key challenge for researchers has been to estimate the proportions of fires that could be classified into one of the three commonly-used descriptive severity classes (low, moderate, high), and how those proportions may have changed over time.

An increasing emphasis in fire research conducted over the past 20 years has specifically aimed at estimating proportions of areas historically affected by low-, moderate-, or high-severity fires, but there remain uncertainties about the actual variability of burn severity historically. Some of this uncertainty is due to methodological limitations, especially in the case of high- and moderate-severity fires, where much of the evidence of past fires is destroyed. This renders fire history studies that exclusively use fire scars less useful under these conditions. However, much progress has been made in recent years by combining fire-scar data with extensive tree age data, tree growth release data, and data on tree mortality events, to provide a more nuanced understanding of the history of fire effects. In addition, aerial photographic reconstructions were employed in the interior Columbia Basin and East-side Forest Health Assessment studies, and these have provided expanded insights into the proportion of patches burned with low-, moderate-, and high-severity fires of those studied ecoregions across the 20th century (Topic H).

[9] A keystone process is one upon which other species and processes in an ecosystem largely depend, such that if it were removed or significantly altered, the ecosystem would change drastically.

Varying degrees of increased continuity of forest in all forest types (i.e., loss of early seral grass- and shrublands, and sparse woodlands and savannas) have been observed with implications for increased vulnerability to larger and more continuous crown fire disturbances, particularly in combination with successful suppression of all but the largest fires. A highly promising area of current research is the integration of dendroecological studies with the existing aerial photographic reconstructions currently covering millions of hectares across the northern Rockies and Inland Northwest. Recent research focusing on proportion of area affected by various burn severities and the emergent patterns represents an important improvement over the former focus almost exclusively on past fire frequencies.

There also was consensus that in many western US forests, there has been dramatically less fire activity over the last century than in prior centuries and millennia, tied to intense and pervasive societal efforts to actively suppress and exclude wildfires. Respondents broadly agreed that patterns of fire occurrence have changed in relation to historical patterns, especially in many dry forests, but also in some other forest types and locations. This is a response to changes in climate and/or fuel properties, recognizing that both extreme fire weather and combustible fuels have always existed to some degree (see Topic F).

There are many existing studies of fire history based on stand-origin mapping over study areas of many tens of thousands of hectares. However, a commonly held view in the fire science community is that even larger areas (i.e., many hundreds of thousands of hectares) are required for effective analyses—combining multi-century fire history data with landscape ecological approaches—to understand past fire patterns and simulation of future fire patterns. A fertile area of future research is analysis of large regional and local landscape historical patterns and patch size distributions of burn severity, and how these varied with topography, climate, prior disturbance, and other influences. Such research is needed because inferences are generally drawn from historical fire frequency, rather than pattern analysis.

Areas of Divergence

Key areas of divergent opinion among respondents included:

- Relative proportions of low-, moderate-, and high-severity fire within western US forests historically.
- Magnitude of changes in fire frequency, severity, sizes, and their consequences for various forest types since the 19th century.
- Magnitude of recent changes in forest patterns relative to historical conditions.
- The urgency, scale and overall need for various active and passive management options.

Key areas of divergent perspective among respondents centered on the relative importance of the various fire attributes that everyone agreed were generally important. For example, whereas numerically dominant perspectives can be identified, there was no consensus about the historical proportions and sizes of differing burn severity classes in some forest types, nor agreement about the magnitude of changes in fire frequencies, severities, and sizes; thus changes in the absolute significance and relative importance of different fire regimes in various landscapes is still debated. It is noteworthy that spatial reconstructions of historical proportions and sizes of differing burn severity classes in various forest types are relatively lacking in the literature for some ecoregions, which is likely a key reason for divergent opinions on this topic.

In particular, perspectives on historical patterns and changes in the occurrence and effects of both low-, moderate-, and high-severity fires in dry and moist mixed-conifer forests were a key area of divergence, with most respondents concerned over the negative effects from historical fire suppression, resultant fuel accumulation, and recent increases in high-severity fire. These observations contrasted with some respondents who highlighted climate and extreme fire weather over fuel accumulation as the main driver of high-severity fire, debated the historical relative importance of low- versus high-severity fire, and emphasized the ecological values and importance of past and present high-severity fires in all forest types, but less so in the driest forest types. Notably, respondents did not try to integrate the concomitant effects of weather, climate, topography, and fuel abundance.

We note that many studies use climate covariates to predict trends in annual area burned. These studies generally do not include fuel covariates, and lacking any evidence of the contribution of fuels covariates, conclude that weather and climate drive area burned. More important are area burned by severity class and changes in patch size distributions of severity classes, which lead to changes in patch size distributions of successional conditions. The lack of data on potential changes in the role of fuels may have fostered disagreements regarding the relative urgency and risks of various active (fuel treatment) versus passive (wildfire only treatment, suppression of human ignitions) management options, the appropriate locations and scale of desirable management actions, and the desirability and trade-offs among alternative forest fire management goals and actions.

Implications

Uncertainties associated with relative proportions of different burn severities and patch-size distributions combine to cloud key points of consensus that have important management implications. There is consensus that various combinations of low-, moderate-, and high-severity fire are important to ecological processes in almost all western US forests. Likewise, there is consensus that these combinations of burn severity, and their variability over space and time, contribute to seral stage pattern and complexity, and the future flammability of the landscape. Therefore, given that landscape patterns of successional and fuel conditions aid in controlling and are to a large extent controlled by fire, and that ecosystem function is altered in the absence of fire, the recent reduction of fire activity in many areas has important ecological implications. Managers are open to using fire on the landscape, but they often are unable to use fire alone. They have intimate knowledge of their landscapes and fuel characteristics, and many acres are not amenable to fire-only prescriptions. Managers wish to use combinations of tools, as is appropriate to the fuel conditions and the land management allocations, to restore more natural patterns of burn severity and of successional conditions that will support them down the road. They can use biophysical and topographic templates to tailor desired treatment patch sizes and intensities to their landscapes. And, they will have to accept some uncertainty about the effectiveness of their fire mitigation procedures under different future climates.

Public land managers throughout the western US are concerned with calibrating fire regimes in many forest types. Central to this idea of calibration is geographically pertinent knowledge of historical patch size distributions of seral stages, burn severity patches, and patterns of lifeform and physiognomic conditions. Nevertheless, paleo studies of fire covering multiple centuries to millennia show significant variability in area burned so that expectation of a long-term stationarity in fire patch sizes is unrealistic. Despite the likely lack of long-term stationarity, these landscape conditions and their variability contribute to the patterns and variability of fire regimes. Specific geographic knowledge of these conditions is often lacking and instead managers often apply knowledge of related or nearby systems, often with less than adequate precision. To learn how to better calibrate relative proportions of each burn severity and patch size distributions, managers should work closely with fire and landscape ecology researchers to improve their local characterizations of these historical conditions. Future proportions of low-, moderate-, and high-severity fire will depend strongly on local context, which includes the HRV, societal and political objectives, prior land-uses, climate and weather, topography, vegetation, and other factors.

Topic D. Knowledge of historical range of variability (HRV) is useful but does not dictate land management goals

Common Ground

Key points of common ground among respondents to the questionnaire included:

- Knowledge of the HRV provides essential context for discussion of land management decisions but it does not set management targets.
- There is no single model of the HRV of forest successional and fuel conditions and fire effects that can be applied across the western US.
- Because the HRV differed greatly from place to place, HRV findings from one area may or may not have relevance to another.
- Understanding the determinants of the HRV is useful in assessing future ecosystem responses to climate change and land-use practices.
- Although appropriate time frames of the HRV are often difficult to define, time frames must be specified for the HRV of particular attributes.
- Deep understanding of the HRV may require application of multiple research methods (see Topic H).

The HRV refers to the variation of ecological conditions and processes over spatial and temporal scales that are essential for understanding current ecosystem conditions[10] and their current departures. While historical patterns of fire and associated vegetation patterns are often the focus of HRV studies, comprehensive HRV studies also examine historical variability of many other factors including climate, impacts of forest insects and pathogens, and land uses. Interpretations of changes in fire regimes may thus be related to numerous potential drivers. These interpretations require consideration of climate variability as well as a broad range of land-use practices such as grazing, logging, mining, and management explicitly aimed at altering fire activity.

The HRV describes *a body of knowledge about historical conditions* without any explicit prescription for how that body of knowledge should be applied. In the sense of understanding how current landscape conditions reflect effects of historical biophysical processes and past human impacts, the HRV provides essential insights for how processes create and maintain spatial patterns of forest and non-forest conditions, and how those patterns in turn drive the processes of interest. Examples of the utility of HRV knowledge include understanding of how

[10] This definition and application of HRV is taken from: Hayward et al. (2012) Challenges in the application of historical range of variation to conservation and land management. Chapter 3 in: Wiens et al. (eds). Historical Environmental Variation in Conservation and Natural Resource Management. P. 32-45. Wiley-Blackwell.

past climate change and land-use impacts have affected modern landscape pattern and structure. Teasing apart the effects of land-use impacts such as grazing, logging, and/or fire exclusion on forest conditions from the effects of climatic variation on wildfire activity and forest conditions requires historical ecological understanding.

The respondents' comments reflected a strong agreement among scientists that knowledge of the HRV provides essential insights for decision-making in land management, in the context of current and future ecosystem responses to climate change. Hypotheses about climatic drivers of future ecological change can be developed and tested with HRV data covering a range of time frames.

Retrospective studies of fire are essential for developing a mechanistic understanding of disturbance-mediated ecological changes, including those driven by climate variability, which in turn supports the development of simulation models of future landscape dynamics driven by climate change. Some respondents stressed relatively abrupt or extreme changes in both historical and modern ecosystem conditions under climate variability as a basis for expecting future "surprises" in ecosystem conditions in the face of climate change. Other respondents suggested that future vegetation predictions from regional and global change models are still crude, particularly if those predictions do not consider fire feedbacks from altered fuel complexes and patchworks, and do not represent adequate advances in understanding sufficient to warrant reduced consideration of the HRV of any geographic area.

Respondents emphasized that knowledge of past natural variability is an essential reference for evaluating impacts of modern land-use practices such as grazing, fire suppression, and logging on current ecosystem conditions and processes. They noted the continuing challenge of distinguishing among the relative effects of past logging or grazing from effects of active fire suppression.

Many respondents stressed that the insights synthesized in an HRV assessment are intended to inform discussions of potential management goals that incorporate social values for decision-making. The value judgments involved in a deliberative decision-making process are improved by knowledge of HRV, but adoption of management goals is not dictated by environmental history.

Areas of Divergence

Key areas of divergent opinion among respondents included:

- In practice and in communicating with the public, static representations of the HRV often continue to be inappropriately emphasized.
- The applicability of HRV knowledge from well-studied regions to similar but less studied forest types in other geographical regions.

The areas of divergence reflected in comments of both survey respondents and in broader discussions with stakeholders appear to reflect different views on how HRV information should be applied to management decision-making. HRV studies are increasingly viewed as scientific and analytical tools useful in decision-making, not as the management goal. In that context, some stakeholders and fire scientists assume that the primary purpose of an HRV study is to reconstruct a set of vegetation parameters (e.g., tree sizes, stand densities, tree spatial patterns) as representing past "natural" conditions and suitable "reference conditions." Other fire scientists stress that such reconstructions may only be "snapshots" in time in the sense that their relevance is time dependent, for example possibly depicting conditions that may have existed ephemerally, but are not fully representative of the range of ecosystem conditions over a longer time period. Still others have shown that HRV conditions, when reflected via a space-for-time substitution sampling methodology, can adequately reflect historically extant variation in forest spatial patterns as reconstructed or simulated by state-transition models. These responses highlighted the importance of comparing alternative methods and time periods that may be used to predict or reconstruct variability of an HRV.

Many respondents emphasized that oversimplified models of the HRV are often applied indiscriminately across a diversity of landscapes so that actual ranges of variability are underappreciated. Numerous respondents identified cases where oversimplified models of HRV did not apply either to an entire study area or were inappropriately applied to landscapes where the model had not been tested through sufficient data collection, independent calibration, or observation. Some respondents noted that divergent views of the HRV reflected the transfer of general models and interpretations from regions that had been well studied, to regions lacking any similar studies that might highlight differences related to unique geography. This is often done based on the assumption that an HRV should be similar in broadly defined cover types.

Implications

The HRV is most useful as a guide to management. Although the HRV can provide invaluable insights about how various processes and patterns interacted in the past, each HRV is but one reference range – it can vary widely across different locations and temporal scales. Managers should exercise caution when applying HRV information collected in other landscapes, recognizing that there is no single HRV model that can be generalized across the entire western US or generally to certain forest types. Despite debates about specific methods and applications of HRV, there was widespread agreement that understanding the climatic, land use, and other determinants of past fire activity and fire effects is useful in assessing future ecosystem responses to climate change and land-use practices.

One of the difficulties facing public land managers is their concern about how to address climate change, wildfire area burned, and burn severity predictions for the mid-21st century, given the high uncertainty associated with those projections, especially projections of future vegetation and lifeform changes, which are thought to be some of the most uncertain. This uncertainty forces managers to generally lean on HRV predictions to hedge their bets going forward. Nonetheless, managers have tools to estimate near-future precipitation, water deficit, plant-available water, and evapotranspiration conditions over the next few decades, and these estimates can be used to condition their understanding of desired forest successional, lifeform, and fuel patterns, and patch-size distributions in light of HRV estimates.

Topic E. Forest structure, composition, and fuels have changed, affecting burn severity and fire extent

Common Ground

Key points of common ground among respondents to the questionnaire included:

- Historical landscape and disturbance ecology strongly influence fuel patterns and legacies of live and dead forests.
- Forest structure and composition have been homogenized in many places by timber harvest, fire suppression, grazing, mining, road-building and other activities.
- Fire behavior is patchy in space and time, and resulting patch-size distributions are important to understanding its effects on the landscape.
- Landscape patch configuration (heterogeneity) is important and is a key determinant of fire regimes, fire behavior and ecosystem function; not every configuration will do.
- Several spatial scales and types of vegetation and fuel heterogeneity exist, and each scale has important and different ecological functions.

In the western US, historical patterns of forest structure, composition, and fuels—collectively making up successional conditions—resulted from recurring wildfire, insect, disease, and weather disturbances that kill trees and regenerate forests. Through time, wildfires repeatedly affected most western forests. Burn severity varied with seasonal weather, previous fires and regional climatic conditions, but also topographic, biotic, and geomorphic conditions. Burn severity patches occurred in predictable frequency-size distributions, which captured the spatio-temporal variability of disturbance and effects on local and regional successional patterns. Within this historical context, respondents generally agreed that fires were prevalent and greatly influenced forests, though fire frequencies and effects varied. Further, respondents all agreed that this historical ecology needs to be incorporated into our understanding and management of forest landscapes.

Respondents identified a number of recent studies showing that successional patterns of many western US forests have been altered by 20th-century management. Management actions included timber harvests, wildfire suppression, domestic livestock grazing, mining, and road and railroad building, which generally fragmented successional patchiness, increased forest area and density, and created novel successional and fuel patterns. Chief among these changes was increased abundance and connectivity of dense, multi-layered young forests, with greater proportions capable of supporting crown fire. However, the degree of these changes has varied across forest types and ecoregions. There was general agreement that these changes occurred in many western ecoregions, especially in the dry ponderosa pine, Jeffrey pine, and in some dry mixed-conifer forests (see Topic B).

Several respondents commented on patch and landscape-level feedbacks, noting that landscape-level feedbacks mediated the frequency-size distributions of future low-, moderate-, and high-severity fire, whereas patch-level feedbacks influenced the likelihood of low- and moderate-severity fires. Prior fires were likely complex patchworks of already burned and recovering vegetation, which increased or decreased the size and severity of future disturbances.

Respondents noted that reconstructed historical landscape patterns, fire history studies, and simulation studies show how landscape successional and fuel patterns and their variability may have supported particular historical fire regimes. Unique ranges of vegetation and fuels patterns were the result of interactions among regional climate, topography, landforms, geology, and biotic communities of an area, along with associated meso- to fine-scale pattern heterogeneity. This pattern of heterogeneity was unique and important to facilitating local variation in burn severity patterns, habitat patterns, and was of central importance at all spatial scales.

Areas of Divergence

Key areas of divergent opinion among respondents included:

- The extent to which future fires and forests are constrained by forest and landscape legacies.
- Importance of bottom-up versus top-down variables in fire regimes.
- The relative amount of forest structural change of an area (e.g., increased density and more complex tree layering leading to increased vertical continuity of fuels that can propagate fire upward).
- Costs and benefits of fuel treatments at necessary spatial and temporal scales.

Respondents disagreed about the extent to which structural change and successional forest patterns have been altered by 20th-century management, as well as the relevance of these legacies for future fire regimes. For example, large landscape assessments in the Inland Northwest showed that the increased abundance and connectivity of dense, multi-layered young to intermediate aged forests, with high crown-fire potential, has occurred in dry, moist, and cold forests. In cold subalpine forests this has occurred via the elimination of formerly complex early-, mid-, and late-seral forest patchworks. In dry and moist forests in the Inland Northwest, this has occurred via increased area of forest (as meadows, sparse woodlands, and some shrub vegetation has been encroached upon by forests), and increased density of a once more-complex patchwork of open and closed canopy forests. In contrast to these patterns, respondents and the peer-reviewed literature for the Colorado Front Range, for example, agreed that for the lower elevation areas of dry ponderosa pine forests there has been a substantial increase in woody fuel connectivity. However, respondents noted that the peer-reviewed literature demonstrates a much smaller shift towards increased woody fuel connectivity in mid-elevation dry mixed-conifer forests and even less in the cold subalpine forests. These respondents noted that for dry mixed-conifer forests of the upper montane zone, abundant research does not support a pattern of significant shift towards a higher percentage of the landscape capable of supporting crown fires today in comparison with historical fire regimes, which also included moderate- and high-severity fires.

Overall, divergence of perspectives on the degree of change in vegetation structure and fire potential often reflects studies conducted in similar forest types but different geographical regions, although in other cases, there are fundamental disagreements over the validity or interpretation of evidence for the same landscape using different methods.

Another area of divergence can be traced to a lack of dialogue and theory integration between climate and landscape ecology researchers. A significant body of landscape ecology research shows that "emergent" properties have central importance to ecosystems and their pattern and process regulation, whereas climate scientists are less focused on local-scale feedbacks and emergent patterns. This creates a fundamental problem in linking climate change and landscape ecology research. Climate models assume that top-down climate covariates drive temperature, precipitation, and solar radiation conditions. Landscape ecology research shows that those top-down inputs can be highly modified by meso- and fine-scale bottom-up environmental controls to produce climatic conditions that are strictly speaking neither the top-down or bottom-up inputs, but are influenced by these inputs. Until the processes that produce such emergence are incorporated into downscaled climate modeling, and until landscape ecology studies incorporate the full suite of realistic climate futures, these uncertainties will remain a problem in applying climate change science to landscapes and their restoration.

Implications

There is consensus that landscape pattern, which is influenced by vegetation, topography, climate, and past fire disturbances, is nearly always an important mediator of fire size and burn severity. A variety of management and land-use activities have altered western US forest landscapes at multiple spatial scales, and essentially created a new landscape template for 21st century fire regimes. Successional and fuel patterns will influence future fires, including size and burn severity of patches. When historical patterns are unknown, efforts to create locally representative reconstructions may be needed.

Forest structure, composition, and fuels have changed to varying degrees in different areas, and in some forest types there is broadly shared common ground that these changes are affecting burn severity and fire extent. While changes observed in some dry forests became a prime motivator for agencies to act, and for Congress to focus financing on restorative actions, there is less common ground about the degree of these changes West-wide in other forest types. However, informed dialogue among scientists and managers, and in some cases additional research, can help to improve common understanding concerning the degree of change and appropriate restorative action for other forest types. Monitoring and adaptive management are needed, especially where reconstructions of representative historical patterns and predictions of future patterns are hard to come by. This is a prime opportunity for scientists to work closely with managers in support of resilience-oriented management. In these cases, a significant monitoring component will facilitate learning. Information gained may be used to initiate restoration of forest structure, composition, and fuels, using the tools that best fit the circumstances. Because simply applying the best available science will not always be sufficient to gain assent from stakeholders and interested parties, collaborative dialogue that factors in local social values and emphases tempered by that science may provide an adequate way forward.

Topic F. Climate and fuels both influence current fire sizes and their severities

Common Ground

Key points of common ground among respondents to the questionnaire included:

- Climate and weather are now and will continue to be primary drivers of fire size and annual area burned.
- Surface and canopy fuels are important drivers of burn severity.

Global and regional climates vary over centuries, decades, and between years, including conspicuous oscillations between the relative dominance of warm-dry versus cool-moist weather patterns. As recently as the late 20th century, a sizable portion of the ecological literature assumed relative stationarity in climate, but increasingly abundant and diverse lines of evidence overwhelmingly demonstrate that the Earth's climate, and that of its many ecoregions, has constantly varied over multiple time scales.

Changes at decadal, centennial, and longer time scales have the potential to redefine biophysical settings. Hence, maps of plant associations, environments, existing and potential vegetation, and physiognomic types are now all seen as shifting patchworks. In landscape ecology, this is an accepted view and is wholly consistent with its body of theory. However, in forest, plant, and rangeland ecology, this view of shifting environmental or biophysical settings has stretched thinking for many practitioners and researchers. Relating projected climate changes to anticipated changes in forest fuel conditions and fire regimes adds further complexity (see Topic G).

Operating within this broader context of changing climate and landscapes, respondents agreed that woody fuel quantity, arrangement, and moisture are important to both the current flammability of western US landscapes, and to the ecological effects of fires. Changes in fuels along with topography drive changes in energy release, fireline intensity, flame length, burn severity, and emissions. Respondents agreed that widespread increases in the area that is forested and in the fuel quantity and vertical and horizontal fuel continuity in many ecoregions and forest types have increased the likelihood of large forest fires and higher burn severities via increased likelihood of crown-fire initiation and spread.

Regional climatic variability and extremes also influence wildfire size and burn severity. Based on the last several decades of research, respondents noted that annual, decadal, and multidecadal climate variability has always been important to fire size, and annual area burned. Respondents also agreed that the largest fires have always been driven by extreme fire weather, and they will continue to be. However, within large historical fires, including those burned under extreme conditions, burn severity was often patchy in response to topography and vegetation (i.e., fuels) conditions. The result was variably-sized patches of low, moderate, and high severity within burn perimeters. These patchy burned areas have changed into the 20th and 21st centuries, and more areas are being burned under high severity than is often typical for the forest types. While this view is supported by many respondents and published studies, there are other studies that question its generality. For example, some research based on historical aerial photography in the northern Rockies on burn area and severity from the 1880s to the early 2000s showed that over this long record, the proportion burned with high severity did not increase, despite extensive area burned in recent decades. Likewise, studies based on satellite imagery, while generally showing trends of increasing burn area since 1984 across the western US, do not show increases in burn severity for all ecoregions or even in a majority of regions. However, we note that in pre-1900 low-severity regime landscapes of the southwestern US and low-elevation Colorado Front Range ponderosa pine ecosystems, the most spatially extensive fire years and likely the largest fires occurred in dry years that followed one or more wet years, which apparently supported buildup and broad-scale continuity of fire-spreading fine surface fuels. Smaller fire sizes and low- and moderate-severity fires are generally associated with milder fire weather and moderating climate conditions.

What has changed most significantly since about 1985 is the frequency of large fires in association with warming temperatures and drought. While some of the increase in the frequency of large fires is expected from increased woody fuel continuity, broad-scale studies based on robust research designs are still needed to tease apart the roles of changing climate and changes in fuels in the observed trends in frequency of large fires. Some respondents argued that the loss of the patchwork created by the historically superabundant small fire-affected patches also has contributed to larger patch sizes of recent forests, and in fact this is a key focus of much current research. In many forests, not just dry mixed-conifer forests, some respondents also noted that fire suppression has resulted in loss of the most numerous smaller and most extensive (in some landscapes) lower-severity fires, which has removed an historical resilience mechanism that once had regulated the frequency and severity of the largest fires by controlling fire growth. Expectations under projections of continued climatic warming include more effective fuel drying during years or seasons of reduced precipitation, as well as more extreme short-term events such as heat waves, driving extreme fire activity. This coupling has the ability to significantly alter the size distribution and burn severity of burned patches and functioning of affected landscapes, including their future physiognomic types[11] and patterns of species composition. What is apparently most important is that increasingly extreme fire weather is increasing the frequency of large and severe fires, and quite small increases in the frequency and extent of large high-severity fire patches can result in tipping points for ecosystems.

These points of common ground coincide with increasing evidence that when recent wildfires severely burn large areas of forest, local elimination of conifer tree seed sources and reduced tree regeneration under emergent warmer-drier conditions can occur. As a result, large areas of forest increasingly are converting to persistent grasslands or shrublands post fire in some regions.

Areas of Divergence

Key areas of divergent opinion among respondents included:

- With respect to current fire regimes, the relative importance of landscape changes in vegetation and fuel properties in comparison with weather and climatic changes.
- The degree to which the frequency of large, high-severity fires and large, severely burned patches within fires has increased, and over what time frames.
- The extent to which landscape tipping points have been reached as a result of high-severity fires.

[11] Examples of physiognomic types include evergreen broadleaf forest, deciduous broadleaf forest, evergreen needle-leaf forest, deciduous needle-leaf forest, grasslands, shrublands. From: 1) Kuchler (1949) A Physiognomic Classification of Vegetation. Annals of the Association of American Geographers, 39(3), 201-210; 2) Box (1981) Predicting physiognomic vegetation types with climate variables. Vegetatio 45: 127-139.

One core area of divergent opinion is the relative importance of landscape change to current fire regimes. Empirical research in some landscapes shows that landscape abundance and horizontal and vertical continuity of woody surface and canopy fuels has increased in many western US ecoregions, which when combined with empirical and modeling research on fire behavior, supports an inference of increased fire intensity, longer flame lengths, increased crown-fire ignition and spread potential, and burn severity (i.e., *fuels affect fire behavior and burn severity*).

On the other hand, much recent research concludes that trends in annual area burned or in numbers of large fires are explained by weather and climatic influences on fuel availability. In these latter studies, drought and related time series are used to predict annual area burned. Models generally show fair to good prediction of a positive climate involvement (i.e., *climate drives the recent increase in area burned*). However, more complex statistical models that show multi-way and multi-scale interactions among fuel properties, fire weather, topography, and climatic predictors of fire extent and burn severity are needed.

A critical limitation on this front has been the lack of quantitative data, for some ecoregions, on changing fuel properties geographically and by forest type. Currently, in some ecoregions, we know more about how area burned and fire extent are influenced by climate than how the ecological effects of fires are affected by both changing climate and fuels. We also know that burn severity varies with fire weather, topography, vegetation, and time since fire (or other disturbances), even when large fires are burning under relatively extreme weather. However, there are few studies that show the relative contributions of each of these factors and climate together to burn severity. Recent reports of increasing burn severity for some ecosystem types are mostly, but not entirely, limited to the 1984-present period, due to the limited temporal depth of Monitoring Trends in Burn Severity (www.MTBS.gov) data. In addition, some respondents were concerned about adequate validation of the MTBS data for that period.

A second core area of disagreement hinges on the degree to which the frequency of large, high-severity fires and large, severely burned patches within fires has increased, and how this differs for dry, moist, and cold forest types. Many respondents believe that the frequency of large fires has increased in association with *both* climatic warming and increased woody fuel abundance and continuity, but as noted, broad-scale analyses of the relative contributions of climate parameters versus altered fuels to observed fire trends remains an important research challenge. Nevertheless, for landscapes with documented large-scale increases in woody fuel connectivity, there is a widely shared concern that increased abundance of large high-severity wildfires has expanded the potential for creating broad-scale shifts in dominant physiognomic types.

Implications

There is broad agreement that both climate and fuels are critical regulators of fire regimes in western US forests. In extreme weather, fires are likely to be large and severe, and managers should be mindful that extreme fire weather is expected to become increasingly common in the 21st century. Under milder conditions, however, fire behavior is mediated by complex interactions among climate, weather, topography, vegetation type, and fuel properties that vary spatially due to successional patch structure and patch size distributions. Further, prior fires (both managed and wild) can alter the extent, burn severity, and patch size distribution of subsequent fires depending on time since fire, topography, climate, and other factors.

In many, but not all, portions of the West (including the Inland Northwest and Pacific Southwest, Colorado Front Range, and monsoonal Southwest), scientists and managers have a reasonably large range of studies documenting changes in forest fuel and seral stage patterns of interior forest types, especially those leading to altered fire regimes. It is likely that restoration activities that seek to reduce fuels and restore successional conditions and their altered spatial patterns can be adequately informed, in particular if appropriate attention is paid to the differences in forest type and habitat.

Topic G. The role of changing climatic conditions is increasingly important

Common Ground

Key points of common ground among the respondents to the questionnaire included:

- Climate variability is a key driver of historical and current fire regimes, with distinctive historical patterns of climatic drivers of fire activity evident in different landscapes.
- The western US has recently been affected by a rapidly warming climate, characterized by reduced snowpack, earlier springs, longer fire seasons, hotter droughts, and more frequent periods of extreme fire weather.
- Recent trends in many western forest regions of more large fires and more area burned are linked to recent climatic trends of hotter droughts and longer, more severe fire seasons.
- Projected climate changes toward substantially hotter and drier conditions in the western US are expected to become increasingly significant drivers of amplified forest fire activity and severity; associated climatic interactions with vegetation and fuel conditions will also increase in significance.
- Climate changes, along with other anthropogenic drivers of global change, affect many vital climate-driven forest processes that will interact with changes in fire activity.

Questionnaire respondents noted that climate variability is now accepted as a driver of both historical and current fire regimes in all western US forests. Distinctive historical patterns of fire activity—driven by periods of hot and dry climate—are evident and well-documented in numerous western US landscapes (see Topic F). This important consensus coincides with the broader scientific consensus that the current western US climate has trended hotter and effectively drier in recent decades. This hotter and drier climate has fostered reduced winter snowpacks, milder winters, earlier springs, more rain-on-snow events, longer fire seasons (at times 40 to 80 days longer), drier fuels, and more instances of extreme fire weather—all generally consistent with regional model projections of future climatic change. Some western ecoregions now have nearly year-round fire seasons.

Consensus also emerged from the questionnaire that these recent climatic trends are linked to changes in fire activity since about 1980-85, contributing to larger fires, more area burned, and more moderate- and high-severity fire in some western US forests. Projected future climate changes toward progressively drier fuels and more extreme fire weather conditions in the western US are expected to amplify forest fire size and area burned. Proportion of high-severity fire may follow different trends as burn severity is more affected by topography, vegetation, and fuel beds, and less by climate than area burned (see discussion about the relative importance of fuel treatments; Topic E). We note that climate and fire weather largely determine the moisture content of vegetation and surface fuels, which has a strong effect on the availability of fuels to burn, energy released by the fuel complex, and resulting flame length, fireline intensity, and smoke emissions. Given projected climate warming and drying in the West, current forest fuel accumulations will be reduced through time by anticipated increases in fire activity (although surface fuel loads typically spike within a decade as standing post-fire snags [i.e., dead boles and branches] fall down amidst diverse vegetation regrowth), by constraints on forest regrowth under a hotter and drier climate, and by forest transitions to non-forest vegetation over increasingly large areas. In some of these areas, afforestation due to lack of fire has occurred, which reduces vegetation flammability and rate of spread. Anticipated future changes in forest fire activity and fire effects ultimately will be modulated by these feedbacks among fire, fuels, vegetation succession, and climate.

Respondents also indicated that emerging climatic changes also widely increase tree physiological stress, and adversely affect tree regeneration, growth and mortality losses, and associated insect and disease outbreaks. Thus, ongoing and future climate-induced changes in forest extent, forest fire extent, severity, and effects must be understood in relation to these additional biotic, abiotic, and anthropogenic factors.

Areas of Divergence

Key areas of divergent opinion among respondents included:

- There remains a divergence of opinion over the relative contributions of climate change and fuel accumulation to current patterns and trends of wildfire activity.
- Effectiveness of fuel treatments under projected climate futures and associated more extreme fire weather.

All respondents agreed that climate change is occurring and likely to continue. The main divergence among respondents involved perceptions of the relative importance of climatic versus fuel factors as drivers of changing fire activity, both now and in the future. This basic divergence in perspectives emerged repeatedly in questionnaire responses, as noted in Topics E and F, despite a general lack of scholarly work to explore joint contributions of climate and fuel to fire extent and burn severity.

This divergence in perspectives about the relative importance of climatic versus fuel factors as drivers of changing fire activity also extends to a related divergence in views on the effectiveness of fuel treatments under projected climate futures and associated more extreme fire weather; this area of divergence is presented under Topic I.

Implications

There is wide agreement that climate has long been a principal regulator of wildfire activity and therefore there is broad consensus that climate change via decreased fuel moisture and more extreme fire weather will considerably impact future wildfire activity. There is also wide agreement that fuels are a principal regulator of wildfire activity and fire effects. Divergent opinions emerge with respect to the relative importance of climate and fuel accumulation. Looking ahead, managers should expect climate change to create conditions of declining favorability to historically dominant forest communities, including warmer droughts, reduced snowpack and other phenomena. These general climatic trends are likely to be conducive to longer fire seasons and greater fire activity in the 21st century. Increasingly extensive vegetation transitions to more drought-tolerant and better fire-adapted species and/or lifeforms are anticipated. Although fuel properties directly influence fire behavior and fire effects, managers require in-depth knowledge of all determinants of fire behavior, including expected climate-related effects on fuel moisture and vegetation and other ecological changes, to determine the extent of possible feedbacks with climate change.

Anticipated changes in western US wildland fire activity have the potential to disruptively challenge the sustainability of historical forest ecosystems and our linked human societies. We expect that a broad range of fire-related adaptation measures will be considered in many western forest landscapes, ranging from increased regulation of human land use activities (e.g., disincentives for exurban development, building codes, seasonal recreation restrictions), implementation of diverse vegetation treatments (including managed wildfire, prescribed burning, and strategically-placed mechanical treatments), to management of forest stand structures, tree species compositions, and genetic variability, in order to foster resilience to growing drought stresses and associated disturbances (fire, insect outbreaks, tree regeneration failures). We expect increased societal attention and preference for such adaptation efforts in order to increase the likelihood of favorable forest adjustments to increasingly novel climate and other emerging environmental stresses.

Topic H. Multiple fire ecology and fire history research approaches can be useful to characterizing fire regimes

Common Ground

Key points of common ground among the respondents to the questionnaire included:

- It is desirable to use multiple methods to reconstruct historical fire regimes. More can be learned using multiple approaches and considering data from diverse temporal and spatial scales.
- Integrating and interpreting findings derived from diverse methods, data sources, and different scales of inquiry can be challenging.

The interpretation of any research evidence and the scope of related inferences is limited by scaling and sampling concerns associated with the methods, and these limitations apply to all research methods. Respondents to our survey strongly agreed with the statement that "*New and important insights should be possible through studies that use and compare alternative sources of data, and results may be used to examine fire history and fire effects in the same study areas.*" Respondents disagreed with the statement, "*Even if we find many different study areas where alternative sources of data are available, there are too many uncertainties or incompatibilities among them to make such comparisons useful.*" Thus, respondents recognized the high potential value of using and considering multiple approaches, data sets, and scales of observation to more robustly assess historical fire regimes. Broadly speaking, this reflects widely-accepted scientific views on the general benefits of using multiple lines of evidence when possible, with increased confidence in conclusions when most results are in agreement.

For this project, we decided to focus on the evidence regarding fire regimes of recent centuries, although substantial paleoecological research using sedimentary charcoal and pollen data has been essential in expanding our understanding of long-term variations in fire regimes. All methods for reconstructing historical fire regimes are necessarily indirect. They may include, but are not limited to, interpreting evidence of past fires or the extent of fire-dependent ecosystems from historical documents, land surveys, aerial photographic reconstructions, fire-scar and growth-release data from tree rings, tree age and death dates from tree-ring data, climatic data linked with past fires, charcoal and pollen deposits, current characteristics of stands (i.e., structure, species, and stand age distribution), fire perimeter mapping, historical timber survey data, and use of statistical distributions for modeling stand-replacing fire. In addition to utilizing multiple methods, the use of clear and shared terminology is needed for effectively combining research approaches to characterize fire regimes. Similarly, the use of diverse archaeological, anthropological, and cultural resource research methods that address the extent and impact of aboriginal fire uses in landscapes can provide useful information in support of restoring culturally important landscapes and their fire-maintained cultural resources.

Respondents noted that multiple methods enhance the potential of inferring the severity and other ecological effects of past fire events, which is central to current debates about the relative proportions of fires of different severity in the past. There are diverse examples where western fire researchers have used multiple methods to characterize historical fire regimes. Commonly, there is general agreement among studies about characteristics of historical fire regimes, particularly for ecosystem types that have had a history dominated by either low-severity fires (e.g., leaving scars but not killing many adult trees) or high-severity fires (killing many adult trees).

In recent decades, we have increased our learning about the strengths and weaknesses of diverse methods and data sources for analyzing high-severity fire, and also the scope of spatial and temporal inference limits for reconstructing historical fire regimes and forest conditions in varied western US landscapes. A particular challenge has been elucidating historical spatial patterns, such as patch sizes, shapes and arrangement. Much of this expanded insight has come on the heels of examining relationships between documented fire histories and associated forest successional or cohort conditions. In particular, further developing studies that cross-walk dendroecological fire histories with aerial photo interpretation and cohort age structure analyses offer much promise. These methods too can be combined with simulation studies that may offer additional insights. Respondents recognized that a more productive approach to multi-methods analysis might be for research laboratories that specialize in one method or another to collaboratively join their strengths in designing, implementing, interpreting, and documenting results of such research through joint work in multiple landscapes.

Areas of Divergence

The areas of divergence in opinion among respondents included:

- The introduction of new methods for reconstructing historical fire regimes has, in recent years, resulted in unresolved debates regarding the limits and usefulness of some new and old methods.

  There currently is significant debate about the validity and thus utility of some new approaches using historical (General Land Office, GLO) and current (USFS Forest Inventory and Analysis, FIA) land and timber survey data to infer the amount of high-severity fire, forest species composition, and the density and age structure of historical forests. Similarly, extrapolating from historical tree-ring and fire-scar point data across much larger areas has been a topic of some debate, but the disagreements are quite different. In the former case, disagreements center around the usefulness of the land survey data to the ends applied. This results from doubts regarding differences in interpretations of historical fire regimes based on tree-ring or other data versus historical land survey data. In some cases these differences are large but in other cases the percentages of a landscape classified as having an historical fire regime of mainly low-severity versus mixed (or higher) severity fire are relatively slight. The validity of reconstructing historical forest conditions and fire regimes in particular from all types of historical land or timber survey data has been critiqued. Such scrutiny of the validity of methods is a normal part of the scientific process, and highlights the need for continued research based on cross-validation from multiple types of data and methods.

Implications

The use of multiple methods for characterizing historical fire regimes, combined with increasingly clear and shared terminology, can improve our understanding of HRV patterns and processes in western forests. However, there can be significant challenges associated with bringing together evidence about historical fire regimes from differing methods and data sources. Each line of evidence has a different scope of spatial and temporal inference, and issues about the nature of the data captured in each sample. In addition, there is substantial skepticism about the utility of some methods for HRV reconstruction purposes, which will have to be resolved. Nonetheless, one new frontier of fire ecology research is the exploration of multi-method approaches by collaborating labs toward more-nuanced understandings of diverse fire regimes. For example, in mixed-severity fire regime forests, by combining time series derived from diverse dendroecological data sources (e.g., fire scars, death dates of trees, establishment of postfire cohorts, growth releases on surviving trees), land survey data, aerial photographic interpretations of successional and past fire severity conditions, landscape panoramic photos, and simulation modeling, stronger inferences may be possible about the ecological effects of past fire events.

Topic I. Many existing fire management tools and strategies can be useful for managing fire going forward

Common Ground

Key points of common ground among the respondents to the questionnaire included:

- Many tools can be useful to fire managers for reducing human vulnerability to fires and increasing ecosystem resilience.
- Managed wildfire is underutilized but viable ecologically and socially in many areas.
- Managing fuels is important and fuels are one contributing factor that can be influenced through management.
- Thinning alone without managing the resulting fuels increases surface fuels and does not mimic many of the ecological effects of fire.
- Firefighter and citizen safety, degree of smoke production, financial costs, and effective scales of treatment must all be considered.
- Land-use and financial incentives could be used to reduce human vulnerability to wildfires in and near the WUI.

Many respondents stressed that a wide variety of tools and policies can be useful to increase forest resilience and reduce human vulnerability to future fires. Suppressing fires to protect highly valued resources is important, but managers need a full suite of active and passive management strategies and tools because different management situations often call for different approaches. There was strong support for managing wildfires to accomplish resource benefits and also support for prescribed burning[12]. We agree. However, there was very little discussion of how and where wildland fire use can be effectively implemented to foster desirable patch size distributions, particularly where climate and forest conditions have changed, and surface and canopy fuels have accumulated over the period of fire suppression. Broad-scale landscape planning for wildland fire use will be essential to better understand special circumstances and clear opportunities for its use.

Wildland or prescribed fire use can be effectively complemented with fire suppression strategies and with thinning to reduce vertically and horizontally continuous fuels that contribute to fire hazard. Tools such as the Wildland Fire Decision Support System are used to make effective fire management decisions considering landscape conditions, jurisdictions, fire weather, values at risk and local management objectives.

Increasing education and outreach, managing post-fire to reduce soil erosion potential where values are at risk, decommissioning roads, creating snags where they are in short

[12] Wildfires are ignited by people or lightning. They may be suppressed, either aggressively or with more limited efforts, depending on management objectives, values at risk, costs, firefighter risk, and other factors. Managed fires are those that achieve resource objectives. They are monitored and parts may be actively suppressed while other parts are managed with less aggressive suppression. Prescribed fires are ignited by management actions under certain, predetermined conditions to meet specific objectives, such as reducing hazardous fuels, improving habitat, managing cultural resources, firefighter training, fire behavior experiments, or restoring forests. Prescribed fires are nearly always conducted under written, approved plans.

supply, and other tools can further help accomplish management objectives, while protecting people and property from fire and fire effects. Other strategies for helping communities become more fire-adapted include altering residential development in highly fire-prone environments, and making existing homes safer from wildfires. Land-use (e.g., applying the national WUI building codes proactively and retroactively, zoning to concentrate development in lower fire danger environments) and financial incentives (e.g., tax, insurance, mortgage restrictions, fees, assistance with fuel treatments around homes and towns, support to mitigate structure ignition vulnerabilities) could be used to reduce vulnerability to wildfires in and near the WUI.

Certainly, fire managers must consider financial costs, firefighter safety, public safety, and smoke. These and other societal and operational management constraints vary geographically, so managers must look for opportunities to adapt and use multifaceted strategies. There was widespread agreement among respondents that the suitability of different tools is highly context specific. In discussing strategies, both the often significant beneficial and detrimental consequences of taking no action must be considered.

There is strong consensus that more fire is needed on the landscape, but not all wildfire behavior or extent will do. Managers need assistance and funding to create landscape conditions that favor more desirable fire behavior at spatial scales and extents that can make a difference to current conditions.

Respondents generally indicated that the scale of landscape change in western US forests is quite broad, and that it could be difficult to overcome (i.e., a high level of landscape inertia), especially with the current level of defunding of public land management agencies. The cost of fire suppression has risen from 17 to nearly 60% of the entire Forest Service budget in the last 25 years, greatly limiting the financial capacity of the agency for proactive work at any meaningful scale. Treatments need to be of sufficient scale and pattern to be effective at restoring patch-size distributions of low-, moderate-, and high-severity fire, and at reducing what is seen by many as an increasing risk of unusually large, high-severity patches within fires. Although fuel treatments can be prioritized across very large landscapes to be potentially effective in managing wildfires to accomplish resource benefits, such treatments must be designed consistently with other ecological and management goals including riparian corridors, habitat for listed species, and the like. Restorative treatments likely need to occur at the scale of the landscape changes to change current fire regime conditions. Due to widespread existing habitat reserve commitments, opportunities for strategically allocated treatments are substantially limited.

Given the profound influence of the type and amount of fuel on fire behavior (Topic F), the type, location, timing, frequency, and maintenance of fuel treatments[13] will all influence their effectiveness. Forest thinning is one commonly applied fuel treatment. Most cutting methods that are applied to reduce future burn severity are thinning treatments where emphasis is on removal of the less fire-resistant trees (usually the smaller ones especially of shade tolerant species). The intensity of thinning determines the amount of branches and tree tops (slash) left behind. There is consensus that follow-up burn treatments of this slash are critical, however, this can be logistically and financially challenging because of highly restrictive smoke management policies. Post-harvest slash burning typically involves burning of piled slash concentrations, and in some cases, broadcast burning of remaining fuels. Prescribed burning

[13] Fuels treatments and fuels management include planned prescribed burns, mechanical treatments such as mastication or thinning, and silvicultural treatments and other treatments designed to change or reduce wildland fuel quantity and arrangement, the intensity of future fires, and increase the ease of fire suppression.

can also be done independently of thinning to reduce surface and ladder fuels, and to reintroduce more natural fire to the ecosystem. There are often significant constraints to this sort of prescribed burning though. For example, where surface fuels are too abundant, and where tree density and layering are significant, burn-only treatments are difficult to execute with any certainty. Burn-only treatments are also highly influenced by favorable fire weather (moderate conditions are best to accomplish goals), availability of fire crews, and smoke management restrictions. Respondents support efforts to overcome these roadblocks so that more fire can be reintroduced. Additional prescribed burn considerations for managers include improving public support for them, helping to design fuel treatments that mimic historical fires, using more fire during the fire season, enlarging the number, size, and positive effects of burns, and decreasing undesired effects of slash burning. Prescribed burns also consume less fuel and are far smaller than large wildfires, thus they produce far less smoke and smoke exposure to the particle sizes that are most harmful to human health.

Many respondents noted that managed wildfire is underutilized. It can be a viable tool ecologically, despite operational constraints. Ideally, this will result in more area burned under less than extreme weather conditions, and more moderate-severity fire effects resulting in heterogeneity that can be more consistent with both the historical range of variability and long-term management goals, if only for altering where and how future fires burn. Practically, there are large areas where mechanical thinning is neither allowed nor feasible, for example in wilderness and roadless areas. Managing wildfire may be useful there for reducing fuel quantity and altering vegetation composition and heterogeneity consistent with management objectives and enabling policy.

Wildfires can sometimes be managed at less cost and less risk to firefighters in areas where other fuel treatments are neither feasible nor desirable. Advanced planning is needed, as is accepting long-term risk and smoke when such fires burn for many days. Public lands are sometimes mapped into zones designated for particular management, included allowing fire. Challenges include societal constraints (e.g., smoke, fear of fire, concerns about shifts in weather, and distrust of managers and scientists) and operational constraints (e.g., costs, long-term risks, timing, and suitable weather). Smoke from fires poses human health hazards and visibility issues. Despite best efforts, some managed wildfires will not go as planned. The biggest challenges are the expanding area of WUI, public and political perceptions of fire and smoke, and unpredictable changes in fire weather. When homes burn, the fear of wildfires and their smoke often fuels political support for aggressive fire suppression, which reinforces the current predicament. But there are beneficial aspects of wildfire smoke too. For example, in northwestern California, Mid-Klamath Basin tribes recognize benefits of canyon smoke inversions for reflecting direct sunlight, and cooling air and river temperatures that can benefit native salmonids. Smoke is also a naturally occurring fumigant that reduces nut, seed, and acorn infestations by forest insects, and along with fire, facilitates seed germination of some native plants if smoke occurs during the natural fire season.

Managing wildfires to accomplish resource benefits may be one important way to achieve relatively widespread vegetation change at the spatial scales and in the short time frame needed to make a difference in the short-term. Depending on the situation, this will typically require strategically pretreating a portion of the landscape using prescribed fire—sometimes coupled with thinning—to reduce vertical and horizontal continuity of fuels, and to anchor managed wildfire or prescribed burning treatments. Such strategies can help manage risks and help society be more comfortable with less aggressive fire suppression, especially in or near the WUI. In remote locations far from the WUI and most vulnerable infrastructure, fairly typical mixes (for the fire regime of interest) of low-, moderate-, and high-severity fires may be a desirable and achievable outcome that is compatible with forest resilience, despite the many challenges managers face in managing wildfires.

Where there is a high concentration of values at risk and sensitive human populations within the WUI, aggressive fire suppression and fuel treatments may be the only socially acceptable strategies. In these situations, managing forests abutting the WUI with thinning and prescribed or pile burning, and aggressive fire suppression, will be appropriate.

Areas of Divergence

Key areas of divergent opinion among respondents included:

- Appropriate locations, scale, and effectiveness of thinning aimed at reducing fire hazard.
- The scale of thinning that can be feasibly and repeatedly implemented relative to the scale of the need.
- Advantages and disadvantages of managed wildfires, including acceptable levels of risk.

Communities often feel a strong sense of urgency and need for hope in the face of threats from wildfires. For areas distant from the WUI and municipal watersheds, some respondents disagreed with the degree of urgency and scale of need for thinning and prescribed burning.

Another area of divergent opinion is the role that forest restoration and fuel treatments might play in charting a new course for forest resilience, especially where public lands are considered. Arguments for restorative treatments range from concerns that prescribed burning or other treatments are needed where the condition of many forests before wildfires can result in undesirable burn-severity and patch-size distributions, to the belief that treatments are not restorative because large areas can only be effectively restored by proactively working with wildfires that are assumed to be “natural.” A range of other arguments falls somewhere on this continuum. One challenge is that fuel treatments may not be performed at a necessary pace and scale, especially when coupled with operational maintenance costs over time. Another view is that fuel treatments are less important in areas that would have experienced some degree of high-severity fire, and these areas may have been widespread and have greater positive influence on biodiversity and wildlife than is currently understood. This is underscored by strong opposition to partial to complete post-fire logging (salvage) of fire-killed trees, because some snag forests provide valuable habitat, and there are concerns about ecological integrity. The crux of this disagreement is whether the dead trees are most useful for their commercial or ecological values. Another view shows that some areas of high-severity fire tend to burn again at high severity, and that efforts to treat fuels and re-create more varied successional and fuels mosaics can help break this cycle. Yet another view asserts that in some instances, burned forests would benefit from removing dead smaller trees that could constitute critical reburn fuels.

Some scientists’ opinions diverge regarding the relative importance of climate and weather (fuel moisture and availability to burn), and fuel quantity (Topic F). For example, for some, there is more acceptance of the utility of fuel treatments within dry forests than in cold subalpine forests. Further, many scientists differed on the scale of treatment needed to influence high-severity fire at landscape scales because of questions about treatment effectiveness given the large amount of fire that burns under extreme weather conditions. Indeed, most current large wildfires are not even finding fuel treatments at the current low level of application. There can also be problems with non-native invasive species increasing in abundance following thinning and/or fire, particularly in lower-elevation forests. Many disagreed on the extent to which levels of high-severity fire and landscapes have changed, and the degree to which fuel treatments far from the WUI are a net benefit. The degree of divergence differs by forest type and landscape context, with stronger agreement about landscape change for dry mixed-conifer forests, and less for landscapes dominated by cold subalpine forests. Another argument by some for not actively managing landscapes outside the WUI is that even where there is strong support for treatments, the cost and difficulty of implementing and maintaining existing treatments may already be too great for society to absorb; this consideration is beyond the scope of this report. Of course, societal cost and practicality must be considered in the context of potential loss of forests, fire-adapted biota and other resource and social values. Further, reduced reliance on fuel treatments might imply an increased use of managed wildfire, but there is currently no consensus framework for weighing the costs and benefits of managed wildfire.

Some divergent opinions derive from establishing forest treatment targets, especially when those targets are not yet socially acceptable. For instance, thinning from below (removing many smaller, fire-intolerant trees while leaving older, larger trees) will reduce fire hazard under many circumstances, and can be a first step in ecological restoration treatments in many dry mixed-conifer forests. However, these treatments must be followed by prescribed burning, and a certain amount of smoke production, to reduce fuels and potentially restore ecosystem processes in the short-term. In some cold subalpine forests, however, where fires are more often limited by weather than by fuels, fuel treatments beyond the WUI may be relatively ineffective when and where fires spread by long-range spotting.

Some respondents noted that current landscape conditions reflect suppression of most fires, effects of past logging, and land uses that have often resulted in landscapes that are more homogeneous fuel-wise, notwithstanding widespread fragmentation by roads. Some respondents argued that these more-homogenous landscapes are more vulnerable now to very large patches burned with high-severity fire relative to historical conditions. Others saw less divergence between present and past high-severity fire potential (see areas of divergence Topics B, E). Careful analysis is needed in each unique geographic location.

Another challenge is that treating large areas is difficult when there is strong level of distrust. Collaboration with diverse groups has in many cases strengthened trust, especially when treatment approaches have been altered through a consensus-building process.

Implications

Managers seeking to reduce human vulnerability to wildfire and enhance forest ecosystem resilience should have available to them a flexible set of management options that includes suppression, thinning and other fuel treatments, prescribed burns and managed wildfires, as well as broad education on both the essential roles of fire and on prevention of undesired human-caused fires. The uses of these various tools should depend highly on management priorities and local context, including vegetation structure and composition, legacies of past fuel treatments and land use, the historical range of variability, presence of houses and resources people highly value, and acceptance by people. In the future, prescribed fire and managed wildfire will be useful to increase or maintain forest structural heterogeneity and restore associated ecosystem processes. Fires can limit the extent and severity of subsequent fires.

Fires respond to interacting influences of climate, weather, fuels, topography, legacies of prior disturbances, and management. The relative importance of these factors varies across landscapes and through time. Those who express that increasingly extreme fire weather with climate change will increasingly override the importance of fuels argue that fuel treatments should be focused around the WUI with limited fuel treatment elsewhere. Their logic is that direct protection of human assets is the top priority on which to focus, and that fuel conditions are less important as fire weather becomes more extreme. Those who emphasize the importance of fuels to fire behavior urge strategic fuels management in both WUI and non-WUI forest landscapes, using a variety of tools and prescriptions as needed across dry, moist, and cold forest types. They also assert that fuels are the main landscape characteristic that management can change. Where fuels and vegetation patterns have changed to foster more contagious fire spread, fires will be widespread and often large when fire weather and fuel moisture are conducive, particularly where grass fuels are continuous.

Going forward, monitoring is important to assure that fire management supports long-term vegetation management goals, particularly in the context of climate change, or to modify management to better align it with goals. We need to learn where fuel treatments are effective under different environmental conditions and where they are not, and then we must adapt management informed by monitoring. Scientist-manager partnerships could be particularly useful here to develop useful monitoring frameworks.

Where managed wildfire is not socially acceptable, more aggressive fire suppression and fuel treatments will be appropriate, along with prescribed burning. Many wildfires will occur and some will be large. With thoughtful management, we may be able to influence their severity and spatial extent under many but not all fire weather conditions.

To address future challenges in the face of expanding WUI, longer fire seasons, and altered forest conditions, managers need many different tools to balance ecosystem needs, costs, risk to firefighters and the need to protect people and property from fire. Federal fire policy allows this flexibility, and fire managers need it if they are to reduce societal vulnerability to fire and smoke, while also limiting costs and risks to fire personnel, and managing for ecosystem values. Addressing the vulnerability of the WUI depends on other approaches as well. Thus, policies to make current WUI communities more fire adapted are critical, as are changes in land use policies that influence where and how future WUI areas develop.

Conclusions

We found much common ground that will be useful to scientists, managers, and others for moving forward. There is wide agreement among scientists that fire is one of the most essential and pervasive influences on the forests of the western US. Further, fires can produce more positive benefits and fewer negative impacts when they burn with an ecologically appropriate mix of low, moderate, and high severity, and in patch size distributions that reflect the natural variability of fire behavior and fire effects.

Many questionnaire respondents suggested that the real challenge is to face the twin realities of increased abundance and connectivity of woody fuels and a changing climate. Not surprising, there were differing opinions about trade-offs between social values and the ecological benefits of fire. For example, smoke from wildfires or prescribed fires is a great concern that can have important influences on how various fire treatments are applied. There was wide agreement on the need for land management strategies that reduce societal and ecosystem vulnerability to negative consequences of wildfire, while providing for the essential role and benefits of fire in forests.

Areas of agreement outnumbered areas of disagreement. Respondents agreed that geographical context is very influential and that human impacts vary, and therefore there is no single one-size-fits-all management prescription. There was strong support for utility of the historical range of variability (HRV) for fostering understanding of how and why ecosystems have changed, and how they respond to fires of varying severity. Despite rapidly changing conditions, HRV will continue to be useful as a guide, but not a prescription for future landscapes. From HRV we can learn how ecosystems respond to wildfires, and the HRV of landscapes and ecosystems forewarns about ecosystem capacities and limitations in response to varied climate and disturbance drivers. As a guide for managing future landscapes, history does not provide precise prescriptions, but does offer precautionary principles. We fully recognize that adaptive resilience for the future will require applying what we learn from history to some future range of variability, where fires burn and ecosystems respond in both similar and different ways. There was strong support for prescribed burning, coupling thinning with prescribed burning, and for managing wildfires to accomplish resources objectives. This is common ground.

Forest structure, composition, and fuels have all changed, affecting burn severity and successional patch size distributions. Climate and fuels together with topography will influence future fires and their effects. There was consensus that many fire management tools and strategies will be useful moving forward, and no tools should be excluded.

We challenge managers and scientists to overcome the tendency to oversimplify historical fire regimes across and within ecoregions and forest types: there is no single model of historical fire regimes. Managers should exercise caution when applying scientific understanding developed in different landscapes and recognize that this may result in erroneous scientific underpinnings and failure to meet local objectives. Rapidly changing circumstances suggest that future management should be highly adaptive, incorporating learning from what works well and poorly. To adopt an adaptive management stance though, managers will need to engage in ongoing monitoring to detect and learn more about the best and poorest methods and outcomes. Scientists can work with managers in these practices, and such partnerships could provide a potent resource for managers. Scientists must also clarify the importance of place when characterizing and presenting knowledge about historical fire regimes, and scientists and managers would both benefit from sharing methodological approaches and collaborating across ecoregions. Scientists and managers should work together with science communication experts to create training and reference materials that capture appropriate levels of simplification and complexity.

Broader discussions center on issues where there is less common ground, including:

High-severity fire

There was strong common ground that for dry pine and some dry mixed-conifer forests, there has been either an observed increase in high-severity fires or an increase in the potential for fires of elevated severity. These changes have occurred as the result of increased area and density of forests, and increased connectivity of woody fuels since the late 19^{th} century. In contrast, for cold subalpine forests the majority of survey respondents agree with the statement that these forests have been less affected by fire suppression. Yet, large-scale landscape assessments of forest spatial patterns in the Inland Pacific Northwest show that recent (i.e., post 1984) patterns of high-severity fire and changes in patch size distributions in many moist mixed-conifer and cold, subalpine forests reflect significant departures from longer-term patterns linked to both climate, increased forest area, and increased density, layering, and connectivity of these forests. Expanded woody fuel connectivity is a result of synchronized successional conditions as a consequence of fire suppression and fire exclusion. Suppression of wildfires in moist and cold forests has yielded much lower prevalence of early seral conditions, and increased connectivity of mid- and late seral conditions, which has concomitantly increased landscape connectivity of conditions that are conducive to initiation and spread of crown fires. Conclusions differ for some cold subalpine forests in the Southern Rockies based on published studies and survey responses. Unfortunately, we don’t have similar information on landscape change across moist and cold subalpine forests in some other ecoregions. Reasons for these different perspectives on the degree of long-term change in the extent of higher severity fire are varied and complex. Some of the variability in scientific perspectives is empirically attributable to geographical differences in the factors determining historical and modern fire regimes. Others reflect disagreements over methods of examining changes in fire regimes and the interpretation of the evidence of past higher severity fire. Still others reflect the goal of influencing management.

Dry mixed-conifer forests (including areas once dominated by pure or nearly pure ponderosa pine), moist forests, and cold forests have all changed in recent decades. The degree of change is not the same everywhere, yet fires interacting with climate and current forest conditions have the potential to create very large patches and a relatively high proportion of areas burned with high severity. This has implications for post-fire tree regeneration (without which forests convert to non-forest), soil burn severity, and related erosion and watershed change, and other ecosystem services valued by society and affected by varying plant successional processes and trajectories. Non-forest vegetation may be maintained by fire where it is not suppressed; even in the absence of fire, however, forests may not regenerate if seed sources are not available. In some areas, forests have increased and non-forest decreased, due to fire suppression.

Both fuel and climate are important as increased woody fuel connectivity in combination with a warming climate trend is setting large areas of landscapes on fundamentally new trajectories where very large patches burn with high severity. Climate is of increasing importance.

Empirical and simulation studies and landscape ecology theory suggest that even small increases in the frequency of the largest high-severity patches can have a semi-permanent influence on future local and regional landscape habitat configurations and wildfire frequency, severity, and spatial extent. Thus, individual fire events can change the broad-scale resistance of the landscape to future wildfires. How these scenarios will play out under continued warming and more fires is highly uncertain.

We suggest that resolving many disagreements depends on greater consideration of specific geographical context. A logical suggestion is to increase in-depth cross-regional field research experiences of the fire research community. Cross-regional comparisons of top-down and bottom-up determinants of fire activity in similar forest cover types is a fertile area of future research to examine how differences in seasonality, productivity, understory fuels, land use history, and other factors may explain some of the reported geographical differences in historical fire regimes in broadly similar forest types. Likewise, systematic regional comparison of the timing and nature of land-use practices by Native Americans and European settlers on fire regimes would improve our understanding of how changes in anthropogenic ignitions, fire exclusion, logging, ranching, mining, and landscape fragmentation may have contributed to geographic differences in historical fire regimes.

There are several reasons for the disagreements about patterns of past fire severity. First, both scientists and managers often uncritically transfer concepts and findings from one place to another (see Topic A). We know that fire effects at a point depend to some degree on the surrounding landscape and forest composition. Some of the disagreement derives from debates over the relative utility and validity of different scientific methods; nonetheless we believe that application of diverse research approaches is useful in HRV reconstructions. We challenge fire scientists who do not share similar perspectives on historical fire regimes in particular ecosystems to engage in civil discourse to better understand the reasons for their disagreement and to objectively communicate those reasons to managers and other stakeholders. We are heartened by the positive outcomes achieved by some previous attempts when small or large groups work together to find common ground.[14]

WUI and beyond

Respondents strongly agreed on the need for fuel treatments and aggressive fire suppression within and adjacent to the WUI. The strategies for managing wildfire will be quite different within and adjacent to the WUI than in areas far from the WUI. However, what fire managers do beyond the WUI has implications for forest resilience, smoke production and its human impacts, water quality, and many other ecosystem services people value.

Fuels management alone, especially if limited to public land, will not be sufficient to address the vulnerability of WUI communities to fires. Fuels management will be important for influencing the resilience of the future forest, and for influencing the behavior of wildfires that approach the WUI. Thus, policies to make current WUI communities more fire adapted (e.g., implementing current WUI codes) are a critical piece of the puzzle, as are changes in land use

[14] See: 1) Kaufmann et al. (2006) Historical fire regimes in ponderosa pine forests of the Colorado Front Range, and recommendations for ecological restoration and fuels management. *Front Range Fuels Treatment Partnership Roundtable, findings of the Ecology Workgroup.* 2) Romme, et al. (2009) Historical and modern disturbance regimes, stand structures, and landscape dynamics in pinon–juniper vegetation of the western United States. *Rangeland Ecology & Management* 62(3): 203-222. 3) Baker et al. (2017) The landscapes they are a-changin' – Severe 19th-century fires, spatial complexity, and natural recovery in historical landscapes on the Uncompahgre Plateau. Colorado Forest Restoration Institute.

policies that influence where and how future WUI areas might develop, and the spatial extent and arrangement of managed and wildfire fuel treatments.

Pattern and Process for Fires in Forest Landscapes

Heterogeneity of fire effects, including the pattern of patches created by fires and other disturbances, is important to forest resilience to future fires (see Topic E). There are potentially profound implications for forest function and carbon sequestration if the proportion of area burned with high-severity fire changes. Even more importantly, as the patch size distribution changes greatly, particularly with respect to the proportion and size of the largest patches burned with high severity, there are multiple ecological consequences. For instance, the proximity of seed sources from surviving trees of fire tolerant species can affect forest regeneration, and the future flammability of the forest. Wildlife habitat use will change, both for those species dependent on hiding and thermal cover adjacent to more open areas and for those thriving in recently burned forest openings. Similarly, the fire refugia that many species depend upon to bridge fire disturbances will all be greatly affected. Further, soil erosion potential is often higher when large patches burn with high severity.

Fires are essential to ecosystem function. Largely missing from many western landscapes are the historically most numerous small- and medium-sized fires that burn under less-than-extreme conditions of weather and fuels. Even when they don’t burn much area individually, such fires cumulatively shape landscape heterogeneity, the resistance of the landscape to wildfire growth, the frequency of large fires, and landscape capacity to respond to future large fires. Simulation modeling in forested landscapes suggests that even relatively small changes in the proportion of large patches alters system behavior. Thus, the patch-size distribution of low-, moderate-, and high-severity fires is of prime concern to policy makers, scientists, and managers.

Climate, Fuels, and the Implications of Landscape Change

Because fuels, weather, and topography dictate fire behavior, fuels management is important to efforts to mitigate fire behavior. However, mechanical treatment of fuels alone is not enough. Thinning without follow-up prescribed burning will typically worsen the problem. More flexible and extensive management of wildfires and prescribed fires will be essential, depending on local objectives and conditions, to increase the footprint of land areas showing reduced surface and canopy fuel abundance and connectivity. More extensive use of prescribed burning combined with thinning will be helpful, where forest fuel conditions (both surface and canopy fuels) are not currently manageable via wildfires and prescribed fires alone. The influence of prior fires on the extent and severity of subsequent fires, even when those fires burn under extreme weather and fuel moisture conditions, is a reflection of the importance of fuels. In some areas, forest conditions are such that some manipulation of fuels is needed so that key ecosystem elements are not lost in extreme fires. Many respondents accept that a proactive approach to fire and fuels management on public lands will reduce overall costs and improve climate change adaptation in the long-term. Some respondents questioned the practicality and effectiveness of fuel treatments under a changing climate; however the literature is clear that fuel reductions reduce flame length and fireline intensity, which reduce the likelihood of high-severity crown fires. Sound, science-based monitoring needs to be coupled with adaptive management to provide locally appropriate stewardship of our forests.

Decades of research in landscape ecology show that emergent properties have central importance to ecosystems and their pattern and process regulation, whereas many recent studies of climate-driven fire and vegetation change are less focused on local-scale feedbacks and emergent patterns. This creates a fundamental problem in linking climate change and landscape ecology research. Climate models assume that top-down climate covariates drive temperature, precipitation, and solar radiation conditions. Landscape ecology research shows that those top-down inputs can be highly modified by meso- and fine-scale bottom-up environmental controls to produce emergent climatic conditions that are strictly speaking neither the top-down or bottom-up inputs, but are influenced by these inputs. Climatic forcing alone poorly explains the shifts in landscape patterns because lagged patterns of historical disturbances continue to influence emergent patterns, under all but the most extreme events. The path forward to more effective projection of future fire and landscape change includes better integration of feedbacks from landscape ecological models into climate-driven models of future fire and landscape change.

In many landscapes, the increased abundance and connectivity of forests and fuels is favoring larger fires, and larger patches burned with higher severity. This is widely shared common ground for ponderosa pine forests and in some ecoregions is also applicable to dry mixed-conifer forests. This view is also commonly applied to moist and cold forests in the Inland Northwest, Pacific Southwest, Northern Rockies, and the Inland Southwest, whereas much less increase in fuel connectivity is believed to have occurred in the cold forests of the Southern Rockies. Regardless of uncertainties about departures from historical landscape conditions, there is a coherent argument based on first principles of fire spread that increasing forest patch heterogeneity could foster resilience to future fires, even as the climate changes. Thus, encouraging heterogeneity at various scales and in various processes is important for biodiversity, reducing connectivity of woody fuels, and increasing resilience with future climate change.

Effective management will depend on both science and trust

Our understanding of historical fire regimes can inform decision-making; indeed, such evidence-based decision-making can build trust. However, fire science points to complex patterns that vary with local conditions, so no single solution, such as logging or limiting all logging, will accomplish desired objectives in all forests. Further, no intervention also has consequences, so all decisions need monitoring to support the assumptions of management. Effective monitoring can improve knowledge, and through collective learning can build common understanding and trust.

Fire management can become more proactive and strategic. Existing tools, such as mechanical fuel treatments, prescribed fire, prevention of accidentally-ignited human fires, and managing wildfires will all be useful, but adaptation and mitigation responses to climate change and changing fire activity will require using these tools in strategic ways to fit area-specific goals. Some past disagreements about fire and fuel management strategies may be due to lack of clarity about specific goals, such as resident and firefighter safety, cost reduction, biodiversity issues, and ecosystem resilience under a changing climate.

The timing of fires is important, particularly in the context of a changing climate. While recognizing that wildfire seasons are long and getting longer, we must also take advantage of the milder fire weather and associated effects of fires in the “shoulder seasons.” Managers may find that both less-aggressive fire suppression and expanded use of managed wildfire under relatively moderate weather conditions can aid them where reducing the vulnerability of people and natural resources to fires is the objective.

One of the grand challenges of fire management is balancing the reality that wildfires will occur and are needed by western forest ecosystems, yet people, property, and economies need protection from the adverse effects of fire. Another grand and fairly urgent challenge is discovering the tipping points of transformative change for various forest landscapes in their respective geographies, where large, high-severity fires (regardless of whether they are considered unprecedented or not) may tip forest ecosystems into persistent non-forest states by constraining tree regeneration opportunities. Particularly as climate changes, we also need a deeper understanding of which landscapes may not be able to sustain forests in the future and how fast such transitions are likely to occur. It is clear that our western history of substantial forest fire activity will continue, one way or another: many fires will occur in the future and some will be large. Ultimately, we must find ways to sustainably use and live with fires that are well-adapted to both ecosystem and societal needs of local landscapes.

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ECOSYSTEMS

Compounded Perturbations Yield Ecological Surprises

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Abstract

All species have evolved in the presence of disturbance, and thus are in a sense matched to the recurrence pattern of the perturbations. Consequently, disturbances within the typical range, even at the extreme of that range as defined by large, infrequent disturbances (LIDs), usually result in little long-term change to the system's fundamental character. We argue that more serious ecological consequences result from compounded perturbations within the normative recovery time of the community in question. We consider both physically based disturbance (for example, storm, volcanic eruption, and forest fire) and biologically based disturbance of populations, such as overharvesting, invasion, and disease, and their interactions. Dispersal capability and measures of generation time or age to first reproduction of the species of interest seem to be the important metrics for scaling the size and frequency of disturbances among different types of ecosystems. We develop six scenarios that describe communities that have been subjected to multiple perturbations, either simultaneously or at a rate faster than the rate of recovery, and appear to have entered new domains or "ecological surprises." In some cases, three or more disturbances seem to have been required to initiate the changed state. We argue that in a world of ever-more-pervasive anthropogenic impacts on natural communities coupled with the increasing certainty of global change, compounded perturbations and ecological surprises will become more common. Understanding these ecological synergisms will be basic to environmental management decisions of the 21st century.

Key words: altered community states; dispersal; multiple disturbances; recovery intervals; scaling disturbances.

Introduction

All natural assemblages are perturbed by both physical and biological forces. These agents of change occur with different intensities, frequencies, and spatial distributions. Some essentially scour the landscape, resetting the successional clock to time zero. More commonly, disturbances leave a residual assemblage that provides a legacy on which subsequent patterns build. We consider the range of single perturbations, from small-scale/frequent disturbances to large/infrequent catastrophes, to be central to much traditional ecology; such directional or cyclical changes stimulated the development of ecology's first paradigm, succession (Cowles 1899; Clements 1905, 1916). A century of accumulated detail on the interplay between pattern and process has provided descriptors for the nature of successional change and system-dependent rates of recovery. There are few surprises embedded here: depending on the time frame of interest, species arrive and depart, canopies or other structures develop, and the system "recovers," converging on the predisturbance state at a rate reflecting the spatial extent and intensity of the disruptive forces. Such patterns have been extensively reviewed (Pickett and White 1985), and variation in recovery dynamics can be attributed to different processes acting independently or in concert (Drury and Nisbet 1973; Con-

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nell and Slatyer 1977). Even large, infrequent disturbances (LIDs) do not appear to override the biotic mechanisms that structure eventual recovery. For example, the 1988 Yellowstone National Park fire, which burned 36% of the park and was an order of magnitude larger than comparable large, infrequent fires, has to date generated no ecological surprises: "the postfire ecosystems are shaping up to be essentially the same as those that prospered before the flames" (Stone 1998: 1527). We argue that cycles of disruption and recovery are the usual state of affairs and submit that rapidly compounded perturbations have more serious implications for long-term alterations of community state, occasionally or even often generating a different assemblage of species.

Physical agents of change are well documented and described by such terms as windstorm, landslide, forest fire, flood, hurricane, and volcanic eruption. Many of these are primarily of terrestrial importance and leave their signature on landscapes as sites with recognizable boundaries and measurable shapes and areas. Biologically based counterparts—clear-cutting of terrestrial forests and trawling on the ocean floor—generate similar map properties. Populations are also subject to biological disturbances that vary from slight to catastrophic. Although these may lack spatially discrete boundaries, their implications for community structure can be at least as profound. Here we combine, when appropriate, biological disturbances like pestilence, population eruptions, invasions, and overharvesting with the more traditional physical forms of disturbance. In so doing, we add an animal and therefore a trophic dimension to a subject traditionally dominated by plant ecologists.

Figure 1 is a heuristic portrayal of our approach. In the top panel, a single large disturbance is followed by eventual return to some baseline condition at which point the assemblage can be considered "recovered." The diverse literature on succession is primarily concerned with this pattern and its underlying mechanisms. The following panels identify our focus. In the middle panel, two large disturbances are shown to occur nearly simultaneously or in close progression. We believe that recovery, if possible, is often substantially delayed under such conditions, and we provide examples below. In the bottom panel, a major disturbance is superimposed on an assemblage already maintained in an altered state, usually by anthropogenic processes. Current examples could include populations depressed by persistent overfishing, whole systems altered by chronic pollutants, or the developing impacts of climate change, such as the apparent increases in frequency and intensity of major storms

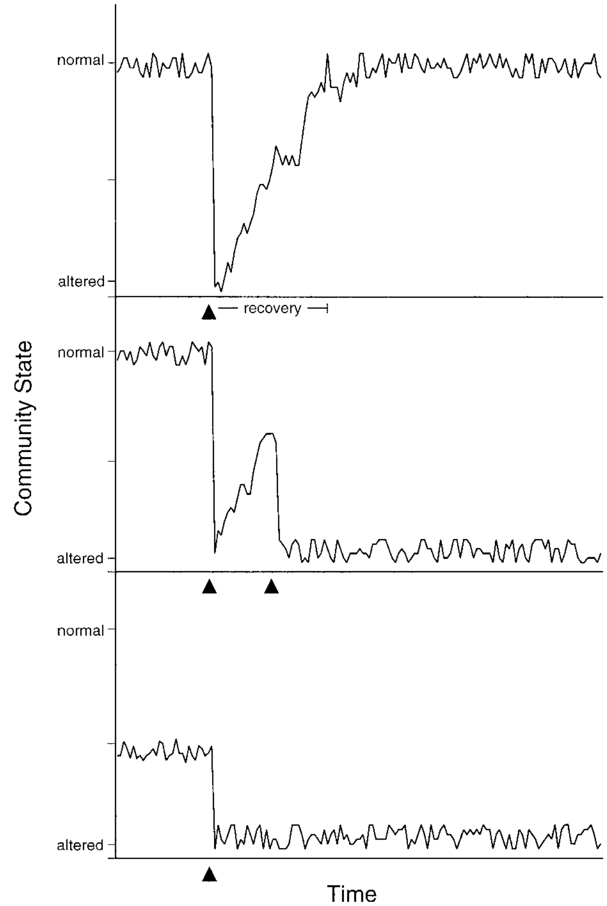

Figure 1. Schematic representation of the effects of large, infrequent disturbances (LIDs) on community state. Top, A normal community is subjected to a single LID and subsequently recovers. Middle, A normal community undergoes a second (or multiple) disturbance(s) before recovery from the first is completed; the combined effects lead to long-term alteration in community state. Bottom, A major disturbance is superimposed on an assemblage already already altered by anthropogenic processes or disease; again the combination of stresses leads to long-term alteration of comunity state. Arrowheads mark the disturbances.

and other climate extremes with increasing temperatures [for example, see Flavin (1996)]. Jansson and Velner (1995: 332), for example, suggest that in the Baltic Sea, a brackish body of water with minimal connection to the North Sea, which has been heavily impacted by eutrophication and toxic inputs, "pollution has reached the point where damage may be irreversible."

The scenarios discussed next include systems that appear, albeit temporarily, to have entered a new ecological domain; that is, they have not recovered. They share two features in common. First, all have been subjected to large (based on duration or spatial magnitude) and severe (quantified as a major mortality event) perturbations that may be physical or biological in origin. Second, these have occurred either simultaneously or in a sequence rapid enough that recovery from the single pulse has not significantly progressed. In some instances, three or more pulses appear to be necessary to initiate the changed state. Another way to describe our concern is to recognize that the community effect of compounded perturbations is multiplicative, not additive. If true and general, ecological surprises should be increasingly commonplace, prediction of recovery rates and trajectories less certain, and management more difficult.

Quantifying Disturbances and Their Frequencies

We adopt the definition of disturbance used by White and Pickett (1985: 7): "A disturbance is any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment." Pickett and White (1985) make the point that *disturbances* span three orders of magnitude of time (years) and ten of space (m^2). Disturbances, while causing spatially identifiable mortality to some species, usually provide open or invadable space for others, often renewing resources in the process. Sometimes, they can be identified by the resultant patchiness, recognizable by shape, size, postdisturbance internal composition, and spatial distribution. These patterns also have a dynamic, especially frequency of formation and rate of return toward the predisturbance state. Turner and colleagues (1997) suggest that it is also necessary to recognize the roles that individuals or species surviving a disturbance event can play in mitigating the event's impact. They suggest that only events characterized by few "residuals" be considered as large. We concur.

Disturbances can mean high mortality, often death of all individuals in the disturbance area. LIDs have different meanings for different ecosystems. For example, a large disturbance in the tidal zone may be on the order of tens of square meters, whereas in a forest it may be thousands. Hence, some scaling relationship must be used to ensure that these terms have equivalent meanings between systems. If, for example, we choose population dynamics as the processes of interest, disturbance size and frequency could be scaled by birth, death, or immigration (dispersal). Dispersal seems a particularly significant scaling metric for it governs the rate of recolonization of the disturbed site. Thus, ruderal (fugitive) species are typically both early invaders and excellent dispersers. Greene and Johnson (1989) applied a scaling metric involving seed terminal velocity, height of seed release, and mean horizontal wind speed to make dispersal comparable between species with different characteristics. For instance, ashfall accompanying the eruption of Mount St. Helens in 1980 greatly reduced many insect and spider populations. Because adult female leafhoppers (*Errhomus*) lack long-distance dispersal, they were slow to return to preeruption abundances in contrast with spiders (Showalter 1985). Similarly, on rocky marine shores, the poorly dispersing brown alga *Postelsia* can be driven locally extinct (Paine 1988 and unpublished), whereas local extinction is highly improbable for the associated barnacle *Balanus glandula,* whose larvae can traverse hundreds of kilometers. Thus, defining when a disturbance is large depends on the interplay between the spatial magnitude of the area disrupted and the dispersal (reinvasion) ability of the species of interest. Similarly, the frequency of disturbance could be scaled by some measures of generation time or age of first reproduction. In contrast, scaling on the basis of size of the dominant organisms does not appear to be a useful metric; for example, consider giant kelp (*Macrocystis pyrifera*) and terrestrial trees: sizes are comparable but time scales for age to first reproduction and life spans differ by orders of magnitude.

Ecological evidence seems to indicate that most LIDs do not override the biotic mechanisms governing species composition: in many disturbances, the postdisturbance composition is similar to the predisturbance composition (Turner and others 1997). This result might have been anticipated if size and frequency of disturbance had been scaled in terms of dispersal distances and generation times instead of the quantity of hectares and years as usually used.

Two other kinds of disturbances lack spatially explicit features but can have equally significant consequences: (a) Populations can be thinned commercially or reduced to mere vestiges of their original abundances by disease. For example, the majority of commercial fish stocks in US and Canadian coastal waters have been overexploited or are currently at maximum sustainable yield levels (NOAA 1993). Rinderpest decimated herds of African ungulates, especially buffalo (Sinclair 1977). The community changes resulting from density shifts of dominating species, often of high trophic status, can be extensive. They are a biologically based disturbance and often leave no immediate spatial signature. (b) Global climate change, a more cosmic form of disturbance, will surely have a substantial though currently unquantified and debated impact. It will provide a background of change in which both physical (for example, fire) and biological (for instance, harvesting and disease) disruptive forces will operate, and it might become the dominant influence on community structure and change in the coming decades despite its subtlety.

When do changes in community composition occur or under what conditions can they be anticipated? We believe that such dramatic shifts are most likely when both the spatial extent and especially frequency of disturbance are at the extremes of normal expectations. Multiple, usually sequential occurrences of these extreme and rare events can produce alternative stable states, that is, abnormal conditions or ecological surprises that defy the norm. The following scenarios describe communities that have been subjected to multiple perturbations and appear to have entered (or be facing) new domains.

Compounded Perturbations in Ecological Time

El Niños, Storms, and Kelp Bed Recovery

El Niño–Southern Oscillation (ENSO) events are large-scale oscillations of the tropical Pacific Ocean–atmosphere system with far-reaching climatic and economic impacts. The 1982–83 El Niño was widely considered the strongest of the century and had a corresponding impact on forests of giant kelp along the coasts of Alta and Baja California. Winter 1982–83 was the most severe storm season in many decades, as atmospheric teleconnections linked to the warming of the eastern equatorial Pacific Ocean affected the Aleutian low-pressure center, generating a large number of severe storms from an unusual southerly direction. These storms devastated kelp forests throughout the range of *Macrocystis pyrifera,* an economically valuable kelp. Anomalous poleward flow of warm, oligotrophic waters that rendered upwelling ineffective led to nutrient depletion, massive loss of kelp biomass, and extensive summer mortality in the southern half of *M. pyrifera*'s range. The severity of the warm-water effects was related to latitude. In South America, anomalies were as high as 11°C during 1982–83; there was mass mortality of *M. integrifolia* and associated animal populations in Peru and northern Chile (Dayton and Tegner 1990). In the southernmost part of the range in Baja California, giant kelp went extinct in some areas and site preemption by lower standing kelps prevented its recovery after the ENSO.

In the San Diego region, the combined effects of the storms (ENSOs can be storm free) and the 4°–5°C warm-water anomalies constituted the most severe disturbance of a giant kelp forest community ever documented, yet recovery was rapid once conditions returned to normal (Tegner and Dayton 1987; Dayton and others 1992). At the other end of the range in central California, the intensity and duration of the warm event were smaller and conditions remained within the suitable range for kelp [reviewed by Tegner and Dayton (1987), Dayton and Tegner (1990), and Dayton and others (1992)]. Kelps are well adapted to winter-storm disturbance, with correspondingly timed reproduction, spore dispersal tied to water movement, and success of the propagules a function of open space. The more problematic warm-water effects result from the severity and duration of the events and probably the frequency, as well.

ENSOs are natural climate variations to which communities have been subjected for at least hundreds of years, but there are major questions about whether the intensity and/or frequency of these events may change as a result of global warming (Trenberth 1995). The observational record indicates that ENSO events have changed in frequency and intensity in the past century, but the frequency of strong events appears unchanged back to 1625 (Enfield 1988). Coupled ocean–atmosphere general circulation models find that ENSOs will continue to exist in a warmer world, but have yet to address frequency and intensity. Trenberth (1995) suggests that because these events have the effect of creating droughts and floods in different parts of the world and global warming tends to enhance the hydrological cycle, there is a real prospect that future ENSOs will be accompanied by more severe droughts and floods. For Northern Hemisphere kelp forests, the question may be whether ENSO thermal additions to already warmer water conditions [for example, see Roemmich and McGowan (1995)] in the future can push kelps beyond the range of recovery, especially in the southern end of the range.

Climatic Extremes and Exotic Species in San Francisco Bay

San Francisco Bay, at the mouth of rivers draining 40% of California, is considered to be the major estuary in the United States most modified by human activity (Nichols and others 1986). Many alterations of the ecology and the bay, such as loss of habitats, water-quality changes, introduced species, and excessive freshwater diversion, date to the 19th century, yet recent disturbances have led to profound changes. In late 1986, the euryhaline Asian bivalve mollusc *Potamocorbula amurensis* was first sampled in San Francisco Bay (Carlton and others

1990; Nichols and others 1990). Within 2 years, it had spread throughout the estuary, on all sediment types and water depths, and reached densities at some sites exceeding 10,000 m^{-2}. This invasion almost certainly resulted from the discharge of seawater ballast from cargo vessels.

Two years of climatic extremes apparently contributed to this remarkable population explosion (Nichols and others 1990). Before *P. amurensis* was discovered, the benthic community in Suisun Bay (northern region of the bay) varied predictably with river inflow: years of normal or high flow were characterized by brackish or freshwater species, and years of low flow by estuarine species. The end of the 1984–85 dry event was marked by an extreme but short-lived flood that eliminated the estuarine species. Thus, when *P. amurensis* was introduced, the Suisun Bay region was inhabited by a disturbed and depauperate community that may have contributed to the initial success of the invader. The invader's timing after the flood guaranteed it months to exploit the available space before the dry-period species would return, and by 1988 the near absence of the dry-period community demonstrated how well *P. amurensis* had displaced the former community. The ability of the invader to live in low-salinity water suggests that it will not be displaced with the return of normal river flow and that the benthic community is permanently altered (Nichols and others 1990).

Carlton and colleagues (1990) predicted significant community changes as this abundant consumer, competitor, disturber, and prey altered the interactive trophic webs in San Francisco Bay; these are beginning to unfold. Within a year, chlorophyll concentration and adult abundance of three common copepod species had declined by 53%–95%; these values persisted through 5 years of study (Kimmerer and others 1994). Before 1987, chlorophyll concentration varied with river flow; after mid-1987, it remained low despite variations in flow. The effect on copepods appears to be via direct clam predation on nauplii; egg production was not affected. Estimates of clam clearance rates are consistent with the reduction in copepod abundance. Although it may be premature to forecast permanent changes in the zooplankton populations, Kimmerer and colleagues (1994) voice serious concern: several species of fish that pass their larval lives in the upper estuary are also in serious decline. Moyle and coworkers (1992) list five species, including those in valuable sport and commercial fisheries, in which poor first-year classes correlate with reduced freshwater outflow, presumably because of decreased survival of larvae and juveniles. A sixth, the delta smelt, which is federally listed as threatened, feeds primarily on copepods, has a narrowly defined habitat in the mixing zone between fresh and salt waters, is an annual species very sensitive to environmental fluctuations, and has declined in concert with increasing water diversions since 1984 (Moyle and others 1992). Thus, the compounded effects of two major disturbances—one biological (the successful establishment of a nonindigenous species) and the other physical (drought followed by an extreme flood)—have initiated sweeping and probably permanent changes in ecosystem structure.

Boreal Forest Wildfires, Forest Fragmentation, and Logging

Fire frequency in the boreal forest is primarily controlled by large-scale climate processes, specifically persistent midtropospheric anomalies that at the surface are expressed as blocking high pressures (Schroeder and others 1964; Newark 1975; Street and Birch 1986; Flannigan and Harrington 1988; Johnson and Wowchuk 1993). In this century, however, agricultural settlement in the southern fringe of the boreal forest and logging further north have resulted in a new multiple-disturbance regime that has caused significant changes in forest composition.

In the last three centuries, fire frequency in the boreal forest has changed several times, each time associated with large-scale climate changes (Johnson 1992). The changes in the early 1700s and the 1800s were a result of the Little Ice Age (Grove 1988). In the boreal forest, this period had more frequent fires than the periods before or since (Bergeron and Achambault 1993). These changes in fire frequency appear to be associated with increased numbers and length of persistent midtropospheric anomalies. Years with large areas burned are known to have more sequences of days during the fire season with warmer, drier weather, which are associated with upper-level ridges. These persistent upper-level ridges over the boreal forest are *teleconnected* (spatially and temporally correlated) to upper-level troughs in the North Pacific and eastern North America. This teleconnection is called the Pacific North America pattern (Rogers 1981; Wallace and Gutzler 1981; Knox and Lawford 1990; Johnson and Wowchuk 1993). Similar patterns have been described in the southwestern United States as a result of ENSOs (Swetnam and Betancourt 1990).

The transition periods between different fire frequencies, for example, at the end of the Little Ice Age, seem to have been periods in which fires occurred more erratically for a decade or more. Often these periods were marked by clusters of years with very large areas burned and many persistent upper-level ridges. One could speculate that the large areas burned in the boreal forest since the 1980s are a result of one of these transition periods, perhaps related to global warming. However, more understanding of these transient processes is required before anything definitive can be said.

These climate-driven changes in fire frequency have generally been part of the ecosystem dynamics of the boreal forest for millennia. In this century, two new classes of large-scale disturbances were added to the climate-driven fire frequency. These anthropogenic forces were agricultural settlement in the southern boreal forest and logging.

Homesteading in the early 1900s led to progressive clearance of forest in the southern fringe of the boreal forest in western Canada (Vanderhill 1958). The effect on the forest north of the fringe was to increase the frequency of fires above that of the natural lightning fire regime. The increase was due to escaped clearance fires spreading from the settlement areas north up to 50–60 km into the unsettled forest. The result of this major increase in fires meant that trees that required longer time to reach sexual maturity, did not have serotinous cones, had little or no vegetative reproduction potential, were greatly reduced in abundance, and in many areas were locally exterminated (Weir 1996). For example, white spruce (*Picea glauca*) and balsam fir (*Abies balsamea*) both became relatively unimportant trees in many forests and trembling aspen (*Populus tremuloides*) significantly increased in abundance. Also, change from a mixed-wood (conifer–deciduous) to primarily deciduous forest has caused many other changes in plant and animal species (Weir 1996). At the same time that this forest was being subjected to an increase in the frequency of fire, high-grade logging (cutting of only large trees) of the white spruce was further reducing this dominant boreal species.

The southern boreal forest today has a significantly different composition than it did a century ago. This change has been due to multiple perturbations: a natural lightning fire regime augmented by settlement fires spreading from adjacent areas and logging.

Early Succession and Exotic Species

Volcanic eruptions clearly embrace the concept of disturbance, either by presenting new landscapes and initiating primary successional processes, or by altering preexisting ones via ashfalls, pyroclastic scorching, and the like. The end result of the recovery/regeneration process seems fairly predictable: Turner and colleagues (1997) compare the assembly of the plant community on Mount St. Helens (WA) following its 1980 eruption with other single large infrequent disasters. On this barren landscape, some degree of successional uncertainty may well characterize the early stages of the recovery process. Morris and Wood (1989) found in experiments on lupine, a nitrogen-fixing pioneer species, evidence that two other invaders could be either facilitated or inhibited. Such stage dependency complicates the successional process; it probably does not alter the ultimate community composition, although insufficient time has elapsed since the eruption to evaluate the consequences of these initial uncertainties.

The Hawaiian Islands are also of volcanic origin and, despite being earth's most isolated archipelago, have been invaded by 4600 exotic plants, 86 of which represent serious threats to the native ecosystem (Vitousek 1990). The successful invasion of a nitrogen-fixing exotic (*Myrica faya*) on the slopes of Hawaii Volcanos National Park provides a classic example. A 1959 eruption deposited 1–2 m of ash on the native vegetation, thinning it substantially. *Myrica* invaded and initiated a series of changes including the identity of the dominant tree, nutrient cycling, and productivity. For instance, Woodward and coworkers (1990) showed that native Hawaiian birds, while visiting *Myrica,* rarely feed on its fruit. Nonnative species visited, fed, and effectively dispersed viable seeds. *Myrica* itself is fecund: Vitousek and Walker (1989) estimate the seed rain at 4.6 million/ha under 21 adult *Myrica*/ha. In addition, the mean adult growth of these invaders is approximately 15 times that of a native tree (Vitousek and Walker 1989). *Myrica* is quadrupling the input of soil nitrogen (Vitousek 1990); earthworms are 2–8 times as dense under it than under native vegetation, which will change litter-processing dynamics and the rate of accumulation of soil organic matter (Aplet 1990).

As Vitousek (1990; Vitousek and Walker 1989) has demonstrated, the changes in whole ecosystem-level function are substantial. One major speculation is that when, or if, *Myrica* is replaced during primary succession, it will be replaced by another exotic. The competitively aggressive strawberry guava is a likely candidate species. Because, in general, sites with more fertile soils—higher concentrations of soil nitrogen in a system where N is a limiting resource—are conducive to invasion, "nitrogen fixation by *Myrica* will ultimately favor invasion by a broader range of exotic species" (Vitousek and Walker 1989: 261). Finally, as these authors note, invasion changes the ground rules governing coexistence of native assemblages recovering from or responding to disturbance. The problem is of great pragmatic importance to conservation biology: it further signals the existence of surprises at ecosystem levels when disturbances, in this case volcanism and biological invasion, are compounded.

Hypoxia in the Northern Gulf of Mexico

Oxygen depletion, long known from confined bodies of water, such as basins, fjords, bays, and estuaries, is increasingly reported from coastal ocean environments (Boesch and Rabalais 1991; Diaz and Rosenberg 1995). This may take the form of anoxia, where dissolved oxygen concentrations are essentially zero and hydrogen sulfide (toxic to metazoans) is detectable, or more commonly hypoxia, where oxygen concentrations on the sea floor are reduced to levels low enough to cause severe stress and mass mortality of benthic and water-column fauna. Varying with the severity of the oxygen depletion, effects on the biota range from avoidance of the affected area, to mortality of more sensitive taxa such as crustaceans and echinoderms, to emergence of the redox potential discontinuity from the sediments, a condition where only chemoautotrophic bacteria can live. Diaz and Rosenberg (1995) report that no other variables of such ecological importance to coastal marine ecosystems have changed so drastically in such a short period of time. Increasing evidence of oxygen depletion in coastal ecosystems is associated with anthropogenic eutrophication and, when eutrophication is coupled with adverse meteorological and/or hydrodynamic events, hypoxic events increase in frequency and intensity.

The inner and middle continental shelf of the northern Gulf of Mexico from the Mississippi River Delta to Texas is the largest and most severely affected area in North America subject to seasonal hypoxia [operationally defined by dissolved oxygen levels less than 2 mg/L or less than 1.4 ml/L (Rabalais and others 1991)]. From 1985 to 1988, hypoxic waters were found from April to October, from 5 to 60 m water depth, from 5 to 60 km offshore, extended up to 20 m above the bottom, and covered up to 9500 km^2. Hypoxia in this region is coincident with strong, salinity-controlled stratification during the warmer months of the year, which restricts reoxygenation of the bottom waters. Large quantities of decomposing phytoplankton biomass fueling intense water-column and benthic respiration rates enhance the effects of stratification on oxygen depletion. Although hypoxia did not cause extensive mortalities on the northern Gulf of Mexico shelf until 1978, it has occurred almost annually since it was first discovered in 1973 (Diaz and Rosenberg 1995). Severity and extent vary interannually with river flow, shelf circulation, and tropical storm mixing (Rabalais and others 1991).

The importance of the extent and duration of the hypoxia on the Louisiana shelf relates to fishery landings from this state, which constitute 28% of the US total (Rabalais and others 1991). Abundances of finfish, shrimp, and swimming crabs are severely depressed in hypoxic areas, and the period of oxygen stress includes critical life-history periods of several commercially important species. The macrofauna either succumb or move to avoid stressful conditions; typically, demersal species have been observed high off the bottom where mortality due to predation is undoubtedly high (Boesch and Rabalais 1991). When hypoxia persists, only highly resistant taxa such as some polychaetes and nematodes survive. Posthypoxia community dynamics depend on the extent of the mortalities, age of affected populations, timing with respect to availability of recruits, size of the affected area relative to short-dispersal recruits, frequency of hypoxic stress, and degree of organic carbon buildup. Because of the recurrent nature of hypoxia in the northern Gulf of Mexico, there are few large or long-lived sedentary species, and the benthic community is dominated by opportunistic species characteristic of an early successional state (Boesch and Rabalais 1991; Diaz and Rosenberg 1995). Intensified commercial fishing on the continental shelf in the 1970s and 1980s has been accompanied by alarming declines in the estimated sizes of remaining fish stocks; although overexploitation is clearly important, Darnell (1992) suggests that habitat deterioration is affecting both nursery areas and food chains for commercial species.

The Mississippi River and its distributaries drain 40% of the coterminous United States. Nitrogen concentrations in the rivers, the major source of "new" nutrients to the offshore phytoplankton, have more than doubled since the mid-1950s (Rabalais and others 1991). Turner and Rabalais (1994) recently demonstrated a close coupling between riverine loading and phytoplankton production; changes in biologically bound silica in the sediments below the river plume were virtually coincident with increases in nitrogen loading. After major flooding in 1993, the Gulf of Mexico hypoxic zone doubled to 18,000 km^2 and has not shrunk much since (Kaiser 1996). On 12–13 August 1996, winds forced oxygen-depleted water from the offshore dead zone below the mouth of the Mississippi River close to shore. This caused a "jubilee" along 36 km of Louisiana coastline, a condition where shrimps, crab, and finfish crowd close to shore to escape the oxygen-deficient water—highly increasing their susceptibility to fishing (Buck 1996). Kaiser (1996) reports that the US federal government may be finally waking up to warnings that the Gulf of Mexico hypoxic or "dead zone" may be one of the nation's worst ecological problems. A multiagency committee has been convened to discuss the problem and recommend management steps, such as voluntary reductions in fertilizer use in the Midwest. Again, compounded perturbations—coastal eutrophication exacerbated by extreme meteorological events—have produced an altered community state.

Phase Shifts in Jamaican Coral Reefs

Like all communities, coral reefs are subject to occasional intense natural disturbances: plagues of the starfish *Acanthaster* (Birkeland 1982), hurricane devastation (Woodly and others 1981), and high temperature stress (Gates 1990). Long-term studies of reef structure along a depth gradient at Discovery Bay, Jamaica, provide a clear and sobering view of phase shifts in assemblage structure associated with a compounded series of severe disturbances.

Baseline data at the main site exist from the 1950s (Goreau 1959), and the site has been repeatedly examined quantitatively since then (Andres and Witman 1995). A sequence of events initiated in the early 1980s, but set against a subtler background of increasing coastal pollution and extreme harvesting pressures on herbivorous fishes, has led to what Hughes (1994) calls "large scale degradation" and vividly portrays as a phase shift in community structure. Depending on the depth at which corals are sampled, percent cover has decreased from 30%–60% in 1977 (Huston 1985) to approximately 5% at depths less than 30 m in 1992 (Andres and Witman 1995). Conversely, algal cover, accounting for less than 5% cover in 1982, comprised approximately 70% cover in 1992. As a result of algal preemption of space, larval recruitment of all corals failed after 1984 (Hughes 1994).

The compounded disturbances—two major hurricanes (Allen in 1980 and Gilbert in 1988) and mass mortality of an ecologically significant grazer, the sea urchin *Diadema,* from 1982 to 1984—occurred well within the normal recovery interval of reefs. Hughes (1994) suggests that reef regeneration was initiated shortly after Hurricane Allen despite reduced grazing fish populations. The *Diadema* dieback delivered the coup de grâce and the recovery trend was reversed. Andres and Witman (1995) suggest that Hurricane Gilbert only slowed the developing domination of algae and therefore failed to enhance coral recovery. Furthermore, these authors imply that if urchin and fish populations had retained their pre-Allen levels, coral recovery and domination of the benthos would have occurred. With continued depression of herbivore populations, recovery of the coral assemblage is not forseeable.

In this case, a variety of disturbances, occurring rapidly relative to coral regeneration capacity, have yielded a novel community state—one that would not have developed if the disturbance events had occurred individually at intervals appropriate for reef recovery.

Discussion

A world of ever-more-pervasive anthropogenic impacts on natural communities coupled with the increasing certainty of climatic response to human activities suggests that compounded perturbations and ecological surprises will become more common. The frequency of disturbances is often scaled in terms of severity and return intervals: for example, the 30-year flood or the 100-year storm. The large number of record weather events in the news in recent years raises the issue of climate change and its interactions with LIDs. We reiterate our belief, and concern, that ecologists must refocus their interest from the ordinary (for instance, recovery sequences between normally spaced disturbances) to the extraordinary (for example, rapid sequential disturbances occurring against a background of increasing climate change).

We do not question the capacity for single LIDs to distort ecosystem structure and functioning temporarily. They obviously do, and merit investigation in their own right. On the other hand, the ecological literature is replete with underappreciated studies of compounded disturbances. Here we identify two more to suggest their ubiquity. Zedler and colleagues (1983) describe the interaction of fire and an annual grass planted to control erosion. Their conclusion that (p. 817) "We doubt if traditional climax-oriented successional theories can be of much use in predicting the outcome" resonates with our theme. In addition, several of the case histories developed in Gunderson and coworkers (1995) suggest that maladaptive management actually decreases ecosystem resilience, thus increasing susceptibility to subsequent disturbances, perhaps even facilitating disturbances and thwarting effective management.

The 1995 Intergovernmental Panel on Climate Change report (IPCC) (Houghton and others 1996), the international consensus summary of climate science, reviews the low probability that observed changes could be caused by natural factors alone, the average rate of warming for the 21st century that is likely to be warmer than any seen in the last 10,000 years, and projections for sea-level rise. Globally, existing data do not offer statistical evidence that extreme weather events or climate variability have increased in the 20th century, although there is clear evidence on regional scales. The IPCC review of model results finds agreement on predictions of general warming with regional variability and an enhanced global mean hydrological cycle. Regional forecasts are more problematic, but the incidence of floods, droughts, fires, and heatwaves is expected to increase in some areas and decrease in others as temperatures rise (Houghton and others 1996).

Our scenarios were chosen to identify the range of possibilities and the resulting ecological surprises. We have made no attempt to be encyclopedic. Rather, our list is characterized by events with blurred or no discernible borders, except as dictated by local geography (San Francisco Bay, for example) or study at a specific location (for example, Jamaica). Missing entirely are those great disturbances with sharp spatial boundaries as often characterize tornado paths or intertidal mussel-bed destruction. Missing also are references to the human condition, for instance, the interplay between malnutrition and disease. The growing evidence for unusual meteorological events leading to outbreaks of disease infectious among humans, such as hantavirus, cholera, and plague (Linden 1996), provides numerous examples. In one well-studied case, Colwell (1996) relates the massive cholera pandemic that struck South America in 1991 to the transport of zooplankton in a freighter's ballast water to the coast of Peru at the onset of an El Niño event. The El Niño brought rain and the influx of nutrients from land as well as warm sea surface temperatures, factors associated with the initiation of plankton blooms. This El Niño event, which lasted from 1990 to June 1995 and is the longest on record since 1882, may also be associated with the cholera bacterium remaining endemic in the region (Colwell 1996).

Present and future global climate-change effects on the environment may be debatable, but this is not the case for increasing human impacts: population growth, urbanization, deforestation, eutrophication, overfishing, loss of habitat, and so on. In ironic contrast with many political leaders, the insurance industry has recognized the significance of the nexus of these issues. Faced with rapidly escalating insured costs for increasing numbers of severe catastrophes over the last decade, that industry is realizing that it must demand political action to protect the climate to prevent its own financial ruin (Flavin 1996; Munich Re 1996). Our perspective is that societies and ecologists must begin to prepare themselves for novel and unanticipated consequences of previously well-understood phenomena. Jamaican coral reefs may not recover, Hawaiian volcanic slopes may develop a novel flora, and San Francisco Bay appears to be acquiring a new and not necessarily desirable invertebrate assemblage, in the process losing native species. Global warming may accelerate the effects of oxygen deficiency and enlarge affected areas in the Gulf of Mexico. These altered and possibly alternative states (Lewontin 1969; Sutherland 1974) may or may not be persistent (stable). On the other hand, mounting evidence suggests that sequential, large-scale disruption of the current state will make these altered states the ecological reality of the future. Understanding the role of compounded disturbances, some natural and others of anthropogenic origin, will be basic to environmental management decisions in the 21st century.

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What's Eating the Pando Clone?

Two Weeks of Cattle Grazing Decimates the Understory of Pando and Adjacent Aspen Groves

by Jonathan B. Ratner, Erik M. Molvar, Tristan K. Meek, and John G. Carter

Western Watersheds Project

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What's Eating the Pando Clone?

Two Weeks of Cattle Grazing Decimates the Understory of Pando and Adjacent Aspen Groves

by Jonathan B. Ratner,[1] Erik M. Molvar,[1] Tristan K. Meek,[1] and John G. Carter[2]

EXECUTIVE SUMMARY

The Pando Clone is an aspen grove on the Fishlake National Forest in south-central Utah that was heralded in 1992 as the world's largest single living organism. Adult trees that are joined by a single rootstock and share identical chromosomes comprise the Pando Clone and, like many aspen groves across the West, it has suffered for a number of years from die-back and failure to regenerate new shoots to replace the aging adult trees for a number of years.

The U.S. Forest Service created fenced exclosures to protect a portion of the Pando Clone from herbivory (browsing - the consumption of woody growth - by mule deer and domestic cattle), and initiated some small-scale vegetation treatments. Aspen regeneration has responded inside the exclosures in both treated and untreated areas, while outside the exclosures, on public lands leased for livestock grazing, regeneration failure and die-backs continue to plague the Pando Clone as well as other aspen groves subjected to the same pattern of livestock and mule deer hebivory, and die-back continues.

Western Watersheds Project initiated a one-year monitoring project in order to quantify ungulate use in the area, using stationary motion-sensing cameras to quantify by species the use of the area and document levels of herbivory by both domestic cattle and mule deer over the 2018 growing season in the unfenced portions of the Pando Clone and in adjacent aspen groves. At our monitoring sites, we documented 4.5 times the amount of cattle use herbivory in two weeks than the mule deer use over six months. Forage utilization by mule deer prior to the onset of livestock grazing was unobservable, while forage utilization by livestock (plus mule deer) during the 2 weeks of cattle grazing consumed 70 to 90 percent of the understory vegetation's annual production.

Cattle have a greater propensity to consume aspen sprouts in autumn, when the nutritional quality of other understory vegetation declines, and the virtual elimination of understory vegetation by this high intensity livestock use may also cause mule deer to switch to aspen shoots, further amplifying the impacts. Our results show that the brief but intense cattle grazing appears to be a major contributor to the decline of the Pando Clone, as well as other aspen groves in the immediate vicinity, in addition to the much lighter continuous herbivory by mule deer. Based on comparisons of the exclosures with the area open to both livestock and mule deer that this high level of use in the unfenced areas effectively eliminates regeneration. A previous study (Rogers and McAvoy 2018) attributed the failure of the Pando Clone to regenerate solely to mule deer, but our results indicate that cattle are also having a major impact on understory vegetation. Our results suggest that livestock herbivory may be having a synergistic interaction with mule deer foraging to suppress aspen sprout growth, and that trampling of soils by livestock may also play a role in depressed aspen recruitment in unfenced portions of the Pando Clone and adjacent aspen stands.

Based on our results, we recommend removal of livestock from the Pando Clone area to protect this globally significant organism, and also recommend that livestock be removed from public land pastures elsewhere where aspen groves show signs of depressed regeneration.

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Introduction

Aspen (*Populus tremuloides*) stands are found across the interior West from southern Arizona to the Canadian Rockies, and typically occur in montane or sage steppe environments, often in association with abundant soil moisture. Aspen groves range along a spectrum from fire-dependent transitional communities that regenerate through periodic fires to stable communities that do not require fire for persistence (Shinneman et al. 2013). Reproduction via seeds happens most commonly in conjunction with severe disturbance such as fire (Long and Mock 2012). More frequently, aspens reproduce by sending up new shoots, or "suckers," from the existing rootstock, and the resulting aspen grove may persist for thousands of years (Jones and DeByle 1985a).

Aspen groves are frequently clones, where a single root system sends up hundreds or even thousands of individual stems (Barnes 1975), with each "tree" being a genetically identical surface expression of one large organism connected through its common root network. Gardner (2013) found that areas with high clonal diversity in aspens occurred in areas with a more frequent fire history, while areas with low aspen clonal diversity, often larger clones, corresponded to areas with less frequent fires. Clones may be long-lived; Kay (2003) hypothesized that aspen clones in north-central Nevada have maintained their presence for thousands of years via vegetative regeneration. As aspen trees age (i.e., exceed 100 years of age), they generally produce relatively few suckers (O'Brien et al. 2010).

For the purpose of clarity, it is useful to define some terms that will be used throughout this report. Aspens growing from a common rootstock are called *ramets*, a term that encompasses fully-grown adult ramets ("trees") as well as immature, regenerating trees rising as *adventitious shoots* ("shoots" or "suckers" in this report). Both new shoots with terminal buds and branches that have lateral buds can be referred to as "stems." The term "seedling" is used in this report exclusively to refer to young aspens growing from a seed, and excludes young aspens growing from adventitious buds on an existing rootstock. Aspen reproduction can be sexual ("seeding") or asexual ("suckering") from buds on the root system. Aspen recruitment occurs when young plants grow above the upper browse level of large herbivores. Clusters of aspen are referred to as "groves." Where such clusters are comprised of genetically identical trees joined by a common root system they are called "clones" and represent a single organism with many adult trees, sometimes thousands. "Regeneration" occurs when the recruitment of aspen suckers replaces the die-off of adult trees.

Aspens commonly grow where soil moisture is relatively abundant. However, in forested areas, sites containing aspens may be wetter simply because they transpire less water into the atmosphere than do conifers (Jones and DeByle 1985b). Aspen groves often contribute more water to drainage systems than do coniferous trees because they transpire only during the part of the year when they have leaves (versus year-round transpiration for conifers) and collect more snow in the understory than do conifers (DeByle 1985c). Aspens also have chlorophyll in their stems, and can photosynthesize throughout winter when leaves are absent (Grant and Mitton 2010). Presumably, water loss is minimal during winter when leaves are absent.

Aspen groves are hotspots of biodiversity and a number of bird species appear to be dependent on aspen habitats. Aspen groves harbored the greatest number of native species (45) of any habitat type in Grand Staircase – Escalante National Monument (Bashkin et al. 2003). Red-naped sapsucker, black-capped chickadee, house wren, warbling vireo, and northern saw-whet owl are closely associated with aspen woodlands (Hejl et al.,1996). Loose and Anderson (1995) found that 30 of 33 woodpecker nests in their Sierra Madre study area were found in aspens, and among these, there was a significant preference for large, old trees. According to Winternitz and Cahn (1983), 40% of species that inhabit aspen are cavity nesters, with a significant preference for large trees over 100 years old and trees infected by heartrot fungus. Heartrot-infected aspens are easier for birds and mammals to hollow out to create cavity nests. Aspens also are of critical importance as a food source for beavers (Williges 1946).

Jones and DeByle (1985c) compiled a thorough analysis of the role of fire in aspen ecology. According to these biologists, almost all even-aged aspen stands in the western U.S. appear to be the result of severe fire, presumably in coniferous forests. In Yellowstone National Park, Romme and Knight (1982) found that fire suppression has led to denser coniferous forests, a decrease in aspen, and an increase in sagebrush in meadow areas. Forest fires can foster aspen regeneration because fallen timber provides refugia for aspen seedlings to escape browsing by ungulates (Ripple and Larsen, 2001).

Strong aspen regeneration typically occurs even after severe burns. An abundance of aspen woodlands in the coniferous forest zone often indicates the prevalence of past stand-replacement fires. But Fornwalt and Smith (2000) noted that old, multi-storied aspen stands can maintain their productivity over time and are in many cases self-perpetuating. Thus, previous assumptions that aspen stands require periodic disturbance to maintain themselves are not universally true, and some stands (particularly old, multi-story stands) perpetuate themselves in the absence of any management treatment.

Although aspen habitats are viewed as valuable grazing resources by the livestock industry, these areas are very sensitive to overgrazing. Meuggler (1985b) reported that heavy grazing by domestic sheep can turn the rich and diverse herbaceous understory that occurs in ungrazed stands into a depauperate understory of grasses. In aspen stands that are overgrazed, invading, unpalatable plants can form a stable grazing disclimax (an unnatural, disturbed plant association that can persist indefinitely), reducing the wildlife habitat value of the grove (Mueggler 1985a). In addition, in older stands, heavy livestock grazing can prevent regeneration and speed the decline of the aspen stand itself (DeByle 1985a). Cole (1993) found that aspen-forb communities are highly susceptible to trampling damage even from human foot traffic. The physical trampling of nests and habitat degradation associated with livestock grazing can be detrimental to ground-nesting birds that prefer aspen habitats, such as the hermit thrush, junco, white-crowned and Lincoln's sparrows, veery, ovenbird, and nighthawk (DeByle 1985b). Because livestock grazing is currently permitted on more than 232 million acres of federal public land in the western United States (Beschta et al. 2013), the potential for ecological damage from livestock grazing is widespread.

The Pando Clone

The name "Pando" comes from the Latin "to spread." Kemperman and Barnes (1976) originally proposed the Pando Clone as a single living thing covering approximately 108 acres in area and made up of approximately 47,000 *ramets*, or stems. Grant et al. (1992) concluded that the Pando Clone represents the largest single organism in the world, with an areal extent of approximately 106 acres (43 ha) and an estimated weight of more than 6,600 tons (6 million kg). The Pando Clone was confirmed through genetic testing to be a single massive organism by DeWoody et al. (2008). Some of the trees in the Pando clone show triploid chromosome patterns (in effect, possessing a third set of chromosomes), and these individuals have a competitive advantage over diploid stems in terms of height and diameter growth (DeRose et al. 2015). This gives these stems an advantage in the 'self-thinning' stage of stand development, when only the most fit stems survive to attain tree form.

The age of the Pando Clone is a matter of substantial scientific debate. Kemperman and Barnes (1976) hypothesized that the Pando Clone was originally established more than 8,000 to 10,000 years before present. Aspen clones in this southern, unglaciated portion of the species' range, including Utah, can be unusually large and of much greater age

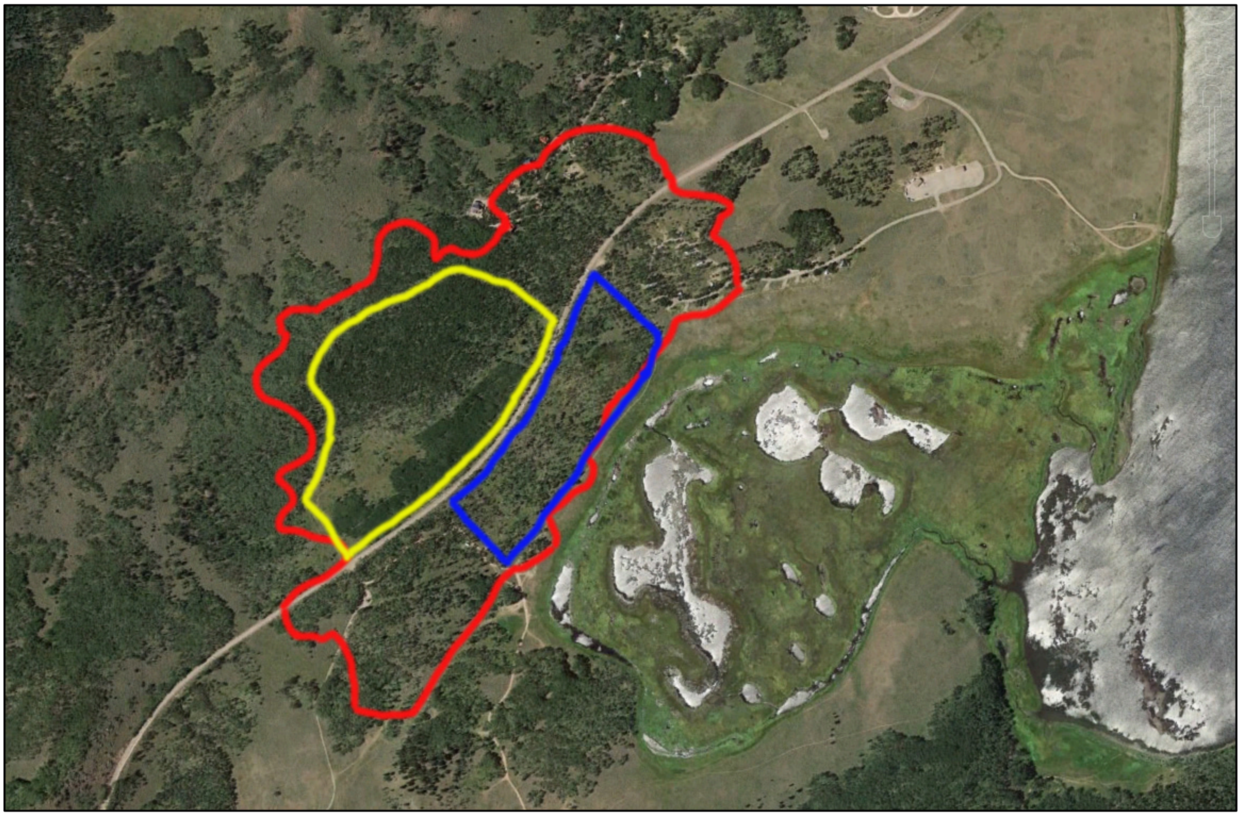

Figure 1. The boundary of the Pando Clone, in red (after Kemperman and Barnes 1976), showing the 2013 exclosure fence (in blue) which more successfully excludes mule deer, and the 2014 exclosure fence (in yellow) which has been less successful in excluding mule deer. State Highway 25 can be seen bisecting the Pando Clone, and Fish Lake appears at the right of the image. Image courtesy Google Earth.

(Barnes 1975). Mock et al. (2008) identified a number of other genetically distinct aspen clones along the fringes of Pando, and hypothesized that the relatively few mutational variants within the Pando Clone may indicate a much less ancient age for Pando. DeRose et al. (2015) hypothesized that the Pando Clone initially became established as recently as the 1880s. However, this conclusion is based on core sampling of existing trees; it is unlikely that the 108-acre root system of the Pando Clone arose spontaneously in a single year or two; indeed it is far likelier that the clone spread gradually over a long span of years. Thus, the definitive overall age of the Pando Clone remains undetermined at this time. Grant and Mitton (2010) estimated the age of the Pando Clone at 80,000 years.

On the Fishlake National Forest, where the Pando Clone grows, aspen cover has declined by 60% from historic levels (Wooley et al. 2008). Fragmentation of the Pando Clone stand itself is currently occurring, due to browsing by herbivores suppressing sapling recruitment, rural real estate development, and a fungal infection called sooty-bark canker (DeWoody et al. 2008). As a result, sapling recruitment in unfenced portions of the Pando Clone is failing to replace aging adult trees. According to Rogers and Gale (2017: 9), "Judging from the near-complete lack of recent recruitment (> 2 m height) and mid-story aspen throughout the study area, it has been many years, likely even decades, since this amount of stand renewal [0.5 ramets per overstory tree] has taken place at Pando." As overstory trees continue to die without replacement by sucker recruitment, the overall size of the Pando clone ultimately will shrink (*id.*). Mule deer and cattle affect the Pando Clone and its ability to regenerate through browsing adventitious suckers and trampling.

Elk do not appear to be affecting the Pando Clone. According to Rogers and Gale (2017: 11, internal citation omitted), "While elk browsing of aspen is a serious concern regionally, we did not see elk or record their scat at Pando." Rogers and McAvoy (2018) reported that "[e]lk sign is evident in the broader area" and used that as a basis for asserting that elk were presently accessing the Pando Clone, but documented no elk sightings and no elk scat during the course of their study.

In 2012, the Forest Service issued a decision to fence 67 acres (22 ha) of the Pando Clone's 108-acre (43-ha) extent (see Figure 1) to prevent herbivory from domestic and wild ungulates and authorized some small-scale, experimental cutting inside the exclosures to stimulate suckering (USFS 2012). The exclosures were built of 8-foot tall woven wire topped with a barbed-wire strand. One exclosure of 17 acres (7 ha) was constructed in 2013 to the east of State Highway 25, and it appears to mostly exclude both mule deer and livestock (Rogers and Gale 2017, Rogers and McAvoy 2018), although Coles-Ritchie documented deer sign and evidence of browsing inside this exclosure. Aspen recruitment is progressing well inside the Pando Clone ungulate exclosure, even though the presence of mule deer has been documented inside the exclosure (Coles-Ritchie 2018). A second exclosure of 37 acres (15 ha) was constructed in 2014 to the west of the highway, incorporating a 22-year-old section of fence, and mule deer appear to be able to enter this fenced exclosure (Rogers and McAvoy 2018). Rogers and Gale (2017) found that the portions of the Pando Clone that had been fenced to exclude large herbivores showed a positive response in terms of regeneration (irrespective of cutting treatments), while the remaining 52 acres (21 ha) of the Pando Clone outside the exclosure showed no improvement. Rogers and Gale (2017) found that fencing alone resulted in an average of 550 regenerating suckers per acre inside the 2013 exclosure, a level sufficient for stand replacement according to earlier scientific findings (Mueggler 1989).

Aspen Declines

The regeneration problems experienced by the Pando clone mirror widespread declines in aspen regeneration, both on the Fishlake National Forest and throughout the West. In addition to the gradual replacement of aspen woodlands through the invasion of conifers in certain areas, aspen die-offs also occur in the absence of conifer encroachment. These die-offs can eliminate adult stems within a period of two years, and are often accompanied by an absence of sapling recruitment (Bartos 2008). On Cedar Mountain in south-central Utah, aspen stands showed depressed sucker recruitment and almost one-fifth showed crown dieback greater than 20% (Rogers et al. 2010). Evans (2010) found that drought weakened aspen on Cedar Mountain, Utah, making them more susceptible to a long-term decline that reduced the area of aspen woodland by 24% over a 23-year span. Many aspen stands in northern Nevada are in poor condition and have not regenerated in more than 100 years, due primarily to heavy livestock browsing (Kay 2003). Brown (1995) attributed the decline of aspen in eastern Oregon and Washington to intensive grazing and fire exclusion. Fairweather et al. (2007) documented a sudden decline of aspens on the Coconino National Forest in Arizona following a severe frost event, followed by a severe drought and an outbreak of tent caterpillars. Smooth brome, an invasive perennial grass, may affect aspen suckering (O'Brien et al. 2010). Overall, multiple factors can contribute to the decline of adult aspens, but reproduction through suckering typically occurs unless it is suppressed be herbivory by non-native livestock or by native herbivores such as deer and elk.

While the gradual decline of aspen groves over time may be widespread, aspen die-offs also occur that eliminate adult stems within a period of two years, with an absence of sapling recruitment (Bartos 2008). Sudden Aspen Death syndrome is associated with aspens at high altitude under water stress (Worrall et al. 2010). The decline of the Pando Clone appears to be of the more gradual variety, rather than Sudden Aspen Death syndrome.

Aspens most commonly reproduce adult stems via suckering from the rootstock; its seeds, while abundant, are short-lived and have demanding germination requirements (Schier 1981, Kay 2003, Long and Mock 2012). As a result, seedling establishment typically occurs only during extremely wet summers (Jones and DeByle 1985b).

Schier (1975) described the dynamics of sucker production as governed by apical dominance, a phenomenon whereby hormones from the terminal buds of aboveground stems (auxins) inhibit hormones in the root system that stimulate sucker growth (cytokinins). When disturbance of the stems reduces the flow of auxins, the cytokinins can initiate the regenerative process. However, when aboveground stems weaken and die, the root system dies back due to a lack of photosynthate being furnished to the roots. Schier (1976) suggested that sucker regeneration is proportional to above-ground disturbance, citing examples from clearcut logging studies where the number of suckers generated is proportional to the number of stems removed by logging. Where suckering is suppressed by ungulate browsing, the die-off of adult aspens can result in areal reductions in aspen habitats across the landscape.

Shepperd (2001) proposed hormonal stimulation, a proper growth environment, and sapling protection as the three elements of an aspen regeneration triangle. Natural disturbances such as fire can stimulate suckering and regenerate aspen stands, but if livestock are not excluded from the aspen grove for several years following fire, their browsing can severely suppress sucker growth (Kay 2003).

The Role of Herbivory in Aspen Declines

Heavy ungulate browsing over extended time periods can cause regeneration failure over spans of many decades, resulting in an even-aged grove of older trees that is less resilient to drought and other stressors (Lindroth and St. Clair 2013). In the Book Cliffs of northeastern Utah, Rogers and Mittanck (2014) found that only three of 77 aspen stands (less than 4%) contained adequate levels of recruitment to perpetuate the stand, due substantially to browsing by wild and domestic herbivores. Herbivory by both domestic livestock (sheep and cattle) and wild ungulates (deer and elk) can suppress aspen shoot recruitment, and thus impair regeneration.

In some circumstances, large herbivore grazing and/or browsing in aspen stands may not put significant pressure on aspen reproduction. For example, Beck and Peek (2005) found only a 3% dietary overlap between spring and summer mule deer and cattle diets in aspen stands, with deer preferring browse and cattle preferring grasses and forbs, and also found that elk and cattle did not have significantly different diets. However, this study found that all of the herbivores studied had a 0% dietary consumption of aspen, with the exception of spring diets in one of three years, which showed <1% aspen contribution to the elk diet. Mower and Smith (1989) found that elk and mule deer diets in northern Utah were quite similar, and although shrubs made up a significant component of both, aspens were not noted as a significant component of the diet. Notwithstanding the preference of native and domestic ungulates for other forage plants, overbrowsing of aspen shoots to the point of regeneration failure is widespread.

Aspens have defensive compounds – phenolic glycosides and tannins – that provide adequate defense against insects and mammalian herbivores when browsing is light, but which is an inadequate defense under heavy browsing, which results in high levels of damage to the trees (Lindroth and St. Clair 2013). Elk may respond negatively to increasing phenolic glycoside content (Wooley et al. 2008). However, the scientific consensus is that while the tannins and phenolic glycosides present in aspens evolved as a defense against herbivory, they are insufficient to prevent browsing by either domestic or wild ungulates.

Deer and Elk Browsing

Aspen communities often are an important source of protein for mule deer in summer, whereas Utah serviceberry and big sagebrush communities may only provide maintenance amounts of protein (Austin and Urness 1985). This dietary advantage of aspen communities may contribute to mule deer preference for them. Severe browsing by elk and deer virtually eliminated sapling recruitment during an aspen die-off in northern Arizona (Fairweather et al. 2007). Additionally, population irruptions of mule deer on the Kaibab Plateau of northern Arizona in the 1920s had, between 1953 and 1962, completely suppressed aspen recruitment on the Kaibab Plateau of northern Arizona (Binkley et al. 2006). In Michigan, Randall and Walters (2011) found that increasing densities of white-tailed deer in aspen stands suppressed suckering and reduced forb density and species richness.

Livestock grazing in aspen groves may come at a cost for resident mule deer. Loft et al. (1991) suggested that as a result of livestock grazing, displacement of mule deer from these habitats and expansion of deer home ranges resulted in a lowering of inclusive fitness for mule deer. According to Loft et al. (1991: 22, internal citation omitted), "Once aspen stands had been occupied by cattle for a few weeks, there was little forage or hiding cover available, and deer essentially quit using the habitat." These studies indicate the likelihood that forage removal by cattle or domestic sheep can alter mule deer habitat selection and/or diet choices.

The likelihood of suppressed aspen regeneration from concentrated elk browsing appears to be greater than for mule deer. Compton (1974) found that elk in the Sierra Madre Range in Wyoming concentrated their summer use in subalpine parks, and found heavy autumn use in aspen cover types. Beck et al. (1997) reported that aspen made up 10% of elk summer diet, versus 3% of domestic sheep summer diet, in north-central Utah. Elk foraging on winter ranges has been shown to depress growth and prevent reproduction of aspen in Rocky Mountain National Park (Baker et al. 1997, Suzuki et al. 1999, Binkley 2008). Aspen are likely to be suppressed where elk density exceeds four elk per square kilometer (Painter et al. 2018). Elk also damage aspens by browsing new shoots, rubbing flexible saplings with their antlers, and by gnawing tree bark to get at the phloem underneath (Fairweather et al. 2007).

In the absence of large native predators, elk can suppress aspen sapling recruitment (Binkley 2008, Beschta and Ripple 2009). Ripple and Larsen (2000) found that due to heavy browsing by elk, following removal of wolves, only 5% of the current overstory aspen in the Northern Range of Yellowstone National Park originated after 1921. Painter et al. (2018) found that the percentage of aspen suckers browsed annually in Yellowstone National Park was 80-100% in 1997-98, decreasing to 30-60% in 2011-15 after the re-establishment of a wolf population. Elk shifted their habitat use and herbivory intensity away from Yellowstone National Park and toward the lower Madison River Valley in response to increasing wolf populations in the Park (Painter et al. 2018). However, in some localities within Yellowstone National Park, elk densities have remained high enough to continue to suppress aspen suckering (*id.*). White et al. (1998) found that elk browsing suppressed aspen recruitment in Canadian Rockies national parks, except under conditions when elk densities were reduced by wolves. There is now a broad scientific consensus that the absence of large native predators can result in depressed recruitment of aspen and other woody species (Beschta and Ripple 2009, Painter et al. 2018).

Browsing Pressure from Domestic Livestock

Livestock often concentrate their grazing activity in aspen groves due to the availability of shade and preferred understory forage species. In the Sierra Nevada mountains, cattle utilized meadow riparian and aspen habitats most strongly, selecting them over other habitats (Loft et al. 1991). According to Kay (2003: 41), "Even on allotments where livestock use has been controlled, aspen stands near water may still be in poor ecological condition because cattle tend to concentrate in those areas." Bailey et al. (1990: 213) found that cattle impacts on aspen are so severe that livestock can be used as a means to suppress aspen reproduction, stating "Overgrazing is generally considered to be detrimental to range stability and productivity over the longer term, but short duration heavy grazing may have a place in forage establishment and control of woody species." These researchers (*id.*: 214) recommended, "Clearly, for immediate control of aspen suckers, top removal or defoliation must be timed similarly to the late grazing treatment in this study. However, aspen suckers are suitable forage for cattle provided they are maintained within reach."

Beschta et al. (2014) found that aspen recruitment rates plummeted in the late 1800s with the onset of cattle grazing on the lands that would become Hart Mountain National Wildlife Refuge in southern Oregon, and increased by an order of magnitude after livestock were removed in 1990. These researchers attributed the decline of aspen groves on Hart Mountain to top-down forcing by cattle browsing, which suppressed aspen sapling recruitment, rather than climate changes. On Monroe Mountain in south-central Utah, Bartos and Campbell (1998a) provided photographic documentation of the effects of livestock preventing aspen regeneration using fenceline contrasts of a previously burned and logged area which remained barren in the presence of livestock and failed to regenerate. Across the fence, on habitats accessible to native herbivores but where livestock were excluded, dense regeneration was evident. Alexander (1995) documented that trampling by livestock broke 40% of aspen samples under both moderate and heavy grazing in his Alberta study; trampling caused damage in the form of basal scars that were present on 25% of surviving aspen saplings. By the second spring of cattle grazing, aspen sapling mortality in this study was 25%, 70%, and 89% for the ungrazed, moderately grazed, and heavily grazed sites, respectively.

Cattle selection for aspen shoots differs by season. According to Kay (2003: 32), "Year-long or season-long grazing is particularly detrimental to aspen, while early-season or dormant-season use may allow aspen to successfully regenerate." According to Jones et al. (2011: 629), "Aspen suckers received no early-growing season use by cattle but received the heaviest late-growing season use of all three vegetation types. Utilization was the same for all vegetation types at mid-growing season. Mean late-growing season use of aspen suckers was greater than 60%, and some stands received 100% use." Jones et al. (2011: 630) observed, "By mid-growing season, the quality of meadow and aspen understory vegetation approached minimum nutritional levels required for cattle." Alexander (1995) found that aspen suckers that have not yet begun to lignify, or become woody (i.e., one-year-old suckers), are a palatable forage for cattle, while two-year-old suckers were "not readily used" by cattle.

Even moderate levels of livestock grazing can suppress aspen regeneration. Alexander (1995) found that moderate and heavy grazing by cattle were equally effective at preventing aspen regeneration, with both moderate and heavy grazing both had a significant negative effect on understory biomass production in aspen stands.

Methods

We quantified ungulate use of the Pando Clone area with two motion-triggered cameras (Cameras 2 and 3) that were placed in portions of the Pando Clone outside the fenced exclosures, and two cameras that were placed in neighboring aspen groves (Cameras 1 and 4) subjected to the same pattern of livestock and mule deer herbivory. The cameras were sited in areas open to grazing and browsing by both domestic livestock and wild ungulates. The cameras were set to take photographs of all motion-triggered events separated by at least 1 minute. Cameras were installed on May 11th, 2018 and retrieved on November 22nd, 2018 to record herbivore activity throughout the growing season. The cameras were more sensitive to motion than expected. As a result, two of the cameras (Cameras 1 and 4, the cameras sited in neighboring aspen groves) ran out of battery power well before the end of the monitoring period, and therefore failed to record photographs during the livestock grazing period. These cameras, when remaining operational throughout the summer and into the fall, provide useful comparisons of forage utilization during cattle-free and cattle grazing periods, but could not be used to compare animal unit equivalents between deer and cattle due to the absence of livestock records.

After retrieval, the photographs were individually examined and the counts of ungulates were tallied for each camera. In order to more accurately compare total use by ungulate species, use was calculated based on body size/forage consumption by the Animal Units Equivalents (AUEs). A literature search found a range of estimates for mule deer, ranging from 0.2 (Pratt and Rasmussen 2001, NRCS 2003) to 0.17 (Ogle and Brazee 2009). For our calculations, we used 6 deer per 1 cattle animal unit (AU) (0.167), which is conservative being based on a 1,000-pound cow with calf. Cattle weights have increased significantly over the last 40 years with current average slaughter weight is presently 1,382 pounds (628 Kg) as of December 2017, (NASS 2018). We graphed the AUE data by week to display use over the monitoring period. It should be noted that the ratio of six deer per cow greatly underestimates the difference. A 1,382-pound (628 kg) cow consumes 3% of its body weight per day, or 41.6 lbs (18.9 Kg) (Ogle and Brazee, 2009). A 150-pound (68 kg) mule deer consumes 1.5 kg/day (UWSP 2019). This current information indicates a mature cow consumes 12.6 times the forage demand of a mule deer. However, we used the lower value to provide a conservative comparison.

We created time lapse videos of the photographs from each camera to help visualize conditions and herbivore use throughout the growing season.

The Interagency Landscape Appearance Method

This method's descriptions classify forage utilization into the following Herbaceous Utilization classes (USFS 1993; *see also* BLM 1996):

1. No Use (0-5%). The rangeland shows no evidence of grazing use; or the rangeland has the appearance of negligible grazing.

2. Slight (6-20%) The rangeland has the appearance of very light grazing. The key herbaceous forage plants may be topped or slightly used. Current seedstalks and young plants of key herbaceous species are little disturbed.

3. Light (21-40%) The rangeland may be topped, skimmed, or grazed in patches. The low-value herbaceous plants are ungrazed and 60 to 80 percent of the number of current seedstalks of key herbaceous species remain intact. Most young plants are undamaged.

4. Moderate (41-60%) The rangeland appears entirely covered as uniformly as natural features and facilities will allow. Fifteen to 25 percent of the number of current seedstalks of forage plants are utilized. (Moderate use does not imply proper use.)

5. Heavy (61-80%) The rangeland has the appearance of complete search. Key herbaceous species are almost completely utilized with less than 10 percent of the current seedstalks remaining. Shoots of rhizomatous grasses are missing. More than 10 percent of the number of low-value herbaceous forage plants have been utilized.

6. Severe (81-100%) The rangeland has a mown appearance and there are indications of repeated coverage. There is no evidence of reproduction or current seedstalks for key herbaceous species. Key herbaceous forage species are completely utilized. The remaining stubble of preferred grasses is grazed to the soil surface.

These videos can be accessed at https://www.westernwatersheds.org/pando-clone-time-lapse/. We also took photographs along the 2013 exclosure fence to document the contrasting rates of regeneration within and outside the exclosure.

Utilization of vegetation by herbivores was estimated using the interagency Landscape Appearance Method descriptions, an estimation procedure used on the Fishlake National Forest (USDI Technical Reference 1734.3), see accompanying box. Utilization was estimated at the photo location on the day prior to livestock entry, 7 days after livestock entry (half of the livestock use period) and again after livestock removal.

Results

The exclosures constructed by the Forest Service in 2013 and 2014 within portions of the Pando Clone provide a clear contrast between natural recovery rates inside the exclosures with the effects of this heavy to severe level of utilization outside the exclosures. The exclosures were built of 8-foot tall woven wire topped with a barbed-wire strand. Figures 2 and 3 are taken from the same location, with one looking into the interior of the 2013 exclosure and the other looking into the grazed allotment, and area used by both deer and cattle.

From the ongoing aspen recovery that has occurred since the exclosures were constructed in 2013 and 2014, and the complete lack of any recruitment of aspen sprouts occurring outside the exclosures, it is clear that current management outside the exclosures prevents the regeneration of the Pando Clone in areas open to livestock grazing. Inside the 2013 exclosure fence, we found successful aspen recruitment is occurring irrespective of any mule deer that may have found a way to enter the exclosure area.

Camera 1 recorded from May 11th, 2018 through August 13th, 2018, prior to the onset

Figure 2. A view inside the 2013 exclosure. Note abundant regeneration 8-12 feet tall after 5 years of exclusion. June 10th, 2019.

Figure 3 (above). Looking into an unfenced portion of the Pando Clone from the same location with no regeneration. June 10th, 2019.

Figure 4 (below). - Fenceline contrast with abundant regeneration inside the 2013 exclosure (left) and no regeneration occurring outside (right). June 10th, 2019.

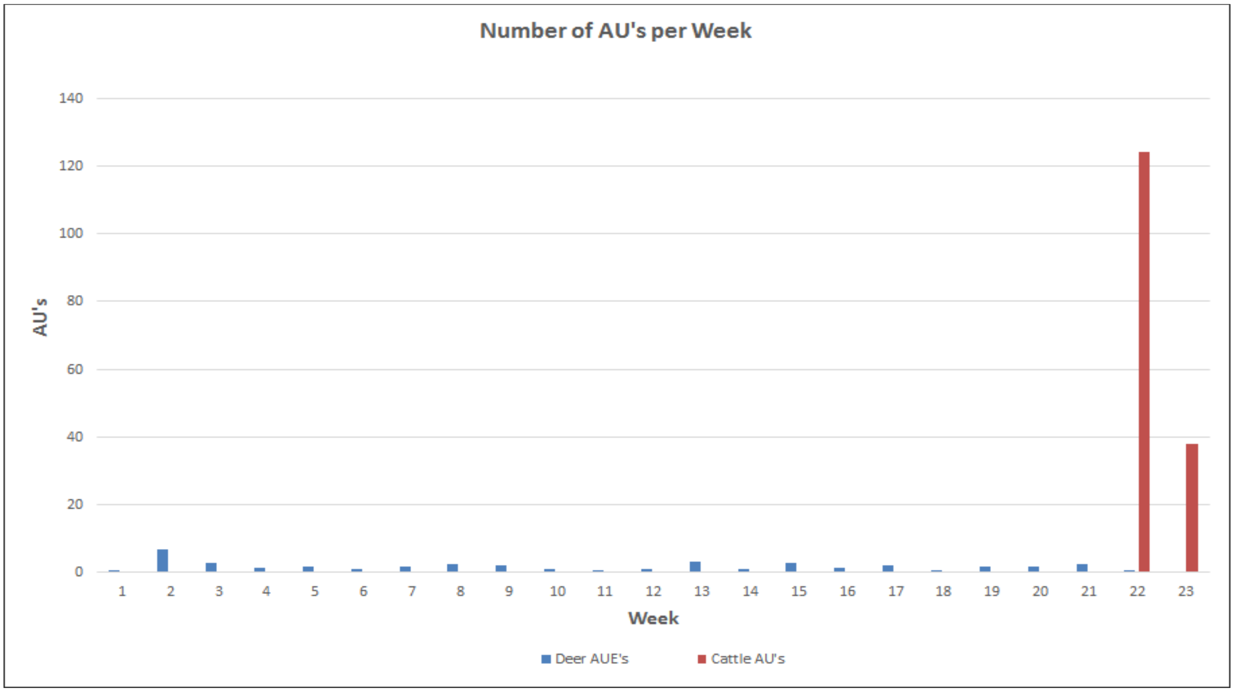

Figure 5. Camera 2, within an unfenced portion of the Pando Clone, deer versus cattle Animal Units by week.

of livestock grazing. Camera 2 recorded from May 11th, 2018 through October 9th, 2018.

Camera 3 recorded from May 11th, 2018 through November 22nd, 2018. Camera 4 recorded from May 11th, 2018 through September 22nd, 2018, prior to the onset of livestock grazing. The livestock use period began on October 4th and ended October 16th for a total of 13 days, during which domestic cattle were the type of livestock present in the project area. The area under study received use by mule deer (*Odocoileus hemonius*) throughout the monitoring period.

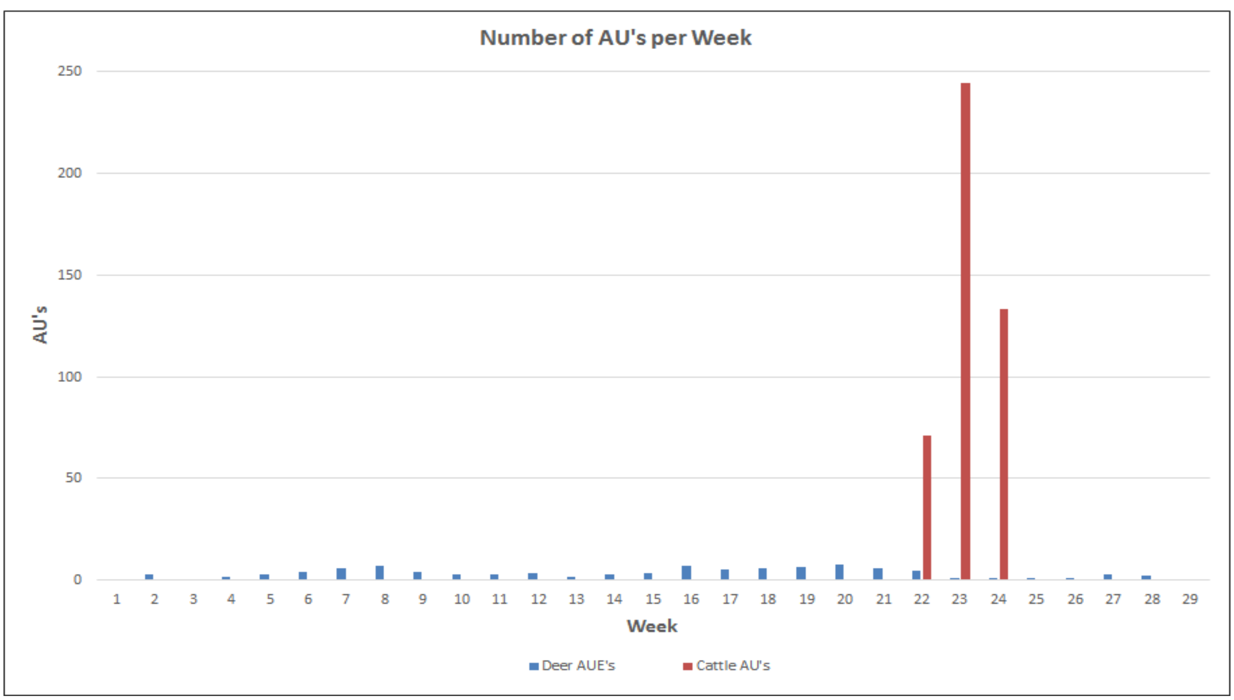

Figure 6. Camera 3, within an unfenced portion of the Pando Clone deer versus cattle Animal Units by week.

Rogers and McAvoy (2018) reported that "[e]lk sign is evident in the broader area" and used this as a basis for asserting that elk might presently be accessing the area. We documented no elk sightings in the Pando Clone, but Camera 1 recorded four elk in one instance in an adjacent aspen grove.

For Camera 1, motion from grass moving in the wind depleted the power supply by August 13th, 2018, so only forage utilization observations could be made. For Camera 2, deer use during the 6-month period totaled 42 AUE's, while cattle use during the 6 days (slightly less than 50% of the cattle use period) totaled 162 AU's. For Camera 3, deer use during the 6-month period totaled 101 AUE's, while cattle use during the 13 days totaled 448 AU's. For Camera 4, motion from grass moving in the wind depleted the power supply by September 27th, 2018, so only utilization observations could be made. On average, the index for animal use documented for cameras that lasted into the livestock grazing season was found to be four times higher for cattle during the 13 days of livestock grazing than for mule deer over the course of the entire growing season. Camera 4, on the eastern shore of Fish Lake, documented a similar result.

During the months prior to the arrival of livestock all cameras documented no observable utilization of the vegetation, whereas within 7 days after the arrival of livestock utilization was in the "heavy" category (61-80% utilization) for Cameras 2 and 3, inside the Pando Clone. After livestock removal, use was in the upper "heavy" to mid "severe" (81-100%) categories at all four sites (see Figures 10, 16, 22, 27, 28, and 29).

Figure 10 shows conditions following livestock removal for Camera 1, in an aspen grove adjacent to Pando. Based on the descriptions in the Landscape Appearance Method this would fit in the upper end of the "heavy" (61-80%) category. Within the Pando Clone, patterns of herbivory by mule deer and livestock were essentially identical to Pando's genetically distinct neighboring groves. For Camera 2, livestock use was near the upper end of the "heavy" category by day 7, with significant utilization on rabbitbrush (*Chrysothamnus* sp.), which has low palatability (see Figure 15). By the time livestock were removed, forage utilization levels, based on

Figure 7. Camera 1 (in an aspen grove immediately adjacent to the southeast corner of the Pando Clone) at deployment. Note mountain lion.

Figure 8 (above). Camera 1, mid-June.

Figure 9 (below). Camera 1, mid-summer.

Figure 10 (above). Camera 1 location on November 22nd, 2018, after livestock removal. Forage utilization shown here is in the upper end of the "heavy" (61-80%) category.

Figure 11 (below). Camera 2, within the Pando Clone, at deployment.

Figure 12 (above). Camera 2, inside the Pando Clone, in mid-June.

Figure 13 (below). Camera 2 in late summer.

Figure 14 (above). Camera 2 just prior to livestock entry.

Figure 15 (below). Camera 2 after 7 days of livestock use.

Figure 16 (above). Camera 2 location on November 22nd, 2018, taken in the general direction the remote camera had been pointed, after livestock removal. Forage utilization shown here is in the mid to upper end of the "severe" (81-100%) category.

Figure 17 (below). Camera 3 (within the southeastern edge of the Pando Clone) at deployment.

Figure 18 (above). Camera 3 in mid-June.

Figure 19 (below). Camera 3, mid-summer.

Figure 20 (above). Camera 3 just before livestock entry. Note for reference the two large bunchgrasses on the left and the scattered fallen limbs on the ground.

Figure 21 (below). Camera 3 after 7 days of livestock use.

Figure 22 (above). Camera 3 after livestock removal. Forage utilization shown here fits in the upper end of the "heavy" (61-80%) category.

Figure 23 (below). Camera 4, above the eastern shore of Fish Lake, at deployment.

Figure 24 (above). Camera 4 in mid-June.

Figure 25 (below). Camera 4, mid-summer.

Figure 26 (above). Camera 4 just before livestock entry.

Figure 27 (below). Camera 4 location, taken in the general direction of the remote camera, on November 22nd, 2018 after livestock removal. Based on the descriptions in the Landscape Appearance Method this level of herbivory fits in the upper end of the "heavy" (61-80%) category.

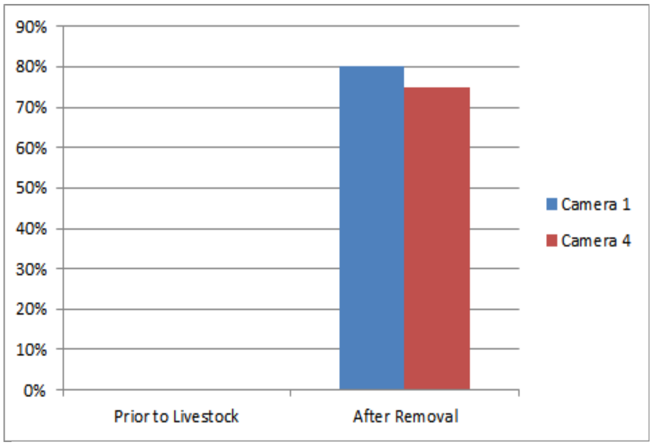

Figure 28. Forage utilization levels before livestock entry and after removal for Cameras 1 and 4, in aspen groves adjacent to the Pando Clone.

the descriptions in the Landscape Appearance Method grazing levels shown by Camera 2 this would fit in the mid to upper end of the "severe" (81-100%) category. By day 7 of livestock use documented by Camera 3 (see Figure 21), the large bunchgrasses had been completely grazed and only a small fraction of the seedheads remained. Note the difference in visibility of the fallen branches at ground level between Figures 20 and 21. Figure 22 shows conditions for Camera 3 following livestock removal. Based on the descriptions in the Landscape Appearance Method this would fit in the upper end of the "heavy" (61-80%) category. By the time livestock were removed, forage utilization levels shown by Camera 2 (see Figure 16) would fit in the mid to upper end of the "severe" (81-100%)

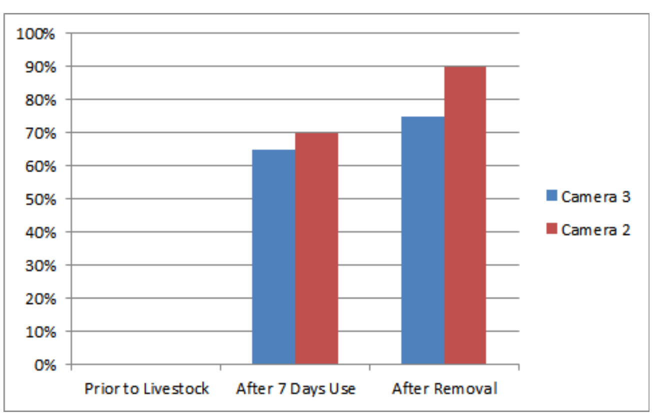

Figure 29. Forage utilization levels before livestock entry and after removal for Cameras 2 and 3, sited within the Pando Clone.

down to the same level as the rest of the forage base.

While we hoped to document direct herbivory by deer and/or cattle on aspen sprouts with the remote cameras, in fact we were unable to document any aspen sprouts at all during the growing season period over which our cameras were deployed. This is consistent with the findings of Rogers and Gale (2017), who also reported essentially no aspen sprouts outside the exclosure fence. Thus, like Rogers and McAvoy (2018), we are unable to measure direct herbivory of aspen by either mule deer or cattle, and are left with making inferences from indirect measures (in the case of this monitoring report, overall forage consumption and animal use). The level of trampling by cattle appears to be heavy in all locations we monitored.

Our findings support the conclusions of Loft et al. (1991), that the presence of livestock results in habitat abandonment by deer. Deer use dropped to nearly zero after the arrival of livestock and only returned after livestock removal and then at much lower levels than prior to livestock entry.

Discussion

We documented levels of herbivory by mule deer that were too light to quantify throughout the summer, measured by the Landscape Appearance Method used by the Forest Service to estimate forage utilization. This was followed by heavy to severe understory utilization by cattle that virtually eliminated understory vegetation during the 14-day period in October when cattle were turned out both in unfenced portions of the Pando Clone, and in neighboring aspen groves subjected to the same pattern of livestock grazing. The level of livestock forage utilization we documented (70 to 90%) was consistent with heavy grazing as defined by Alexander (1995), who classified 73% forage utilization by cattle as "heavy" and found this level – entailing the browsing of 95% of aspen saplings – to be sufficient to suppress aspen regeneration (see Figures 28 and 29). This is supported by scientific observations at Pando itself. Rogers and Gale (2017: 11) concluded, "A key message, then, is that while we cannot state unequivocally that there are 'too many' herbivores at Pando, we do know that there are too many for current conditions."

These heavy to extreme levels of forage utilization exceed the Forest Service allowable utilization level of 50% (USDA 2018). In addition, these levels are much greater than the 25% level supported by leading range scientists (Galt et al 2000).

By quantifying the ungulate use of the Pando area and tracking utilization over the study period, our data and analysis demonstrates, based on Animal Unit Equivalents, that more than 4 times the animal use occurs in the unfenced portion of the Pando Clone and in neighboring aspen groves from livestock than for mule deer. Nearly all of the observable forage utilization in the understories of aspen groves in this area during the monitoring period was the result of livestock. According to Rogers and Gale (2017: 6),

> we counted only one mule deer scat pile, but 219 cattle deposits in Year 1. In Year 2, we counted no scat piles of any species within the fence, but 72 cattle and five deer piles outside the exclosure. By Year 3, cattle deposits were 64 and deer scat was 14, all outside the exclosure.

Our results are consistent with these findings.

Our findings contrast with Rogers and McAvoy 2018, which concluded that mule deer are the primary factor in regeneration failure in the Pando Clone. The Rogers and McAvoy study used "browse level, and feces counts as a surrogate for ungulate presence." Its analysis identified deer presence (indexed by density of pellet groups) as the key factor relating to failure of aspen sprouts to recruit. Cattle presence as indexed by feces was negatively related to both recruitment and aspen density but was not identified as a major factor by this exploratory analysis. It is troubling that while pellet groups were negatively related to aspen regeneration in the Rogers and McAvoy study, browse level was not a significant factor. Browsing of aspen saplings would presumably be the direct means by which either mule deer or cattle would directly affect sapling survival and recruitment.

In addition, Rogers and McAvoy's identification of cattle concentration as an unimportant factor in aspen recovery runs contrary to earlier findings that aspen recruitment is lowest in portions of the Pando Clone accessible to livestock, and higher in fenced areas, whether these are accessible to mule deer or not (Rogers and Gale 2017, Coles-Ritchie et al. 2018). Rogers and McAvoy (2018) concluded that deer were the cause of regeneration failure. But in their analysis of regeneration, the 2014 exclosure was accessible to deer but not cattle, and had a browse level of 24%, while in the unfenced area, where both deer and cattle were present, the browse level was 55%. Furthermore, aspen recruitment was highest in the 2014 exclosure (1,204 stems/ha) in the presence of deer and lowest in the 2013 exclosure from which both deer and livestock were absent, further muddying this conclusion.

The season of livestock grazing can also have a major impact on regeneration. Livestock show greater preference for browsing aspen shoots in autumn than in spring (Fitzgerald et al. 1986). Aspen suckers have higher nutritional quality than other forage types throughout the year, but cattle focus their foraging on meadow and understory vegetation in early and late summer, increasing utilization of aspen suckers only late in the growing season when other forage types were of low nutritional quality and depleted by grazing (Jones et al. 2011). However, experimentally browsed aspens showed greater growth when browsed in the autumn than when browsed in early or late summer (Jones et al. 2009). Balancing aspen's greater resilience to livestock grazing in fall with the far greater tendency of cattle to select aspen browse at this same time of year thus becomes critical.

Late-season grazing by cattle (just before leaf drop) is the most effective season for cattle grazing to suppress aspen regeneration, and livestock grazing during this time of year can eliminate aspen seedling recovery after six consecutive seasons of grazing post-fire (Bailey et al. 1990). Jones et al. (2011) recommended avoiding late-season grazing by cattle in aspen stands to minimize browsing on aspens, and recommended that mid- and late-season grazing by cattle not occur in consecutive years. Jones et al. (2011) recommended avoiding late-season grazing by cattle in aspen stands to minimize browsing on aspens, and recommended that mid- and late-season grazing by cattle not occur in consecutive years. In the case of the Pando Clone, livestock use this pasture in the fall every year, at the very time of year when the greatest selection by cattle for aspen shoots occurs. In this case, cattle were turned out in the Pando Clone in early autumn, precisely the season when the tendency of cattle to browse on aspen saplings would be expected to be greatest based on the science.

In our monitoring, we found livestock use in the "heavy" to "severe" categories that would result in complete use on any aspen suckers that had emerged. Our cameras were unable to detect aspen sprouts – or either mule deer or cattle herbivory on them – but the end result was that aspen sprouts were virtually completely suppressed outside the exclosure fences, based on the absence of any aspen sprouts visible in our photographs. This finding is consistent with other reports documenting little or no aspen recruitment outside exclosure fences that prevent grazing by livestock (but do not always prevent access by mule deer).

Cattle grazing in parts of the Pando Clone outside the exclosure, and in neighboring unfenced aspen groves, may also have a synergistic effect with the herbivory by native mule deer, resulting in impacts to aspen recruitment that may be greater than simply adding the two types of impact together. Wild herbivores may be drawn to ungrazed areas where livestock have been excluded (O'Brien et al. 2010). Aspen habitats are preferred by mule deer when cattle are absent, but preference declines under moderate to heavy grazing to the point where deer use aspen habitats roughly in proportion to their availability (Loft et al. 1991). Mueggler and Bartos (1977) studied an exclosure accessible to deer but not livestock in which production of forbs, or broad-leaf understory herbs, occurred inside the exclosure. This abundance of forage likely concentrated deer foraging activity inside the exclosure, to the detriment of aspen suckers, which failed to survive to reach tree status between 1905 and 1934, based on subsequent tree-ring analysis.

Austin and Urness (1985) reported that aspen proportion in mule deer summer diets ranged from 0.2 – 3%, but increased to 9% in September. The heavy level of understory utilization by cattle in the unfenced parts of the Pando Clone and in nearby aspen stands (70-90% as found in our study) during a time of year when deer intrinsically increased their herbivory on aspen saplings may, through competition, further increase mule deer browsing on aspen shoots by leaving behind few alternative sources of forage.

Kay and Bartos (2000) studied exclosures on the Dixie and Fishlake National Forests that excluded deer and livestock both, or livestock only. Complete failure of new regeneration occurred in the presence of both livestock and deer herbivory outside the exclosures at 4 of the 5 sites where portions of the exclosures prevented access by both deer and livestock, and at 3 of the 8 sites having livestock-only exclosures new regeneration failed in areas where the livestock were excluded. Kay and Bartos found that excluding livestock and/or native herbivores increased recruitment of aspen saplings in the 2-meter to 5-centimeter diameter-at-breast-height range, with an average of 4,474 surviving aspen ramets under livestock and cervid exclusion, 2,498 ramets surviving by excluding livestock only, and an average of 1,012 surviving ramets outside the exclosures, where aspens were subject to herbivory by cattle, sheep, deer, and/or elk. Rogers and Gale (2017) documented a more than fourfold increase in aspen regeneration inside the Pando Clone's fenced exclosure compared with outside.

In this monitoring project, we found little visual evidence of aspen recruitment outside the exclosure fences, indicating either that aspen sprouts were browsed away prior to the onset of the growing season for grasses, or that deer and/or livestock herbivory was eliminating them prior to the point at which they would become visible to the camera. Given the extreme level of understory herbivory by cattle during the 13-day grazing period that we recorded in 2018, it is entirely possible that mule deer returning to the Pando Clone following cattle grazing would have found little understory forage, increasing the likelihood of 100% utilization of aspen sprouts that emerged prior to the onset of the following season. In this way, the overgrazing by cattle that we recorded within unfenced portions of the Pando Clone may be interacting with herbivory by mule deer to eliminate aspen recruitment outside the ungulate exclosures.

Bailey et al. (1990: 214) found fall cattle grazing to be an effective tool for eliminating aspen regeneration:

> Suckers defoliated by grazing in August, late in the growing season, were nearly eliminated after only 1 defoliation (FitzGerald and Bailey 1984) whereas suckers defoliated earlier in the season continued to regenerate and took 7 years to decline to 7% of original stem densities…. Schier (1976) indicated that repeated removal of tops and consequent initiation and growth of new suckers leads to a gradual depletion of nonstructural carbohydrates in the roots. Exhaustion of carbohydrates by annually repeated destruction of growing points appears to take from 6 to 8 years…. Clearly, for immediate control of aspen suckers, top removal or defoliation must be timed similarly to the late grazing treatment in this study.

These authors conclude by stating, "If the first priority is to nearly eradicate regenerating aspen suckers, then late season, short duration heavy grazing should be applied." Unfortunately, this is exactly what is happening within the unfenced Pando Clone and surrounding aspen groves.

Trampling damage by ungulates has often been implicated as a potentially significant cause of aspen regeneration failure (Schier 1981, DeByle 1990, Brown 1995). With regard to cattle, Weatherill et al. (1969: 5) concluded that "[c]onsumption reduces photosynthesis, trampling may break stems and leaves, while soil compaction can injure root systems and decrease soil aeration and water holding capacity." While Dockrill et al. (2004: 261) found that damage from cattle due to direct browsing and trampling damage killed individual aspen sprouts, these researchers concluded that "[h]igh mortality among stems without observed injuries might have been indirectly associated with cattle damage resulting from soil compaction, reduced root oxygen and subsurface severing of lateral roots." Because adventitious buds forming on lateral roots are the genesis of aspen sprouts, and because the level of trampling by cattle appears to be substantial based on our monitoring, more detailed study of the effect of trampling by livestock on the roots, adventitious buds, and initiation of suckering in the Pando Clone is necessary prior to concluding that herbivory by deer or livestock (or some synergistic combination of the two) is primarily responsible for the failure of sprout recruitment outside fenced exclosures.

Livestock appear to have the heavier impact than mule deer on aspen regeneration, based on exclosure studies that differentially exclude cattle and wild cervids. Based on a study of 30 grazing exclosures in aspen habitats in Nevada, Kay (2003: vi) stated,

> The [declining] status and trend of aspen communities in north-central Nevada, however, is not related to climatic variation, fire suppression, or browsing by mule deer. Instead, the condition of individual aspen communities is related to past and present levels of livestock grazing. That

> is, aspen is declining throughout most of north-central Nevada due to repeated browsing of aspen suckers by cattle and/or domestic sheep – repeated browsing eliminates sucker height growth, which prevents their maturation into aspen saplings and trees. Without stem replacement, aspen clones are consigned to extinction.

Livestock in mountain ranges of central Nevada contributed to poor aspen clone condition, and grazing by sheep and cattle accounted for 99.5% of the grazing pressure based on feces counts (Kay 2001).

While mule deer have been implicated as the cause of regeneration failure in the Pando Clone (Rogers and McAvoy 2018), the bulk of science thus far published (reviewed herein) does not necessarily support this conclusion, and our own monitoring photos show quite clearly that cattle, rather than mule deer, are having the heaviest impact on understory vegetation in the Pando Clone and on the understories of neighboring aspen groves.

Recommendations

We recommend eliminating livestock grazing during all seasons for the entire Pando Clone, and for aspen habitats generally, livestock should be removed if aspens are experiencing regeneration failure. This should be done until aspen regeneration is above browse height, and will require periodic repetition to prevent future aspen sprout suppression. Kay (2003) recommended fencing critical aspen stands or restricting livestock to only early-season grazing. According to Beschta et al. (2014: 36, internal citations omitted), "Our results indicate that for areas grazed by livestock and where aspen recruitment is either absent or occurring at low levels, implementing strategies that eliminate or minimize the effects of livestock herbivory may be needed. Given the vast amount of public land annually utilized by domestic ungulates and the large losses in aspen those lands have experienced to date, reducing livestock grazing effects within and across ecoregions may be required for attaining ecological restoration of herbivore-altered plant communities." According to Alexander (1995: 120), "even though aspen sucker density was still high after two years [cattle] grazing, it was the author's opinion that if the grazing treatments were continued, the prognosis for successful aspen forest regeneration would be poor."

Mechanical treatments such as coppice logging do not appear to be warranted in the Pando Clone based on the science. Aspen stands can reach high densities without stagnating because they are self-thinning (DeByle 1984). Thus, the thinning or logging of aspen stands is unwarranted from a silvicultural perspective. Bird species richness increases with aspen patch size (Johns 1993), suggesting that fragmenting aspen stands into progressively smaller patches through clearcutting may lead to a loss of bird diversity. In the Pando Clone, coppice logging of aspens might also inadvertently cause a loss of genetic diversity by completing the dominance of triploid aspens (DeRose et al. 2015). The successful regeneration of aspen saplings inside the Pando Clone's exclosure fence in the absence of mechanical treatments is proof positive that mechanical interventions are unnecessary.

The idea of eliminating grazers from aspen stands struggling to reproduce is not a new concept. Mueggler (1989) recommended protecting aspen groves with exclosures where the stand is heavily grazed or browsed. According to Shepperd (2001: 363), "Fencing is the only guaranteed means of directly protecting sprouts from browsing animals." O'Brien et al. (2010: 28) recommended, "In situations where the relative impact of domestic livestock versus wildlife has not been determined, a livestock exclusion fence alone (followed with monitoring) may be a reasonable first choice."

The significant role of cattle grazing in the Pando Clone has been acknowledged by scientific researchers. According to Rogers and Gale (2017: 11), "While we know that mule deer are responsible for a portion of aspen sucker browsing, cattle reduction and exclosure seem to also play an important role as evidenced by the combination of scat counts, browse levels, and overall regeneration response inside and outside our study area."

At a minimum, the existing exclosures should be expanded to encompass the entire perimeter of the Pando Clone, plus a quarter-mile buffer to allow for expansion, and livestock grazing should cease in this area. A better solution would be to permanently close the Dry Ponds pasture and any other pasture that encompasses the Pando Clone, to livestock grazing. Further research is needed to determine thresholds at which mule deer and/or cattle density reduce aspen recruitment below self-sustaining levels, and the degree to which soil trampling by livestock contributes to sprout suppression and root damage in aspen clones.

Aspens and mule deer have been evolving together for thousands of years. In light of our findings that heavy cattle utilization of aspen understories in the unfenced portions of the Pando Clone and in neighboring aspen stands, and the likelihood that this heavy level of grazing could work synergistically with mule deer browsing to suppress aspen regeneration, previous hypotheses that mule deer browsing alone is responsible for the decline of Pando Clone sucker establishment appear highly unlikely. Taken together, the evidence brought forward thus far suggests that livestock grazing and/or trampling may be the critical factor(s) tipping browsing pressure over the threshold at which aspen regeneration begins to fail. Removing livestock grazing from the pastures south of Fish Lake and measuring suckering and recruitment for a period of 5 years would be a logical method to determine whether the primary driver of the failure to recruit is deer or livestock.

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forest policy

Implementing the 2012 Forest Planning Rule: Best Available Scientific Information in Forest Planning Assessments

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National forests and grasslands in the United States are governed by land and resource management plans that should be updated every 15 years to reflect changing social, economic, and environmental conditions and to address new priorities. A new forest planning rule finalized in 2012 introduces new planning approaches and requirements, and several forests have completed the forest assessment phase of their planning process. Using document analysis and interview data, we analyzed four completed forest assessments to gain insights into early forest planning efforts under the 2012 rule. We found that forest assessments address the required topics, although the organization and depth of treatment varies across cases; government sources and academic publishers are relied on most often as sources of scientific information; and approaches to best available scientific information rely on peer-reviewed information, agency technical reports and syntheses, and personal expertise and judgement.

Keywords: early adopter, expertise, US Forest Service

Management of the 154 national forests and 20 grasslands in the United States is governed by land and resource management plans (also called forest plans), as required by the National Forest Management Act of 1976 (NFMA; 16 U.S.C. 1604). The forest plan functions as a guiding document that outlines goals, objectives, and strategies for management of the unit. Periodically, the rule related to forest planning is revised to reflect societal changes, new approaches and technologies, and scientific discoveries. For many years the US Forest Service (USFS), which manages the system of national forests and grasslands, has operated under a planning rule finalized in 1982 (47 FR 43026) despite several efforts (2000, 2005, and 2008) to revise and improve the rule (Schultz et al. 2013). A new planning rule issued in April 2012 (77 FR 21161) introduces several significant changes, including a renewed emphasis on collaboration, improved transparency, and a strengthened role for public involvement throughout the planning process. Of interest for our study is the requirement to use the best available scientific information (BASI) to inform the assessment, plan revision decisions, and monitoring program.

To date, little research has addressed implementation of the 2012 planning rule. Schultz et al. (2013) examined approaches to wildlife conservation planning under the new rule, raising concerns regarding potential extirpation of species. Another study analyzed public participation processes in 12 national forests (University of Montana 2015), and Schembra (2013) explored the role of standards and guidelines and how they are used in planning activities. Forest planning under the 2012 rule consists of three phases (assessment, plan development, and monitoring). The assessment phase is important, as it assembles relevant scientific information that planners will rely on to make decisions on forest management in the plan development phase. Our study contributes to this growing body of knowledge by examining the assessment phase of the forest planning process.

Eight "early adopter" national forests, along with several other forests, are currently developing their forest plans using the 2012

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rule. These forests were designated as early adopters because they provide important benefits, had strong existing collaborative networks in place, and needed to revise their forest plans (USDA Forest Service 2012a). The eight early adopter forests are: Cibola (NM), Chugach (AK), El Yunque (PR), Nez Perce and Clearwater (ID), and three forests that are coordinating planning on a regional basis: Inyo, Sequoia, and Sierra (CA).

Although implementation is still in early stages, several of the early adopter forests have completed their forest assessments and draft forest plans, which presents an opportunity to study implementation of the planning process under the new rule. One forest (the Francis Marion in SC) has completed the full plan revision process as of this writing. We examined four forests that have completed their assessments, including three forests identified by the agency as early adopters and one forest that is keeping pace with this group. The study explored three questions: 1) What does the 2012 planning rule require regarding the structure, content, and process for forest assessments? 2) How have forests implemented the directives related to forest assessments under the 2012 planning rule? 3) How are forests approaching the requirement for the use of best available scientific information in their assessments?

Forest Planning under the 2012 Rule

The 2012 planning rule suggests an adaptive approach to forest planning, instructing managers to 1) *assess* forest conditions; 2) *revise or amend* plans if the assessment indicates a need for change; and 3) *monitor* plan implementation (36 CFR 219.5). The process is cyclical, with monitoring data feeding back into the assessment of conditions in the management unit (USDA Forest Service 2012b). During the assessment phase, planners are expected to "rapidly evaluate existing information about relevant ecological, economic, and social conditions, trends, and sustainability, and their relationship to the land management plan within the context of the broader landscape" (36 CFR 219.5(a)(1)). The second phase of the planning process is plan development, amendment, or revision, where planners use the results of the assessment to establish a need for change and generate planning alternatives (36 CFR 219.5(a)(2)), and the public has the greatest opportunity for input. The plan development phase includes environmental impact assessment, public input, and plan publication (36 CFR 219.5(a)(2)). The third phase (monitoring) is an opportunity to track and measure management effectiveness over time (36 CFR 219.5(a)(3)). The planning *process* under the 2012 rule is similar to the process specified under the 1982 rule, but differs in terms of the specific elements required for the *assessment* (2012 rule) and the *analysis of the management situation* (the assessment's counterpart in the 1982 rule).

We focused our study on the assessment phase of the planning process. The assessment phase is important because it requires the forest to assemble and synthesize the most recent, relevant, and highest-quality science on social, ecological, and economic conditions to inform the plan development. Not only does this provide planners an opportunity to evaluate changes in biophysical and socio-economic conditions based on the latest monitoring data, it also represents a chance to reflect on new concepts, models, and methods that result in new scientific information about the local forest environment. Under the 2012 planning rule, the assessment phase identifies existing conditions, trends, risks, uncertainties, and information gaps that are relevant to land and resource management issues in the unit (36 CFR 219.5–219.6). In the assessment phase, the planning unit is not required to generate new studies or information, but is expected to obtain pre-existing information that is publicly available or voluntarily provided (36 CFR 219.6). Information can come from government and nongovernment sources, and the rule instructs the Forest Supervisor to provide opportunities for stakeholders to provide information for the assessment (36 CFR 219.6). The primary product of the assessment phase is an assessment document that evaluates existing information for 15 specific topic areas (Figure 1). Although the general topic areas are mandated by the 2012 rule, the Forest Supervisor has discretion to determine the scope, scale, and timing of the assessment, assuming the other requirements in the planning rule are followed (36 CFR 219.6).

Role of Science in Natural Resource Management

Historically, natural resource management in the United States was guided by the idea of scientific management and Progressive-era approaches (Taylor 1896). In particular, Samuel Hays's "gospel of efficiency" relied on a rational and scientific method of making decisions through a single, central authority. The thought was to avoid conflict via a scientific approach to social and economic issues (Hays 1959, p. 267). The US Forest Service exemplifies the approach of technical rationality and empirical science as the basis for sound resource management practices (Wellman 1987; Kaufman 1960). Foresters and natural resource managers

Management and Policy Implications

Although implementation of the US Forest Service's 2012 planning rule is still in the early stages, several national forests have completed the assessment phase and moved on to the next phase of forest planning. Our analysis of forest assessments from several "early adopter" forests illustrates that forest planners are making serious efforts to address required topics and rely on the best available scientific information. Assessment reports were disproportionately heavy in science related to terrestrial and aquatic ecosystems, and more limited in treatment of infrastructure, land ownership and access patterns, cultural heritage, and areas of tribal importance. Ensuring that assessment teams include broad and diverse disciplinary experts will help address this challenge, recognizing that some forests may not have access to necessary disciplinary specialists. It is also possible that some of the topics (e.g., ecosystem services, tribal and cultural resources, land status and use patterns) simply do not have as much relevant and available information as other topics. Assessment teams may want to consider additional ways to interact with scientists and others to create functioning communities of practice related to science exchange for forest planning. In the same way, agency scientists may consider forging new and enduring relationships with planners and managers that could generate new science that is of immediate relevance. We found similarities across all forests in the most common approaches to identifying BASI in addition to other approaches such as data sharing meetings, a wiki review site, and requests for a science synthesis. Information from non-peer-reviewed sources was more difficult for planners to assess and evaluate. Sharing best practices, along with revised guidance for planning rule implementation, may help national forest planners improve the utility, efficiency, and quality of forest assessments.

- Terrestrial ecosystems, aquatic ecosystems, and watersheds
- Air, soil, and water resources and quality
- System drivers, including ecological processes, disturbance regimes, and stressors
- Baseline carbon stocks
- Threatened, endangered, proposed and candidate species; potential species of concern
- Social, cultural, and economic conditions
- Benefits people obtain from the planning area (ecosystem services)
- Multiple uses and their contributions to economies
- Recreation settings, opportunities, and access, and scenic character
- Renewable and nonrenewable energy and mineral resources
- Infrastructure (recreational facilities, transportation and utility corridors)
- Areas of tribal importance
- Cultural and historic resources and uses
- Land status, ownership, use, and access patterns
- Existing designated areas including wilderness and wild and scenic rivers; need and opportunity for new designations

Figure 1. Topics for forest plan assessments (36 CFR 219.6)

are expected to incorporate state-of-the-art scientific knowledge to manage public lands (Lachapelle et al. 2003). However, the role of science in natural resource decision-making has become much more complex (Mills and Clark 2001). Recent literature acknowledges that no important policy issue or decision is purely technical, that established practices are problematic, and that politics are unavoidable (Brunner et al. 2005). In spite of this, numerous policies reflect the scientific management paradigm in their calls for best available science.

In the United States, many policies and statutes contain references to best available science, including the Marine Mammal Protection Act of 1972, the Endangered Species Act of 1973, and the Safe Drinking Water Act of 1974. Despite references to the concept of best available science, these policies do not include specific definitions of its properties, standards, or practical application in the decision-making process (Doremus 2004; Smallwood et al. 1999), leading to different definitions of what it means. Ryder et al. (2010) identify attributes of best available science from published literature that span topics such as endangered species legislation, protection of conservation areas, forest management, water resource management, and ocean fisheries. The paper highlights the diversity of attributes assigned to best available science, and demonstrates that no single attribute is common to all studies, suggesting that best available science is context specific (Ryder et al. 2010). Moreover, as Lowell and Kelly (2016) observe, the ability to use best available science may be inhibited by institutional constraints within particular agencies limited by time or organizational capacity. Other literature has attempted to assign descriptors to the concept. For example, "best" often connotes scientific information with the greatest degree of excellence and authenticity based on sound logic (Moghissi et al. 2010), or that there is no better scientific information, and suggests the use of the most relevant and contemporary data and methods (National Research Council 2004). "Available" connotes scientific information that is accessible and attainable (Moghissi et al. 2010), or that decisions can be consistent with the scientific information that is available even though data gaps exist (National Research Council 2004). "Science or Scientific information" is defined as knowledge that emerges from a process of observation, identification, description, and testing of explanatory hypotheses about fundamental principles that govern cause-and-effect (National Research Council 2004). The National Research Council report includes guidelines for effectively using best available science, including concepts of relevance, inclusiveness, objectivity, transparency and openness, timeliness, and peer review. Finally, Charnley et al. (2017) analyzed a science synthesis for three national forests and suggest criteria for evaluating "best available *social* science," which may be different from the criteria used to evaluate best available biophysical science.

A key aspect of the 2012 planning rule is that it requires the planning process to draw on the best available scientific information (36 CFR 219.3). The preamble to the planning rule notes that there is a range of information that can be considered BASI, stating:

> In some circumstances, the BASI would be that which is developed using the scientific method, which includes clearly stated questions, well-designed investigations and logically analyzed results, documented clearly and subjected to peer review. However, in other circumstances the BASI for the matter under consideration may be information from analyses of data obtained from a local area, or studies to address a specific question in one area. In other circumstances, the BASI also could be the result of expert opinion, panel consensus, or observations, as long as the responsible official has a reasonable basis for relying on that scientific information as the best available. (77 FR 21192 [April 9, 2012])

Planning Directives are agency guidance documents that direct implementation of rules such as the 2012 planning rule, and directives for assessments are in Chapter 10 of the Land Management Planning Handbook (USDA Forest Service 2015a). The definition of BASI is contained in the "zero code" chapter of the handbook and specifies three primary criteria for determining BASI: accuracy, reliability, and relevance (FSH 1909.12.07.12), in addition to referencing the Data Quality Act (PL 106–554) for guidance on evaluating available information (Figure 2). Available is defined as information that currently exists in a form useful for the planning process without further data collection, modification, or validation (FSH 1909.07.01).

The directives also provide guidance regarding sources of scientific information. The sources mentioned in the guidance include peer-reviewed articles, scientific assessments, other scientific information (expert opinion, panel consensus, inventories, or observational data), data prepared and managed by the Forest Service

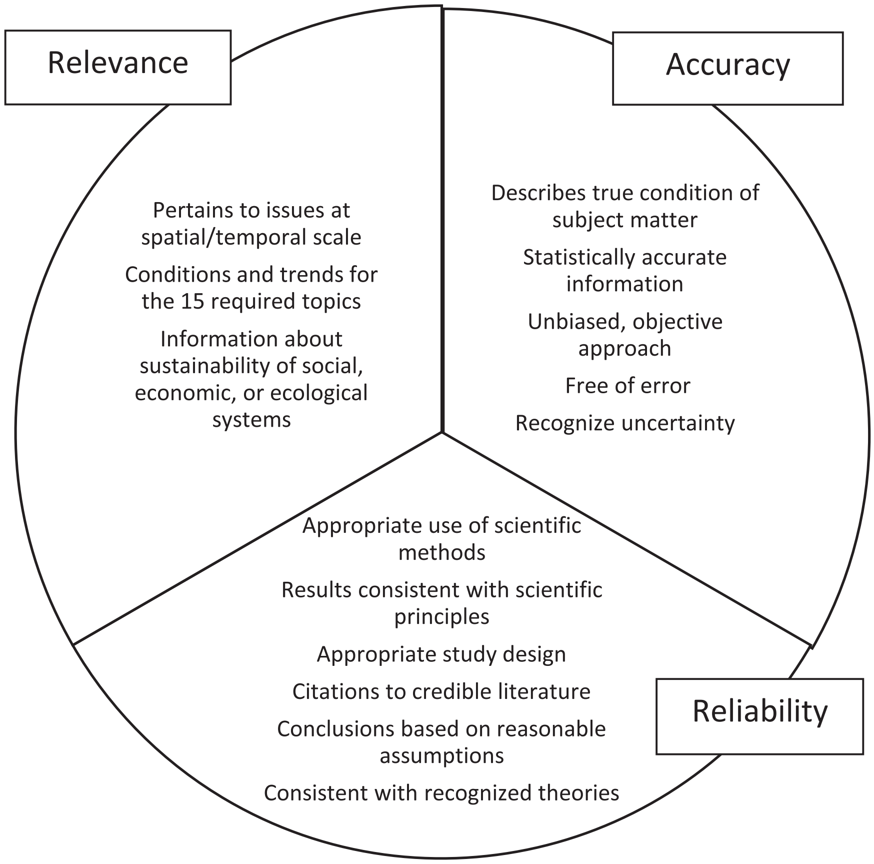

Figure 2. Criteria for determining best available scientific information (BASI). Source: Forest Service Handbook 1909.12.07.12

or other federal agencies, information prepared by universities, national research networks, and other reputable scientific organizations, and data or information from public and governmental participation (FSH 1909.12.07.13).

At the US Forest Service, two regional science synthesis efforts were initiated to assist forest planners in identifying BASI for their assessments. The first synthesis included the Sierra Nevada, southern Cascades, and Modoc plateau areas of California, and informed plan revisions on three national forests (Long et al. 2014). The second synthesis is currently underway as part of the Northwest Forest Plan area planning process, which covers 17 national forests and five Bureau of Land Management units across parts of the Cascade and coastal ranges of Washington, Oregon, and northern California. Once drafted, the synthesis report underwent independent third-party peer review, in addition to public review, and is currently under revision (Spies et al. 2017). Science synthesis efforts represent a noteworthy approach to developing BASI for use in forest assessments, creating a role for public engagement, and for employing a bioregional approach to assembling the latest science for use by multiple forests.

Methods

We used an exploratory case study approach to examine four national forest planning units that were revising their forest plans under the 2012 rule. Information on the USFS website helped us determine the planning status of each national forest as of spring 2015. The primary selection criterion was completion of the assessment process by spring 2015. We also strove to select national forests from different regions. Based on these criteria, we selected the Chugach National Forest (Alaska), Cibola National Forest (New Mexico), Inyo National Forest (California), and the Nantahala and Pisgah National Forests (North Carolina). Table 1 displays characteristics of each national forest planning unit in our sample.

Our research approach relied on content analysis of documents and interview data. We began by conducting a chapter-by-chapter analysis of each forest's assessment report to identify and characterize the information presented. We recorded page counts for each of the 15 assessment topics specified in the 2012 rule. In some cases, the chapters directly aligned with the required topics (Figure 1). In other cases, we had to make a more subjective characterization of the chapter contents. We also noted and analyzed any references to the use of best available science.

Second, as part of the document review, we analyzed data sources used in the assessment. For each assessment report, we identified all of the items cited in the reference section. We then coded each cited item according to the type of publishing entity and the type of document. Every cited item was placed in one category for each coding exercise. For each cited item, we determined the appropriate categories by examining the information in the citation entry and (when necessary) directly reviewing the item or gathering information on the publishing entity. We grouped publishing entities into five types: government; non-government; scientific, scholarly, or peer-reviewed; universities; and unknown or other (Table 2). This categorization approximates the rigor of scientific review, but there is overlap in categories. Most scholarly journals require a double-blind peer-review process, where reviewers and authors are

Table 1. Characteristics of national forests in the study.

Management unit(s)	Geography	Total acreage* (millions of acres)	Notes on use and resources	Designated early adopter?	Most recent previous plan revision	Notes on current plan revision
Chugach National Forest Alaska Region (R10)	Southcentral Alaska: major geographic areas are Cooper River, Prince William Sound, and eastern Kenai Peninsula	6.26	Subsistence, timber, recreation, mining. Human use concentrated in Kenai area. Very limited road coverage and use in other areas. Habitat for all 5 Pacific salmon species	Yes	2002	Managed by a planning team housed within unit
Cibola National Forest Southwest Region (R3)	West-Central New Mexico: Eight noncontiguous parcels organized around distinct mountainous areas known as "sky islands"	2.11	Recreation, timber, cultural heritage, range. Surrounding region experiencing population growth and demographic changes. Pinyon-juniper & ponderosa pine are predominate vegetation types	Yes	1985	Managed by a planning team housed within unit. Does not include 4 associated national grasslands
Inyo National Forest Pacific Southwest Region (R5)	Eastern California & West Nevada: Two noncontiguous parcels at intersection of Sierra Nevada, Great Basin, and Mojave Desert areas	2.07	Water supply, hydroelectricity, recreation, timber, range. Nearly 47% of total area is wilderness. Focus on wildland fire management. Substantial variation in vegetation type, habitat, and elevation	Yes	1988	One of three early adopters in R5. Coordination through a regional planning team, with separate planning teams for each unit. Each unit releases its own assessment & forest plan. Joint EIS for 3 units
Nantahala & Pisgah National Forests Southern Region (R8)	Western North Carolina: Blue Ridge region of Appalachian Mountains	2.48	Timber, recreation, cultural/historical heritage, water development. Located in Blue Ridge National Heritage Area. Hardwood forest with high species diversity	No	1987	Both units will use same revised plan. Managed by planning team housed at NF in NC headquarters

*Total acreage includes NFS-owned land and acreage under other ownership within each unit. Source: USDA Forest Service 2015b.

Table 2. Categories for coding type of publishing entity.

Publishing entity	Description of coding criteria
Government	Federal, tribal, state, or local governments in the United States; foreign governments; international intergovernmental groups such as the United Nations and affiliates. Includes peer-reviewed and non-peer-reviewed materials
Non-government	Materials not published by a government agency, university, or peer-reviewed entity. Includes businesses, consulting firms, and advocacy groups
Scientific scholarly or peer reviewed	Associations, societies, journal publishers, university presses, or other entities that produce peer-reviewed scientific or scholarly material
Universities	Materials from universities that may or may not be subject to rigorous academic peer review. Includes university or college departments, programs, laboratories, and centers, and theses and dissertations from universities
Unknown or other	News organizations or other undefined groups; disposition of publisher could not be determined

unknown to each other. University and government agency scientific documents often require peer review, but the level of rigor of the review may be variable. It was not possible to discern the level or type of peer review or scientific rigor for each category.

For the type of document, we sorted the references into 12 categories: academic book; non-academic book; conference proceeding; correspondence; database; scientific journal; news; technical report; statute or regulation; thesis or dissertation; website; and unknown (Table 3).

Our final data collection activity was qualitative interviewing with members of the planning teams at three of the forests in our study. We conducted nine semi-structured interviews (nine people in total; three interviews each from three forests). Unfortunately, we were not able to recruit interview participants from the Cibola planning effort. Potential interview participants were identified through the list of preparers included in each assessment document. Interviewees were subject matter experts who had contributed material to the assessment reports, along with planning staff officers or coordinators. Interview questions explored the overall structure of the assessment process, the role of the planning directives, the overall organization of the forests' plan revision efforts, and approaches to identification and use of best available science. Interviews were audio-recorded, transcribed, and analyzed using content analysis with a coding framework developed by the study team. Content analysis is a method that uses codes, or labels that assign meaning to descriptive or inferential data collected during a study (Miles et al. 2014). The codes are used to retrieve and organize similar data and aid the researcher in relating data to research questions, theoretical concepts, and themes (Araujo 1995; Miles et al. 2014).

Results

We present results of our analysis in three sections: 1) required topics; 2) sources and types of information; and 3) identifying and using BASI.

Required topics in the forest assessment

The number and percent of pages devoted to each required topic is presented in Table 4. We did not include introductory front matter in the page counts. A 0* entry means that the assessment report did not

Table 3. Categories for coding type of document.

Document type	Description of coding criteria
Academic book	An item printed, bound, distributed as a book, or released as an e-book by a peer-reviewed/scholarly entity
Non-academic book	An item printed, bound, distributed as a book, or released as an e-book by an entity whose primary orientation is not peer reviewed/scholarly
Conference proceeding	Papers, abstracts, and talks presented at a conference and published in a conference proceeding collection
Correspondence	Letters or emails written by individuals of any affiliation
Database	Raw data or data analysis tools/software; online databases
Scientific journal	A peer-reviewed article in a scholarly journal
News	Articles in newspapers (print or online) and news magazines
Technical report	Technical and research reports, white papers, policy papers, fact sheets, briefings
Statute, regulation, and planning documents	Federal, state, or local laws and rules; EISs; management plans; strategic plans
Thesis or dissertation	Advanced degree projects and papers
Website	One or more webpages on a non-database website, including encyclopedias with narrative entries
Unknown	The type of document could not be discerned

Table 4. Page counts and percentages of total pages for 15 required assessment topics.

		Number of pages (pct. of total pages in report)				Pct.
Topic #	Assessment topics (per 36 CFR 219.6)	Chugach	Cibola	Inyo	N&P	Avg.
1	Terrestrial ecosystems, aquatic ecosystems, and watersheds	66 (22.9%)	51.5 (11.2%)	38.5 (21.0%)	29 (15.7%)	**17.7**
2	Air, soil and water resources and quality	17 (5.9%)	88 (19.2%)	9 (4.9%)	19 (10.3%)	**10.1**
3	System drivers (processes, disturbance regimes, and stressors)	40 (13.9%)	21 (4.6%)	15 (8.2%)	7 (3.8%)	**7.6**
4	Baseline carbon stocks	7 (2.4%)	6 (1.3%)	4 (2.2%)	7 (3.8%)	**2.4**
5	Threatened, endangered, candidate species; potential species of conservation concern	12 (4.2%)	36 (7.9%)	24 (13.1%)	4 (2.2%)	**6.8**
6	Social, cultural, and economic conditions	21 (7.3%)	71 (15.5%)	14 (7.7%)	8 (4.3%)	**8.7**
7	Benefits obtained by people (ecosystem services)	49 (17.0%)	0* (0.0%)	2.5 (1.4%)	4 (2.2%)	**5.1**
8	Multiple uses and their contributions to economies	0* (0.0%)	26 (5.7%)	15 (8.2%)	17 (9.2%)	**5.8**
9	Recreation settings, opportunities, and access, and scenic character	29 (10.0%)	39 (8.5%)	15.5 (8.5%)	21 (11.4%)	**9.6**
10	Renewable and nonrenewable energy and mineral resources	17 (5.9%)	18 (3.9%)	3.5 (1.9%)	8 (4.3%)	**4.0**
11	Infrastructure	2 (0.7%)	12 (2.6%)	9.5 (5.2%)	10 (5.4%)	**3.5**
12	Areas of tribal importance	2 (0.7%)	13 (2.8%)	4.5 (2.5%)	3 (1.6%)	**1.9**
13	Cultural and historical resources and uses	3.5 (1.2%)	40 (8.7%)	7 (3.8%)	23 (12.4%)	**6.6**
14	Land status and ownership, use, and access patterns	8 (2.8%)	17 (3.7%)	7 (3.8%)	9 (4.9%)	**3.8**
15	Designated areas, potential/need for new designations	15 (5.2%)	20 (4.4%)	14 (7.7%)	16 (8.7%)	**6.5**
	TOTAL	**288.5**	**458.5**	**183**	**185**	**100**

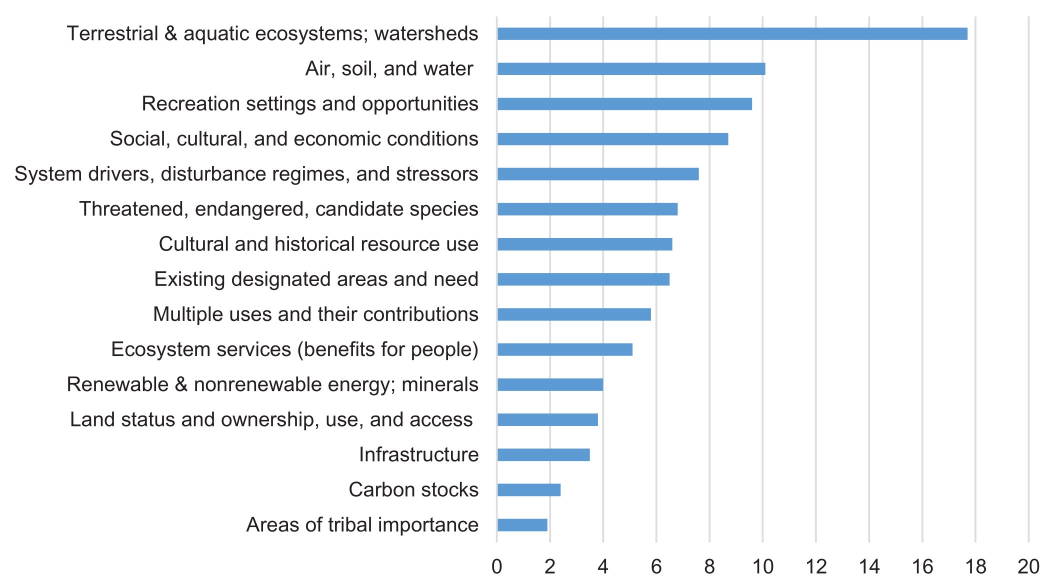

Figure 3. Average percentage of pages devoted to each topic in each forest assessment for all forests combined

have any pages that were specifically devoted to the topic, but references to the topic were instead interspersed throughout the report and it was too difficult to separate them from other topic page counts.

Two of the national forests (Inyo and Nantahala-Pisgah) published assessment reports that consisted of 15 chapters that directly reflected each of the required topics. Meanwhile, the Chugach and Cibola took a different approach; some of the chapter topics aligned with the topic requirements in the 2012 rule, but other required topics were broken up and distributed among multiple chapters. For example, the Chugach had one chapter for areas of tribal importance and one chapter for land status and ownership, but divided the terrestrial and aquatic ecosystems and watersheds

Table 5. Percent allocation of predominant topics among four forest assessments.

Rank	Chugach topics	Pct.	Cibola topics	Pct.	Inyo topics	Pct.	N&P topics	Pct.
1	Terrestrial and aquatic ecosystems	23%	Air, soil, and water	19%	Terrestrial and aquatic ecosystems	21%	Terrestrial and aquatic ecosystems	16%
2	Benefits obtained by people (ecosystem services)	17%	Social, cultural, and economic conditions	16%	Threatened and endangered species	13%	Cultural and historic resources	12%
3	System drivers, disturbance regimes, and stressors	14%	Terrestrial and aquatic ecosystems	11%	Recreation settings and opportunities	9%	Recreation settings and opportunities	11%
4	Recreation settings and opportunities	10%	Cultural and historic resources	9%	System drivers, disturbance regimes, and stressors	8%	Air, soil and water	10%
5	Social, cultural, and economic conditions	7%	Recreation settings and opportunities	9%	Multiple uses	8%	Multiple uses	9%
Total		71%		63%		59%		59%

into five chapters, one each for watersheds, fish, wetlands, vegetation, and wildlife, and these chapters were integrated with material discussing soils and carbon stocks. Two forests did not have any pages specifically devoted to one required topic each (benefits obtained by people for the Cibola, and multiple uses for the Chugach), but these subjects were still referenced in the context of the other topics.

For all four assessments combined, the required topic with the largest average percentage of pages was terrestrial and aquatic ecosystems and watersheds (17.7%), followed by air, soil, and water resources (10.1%) and recreation opportunities (9.6%) (Figure 3).

Terrestrial and aquatic ecosystems and watersheds comprised the largest section of the assessment for three of the four forests. Air, soil, and water was especially prominent for the Cibola National Forest, and all of the forest assessments covered recreation evenly. In contrast, the three required topics with the smallest page counts, on average, were areas of tribal importance (1.9%), carbon stocks (2.4%), and infrastructure (3.4%). Benefits obtained by people (ecosystem services) had the most variable coverage, with one of the shortest sections for three of the four forest assessments, but the second longest topic for the Chugach National Forest. In all four assessment documents, benefits obtained by people were mentioned throughout the document in sentences or paragraphs at too fine a scale for this analysis to count.

We found some variation among the forest assessments in terms of the extent to which a forest focused on a particular topic (Table 5).

For the Chugach National Forest, the top five topics comprised more than 70% of the assessment, with the bulk emphasizing terrestrial and aquatic ecosystems, which reflects the importance of salmon habitat. The Chugach was the only forest to emphasize ecosystem services as a predominant framework to capture benefits obtained by people. However, other forests may have captured this topic under the category of multiple uses. Disturbance regimes (fire and invasive species) were also important for the Chugach. The Cibola National Forest was unique in their emphasis on air, soil, and water as well as social, cultural, and economic conditions and cultural and historic sites. Because water access is very important in the southwest, the predominance of this topic is not surprising. For the Inyo National Forest, the topic of threatened and endangered species was prominent, while topics related to recreation and disturbance regimes (fire, invasive species, and other ecosystem stressors) were also important. Meanwhile, cultural and historical resources were prominent in the Nantahala and Pisgah National Forests, along with recreation.

Although the 2012 rule provides a list of 15 distinct required topics, these topics overlap and are not discussed in complete isolation from one another. As we found in our analysis, it is difficult to discuss multiple uses without also discussing benefits obtained by people; air, soil, and water resources; recreation; and terrestrial and aquatic ecosystems and watersheds. In our analysis, we often found that an assessment chapter devoted to a required topic also contained information that closely resembled material discussed elsewhere. In particular, we found the chapters on multiple uses and benefits obtained by people to be largely redundant, given the other topics that were also included in the report.

Sources and types of information in the forest assessment

To understand the sources and types of information used in the assessments, we conducted a systematic examination and tally of citations by publication source and type. Overall, government sources were the most commonly cited information source (51.8%), followed by scientific scholarly publications (30.7%) (Table 6).

Table 6. Citations based on information source for forest assessments.

	Count (Percent)				
Publishing entity	Chugach	Cibola	Inyo	Nantahala & Pisgah	TOTAL (Mean)
Government	239 (53.6%)	159 (49.8%)	131 (49.8%)	109 (54.0%)	**638 (51.8%)**
Scientific scholarly or peer reviewed	155 (34.8%)	82 (25.7%)	82 (31.2%)	63 (31.2%)	**382 (30.7%)**
Non-government	21 (4.7%)	39 (12.2%)	24 (9.1%)	18 (8.9%)	**102 (8.7%)**
Universities	30 (6.7%)	39 (12.2%)	19 (7.2%)	11 (5.5%)	**99 (7.9%)**
Unknown or other	1 (0.2%)	0 (0%)	7 (2.7%)	1 (0.5%)	**9 (0.9%)**
TOTAL	446	319	263	202	**1230**

Table 7. Citations based on document type for forest assessments.

Document type	Count (Percent)				
	Chugach	Cibola	Inyo	Nantahala & Pisgah	**TOTAL**
Technical report	174 (39.0%)	121 (37.9%)	108 (41.1%)	73 (36.1%)	**476 (38.5%)**
Scientific journal article	129 (28.9%)	47 (14.7%)	63 (24.0%)	48 (23.8%)	**287 (22.8%)**
Academic book	28 (6.3%)	36 (11.3%)	20 (7.6%)	15 (7.4%)	**99 (8.2%)**
Statute, regulation, or planning document	43 (9.6%)	26 (8.2%)	23 (8.8%)	12 (5.9%)	**104 (8.1%)**
Website	33 (7.4%)	42 (13.2%)	3 (1.1%)	13 (6.4%)	**91 (7.0%)**
Database	17 (3.8%)	25 (7.8%)	17 (6.5%)	18 (8.9%)	**77 (6.8%)**
Conference proceeding	10 (2.2%)	3 (0.9%)	6 (2.3%)	18 (8.9%)	**37 (3.6%)**
Non-academic book	4 (0.9%)	9 (2.8%)	10 (3.8%)	0 (0.0%)	**23 (1.9%)**
Correspondence	0 (0.0%)	7 (2.2%)	5 (1.9%)	4 (2.0%)	**16 (1.5%)**
Thesis or dissertation	8 (1.8%)	2 (0.6%)	2 (0.8%)	1 (0.5%)	**13 (0.9%)**
News	0 (0.0%)	1 (0.3%)	3 (1.1%)	0 (0.0%)	**4 (0.4%)**
Unknown	0 (0.0%)	0 (0.0%)	3 (1.1%)	0 (0.0%)	**3 (0.3%)**
TOTAL	446 (100.0%)	319 (100.0%)	263 (100.0%)	202 (100.0%)	**1230 (100.0%)**

A large portion of the government sources included US Forest Service publications (average of 28%), which were more commonly cited than other federal government sources (average of 12%) or state and local governments (average of 11%). Some variation exists among the forests in our sample, but the trends were consistent in terms of reliance on government sources and scholarly peer-reviewed publishers for the majority of citations (82.5% combined average for both categories). The Chugach relied to a greater degree on scholarly publications than other forests. The Cibola had the highest proportion from non-governmental organizations and trade groups (12.2%). The Inyo and the Nantahala and Pisgah mirrored the group average.

Next, we explored citations by the type of document referenced. We found that technical reports were the most common type of document cited in the assessments, with an average of 38.5% (Table 7).

The technical report classification is broad and includes technical and scientific reports, policy briefings, white papers, and other types of information (sometimes referred to as gray literature). All four forests were consistent in the ratio of technical reports cited. The second most common document type was the scientific journal article, with an average of 23%, although the Cibola assessment featured far fewer than the other forests. All of the forests cited a wide variety of regulations, statutes, and planning documents, (e.g., water quality regulations, county comprehensive plans, environmental impact statements, state resource management plans, and forest plans). The Cibola assessment featured the greatest variety of document types, relying on websites and academic books more than the other forests. The Nantahala and Pisgah assessment relied more heavily on conference proceedings. The least commonly cited document types, on average, were news articles (0.4%), theses or dissertations (0.9%), and correspondence (1.5%). Although there is a separate category for websites, documents in many of the other categories were readily available online.

Table 8. Approaches to identifying and using BASI from interview data.

BASI approach	Chugach	Nantahala/ Pisgah	Inyo
Literature review (e.g. Google Scholar for scholarly literature)	x	x	x
Forest Service reports, monitoring data	x	x	x
Personal expertise/training/judgement	x	x	x
Existing dataset/database	x		x
Nearby Forest Service research station		x	x
Nearby university		x	
Host data sharing meeting (partners and stakeholders)		x	
Meet with scientists		x	
Post draft documents on wiki site for public review/editing			x
Other public review opportunity		x	
Gray ("non-peer-reviewed") literature, traditional knowledge			x

Identifying and using best available scientific information in the forest assessment

In interviews, respondents were asked how they identified and obtained BASI for their assessment. Table 8 displays the different approaches used by three of the four forests.

Literature reviews and searches, Forest Service reports and datasets, and personal scientific expertise were mentioned by all nine respondents as primary ways that they identified and obtained BASI. Literature reviews focused on identifying peer-reviewed journals, conference proceedings, or agency reports. Existing datasets and nearby Forest Service research stations and universities were also relied upon. The Sierra Nevada science synthesis effort, which informed the Inyo National Forest assessment, took nearly 18 months to complete (Long et al. 2014). The Inyo also posted draft documents on a wiki site for public review and editing. All nine interviewees stated that their assessment team used the Draft Planning Directives, but also mentioned that the directives were not clear, save for the focus on organizing around the 15 topics. No respondent mentioned specific guidance beyond the draft directives on how to identify BASI. The final directives do specifically address the definition of BASI, as discussed above (Figure 2). Gray literature and traditional knowledge presented challenges, as it at times conflicted with peer-reviewed information. Two respondents mentioned that they wanted to incorporate this type of information, but were unsure how to do so.

Assessments must document what information was determined to be BASI, explain the basis for that determination, and explain how the information was applied to the issues considered (36 CFR

219.3). Our analysis of the assessment documents reveals that all documents discuss the use of high-quality and valid scientific information, citing criteria such as clearly defined and well-developed methodology; standardized methodology; logical conclusions; and reasonable inferences (Chugach National Forest 2014; Inyo National Forest 2014; Nantahala and Pisgah National Forests 2014; Cibola National Forest and National Grasslands 2015). The assessments for all forests mention their reliance on information relevant to their specific forests and issues. Only the Nantahala-Pisgah assessment presented a hierarchy of information sources, with peer-reviewed journal articles the highest, followed by government documents and reports, monitoring datasets, theses and dissertations from universities, and expert opinion where facts were not known through the other sources.

Discussion

The 2012 forest planning rule requires that each national forest or grassland conduct a scientific assessment to guide plan development. We found that assessment reports were disproportionately heavy in science related to terrestrial and aquatic ecosystems, and more limited in treatment of infrastructure, land ownership and access patterns, cultural heritage, and areas of tribal importance. Recreation was the only topic to receive consistent attention across all four forests, although the topic was overshadowed by terrestrial and aquatic ecosystems. We may only speculate about why terrestrial and aquatic ecosystem information was the most prevalent in all four forests, but it is consistent with agency administrative hiring practices since the 1980s that have emphasized recruitment of ecologists, biologists, and other biophysical scientists, compared to social scientists, for example (Thomas and Mohai 1995). The abundance of agency specialists in these topic areas may reinforce the relative importance of terrestrial and aquatic ecosystems compared to other topic areas, such as recreation, social science, or cultural resource management. This has been confirmed by a national assessment of interdisciplinary planning team composition (Cerveny et al. 2011). Ensuring that assessment teams include broad and diverse disciplinary experts will help address this challenge, recognizing that some forests may not have access to necessary disciplinary specialists. It is also possible that some of the topics (e.g., ecosystem services, tribal and cultural resources, land status and use patterns) simply do not have as much relevant and available information as other topics.

The benefits obtained by people (ecosystem services) topic received little or no explicit coverage in all but one assessment. The limited coverage of ecosystem services may make sense because it was not even considered an area of research until the late 1990s, so there would be less existing information on certain important ecosystem service topics (e.g., pollination, stormwater attenuation, medicinal resources, and spiritual and historical significance) compared to recreation, threatened and endangered species, and other traditional assessment topics (Blahna et al. 2017). Previously, "forest benefits to people" were considered elements of "multiple use" and planners might have addressed these benefits under the "multiple use" topic. Ecosystem services (ES) are often categorized into four classes: provisioning, regulating, cultural, and supporting. Timber, recreation, wildlife, and other traditional forest planning topics all fall into one of these four classes. Another reason for lack of coverage of ecosystem services may be that planners could not differentiate the normal assessment topics from the ecosystem service classes. Efforts to help planning team members understand ecosystem services approaches and how they can be used to inform the planning process may be warranted, and the rule's current requirement for only using existing data in assessments may need to be revisited (Blahna et al. 2017). For example, implementation teams working on ecosystem services may consider the benefits of providing specific tools, frameworks, and guidelines for integrating ecosystem services models into the forest planning process. In addition, critical issues and topics (e.g., newly listed threatened or endangered species, or changing recreation behaviors) that forest plans need to address may change from one planning cycle to the next.

The specific required topics may not be universally appropriate for every planning unit. Planners felt obligated to address all 15 topics, but the lack of coverage for some topics suggests that the topic was not deemed relevant or meaningful for their plan, there was no available data on the topic, or it was unclear how the topics could be covered. Variability in application of the directives, and acknowledgment of local context and conditions, is consistent with the overall Forest Service approach toward decentralized decision-making (Kaufman 1960; Tipple and Wellman 1991; Koontz 2007) and localized interpretation by planning teams, similar to "street-level" bureaucrats who create de facto policy through everyday practice (Sabatier et al. 1995; Lipsky 2010; Trusty and Cerveny 2012). Kaufman (1960) observes the traditional Forest Service practice of maintaining control of heterogeneous and geographically dispersed management units by issuing centralized directives that provide parameters (or "side boards") within which line officers have some leeway to make decisions. This tendency toward uniformity and "pre-formed" decisions may result in some inefficiencies and omissions. The implied obligation to cover all 15 topics may have resulted in some assessments that distract from the most important management issues for the unit. This will be especially important during the next stage of planning—revision or amendment—where the assessment data will be used to analyze different management scenarios. Approaches for identifying and analyzing the most relevant assessment data that address the key environmental problems or social conflicts that confront each planning unit will be needed (Blahna et al. 2017). This is especially important for topics like human benefits (ecosystem services) and multiple uses, which cut across all of the other topical areas and are not as easily categorized in assessments. Recent efforts to engage the public in science synthesis efforts in support of forest planning suggest that there may be an important role for the public to help prioritize forest assessment topics.

The most common sources of information were government sources, followed by scholarly academic sources. Many of the agency sources were peer-reviewed scientific studies, which appear to be especially useful because of the topical specificity or geographic focus (relevance). Although not all technical reports are peer reviewed, they may be more accessible and usable compared to scholarly journal articles, which may require planning team members to interpret the findings and make inferences for relevance to local conditions. This finding is consistent with previous research examining the information needs and sources of Forest Service fire managers (Ryan and Cerveny 2011) and recreation managers (Ryan and Cerveny 2010). Fire managers relied heavily on agency information sources. Although managers in the study noted the availability of high-quality, relevant information, they faced significant barriers in terms of time, funding,

and personnel to access and use that information. Similarly, recreation managers also relied on agency information sources, but indicated strong preferences for enhanced interactions with agency scientists, including collaborative research, conferences, and a desire for agency researchers to reach out more directly to managers to ensure their research was relevant and useful. With regard to forest assessments, engagement with scientists is particularly important for topics where little research is available. Assessment teams may want to consider additional ways to interact with scientists and others to create functioning communities of practice related to science exchange for forest planning. In the same way, agency scientists may consider forging new and enduring relationships with planners and managers that could generate new science that is of immediate relevance.

The 2012 planning rule and its directives provide criteria for BASI, and we found similarities across all forests in the most common approaches to identifying BASI, in addition to other approaches, such as data sharing meetings, a wiki review site, and requests for a science synthesis. Information from non-peer-reviewed sources was more difficult for planners to assess and evaluate, and it is not clear how this information was incorporated into each assessment. Teams may not have the capacity to separately evaluate and assess the many different types and sources of information, and so they rely on hierarchical ranking approaches (peer-reviewed sources being highest rank) to streamline the evaluation. Planning teams clearly value peer-reviewed and agency-generated information, and it may be that they are simply identifying information that is "available" and using the "best" of that based on their judgments. This may result in situations where the science expertise on each team could influence BASI decisions. As discussed above, consideration of the makeup and membership of the assessment team is important here, as well as increased transparency regarding the process for determining science relevance and quality.

Conclusion

Implementation of the US Forest Service 2012 planning rule is still in its early stages. Our study illustrates that forest planners use a variety of approaches to address required topics, and do rely on BASI as they develop their forest assessments. While each national forest assessment included the 15 required topics, we found considerable variation in coverage, which suggests that planners may emphasize topics most relevant to their forest, or that variation exists in terms of what science or planning team expertise is available or deemed desirable. The predominance of science related to terrestrial and aquatic ecosystems in the assessments compared to other topics warrants further inquiry in order to learn whether this asymmetry is based on policy, availability of information, existing expertise, or other factors. Efforts to include the public in the process of prioritizing topics for the assessments could also be evaluated. The reliance on government sources for scientific information suggests that agency-supported science is either more accessible or more relevant to the planning team. It also suggests that there may be benefits to bolstering "communities of practice" for key topical areas covered by forest assessments that bring together university and agency scientists with managers.

The appearance of science in an assessment report is important, but the actual *use* of science in planning may be more important. Although our findings are not generalizable to all national forests, they do provide an understanding of plan assessment activities for those in the early phases of forest planning, whose efforts are likely to inform and influence other national forests. Our goal was to provide an early glimpse of plan revision efforts in order to highlight important lessons learned and create a foundation for future research. For example, do planners find that the required topics provide useful guidance for developing their assessments? How can planners become more confident in knowing what BASI is, and how to identify and use it? Is additional guidance needed for incorporation of traditional knowledge and other information? Of particular interest is whether the "science synthesis" information is useful to forest planners in addressing their forest assessment needs, given the significant agency resources devoted to developing science syntheses. Finally, how is information from the assessment used in forest plan revision (development and selection of management options) and monitoring efforts? While draft environmental impact assessment (EIS) reports are available in various stages, as of this writing only one final Record of Decision (ROD) has been issued for a forest plan undergoing revision under the 2012 rule. Thus, it remains to be seen how scientific information will be incorporated in development of alternatives, impact statements, and final management decisions.

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LETTER

How will climate change affect wildland fire severity in the western US?

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Abstract

Fire regime characteristics in North America are expected to change over the next several decades as a result of anthropogenic climate change. Although some fire regime characteristics (e.g., area burned and fire season length) are relatively well-studied in the context of a changing climate, fire severity has received less attention. In this study, we used observed data from 1984 to 2012 for the western United States (US) to build a statistical model of fire severity as a function of climate. We then applied this model to several ($n = 20$) climate change projections representing mid-century (2040–2069) conditions under the RCP 8.5 scenario. Model predictions suggest widespread reduction in fire severity for large portions of the western US. However, our model implicitly incorporates climate-induced changes in vegetation type, fuel load, and fire frequency. As such, our predictions are best interpreted as a *potential* reduction in fire severity, a potential that may not be realized due human-induced disequilibrium between plant communities and climate. Consequently, to realize the reductions in fire severity predicted in this study, land managers in the western US could facilitate the transition of plant communities towards a state of equilibrium with the emerging climate through means such as active restoration treatments (e.g., mechanical thinning and prescribed fire) and passive restoration strategies like managed natural fire (under suitable weather conditions). Resisting changes in vegetation composition and fuel load via activities such as aggressive fire suppression will amplify disequilibrium conditions and will likely result in increased fire severity in future decades because fuel loads will increase as the climate warms and fire danger becomes more extreme. The results of our study provide insights to the pros and cons of resisting or facilitating change in vegetation composition and fuel load in the context of a changing climate.

Introduction

Fire regimes in North America are expected to change over the next several decades as a result of anthropogenic climate change (Dale *et al* 2001). Fire activity (i.e., annual area burned and fire frequency) is expected to increase in many regions (Krawchuk *et al* 2009, Littell *et al* 2010) and new research shows that fire seasons are now starting earlier and ending later compared to previous decades (Jolly *et al* 2015). However, the effect of climate change on one very important fire regime characteristic—*fire severity*—is not well-studied or understood (Flannigan *et al* 2009, Hessl 2011). In the context of this paper, we define severity as the degree of fire-induced change to vegetation and soils one year post-fire (Key and Benson 2006, Miller and Thode 2007). For example, a stand-replacing fire in upper-elevation conifer forest is considered high severity because the site has drastically changed one year post-fire compared to pre-fire conditions, whereas a surface fire in a grass-dominated ecosystem is considered low severity because the vegetation is nearly fully recovered one-year post fire.

The severity at which a site burns influences vegetation response and successional trajectory (Barrett *et al* 2011), faunal response (Smucker *et al* 2005), carbon emissions (Ghimire *et al* 2012), and erosion rates and sedimentation (Benavides-Solorio and MacDonald 2005). Furthermore, human safety and infrastructure are influenced by the severity at which a site burns (Miller and Ager 2013), and management responses to fire and allocation of firefighting resources are also influenced by the expected fire severity (e.g., Calkin *et al* 2011). As such, there is a need to better understand how fire severity will respond to a changing climate (e.g., Miller *et al* 2009).

At fine temporal scales, fire severity depends on factors that are highly variable over time, such as fire spread rate and direction (e.g., heading versus backing fire) and weather (Finney 2005, Birch *et al* 2015). At broader temporal scales, however, climate (in terms of climatic normals) is a major influence through its interactive effect on productivity (and hence amount of biomass) and moisture availability (i.e., wet versus dry ecosystems) (Parks *et al* 2014b, Whitman *et al* 2015). Consequently, because fire regimes are intrinsically defined by the characteristics of fires that occur over extended periods of time (years to centuries) (Morgan *et al* 2001), evaluations of fire severity over gradients of observed and predicted climatic normals allows for a formal assessment of how fire severity may respond to climate change.

We seek to quantify how fire severity in the contiguous western United States (US) (hereafter the 'western US') may respond to climate change. We use statistical relationships between observed climatic normals and fire severity (Parks *et al* 2014b, Kane *et al* 2015) to conduct a formal evaluation of future fire severity patterns. Because the relationship between climate and fire regimes is known to be weak in areas of high human impact (Parks *et al* 2014b), we used data from areas with low anthropogenic influence to build a statistical model of fire severity as a function of climatic normals over the 1984–2012 time period. We then predicted contemporary (1984–2012) and future (mid-century; 2040–2069) fire severity using climate data from numerous global climate models (GCMs) for the western US. As far as we know, this study is the first to examine how fire severity may respond to a changing climate over such a broad spatial extent. The results of this study will advance our understanding of fire regimes in the western US in the context of a changing climate and will assist policy makers and land managers to better manage for resilient landscapes.

Methods

Consistent with major fire severity mapping efforts (Key and Benson 2006, Eidenshink *et al* 2007), we define fire severity as the degree of fire-induced change to vegetation and soils. We built a statistical model of fire severity as a function of climate by first partitioning our study area (the western US; figures 1(a) and (b)) into 500 km^2 hexagonal polygons (i.e., 'hexels'). Within each hexel, we summarized fire severity using the delta normalized burn ratio (dNBR) (Key and Benson 2006), a satellite index (resolution: 30 m) that differences pre- and post-fire Landsat TM, ETM+, and OLI images and has a high correspondence to field-based measures of severity such as the composite burn index (CBI; $R^2 \geqslant 0.65$) (van Wagtendonk *et al* 2004, Parks *et al* 2014a). The CBI is a post-fire assessment in which individual rating factors in each of several vertically arranged strata (soil and rock, litter and surface fuels, low herbs and shrubs, tall shrubs, and trees) are assessed on a continuous 0–3 scale indicating the magnitude of fire effects. A rating of 0 reflects no change due to fire, whereas 3 reflects the highest degree of change. Factors assessed include soil char, surface fuel consumption, vegetation mortality, and scorching of trees. Ratings are averaged for each stratum and then across all strata to arrive at an overall CBI rating for an entire plot. The CBI indicates that, as dNBR values increase, there is generally an increase in char and scorched/blackened vegetation and a decrease in moisture content and vegetative cover (Key and Benson 2006). Measurements of fire severity (dNBR and CBI) are generally conducted one year after fire, so any regrowth that occurs within one year will result in reduced severity compared to assessments conducted immediately post-fire; this is particularly relevant for species that recover quickly after fire (e.g., resprouting shrubs, grasses).

Fire severity (i.e., dNBR) data were obtained from the Monitoring Trends in Burn Severity project (Eidenshink *et al* 2007) for all fires ⩾400 ha for the 1984–2012 time period. Raw dNBR values obtained from MTBS were adjusted using the 'dNBR offset' (Key 2006), which accounts for differences due to phenology or precipitation between the pre- and post-fire images by subtracting the average dNBR of pixels outside the burn perimeter. This adjustment can be important when comparing severity among fires (Parks *et al* 2014a). A mean dNBR was calculated using all pixels of all fires that intersected each 500 km^2 hexel; pixels classified as nonfuel were excluded in the calculation of the mean. We square-root transformed mean dNBR values to linearize the relationship to the CBI (figure S1).

We summarized climate normals within each hexel using five variables with known links to fire regimes (e.g., Littell and Gwozdz 2011, Abatzoglou and Kolden 2013, Parks *et al* 2015b): actual evapotranspiration (AET), water deficit (WD), annual

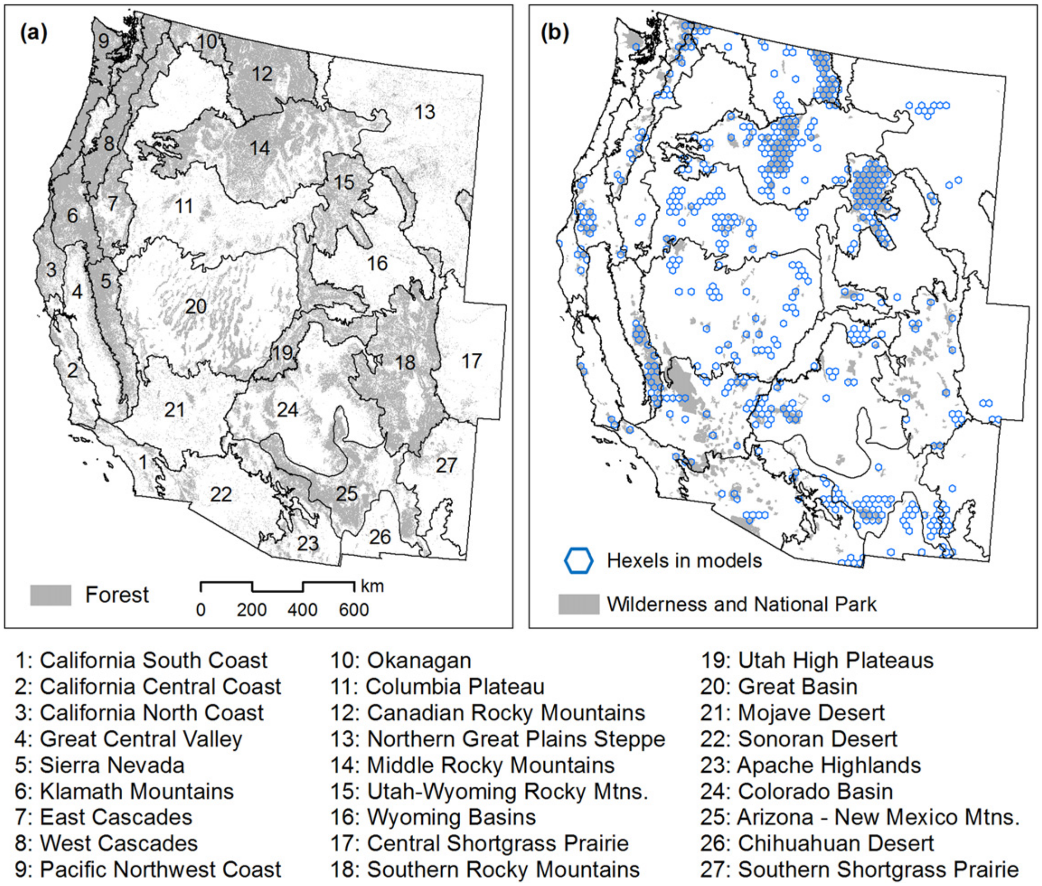

Figure 1. Study area of the western US for which we predicted changes in fire severity under a future climate. Map showing ecoregion boundaries (The Nature Conservancy 2009) and forested areas (in gray) (a) and showing designated wilderness areas and national parks (in gray) as well as the 544 hexels (in blue) used to build the models of fire severity as a function of climate (b). Ecoregion names and boundaries provided for context.

precipitation (PPT), soil moisture (SMO), and snow water equivalent (SWE). Gridded monthly temperature and PPT data were obtained from the parameter-elevation regression on independent slopes model (PRISM; Daly *et al* 2002), which uses weather station data and physiographic factors to map climate at a spatial resolution of ~800 m. In addition, daily and sub-daily surface meteorological variables (~4 km resolution) describing temperature, humidity, winds, solar radiation, and precipitation were produced following Abatzoglou (2013). These data were collectively used to compute climatic water balance following Dobrowski *et al* (2013) to estimate AET, SWE, SMO, and WD. This water balance model operates on a monthly time-step and accounts for atmospheric demand (via the Penman–Monteith equation), soil water storage, and includes the effect of temperature and radiation on snow hydrology via a snow melt model. Each variable was averaged within each hexel for the years 1984–2012, thereby matching the years of the fire severity data. We similarly summarized these five climate variables representing mid-21st century (2040–2069) conditions using 20 global climate models (GCMs) for the RCP8.5 emissions scenario (table S1). These tables were statistically downscaled to the same grid as observed data using the multivariate adapted constructed analogs approach (Abatzoglou and Brown 2012).

Because the relationship between climate and fire is weaker in landscapes that are highly influenced by humans (Parks *et al* 2014b), we built our model using data from a subset of hexels with low human influence (figure 1(b)). We selected only those hexels that were comprised of at least 50% designated wilderness or national park or had an average 'human footprint' (Leu *et al* 2008) $\leqslant$ 2.5 (on a scale of 1–10). We further limited our dataset to include only those hexels with at least 400 ha of total burned area from 1984 to 2012. These selection criteria resulted in 544 hexels that, despite representing a small proportion of our study area (8.7%), are climatically representative of much of the western US, with the notable exception of the wet regions of the Pacific Northwest (figure S2).

Using data from the subset of 544 hexels, we modeled fire severity (dNBR) as a function of contemporary climate (1984–2012) using boosted regression trees (BRT) ('gbm' package) in the R statistical environment (R Development Core Team 2007). BRT is a nonparametric machine-learning approach that does not require *a priori* model specification or test of hypothesis (De'ath 2007). The BRT algorithm fits the best possible model to the data structure, including complex interactions among variables. It does so by building a large number of regression trees, whereby, through a forward stage-wise model-fitting process, each term represents a small tree built on the weighted residuals of the previous tree. The stage-wise procedure reduces bias, whereas variance is decreased through model averaging. The BRT method also employs 'bagging', the use of a random subset of samples, which typically improves model predictions. Comparisons to other modeling techniques indicate that BRT models consistently produce robust predictive estimates (Elith *et al* 2006). We followed the recommendations of Elith *et al* (2008) for selecting BRT options; we set the bagging fraction to 0.5, learning rate to 0.005, and tree complexity to three. We used a custom script from Elith *et al* (2008) to determine the necessary number of trees, thereby reducing the potential for overfitting. We evaluated the model fit using the (a) correlation between predicted and observed fire severity and (b) ten-fold cross-validated correlation between predicted and observed fire severity.

We used the model to predict contemporary (1984–2012) fire severity (dNBR) for all hexels in the western US. However, interpreting dNBR and changes in dNBR under a changing climate is challenging because dNBR units have no direct ecological interpretation. As such, we rescaled these predictions to correspond to the ecologically relevant composite burn index (hereafter 'inferred CBI') that ranges from 0 to 3 (Key and Benson 2006): the lowest predicted severity was given an inferred CBI of 0.1, which is the threshold for 'unchanged' (Miller and Thode 2007), and the highest predicted severity was given an inferred CBI of 3.0. We were then able to infer the CBI of all remaining predictions because the square-root transformation of dNBR linearized the relationship to CBI (figure S1). Consequently, we generated a map representing the inferred CBI for the western US under contemporary climate.

We then predicted fire severity for the mid-21st century (2040–2069) as projected by each GCM using the BRT model. We inferred CBI as previously described using the linear relationship between dNBR and CBI of the observed predictions to make the inferences. Note that the predictions for all hexels in the western US were 'clamped' to avoid predicting outside of the observed range of severity values; all predictions >3 and <0.1 were given values of 3.0 and 0.1, respectively. For each BRT prediction (one for each GCM), we then quantified the predicted change in fire severity by subtracting the inferred CBI of contemporary climate from the inferred CBI of mid-21st century

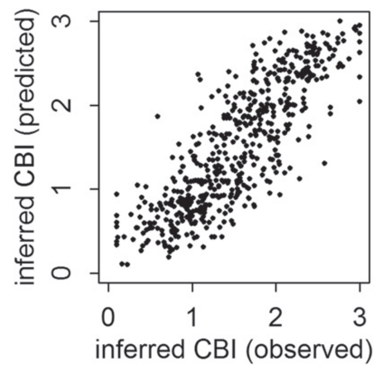

Figure 2. Plot showing predicted versus observed inferred CBI. Inferred CBI of observed data was calculated using similar methods to that of the inferred CBI of predicted data but were bounded by the 0.5 and 99.5 percentile dNBR to ensure extreme values did not overly influence this interpretation. $R^2 = 0.64$.

climate. We summarized the results by generating maps of (1) contemporary fire severity, (2) predicted mid-21st century fire severity (averaged over 20 GCMs) and, (3) the average change (for all 20 GCMs) in fire severity (i.e., inferred CBI) between contemporary and mid-century time periods.

Results

The correlation between predicted and observed dNBR among the 544 hexels was 0.80 and the cross-validated correlation was 0.72. A plot showing predicted versus observed inferred CBI also indicates a good fit ($R^2 = 0.64$; figure 2). Water deficit and PPT were the most influential variables (relative influence = 41.5% and 29.8%, respectively) (figure 3(a)). Fire severity generally decreased with WD and increased with PPT (figures 3(b) and (c)). The map of predicted contemporary (1984–2012) fire severity indicates that cooler and wetter forested ecoregions (e.g., Pacific Northwest, Northern Rocky Mountains, and Southern Rocky Mountains) experience more high severity fire (inferred CBI $\geqslant$ 2.25) compared to warmer and drier forested ecoregions (e.g., Arizona - New Mexico Mountains) (figure 4(a)). Non-forested ecoregions for the most part experience fairly low fire severity (inferred CBI < 1.25). The map of mid-21st century fire severity shows a similar pattern in that the cooler and/or wetter regions generally have higher severity than elsewhere (figure 4(b)), but for the most part, fire severity is predicted to decrease over much of the western US (figure 4(c)). The results of current, future, and predicted changes in fire severity are strikingly similar when we measured fire severity using a relativized metric (the relativized burn ratio; RBR) (Parks *et al* 2014a) instead of dNBR (figure S3).

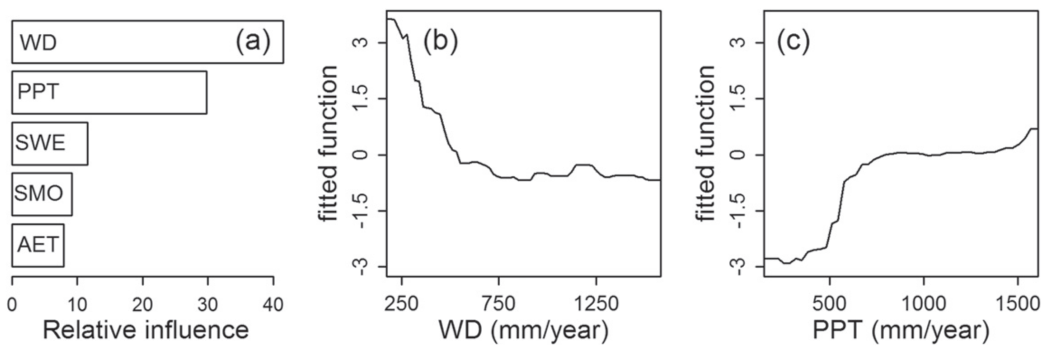

Figure 3. Variable importance in the BRT model (a) and partial dependence plots showing the relationship between dNBR and the two most influential variables (WD and PPT) (b), (c). Note that the partial dependence plots do not reflect interactions between variables and therefore simplify the relationships.

Discussion

Our models based on contemporary fire–climate relationships predict a widespread reduction in fire severity for large portions of the western US by the mid-21st century. Only a very small proportion of the western US is predicted to experience an increase in severity. Our prediction contrasts with those based on the direct influence of climate on fuel moisture and associated fire danger indices that occur at seasonal time scales (Fried *et al* 2004, Nitschke and Innes 2008). Our use of broad-scale climate as a proxy for vegetation composition and fuel load instead emphasizes the indirect influence that climate has on fire regimes (Miller and Urban 1999, Higuera *et al* 2014). Specifically, the predicted decrease in fire severity can be attributed to climatic conditions associated with higher WDs (figures 5(a) and (b)), lower productivity, and less burnable biomass (Zhao and Running 2010, Stegen *et al* 2011).

Our approach and findings are based on an implicit assumption that vegetation composition and fuel load will track changes in climate. Indeed, this is a common assumption that underlies numerous climate change studies, including those that use distribution models to project shifts in habitat ranges (Engler *et al* 2011) and fire activity (Krawchuk *et al* 2009, Moritz *et al* 2012). Specifically, our predictions of overall lower fire severity implicitly assume that vegetation composition and burnable biomass will reflect lower productivity associated with warmer and drier climates (e.g., increased WD; figure 5(b)). As such, our predictions are best interpreted as a *potential* reduction in fire severity, a potential that may not be realized where there is disequilibrium between climate and vegetation. Disequilibrium dynamics are the result of many factors and signals that directional changes in climate may not result in immediate changes in vegetation composition and fuel load (Sprugel 1991, Svenning and Sandel 2013). For example, leading-edge disequilibrium can arise when species are dispersal limited or don't reach reproductive maturity for many years (Svenning and Sandel 2013). Trailing-edge disequilibrium can arise because some species are long-lived and have deep roots, thereby facilitating survival and persistence under substantial inter-annual and decadal fluctuations in climate even though seedlings of the same species are unable to survive (Grubb 1977, Jackson *et al* 2009). To compound this, human-induced disequilibrium has also substantially affected most ecosystems in the western US (and globally) (Parks *et al* 2015b), in that natural disturbances such as fire have been excluded by factors such as livestock grazing, fire suppression, and landscape fragmentation (Marlon *et al* 2008). Both climate- and human-induced disequilibrium underlie present-day concerns about restoration of fire-adapted ecosystems after a century of fire exclusion (Stephens *et al* 2013, Hessburg *et al* 2015).

Consequently, our predictions are more likely to hold up in the presence of an active disturbance regime that catalyzes climatically driven changes in vegetation composition and fuel load (Flannigan *et al* 2000, Turner 2010). Disturbance catalysts are critical components for maintaining a dynamic equilibrium between vegetation and climate and appear to already be occurring with increasing frequency in some regions. For example, many studies have concluded that fire activity has increased in recent years (Westerling *et al* 2006, Kelly *et al* 2013) and widespread tree mortality has been attributed to drought and insect outbreaks (Allen *et al* 2010, Bentz *et al* 2010). In areas recently affected by these disturbances, the post-fire species and vegetation densities may be more tailored to the emerging climate (Overpeck *et al* 1990, Millar *et al* 2007). Although generally considered undesirable, disturbance-facilitated conversions from forest to non-forest vegetation are likely to occur in some situations (Stephens *et al* 2013, Coop *et al* in press), especially when compounded by human-induced disequilibrium.

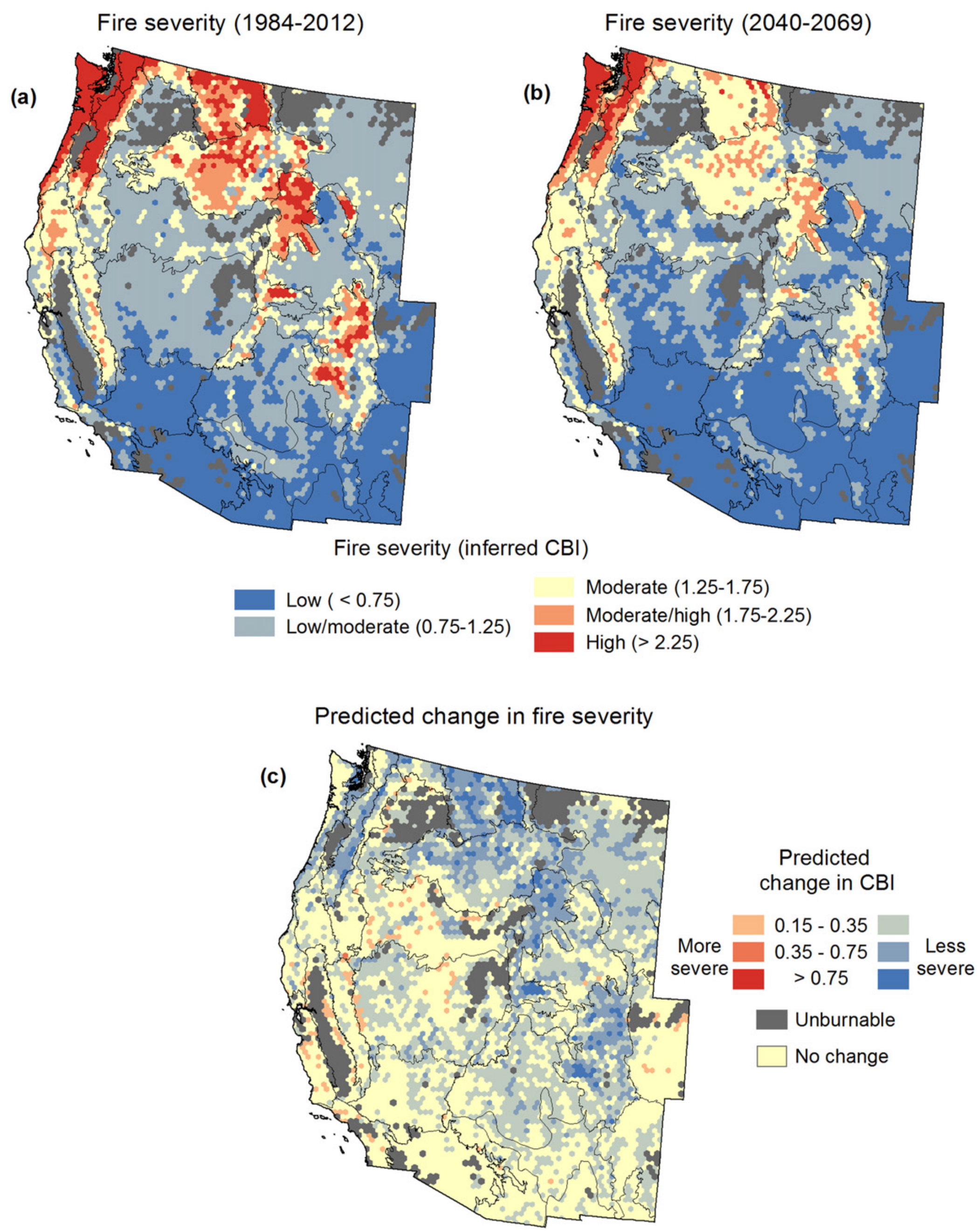

Figure 4. Predicted fire severity under observed (a) and mid-century climate (b). Mean change in fire severity among the 20 predictions (one prediction for each GCM) (c).

Most forested regions in the western US are currently experiencing a 'fire deficit' (Marlon *et al* 2012, Parks *et al* 2015b) because human activities and infrastructure (e.g., fire suppression and roads) exclude fire as an important disturbance agent. Consequently, human-induced disequilibrium between vegetation and climate, coupled with a changing climate, has important implications for future fire severity. We posit that such amplified disequilibrium will likely result in *increased* fire severity in future decades as fuel loads increase, fire seasons lengthen, and fire danger becomes more extreme (Collins 2014, Jolly *et al* 2015). This supposition is consistent with the findings of other studies that found a climate-induced increase in fire severity when assuming static vegetation (Fried *et al* 2004, Nitschke and Innes 2008). Continuing to resist catalysts of vegetation change only increases the probability of undesirable effects given that fire is inevitable (North *et al* 2009, Calkin *et al* 2015). An alternative to this unsustainable cycle is to actively facilitate transition of ecosystems to conditions that are more suited to the future climate by means of managed wildland fire or other restoration treatments (Millar *et al* 2007).

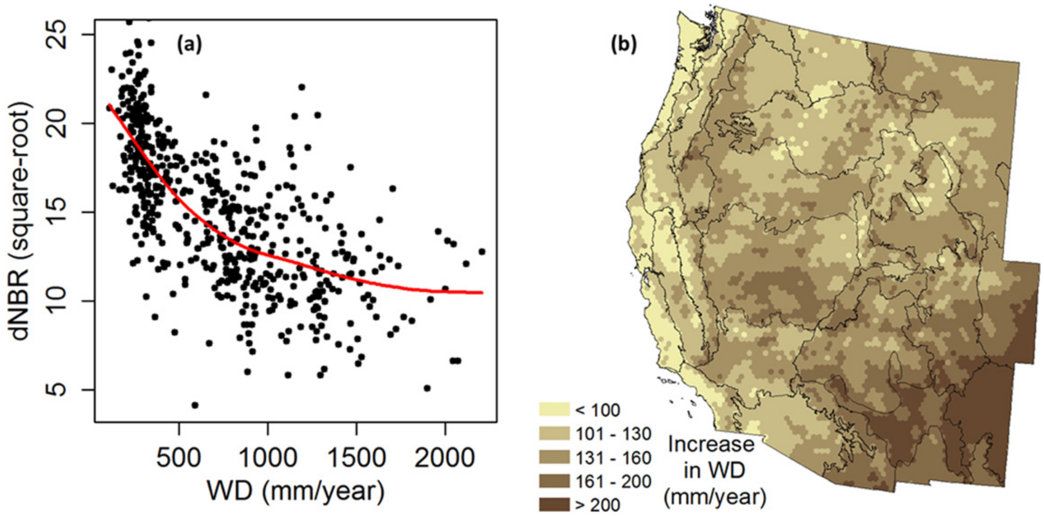

Figure 5. Plot of observed fire severity as a function of observed (1984–2012) water deficit (WD; the most influential variable in the BRT model) for the 544 hexels used to build the model (a). The red line shows the model fit according to a generalized additive model. Map of predicted increase in WD from contemporary to mid-century (2040–2069) climatic conditions (b); values depict the multi-model average change between time periods. According to this simple relationship, increased WD due to climate change will result in decreased fire severity. Note that the relationship flattens out which suggests a weaker response in dry ecosystems.

Our study complements and expands our understanding of controls on fire regimes and how they may respond to a changing climate in the western US. Specifically, predicted increases in fire activity (Littell *et al* 2010, Moritz *et al* 2012) imply that less biomass will be able to accumulate between successive fires, resulting in less biomass available for combustion and a reduction in fire severity. Furthermore, predicted increases in WD (figure 5(b)) are expected to increase water stress and decrease productivity in the generally water-limited western US (Chen *et al* 2010, Williams *et al* 2013), ultimately reducing the amount of biomass available to burn and resultant fire severity. It should be noted, however, that temperature-limited ecosystems (i.e., alpine environments) will likely experience an increase in productivity (and fire severity) under a warmer climate (Grimm *et al* 2013, Goulden and Bales 2014).

Our study relied on observed and predicted climatic normals (i.e., multi-decadal averages) to predict potential changes in fire severity. This is in contrast to other climate change fire studies that used annually or seasonally resolved climate (observed and GCM projections) and fire data to make predictions of potential changes in *fire activity* (i.e., fire frequency or area burned) (Littell *et al* 2010, Stavros *et al* 2014). The latter approach is often used because of the noted importance of climatic extremes on fire regimes (e.g., Westerling *et al* 2006). Although we could have built our model of fire severity using annually resolved data, we posit, for the purpose of predicting future fire severity, using long term averages (e.g., 1984–2012) is more appropriate for at least three reasons. First, although several studies have shown that fire severity responds to annual, seasonal, or daily variability in climate or weather, the relative influence of this variability can be fairly weak (Dillon *et al* 2011, Birch *et al* 2015). This is in contrast to broad temporal scales where the relationship between fire severity and climate has been found to be much stronger (Parks *et al* 2014b, Kane *et al* 2015). Second, because models built at a fine temporal resolution are more focused on the direct influence of climatic variability on fire weather and fuel moisture, they generally fail to incorporate climate- or fire-induced changes in vegetation composition or fuel load (Allen *et al* 2010, Parks *et al* 2015a). We suggest that predictions based on climatic normals implicitly incorporate such changes (Kelly and Goulden 2008, Marlon *et al* 2009). Lastly, GCMs may not adequately simulate annual climatic variability and thus are better suited for predicting long term trends (Stoner *et al* 2009).

Our model used broad scale data and the predictions of widespread reduced fire severity under future climate should be interpreted accordingly. For example, fire severity and climate vary at scales finer than the spatial resolution of the hexel used in this study (Schoennagel *et al* 2004). As such, our analysis does not likely capture finer-scale changes in fire severity that could occur. For example, in alpine environments where localized upward shifts in treeline under a warmer climate are expected to contribute to increases in biomass (Higuera *et al* 2014), fire severity might be expected to increase. Although our model of fire severity (dNBR) as a function of climate performed reasonably well (see section Results), we acknowledge that further error may be introduced due to error in the relationship between CBI and dNBR. However, we posit that the improved ecological interpretation attained by converting dNBR to CBI outweighs any increased error in our predictions.

Our measure of fire severity relied on dNBR (a unitless ratio) and CBI (a composite rating) and, consequently, there is no definable unit of measurement (e.g., grams of carbon consumed m^{-2}). Instead we infer changes in CBI, which integrates several strata (e.g., soil and shrubs) and scales severity from 0 to 3. This is admittedly a somewhat vague framework for assessing potential changes in fire severity, but takes advantage of the widespread availability of satellite-inferred metrics of fire severity and their documented correlation to the CBI. We suggest future research efforts involving fire severity and climate change aim to use more definitive and quantitative units of measurement. On a similar note, fire severity has ecological significance beyond what can be inferred from dNBR and is the result of many complex physical, biological, and ecological factors (Morgan *et al* 2014). For example, in ecosystems that are ill-adapted to fire (e.g., the Mojave Desert), dNBR values may be irrelevant, as any and all fires might be considered 'severe' (Brooks and Matchett 2006). Accordingly, although we used dNBR and CBI as a convenient and standardized way to assess fire severity, predictions for some ecoregions should be carefully interpreted.

Our model does not consider plant physiological responses to a CO_2 enriched atmosphere (e.g., improved water use efficiency and plant productivity) that could lead to increases in fire severity (Drake *et al* 1997, Keenan *et al* 2013). Given that today's atmospheric CO_2 concentration is the highest it's been for at least 650 000 years (Siegenthaler *et al* 2005), this could be a particularly important consideration for extreme water limited ecosystems such as grasslands, where woody plant encroachment could cause changes in biomass amount and structure (Morgan *et al* 2007, Norby and Zak 2011). Consequently, other research approaches using tools such as dynamic global vegetation models may predict different outcomes (Thonicke *et al* 2001).

Although we relied on data from protected areas and other areas of low human influence and thus underrepresented certain climatic environments (see Batllori *et al* 2014), these data represent a surprisingly broad range of ecosystem types in the western US ranging from warm desert (Death Valley National Park (NP) to dry conifer forest (Gila Wilderness) to cold forest (Yellowstone NP) (figure S2). As such, we suggest that under-represented climates have only a marginal effect on our results (see figure S2). Indeed, our analysis (figure S2) indicates that the data we used to build the model adequately represents the climates of most of the western US with the most notable exception being those in the Pacific Northwest where fires were historically and are currently infrequent (Agee 1993).

Conclusions

Our study predicts an overall decrease in fire severity for much of the western US by mid-century (2040–2069) due to changing climatic conditions. These predictions are best interpreted as *potential* decreases in severity that may not be realized unless vegetation composition and fuel load change in parallel with climate. Disequilibrium between plant communities and climate will only escalate, particularly in forested areas, unless natural disturbances and management activities (i.e., prescribed fire and restoration treatments) act as catalysts of vegetation change and push plant communities towards a state of equilibrium with climate. A high degree of disequilibrium between plant communities and climate is generally considered undesirable because the result may be an uncharacteristically severe wildland fire that causes abrupt ecosystem state shifts from, for example, forest to non-forest vegetation (e.g., Coop *et al* 2016).

Our findings support a passive management approach to ecosystem restoration (Arno *et al* 2000), whereby natural disturbance regimes are used to facilitate the transition of plant communities towards a state of equilibrium with the emerging climate. Active restoration treatments may also aid in facilitating these changes in certain situations (Millar *et al* 2007, Stephens *et al* 2010), but the current pace and scale of such treatments is insufficient to make a meaningful impact across the vast forested regions of the western US (North *et al* 2012). In addition, legal (e.g., designated wilderness) and logistical constraints (e.g., steep slopes) make certain activities (mechanical thinning) infeasible across a large proportion of land in the western US (North *et al* 2014). Achieving landscape resilience in a changing climate will likely require increased use of managed wildland fire, especially when weather conditions are not extreme (North *et al* 2015), and in fact, resisting change via activities such as aggressive fire suppression may be counter-productive in the long-run (Calkin *et al* 2015). As such, the results of this study provide insights to policy makers and land managers in the western US as to the pros and cons of resisting or facilitating change in vegetation composition and fuel load in the context of a changing climate.

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California Spotted Owl, Songbird, and Small Mammal Responses to Landscape Fuel Treatments

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A principal challenge of federal forest management has been maintaining and improving habitat for sensitive species in forests adapted to frequent, low- to moderate-intensity fire regimes that have become increasingly vulnerable to uncharacteristically severe wildfires. To enhance forest resilience, a coordinated landscape fuel network was installed in the northern Sierra Nevada, which reduced the potential for hazardous fire, despite constraints for wildlife protection that limited the extent and intensity of treatments. Small mammal and songbird communities were largely unaffected by this landscape strategy, but the number of California spotted owl territories declined. The effects on owls could have been mitigated by increasing the spatial heterogeneity of fuel treatments and by using more prescribed fire or managed wildfire to better mimic historic vegetation patterns and processes. More landscape-scale experimentation with strategies that conserve key wildlife species while also improving forest resiliency is needed, especially in response to continued warming climates.

Keywords: adaptive management, mixed conifer, restoration, Sierra Nevada, wildlife conservation

The role of wildfire in many of the world's forests that are adapted to frequent, low- to moderate-intensity fire regimes has been altered through fire exclusion, timber harvesting, livestock grazing, and urbanization (Agee and Skinner 2005, Collins et al. 2010). In the western United States, these land-use practices have affected forest structure and species composition, increasing surface fuel loads, tree density, the dominance of shade-tolerant tree species, and forest homogeneity (Hessberg et al. 2005, North et al. 2009, Chiono et al. 2012). As a consequence, many forests in the western United States are experiencing higher-severity burns—in some cases, producing large patches of tree mortality that can severely hinder the reestablishment of conifer forests (Roccaforte et al. 2012, Collins and Roller 2013). Consequently, one of the primary focuses of contemporary forest management is the treatment of fuels and vegetation to reduce fire hazards, especially as climate continues to warm (Stephens et al. 2013).

There is increased recognition that forests adapted to low- to moderate-intensity fire regimes experienced some high-severity fire (Perry et al. 2011, Marlon et al. 2012). Patchy, high-severity fire provides opportunities for early-seral habitat development and the production of large pieces of deadwood resources that are important to many wildlife species (Fontaine and Kennedy 2012). As such, forest fuel treatments should not be used to eliminate all high-severity fire. Rather, treatments should allow for patterns of fire effects that approximate those occurring under more natural forest conditions. What little information we have on fire patterns under these conditions suggests that high-severity fire constitutes fairly low proportions of the overall burned area (5%–15%) in these forest types, which is generally aggregated in relatively small patches (smaller than 4 hectares [ha]), as is the case in the upper mixed-conifer forests in Yosemite National Park (Collins and Stephens 2010, Mallek et al. 2013).

Forest management involving habitat used by wildlife species at risk has been one of the principal challenges to US federal land managers for the last 25 years. In the Sierra Nevada, an ongoing debate is focused on several species that use old-growth forest, including the California spotted owl (CSO; *Strix occidentalis occidentalis*) and the Pacific fisher (*Martes pennanti pacifica*). Forest managers need information on appropriate levels of forest manipulations to create the desired balance between habitat conservation for wildlife populations and modifications of forests to improve their resilience to large high-severity fires that could prove more expensive and detrimental than the short-term effects of restoration treatments.

Fuel-reduction treatments reduce the potential impacts of wildfire by reducing the only aspect of the fire behavior

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Figure 1. Fuel treatments implemented in the Meadow Valley project area. (a) Pretreatment mixed-conifer forest. (b) Whole-tree harvester cutting small trees (thinning from below). (c) Small trees, tree tops, and limbs being chipped and shipped by truck to a bioenergy plant to produce electricity. (c) Posttreatment defensible fuel profile zone, taken from the same perspective as in panel (a). Photographs: Keith Perchemlides.

triangle (i.e., topography, weather, fuel) that can be modified by managers: the quantity and continuity of fuel. A number of techniques are employed to reduce fire hazards, and each technique has associated effects on forest structure (Agee and Skinner 2005). Mechanical treatments can reduce stand density, basal area, and ladder and canopy fuel. To reduce accumulated surface fuel and to offset the detritus added from harvest operations, prescribed fire is sometimes used following forest thinning to reduce fire hazards, but whole-tree harvesting (i.e., complete tree removal, with the materials chipped and trucked to a processing facility; figure 1) can also effectively keep much of the harvest detritus from being added to the forest floor. Broadcast burning alone is very effective in elevating canopy base height and in reducing surface fuel (Agee and Skinner 2005).

Recent research confirms the ability of fuel treatments to alter potential fire behavior (Fulé et al. 2012) and actual wildfire effects (Safford et al. 2012). Research has also determined that fuel-reduction treatments achieve their objectives with generally positive or neutral ecological effects (Stephens et al. 2012); however, almost all research on the effects of fuel treatments has been performed at the stand scale (10–25 ha). Given the large home ranges of many key wildlife species commonly at the crux of forest management issues in the western United States (e.g., the CSO, the northern spotted owl [*Strix occidentalis caurina*], the Pacific fisher), it is important to understand fuel-treatment impacts at larger spatial scales. This is particularly relevant because many fuel-treatment projects are being proposed—and, in a few instances, implemented—at landscape scales (15,000–40,000 ha; Ager et al. 2007, Collins et al. 2010).

Fuel treatments directly alter wildlife habitat by removing both aerial (trees) and ground (coarse wood, shrubs) cover. These altered conditions can affect both habitat suitability, which influences the number of individuals that an area can support, and habitat quality, which directly affects the fitness

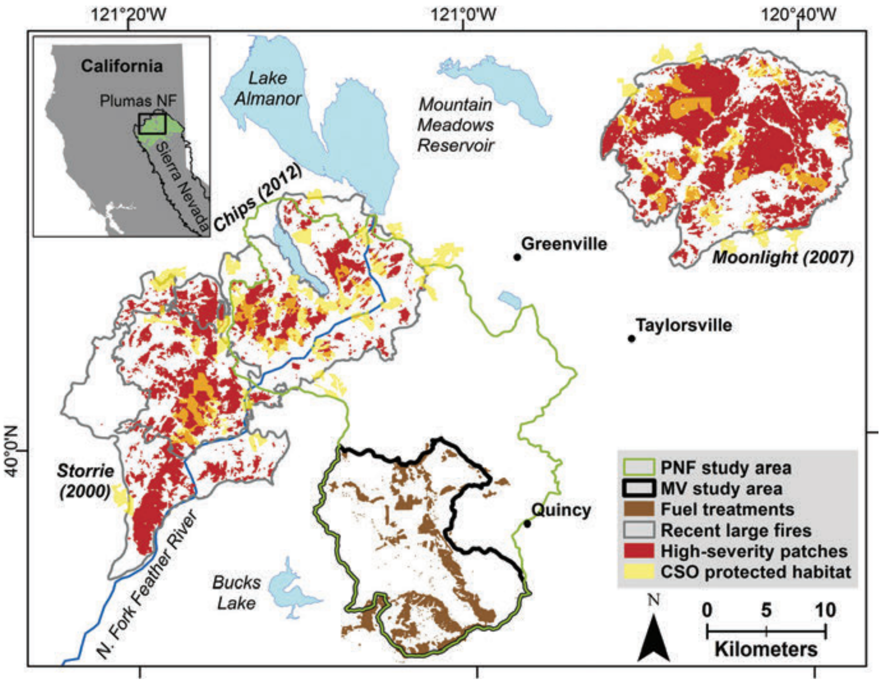

Figure 2. Meadow Valley study area with completed landscape fuel-treatment network. Recent large wildfires and the resulting patches of high-severity fire effects are also indicated. Three wildfires are shown: Storrie (2000), Moonlight (2007), and Chips (2012). These were selected on the basis of the following criteria: proximity to the study area (closer than 25 kilometers), vegetation type (conifer dominated), size (larger than 10,000 hectares), and age (since 2000). Abbreviations: CSO, California spotted owl; MV, Meadow Valley; N, north; NF, national forest; PNF, Plumas National Forest; W, west.

and productivity of individuals. Because more-suitable habitat for certain at-risk wildlife species is associated with greater aerial and ground cover, the effects of fuel treatments are generally perceived as negative. However, large patches of wildfire-caused tree mortality can also negatively affect both habitat suitability and quality (Tempel et al. in press). To the extent that fuel treatments reduce the potential for large patches of tree mortality in wildfire, there may also be an indirect benefit of fuel treatments to certain species' habitat. Finding a balance between these influences is a crucial management need.

Over the past decade, we have studied the ecological effects of one of the few completed landscape-level fuel-treatment networks in western US forests. Here, we distill the results of these efforts. We quantify change in vegetation structure and modeled fire behavior as a result of fuels treatments and assess treatment effects on the CSO, songbirds, and small mammals. Modeling studies have been published in which the trade-offs in these systems have been conceptually examined (Lee DC and Irwin 2005), but this is one of the first studies in which these questions have been empirically examined at landscape scales.

Study area and design

Our study area is located in the Meadow Valley area of the Plumas National Forest, situated in the northern Sierra Nevada, at 39 degrees (°) 56 minutes (′) north, 121°3′ west (figure 2). The climate is Mediterranean, with warm, dry summers and cool, wet winters, which is when most precipitation (1050 millimeters per year; Ansley and Battles 1998) occurs. The core study area is 19,236 ha, with elevations ranging from 850–2100 meters (m). The vegetation is primarily mixed-conifer forest, consisting of white fir (*Abies concolor*), Douglas-fir (*Pseudotsuga menziesii*), sugar pine (*Pinus lambertiana*), ponderosa pine (*Pinus ponderosa*), Jeffrey pine (*Pinus jeffreyi*), incense-cedar (*Calocedrus decurrens*), California black oak (*Quercus kelloggii*), and other less common hardwood species. White fir is the most abundant tree, although large (e.g., larger than 1 m in diameter) stumps of pines encountered frequently in the forest attest to a change in composition and structure in recent history. Red fir (*Abies magnifica*) is common at higher elevations, where it mixes with white fir. In addition, a number of species are found occasionally in or on the edge of the mixed-conifer forest, including western white pine (*Pinus monticola*) at higher elevations, lodgepole pine (*Pinus contorta* var. *murrayana*) in cold air pockets, and western juniper (*Juniperus occidentalis*) on xeric sites. California hazelnut (*Corylus cornuta*), dogwood (*Cornus* spp.), and willow (*Salix* spp.) are found in moister riparian areas. Montane chaparral and some meadows are interspersed in the landscape. Tree density varies as a result of recent fire- and timber-management history, elevation, slope, aspect, and edaphic conditions. Historical fire occurrence, which can be inferred from fire scars recorded in tree rings, suggests that the fire regime was predominantly frequent, low- to moderate-severity fires, at intervals ranging from 7–19 years, with the last widespread fires occurring 85–125 years ago (Moody et al. 2006).

Fire activity in the last 15–20 years has been notably higher in the northern Sierra Nevada than in the rest of the range (Collins 2014). Since 2000, there have been three megafires (covering more than 10,000 ha; Stephens et al. 2014) within 25 kilometers (km) of our study area, burning a total of 73,000 ha (figure 2). These fires burned predominantly in mixed-conifer forests, encompassing approximately 60 CSO protected activity centers (figure 2). Cumulatively, 34% of the area burned in these three fires suffered high-severity fire (more than 95% dominant tree mortality; figure 3a; Miller et al. 2009). More important than the total proportion of area severely burned is the distribution of high-severity patches over the burned area, because this can limit tree seed

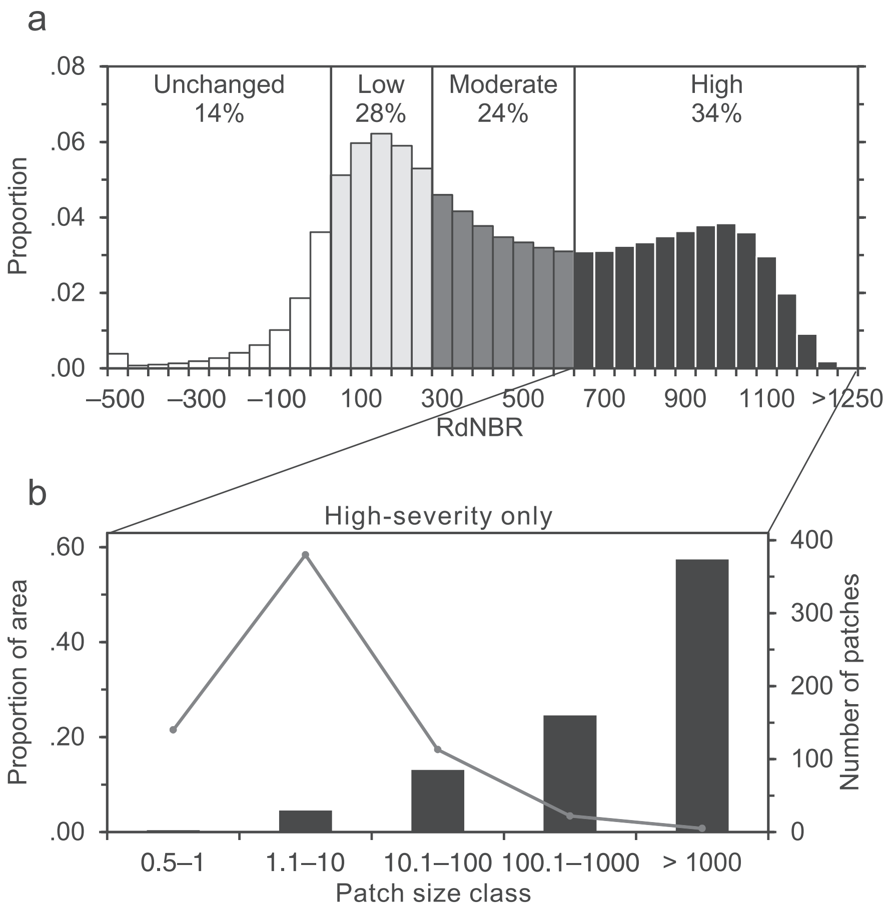

Figure 3. (a) Fire severity distribution for the three recent large fires in the Meadow Valley study area (see figure 2). The fire-severity estimates are based on the relative differenced normalized burn ratio (RdNBR; Miller and Thode 2007). (b) The proportion of total high-severity area (bars) and the number of patches (line) as a function of patch size class.

dispersal from wind and animals (Perry et al. 2011, Collins and Roller 2013). Large patches (defined here as larger than 1000 ha) accounted for a disproportionate amount of the total high-severity-fire area in the recent wildfires near the study area (figure 3b).

The projects that contributed to the fuel-treatment network are part of the larger Herger-Feinstein Quincy Library Group Pilot Project (USHR 1998). This project was directed by the US Congress to involve local communities in forest management. The project objectives included improving forest health, reducing uncharacteristic high-severity fire, conserving wildlife habitats, and stabilizing economic conditions in local communities. The projects in Meadow Valley encompassed a range of treatment types and intensities reflecting changes in regional management directions and differing land-management constraints across a complex landscape (Collins et al. 2010, Moghaddas et al. 2010). The primary fuel treatment used in Meadow Valley was defensible fuel profile zones (DFPZs), which are areas approximately 0.4–0.8 km wide in which surface, ladder, and crown fuel loads are reduced with a combination of moderate thinning from below (Moghaddas et al. 2010) and prescribed fire treatments (figure 1).

The DFPZs were excluded from portions of the landscape set aside as reserves and from designated CSO protected activity centers, which are 121-ha areas of high-suitability nesting habitat designated by forest biologists. In addition, the project predominantly excluded all riparian habitat conservation areas or stream buffers intended to protect riparian and aquatic resources (figure 4). The activities conducted in the DFPZs were chainsaw thinning and pile burning of trees up to 30 centimeters (cm) in diameter at breast height (dbh); mastication: primarily shrubs and small trees were shredded and chipped in place, with the material left on site; prescription burning: stands were burned under conditions of moderate relative humidity and fuel moisture; and a combination of mechanical thinning and prescription burning of trees up to 51 or 76 cm dbh, depending on whether the stands were in the wildland–urban interface, using a whole-tree harvest system (figure 1) to achieve a residual canopy cover of approximately 40%, and some were underburned (Moghaddas et al. 2010). In addition to the DFPZs, group-selection treatments were implemented as part of the project. The group-selection treatments included the removal of all conifers up to 76 cm dbh within an area of 0.8 ha, followed by residue piling and burning, then either natural regeneration or replanting to a density of 270 trees per ha with a mix of sugar pine, ponderosa pine, and Douglas-fir. These treatments collectively covered 3688 ha (3448 ha in the DFPZs, 240 ha in the group-selection treatment), or 19% of our study area, and were implemented between 2003 and 2008.

Forest structure and microclimate

Although they are designed to reduce fire hazards, forest treatments alter stand conditions directly by reducing tree density and canopy cover, and indirectly by altering microclimate conditions affecting the understory community. To assess these changes we measured stand structure, light, understory plant cover, micro-meteorological variables, soil moisture, and fuel moisture in replicated control, thinning, and group-selection treatments plots embedded within the landscape-level treatments (see Bigelow al. 2009, 2011, Bigelow and North 2012 for detailed methods).

The mean forest canopy cover was 69% (standard deviation [SD] = 7%) before treatment; after treatment it was 53%

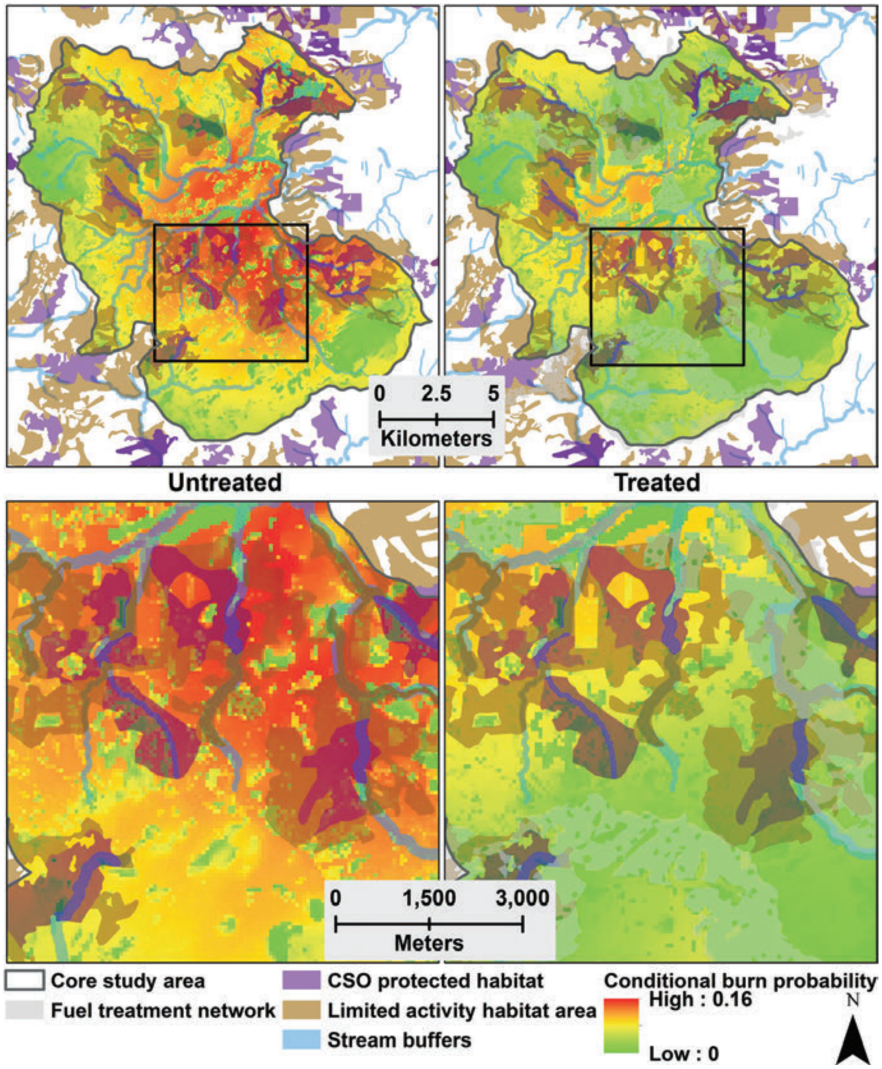

Figure 4. Hazardous fire potential across the Meadow Valley study area for the untreated and treated landscape conditions. This fire potential is based on the conditional burn probability of fire occurring with flame lengths greater than 2 meters, which is consistent with tree torching (see Collins et al. 2013 for specific details). Land designations that often limit or exclude active forest management (e.g., California spotted owl [CSO] protected habitat, stream buffers) are also shown to illustrate off-site effects of the landscape fuel-treatment network. The black square in the upper panels indicates the focal area shown in the bottom panels.

(SD = 7%) in thinned stands and 12% (SD = 6%) in the group-selection openings (Bigelow et al. 2011). These differences were reflected in growing-season understory light, which averaged 17% of full sun before treatment and increased to 26% in thinned stands and 67% in group-selection openings. Models of regenerating tree growth and light availability demonstrated that the height growth rates of shade-intolerant yellow pines (ponderosa and Jeffrey pines) and shade-tolerant white fir were equal at 41% of full sun. Light levels greater than this correlated exponentially with the height growth of the pines. The group-selection treatments provided ample light to recruit shade-intolerant species to the canopy, but only 8% of the sample locations in the thinning treatments had light levels exceeding the 41% crossover point, which suggests that these treatments would not substantially contribute to pine restoration across the landscape. An analysis of hemispherical photographs showed that the treatments decreased canopy closure following thinning. At the plot (1-ha) scale 3 years after treatment, cover of understory plant life-forms only changed under group selection ($p < .05$). Shade-tolerant conifers decreased, and graminoids, forbs, and broad-leaved trees (mainly California black oak and dogwood) increased (figure 5). There was no increase in exotic plant species cover with any of the treatments (Chiono 2012).

Changes in abiotic conditions followed differences in canopy cover for only some of the variables measured (Bigelow and North 2012). Soil moisture increased and duff moisture decreased in the group-selection treatments relative to the thinned and pretreatment conditions. Wind gust speeds (measured 2.5 m above ground) averaged 31% higher in the thinned stands than in the controls, but this was far less than the 128% increase in the group-selection openings. However, there was no difference in air temperature or relative humidity among the treatments, possibly because the increase in understory wind increased air mixing and eliminated any gradients in air temperature and humidity that might have resulted from increased irradiance.

Treatment increased within-stand variability for some vegetation and microclimate conditions but, in general, did not create the landscape-level heterogeneity characteristic of historic forest conditions in the Sierra Nevada (North et al. 2009). Mixed-conifer forests support the highest vertebrate diversity of California forests (Verner and Boss 1980), and studies suggest that this may result from habitat variability associated with the observed range of tree species diversity, canopy cover, microclimate, and deadwood conditions (Rambo and North 2009, Ma et al. 2010, White et al. 2013). This historic forest heterogeneity appears to reflect differences in fire intensity and site productivity associated with local and large-scale changes in slope, aspect, soil, and slope position (North et al. 2009, Lydersen and North 2012). On average, more mesic sites (e.g., drainage bottoms and north-facing slopes) historically supported greater stem density, canopy cover, and tree basal area, whereas drier and

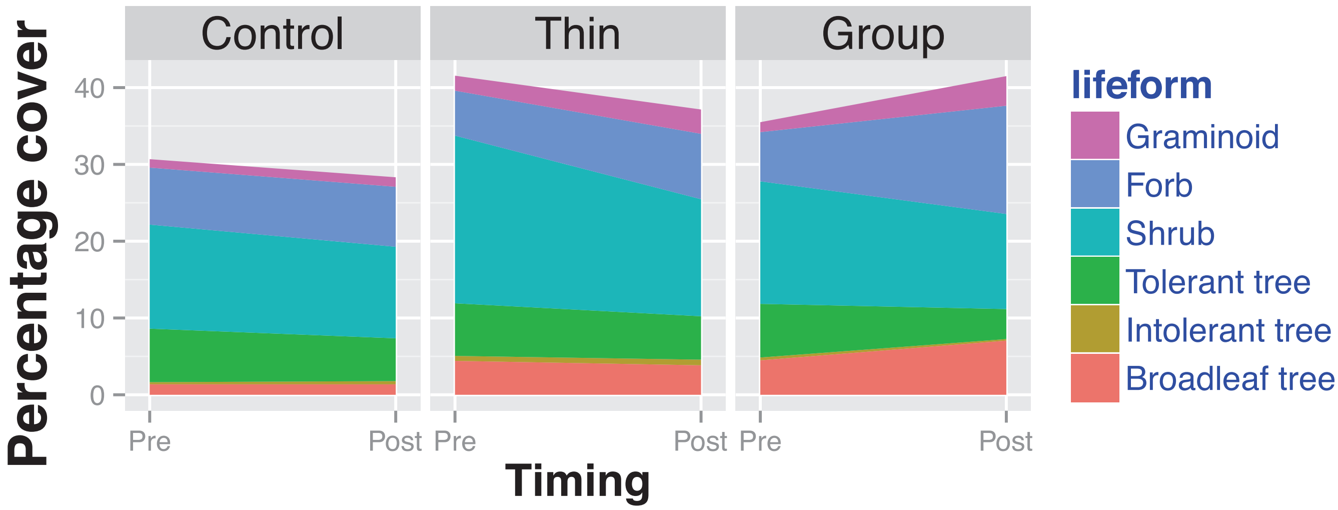

Figure 5. The percentage cover of plant life forms before* (pre) *and 3 years after* (post) *fuel-reduction thinning and group-selection treatments (*n *= 300 subplots per treatment) that were implemented in 2007 in Meadow Valley. Changes in understory cover in thinned stands were not significant* ($p > .16$)*. Graminoids, forbs, and broadleaf trees increased and shade-tolerant conifers decreased* ($p < .05$) *in group selection openings.

steeper areas burned more frequently and intensely, creating more-open, pine-dominated forests (North et al. 2009). Although the Meadow Valley treatments did increase within-stand heterogeneity, they were not explicitly designed to vary with site topography or local productivity to produce this historic landscape variability.

Potential fire behavior

We employed a spatially explicit fire behavior model (Finney et al. 2007) to simulate fire spread across the Meadow Valley area. We simulated 10,000 individual fire events, with random ignition locations, and compared patterns of burn probability based on the number of times a particular area burned with the given ignition locations and simulated flame lengths for the study area prior to and following the implementation of landscape fuel treatments. Each fire event simulated burning for 240 minutes (one 4-hour burn period) under 97th percentile fuel moisture and wind conditions. These are the conditions associated with large-fire growth in this region (Collins et al. 2013). The burn period duration was selected such that the simulated fire sizes (for one burn period) approximated large-spread events observed (daily) in nearby recent wildfires (Collins et al. 2013). One of the primary assumptions with this approach is that, during these large-spread events (burn periods), fire suppression operations have limited impact, which is consistent with observed large-fire occurrence throughout the western United States (Finney et al. 2007). We summarized the burn probabilities across the Meadow Valley area into land allocations determined by the US Forest Service (USFS; Moghaddas et al. 2010).

The simulated fire behavior indicated that the landscape-scale network of DFPZs and prior fuel treatments were effective at reducing conditional burn probabilities across all land-allocation types, except the small area of off-base lands (figure 4; Moghaddas et al. 2010). Because burn probabilities are correlated directly and positively to fire size (Finney et al. 2007), it is clear that the pretreatment landscape was more conducive to large-fire growth than the posttreatment landscape was (Moghaddas et al. 2010, Collins et al. 2013). Although the influence of the treatments on the modeled burn probabilities of each land allocation varied, the untreated stands (e.g., those designated for protected CSO habitat, riparian and aquatic resources, and reserve lands) and the remaining private and unclassified lands all experienced reduced burn probabilities from the application of fuel treatments at the landscape scale (figure 4; Moghaddas et al. 2010). A similar reduced burn severity immediately adjacent to treated areas has been reported for actual fires across the western United States (Finney et al. 2005).

The substantial reduction in both the total area and the area burned at higher flame lengths under a posttreatment wildfire scenario was notable, given that only 19% of the study area had been treated (Moghaddas et al. 2010, Collins et al. 2013). Both the orientation of the treatments (approximately orthogonal to the predominant wind direction throughout the duration of the simulated fire), and the long, continuous shape of the DFPZs resulted in potential wildfires' intersecting fuel treatments in multiple places. In addition, the treatments were somewhat concentrated in the southwestern portion of the study area (figure 2), which is the dominant direction of strong winds during the fire season (Collins et al. 2013). In combination, these factors limited the ability of the simulated fire to both circumvent the treated areas and to regain spread and intensity after encountering the treatments. These results are important to managers, because similar installations of fuel and restoration treatments are needed in many Sierra Nevada

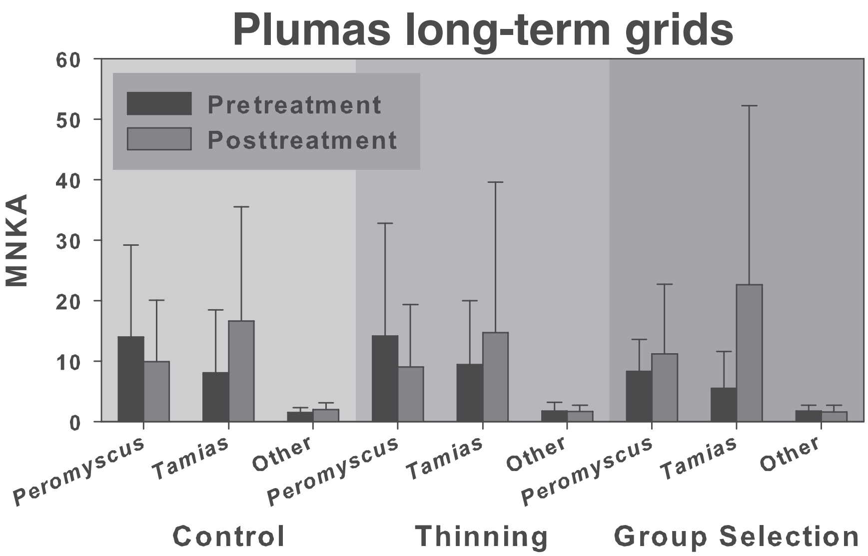

Figure 6. The mean minimum number of small animals known alive (MNKA), recorded before and after fuel treatments in the Plumas National Forest study area. For ease of presentation, we present three species groups (**Peromyscus boylii** ***and*** **Peromyscus maniculatus*****;*** **Tamias quadrimaculatus** ***and*** **Tamias senex*****; all other species; see Kelt et al. 2013 for details). The bars represent the means of the replicate sampling grids. The error bars represent the positive standard deviation.***

mixed-conifer forests, where the present treatment rates are very low (North et al. 2012).

Small mammals

The northern Sierra Nevada supports a diverse fauna of small mammals that play key ecological roles as consumers, seed and fungal dispersers, and prey for both terrestrial and aerial predators (Hallett et al. 2003, Kelt et al. 2013). We studied small mammals in the Meadow Valley study area and the greater Plumas National Forest study area (PNFSA; figure 2), with a particular focus on two species that are key prey of the CSO (Gutiérrez et al. 1995): the dusky-footed woodrat (*Neotoma fuscipes*) and the northern flying squirrel (*Glaucomys sabrinus*). Results on focal species efforts have been reported elsewhere (Innes et al. 2007, Smith et al. 2011), but one finding merits emphasis here. California black oak, the primary hardwood in mixed-conifer forests, is an important habitat element for both the woodrat and the flying squirrel. Woodrat density was positively correlated with black oak density (Innes et al. 2007), and both species strongly preferred black oaks for nest sites (Innes et al. 2008, Smith et al. 2011). California black oak may be important for other wildlife species as well (Zielinski et al. 2004), but its persistence in our study landscape is in doubt. California black oak is shade intolerant, and across our study area, there were few thriving seedlings and many mature trees in decline as adjacent conifers overtopped them. California black oak trees were present in only 133 of 602 plots placed randomly in the PNFSA and were in a codominant canopy position in less than 10% of the plots in which it was present (see supplement S1).

Our broader studies on the management needs of entire small mammal assemblages included two complementary efforts. We sampled small mammals annually for 8 years on replicate trapping grids in treated and untreated mixed-conifer forests dominated by white fir in order to evaluate the responses of the small mammal community to canopy thinning (Kelt et al. 2013). To determine whether the habitat associations of the mammals in these forests were similar to those of mammals in other forest types, we expanded our efforts to include stratified random sampling of the PNFSA that encompassed the Meadow Valley study area (figure 2).

Whereas canopy thinning in white-fir-dominated mixed-conifer forests caused some significant changes in forest structure, small mammal assemblages were similar before and after canopy thinning and group selection (Kelt et al. 2013), which suggests a minimal response in the short-term to these treatments (*contra* Suzuki and Hayes 2003, Gitzen et al. 2007, but see Carey and Wilson 2001). Although each treatment may have elicited somewhat different responses (figure 6), the variance across replicate plots eroded any such differences even in the face of the substantial variation in canopy cover. The lack of a short-term response may not be surprising in a system characterized by high interannual variation in weather and in a system dominated by generalist species; we look forward to resampling these sites after 10–15 years to assess potential longer-term responses. Because our manipulative experiment was focused on white-fir-dominated mixed-conifer forests, we pursued a more general assessment of mammalian responses to habitat and environmental variation across the entire PNFSA, capitalizing on a series of point-count transects established throughout the forest in a stratified (by forest type) random manner (see the "Songbirds" section below). We sampled eight randomly selected points on each of 74 transects to characterize how small mammals respond to broader variation in forest structure.

We assessed assemblage-wide responses to this variation with ordination (canonical correspondence and canonical correlation) and species-specific responses with multiple stepwise regression. All data were standardized (both rows and columns) by centering and normalizing, and the mammal data were log-transformed to prevent domination of the axes by common species. The results from all of the analyses were qualitatively identical to those of the Meadow Valley experimental grids, which indicates minimal responses of small mammal assemblages to variation in forest structure or composition. Although the spatial arrangement of the

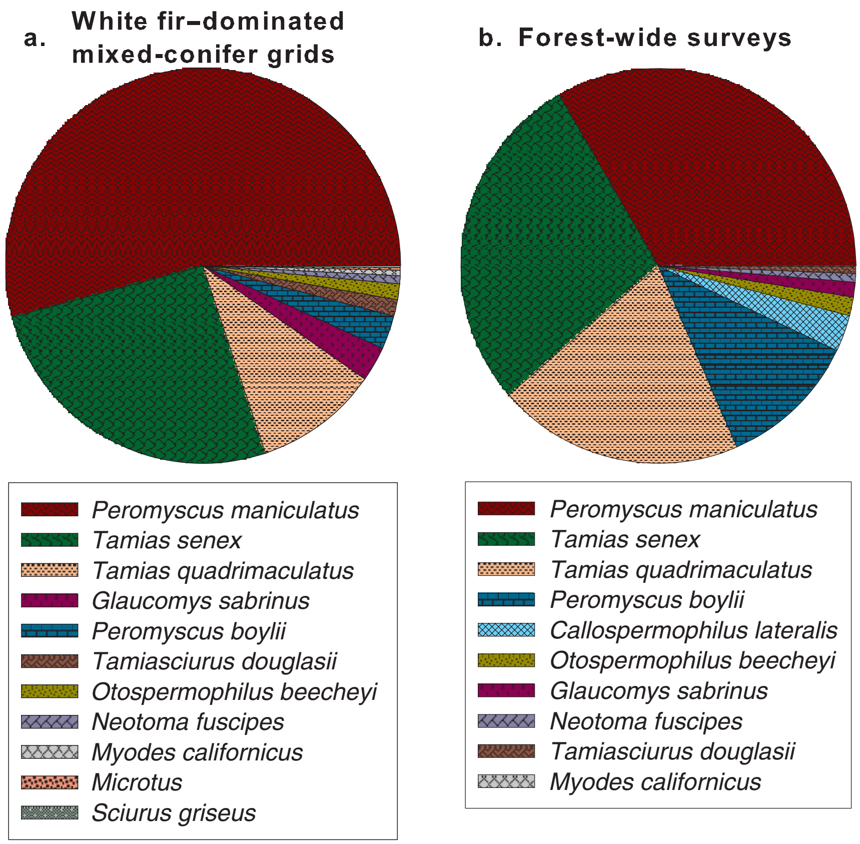

Figure 7. Small mammal composition at two spatial scales in the Plumas National Forest study area. At both scales, captures were dominated by three species. At the forest scale, only one other species was highly represented. All other species at both scales were only minor elements.

small mammal species in the ordination space was ecologically reasonable (e.g., woodrats and brush mice [*Peromyscus boylii*] associated with oaks, and chipmunks [*Tamias*] and Douglas squirrels [*Tamiasciurus douglasii*] associated with conifers and with a high basal area of trees and snags), ordination explained only a small proportion of variance in the distribution of small mammals. Similarly, regression failed to produce compelling associations for any species (or for community metrics such as species richness or diversity). The coefficients for both sets of analyses were universally low (Kelt et al. 2013).

In trapping efforts on the Meadow Valley experimental grids and in the larger PNFSA (figure 2), our captures were overwhelmingly dominated by 3–5 species (figure 7). Deer mice (*Peromyscus maniculatus*) dominated the captures at both spatial scales, comprising a full 55% of the captures on the Meadow Valley experimental grids and just over one-third of the captures in the PNFSA. Two species of chipmunk (*Tamias quadrimaculatus*, *Tamias senex*) represented an additional 40%–44%, and brush mice were an additional 8% in the PNFSA. Therefore, our samples were dominated by ecological generalists known to be tolerant of diverse habitats. What appears to be missing is a reasonable representation of species with more restricted niche requirements. Our sampling was not designed to sample shrews (*Sorex*), but California red-backed voles (*Myodes* [formerly *Clethrionomys*] *californicus*) may have been more common in this region in the 1940s and 1950s (Kelt et al. 2013) and should have been present in our study. This species forages on fungi, however, and requires large downed woody debris and a closed-canopy forest to allow sufficient moisture retention to promote fungal growth (Alexander and Verts 1992). In 177,216 trap nights of effort, we captured only 11 *Myodes* (all but one on Meadow Valley experimental grids). Other species that are mesic habitat specialists were not sampled (e.g., *Zapus trinotatus, Sorex palustris*).

It is not clear whether the taxonomically depauperate assemblage structure documented in our study represents a relatively recent reduction or is more historic for this region. No data on mammal assemblages exist prior to European settlement and the beginning of widespread changes to the Sierra Nevada forest ecosystems (Merchant 2012). However, one implication of this research is that, in spite of nearly a kilometer of vertical elevation relief and diverse forest types from ponderosa pine to red fir, the current forest conditions support a relatively homogeneous small mammal community dominated by ruderal species. It is unclear whether this reflects a legacy of fire exclusion and the resulting accumulation of fine woody debris or, perhaps, a response to a history of logging and fire suppression in this region. In contrast, other recent work in Yosemite (Roberts et al. 2008) confirms that small mammals respond strongly to variation in burn history. Taken together, these results support the fundamental ecological role of fire and broadscale forest heterogeneity in managing mixed-conifer forests in the Sierra Nevada (North et al. 2009).

Songbirds

To evaluate the effects of the Meadow Valley fuel-treatment network on songbirds, we compared avian community diversity before and after treatment. From 2004 to 2011, we surveyed the breeding community in and adjacent to Meadow Valley, using standardized point-count surveys with a 50-m radius (Ralph et al. 1995). Surveys were conducted at 51 stations where DFPZs were implemented (treated) and 201 stations where no treatments were implemented (untreated), proportional to the 19% of the study area treated. An additional 180 stations were surveyed in adjacent untreated PNFSA (figure 2) watersheds (the reference group). We used geographic information systems to establish locations

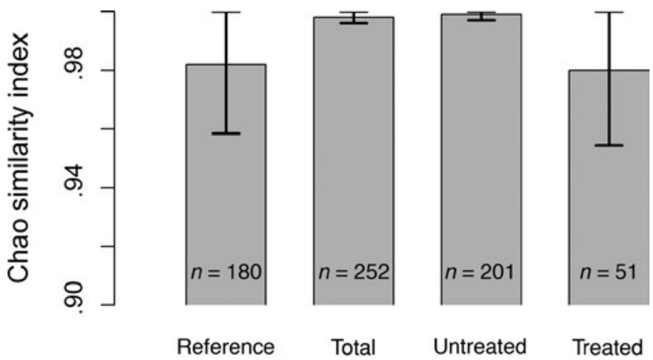

Figure 8. Chao similarity index for the avian community (60 species) before and after treatment at treated and untreated locations in the Meadow Valley study area and reference locations in the adjacent Plumas National Forest study area that also received no treatment. This metric ranges from 0–1, with 1 representing perfect similarity (all species and relative abundances shared among both samples). The error bars are 95% confidence intervals.

for the untreated and reference stations from a randomly selected origin (constrained by slopes lower than 35% and on USFS land) along a random compass bearing in a linear array of 4–12 points. The treated stations were placed within proposed DFPZ treatments across the breadth of treatment types and geography described above. All of the stations were a minimum of 250 m apart.

We surveyed all of the stations in both 2004 and 2005, prior to treatment, and for 2 years after all treatments were implemented (2010–2011). In each year, we surveyed every station twice during the peak of the breeding season (15 May–10 July), with a minimum of 10 days between visits. We limited our analyses to the 60 species breeding in upland habitats that were reliably recorded with point counts (Hutto et al. 1986). The results were summarized at the level of the three treatment groups described above (treated, untreated, reference) and for treated and untreated locations in Meadow Valley combined. For all of the analyses, we summed detections across four surveys (two visits per year over 2 years) for the pre- and posttreatment periods. We compared avian assemblages before and after the treatment with Chao–Jaccard's similarity index (Chao et al. 2005), calculated using EstimateS (version 9.1, University of Connecticut, Storrs). Chao–Jaccard similarity is sensitive to changes in species composition and abundance. Differences in avian diversity were evaluated using the exponent of the Shannon index (Nur et al. 1999). For both analyses, 95% confidence intervals were derived from estimated standard errors from 1000 bootstrap samples.

Our results indicate little change in the Meadow Valley avian communities in response to treatment. The communities were similar across the treated, untreated, and reference samples (figure 8). There was some evidence that the treated areas were less similar to each other than were the untreated areas, but this was not statistically significant ($p > .05$). Avian diversity (the Shannon index) was lowest for the treated sample prior to treatment but increased more in the posttreatment period, such that the Shannon index after treatment was equivalent in the treated and untreated samples (figure 9).

Evaluating the effects of fuel treatments with coarse metrics such as similarity and diversity can cause one to overlook large effects on select species (Hurteau et al. 2008). Numerous studies in seasonally dry fire-prone US forests have shown that fuel treatments can result in at least modest changes in the abundance of a broad range of avian species (Fontaine and Kennedy 2012). We recently reported that mechanical fuel-reduction treatments in the northern Sierra Nevada (including Meadow Valley) resulted in modest decreases in the abundance of a few closed-canopy associates and increases in some edge and open forest associates (Burnett et al. 2013). None of the 15 species evaluated in that study showed a significant decline following the construction of shaded fuel break DFPZ treatments—the primary treatment used in the Meadow Valley study area. With the moderate portion of the landscape treated, small differences in avian community similarity and diversity resulting from treatment, and the results from our previous evaluation of individual species response, we conclude that the effects of the Meadow Valley fuel-treatment network on the songbird community were minimal.

The fuel treatments implemented in Meadow Valley were typically less intense than those shown to result in large changes in avian communities (for a review, see Vanderwel et al. 2007). The treatments were applied to 19% of the landscape, and the prescriptions left relatively high canopy cover. Fire suppression and silvicultural practices over the last century have reduced forest heterogeneity and increased stand density (Scholl and Taylor 2010, Collins et al. 2011). In the Sierra Nevada, most fuel treatments changed the forest structure moderately from historic forest conditions (North et al. 2007). The Meadow Valley mechanical treatments primarily removed ladder fuels, which reduced crown fire potential but did not substantially alter the existing habitat features associated with songbirds, such as shrub cover or large overstory trees.

Our results should be considered in the context of the conditions that existed in the study area prior to the implementation of the landscape treatments. If an objective of these treatments was to maintain the existing avian assemblage and diversity, they appear to have been successful. However, a frequently stated objective for fuel reduction is to act as a surrogate for the natural fire regime (Stephens et al. 2012). Therefore, the maintenance of the pretreatment wildlife community may not always be the most desirable outcome in landscapes such as Meadow Valley and the larger PNFSA, where fire has been excluded for 85–125 years (Moody et al. 2006). Creating or enhancing

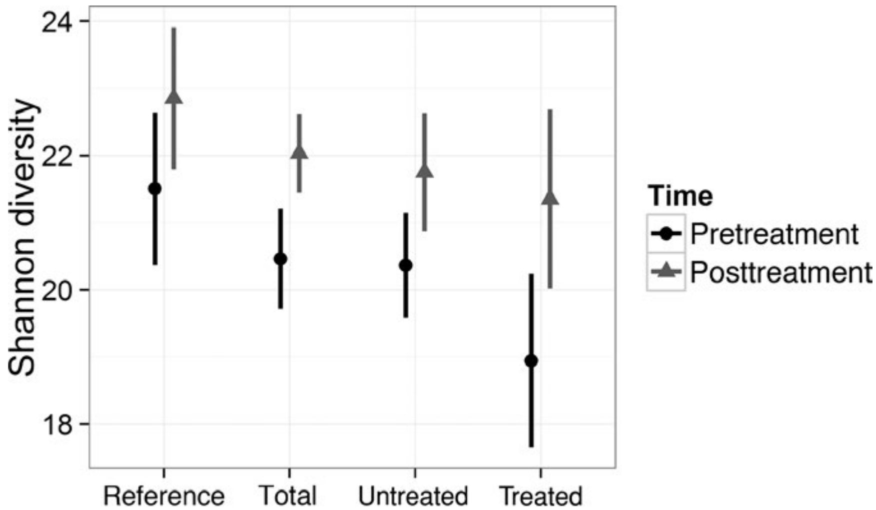

*Figure 9. Shannon diversity index of avian diversity before (*pretreatment*) and after (*posttreatment*) fuel treatments were implemented at treated (*n = 51*) and untreated (*n = 201*) locations and the first two combined (*Total; n = 252*) in the Meadow Valley study area and in reference locations in the adjacent Plumas National Forest study area, which received no treatment (*n = 181*). The error bars are 95% confidence intervals.*

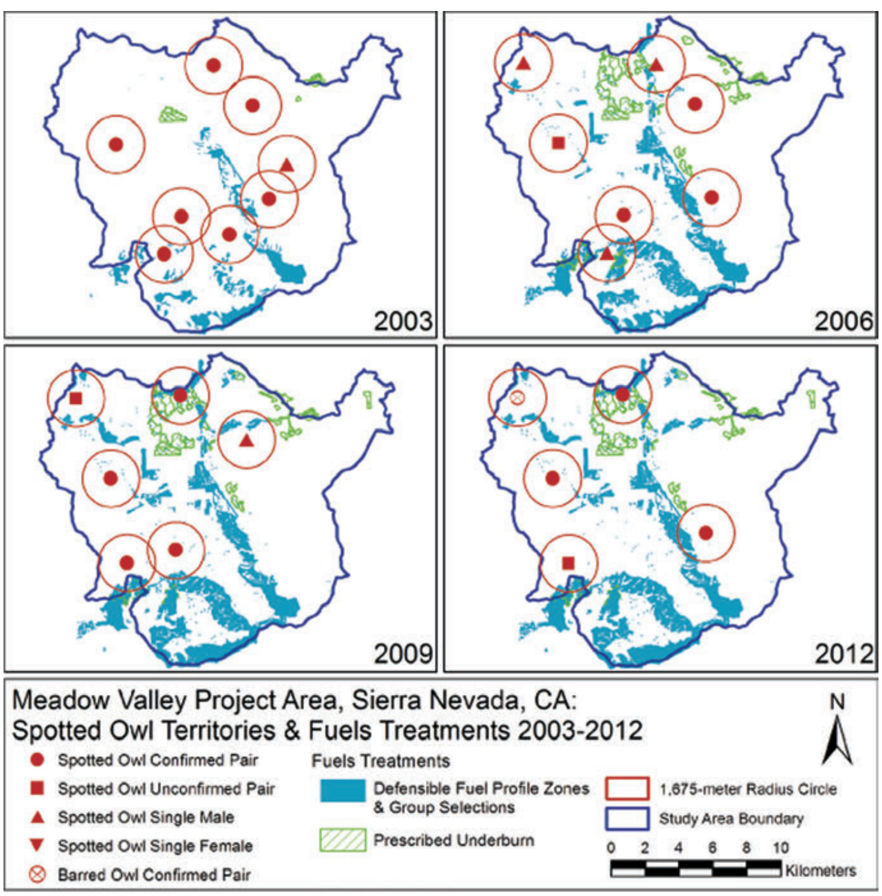

Figure 10. Distribution of territorial California spotted owl sites and landscape forest fuel treatments within the Meadow Valley study area from 2003 to 2012.

conditions for species associated with postdisturbance habitat, some of which have experienced recent declines, may be a prudent approach for achieving some biological diversity objectives (Fontaine and Kennedy 2012). If fuel-reduction treatments are to be a complementary tool to fire in achieving biological objectives, we suggest that they be designed to further increase landscape heterogeneity in fire-excluded forests.

California spotted owls

Modeling studies have projected that fuel treatments on a portion of the landscape (20%–35%) may have minimal effects on owl habitat and that the longer-term benefits of reduced wildfire risk may outweigh the short-term treatment effects on owl habitat (Ager et al. 2007, Roloff et al. 2012). However, no empirical data are available to assess the effects of landscape fuel treatments on the CSO and its habitat.

We used standardized surveys and color banding of individual owls to monitor the distribution, occupancy, survival, and reproduction of CSO sites annually across 1889 square kilometers between 2003 and 2012 in the Plumas and Lassen National Forests. Within this area, four areas were identified for implementation of landscape-scale fuel and restoration treatments. Our initial objectives were to establish baseline values for CSO distribution and abundance and to monitor the owl's response in the treated and untreated landscapes in posttreatment years. However, complete implementation of the fuel-treatment network only occurred on one (Meadow Valley; figure 10) of the four landscapes because of legal challenges to the proposed US Forest Service management strategy.

In the Meadow Valley study area, the number of territorial owl sites declined after treatment. Prior to and throughout the implementation of the treatment, the number of owl sites ranged from seven to nine. Between the final year of the DFPZ and group-selection installations (2008) and 2 years after treatment (2009–2010), the number of owl sites declined by one (six territorial sites), and by 3–4 years after treatment (2011–2012), the number of sites had declined to four—a decline of 43% from the pretreatment numbers (figure 11). These results mirror similar declines of the CSO in the larger Plumas-Lassen CSO study area over the past 20 years (Conner et al. 2013) but suggest a greater magnitude of decline within Meadow Valley (figure 11).

The CSO nests and roosts in dense, multilayered, mature forest patches, and the adult survival and territory occupancy of these owls is positively correlated to the amounts of mature forest in core areas around CSO sites (Dugger et al. 2011). For foraging, however, the CSO uses a broader range of vegetative conditions. Radio-telemetry conducted in Meadow Valley indicates that the CSO avoids foraging in DFPZs in the first 1–2 years after fuel treatments and that the owl's home range size was positively correlated with the amount of treatment within the home range (Gallagher 2010). Barred owls (*Strix varia*) began to colonize the Meadow Valley study area in 2012 and are likely to become a threat to the CSO and a confounding factor to be accounted for in assessments of forest management effects (Keane 2014).

Although inference must be tempered from a single study, the Meadow Valley area is the first large area to receive full the implementation of landscape-scale DFPZ and group-selection treatments in which CSOs were monitored annually both before and after treatment. CSOs are long-lived (up to 20 years) and exhibit high site fidelity as adults, although there is high annual variation in reproduction associated with weather and food (Gutierrez et al. 1995). Given these traits, individual CSOs may exhibit both short- and long-term responses to fuel treatments or wildfire, and understanding both is important to land-use managers. Our results documented a decline in CSO territories as a result of landscape fuel treatments, but the factors driving the decline remain unknown.

Conclusions

This study has shown that coordinated landscape-scale fuel treatments can substantially reduce the potential for hazardous fire across a large montane region, even when a moderate proportion of the area that could not be treated because of management constraints. In many cases, lands with designated management emphasis, such as wildlife habitat reserves and stream buffers, are distributed throughout the landscape. Creating fuel treatments that exclude these lands can result in a patchwork of treated areas heavily dissected by, for example, untreated stream buffers. Hazardous fire potential decreased in untreated areas, but that effect is not stable over time. Even if the existing network was maintained in a "treated" condition (i.e., periodic prescribed fire to keep surface and ladder fuels low) hazards will continue to increase in untreated areas because of stand development (Collins et al. 2013).

Our results indicate negative CSO responses to treatments, supported by the avoidance of DFPZs by foraging owls, larger owl home ranges associated with increasing amounts of treatment within the home ranges, and a 43% decline in the number of territorial CSO sites across the Meadow Valley study area within 3–4 years of the implementation of landscape treatments. In addition to changes in the number of owls, we also observed spatial redistribution of owl sites over time across the landscape (figure 10). The specific mechanisms driving these observations are unclear, but given the region-wide decline in the CSO population (Conner et al. 2013) and the increasing barred owl populations, it is difficult to disentangle fuel treatment effects from background or external pressures. Despite the challenges of working at landscape scales, studies such as this provide opportunities for addressing scale-dependent ecological phenomena, such as population-level species responses and responses to management strategies that cannot be addressed at smaller spatial scales.

To date, little discussion has been focused on what may constitute sustainable, viable CSO populations under various landscape conditions designed to address projected fire and climate scenarios. Furthermore, there is not a clear understanding of the balance between the potential short-term impacts from treatments and the longer-term benefits provided by introducing landscape heterogeneity (North et al. 2009), reducing potential for severe fire (Ager et al. 2007, Collins et al. 2013), increasing the potential for more desirable fire effects (North et al. 2012), and increasing resilience to climate change (Stephens et al. 2010). The Meadow Valley study is an important step in learning about the responses of wildlife species to fuel-reduction treatments.

Recent research in Yosemite National Park suggests that CSOs are not adversely affected by low- to moderate-severity fire (Roberts et al. 2011, Lee et al. 2013). Studies of the CSO both in Yosemite and in Sequoia and Kings Canyon National Parks have not shown population declines that have been found in several national forests in California. There are many differences between the two ownerships: National forest lands generally contain younger forests and lack the large tree structures associated with preferred owl habitat. With continued fire suppression, national forest lands continue to develop dense, small-tree stand conditions, reducing the habitat heterogeneity associated with a variety of small mammals that constitute the CSO's prey base. Because of these differences, it is difficult to determine whether more recent mechanical treatments or existing fire-suppressed conditions might be associated with declining CSO populations. Uncertainty also persists regarding the potential thresholds at which the amounts and patch sizes of high-severity fire reduce the postfire probabilities of CSO occupancy, survival, and reproduction. This is a significant information gap, given the trend for increasing amounts and patch sizes of high-severity fire in many Sierra Nevada forests (Miller et al. 2009). Unfortunately, only one CSO pair in Meadow Valley used an area that received prescribed burn treatments, but unlike those in some of the mechanically treated areas, these owls continued to occupy the burned area through the duration of the study and foraged within the burn-treatment areas (Gallagher 2010). The introduction of barred owls to Meadow Valley adds another important factor that may

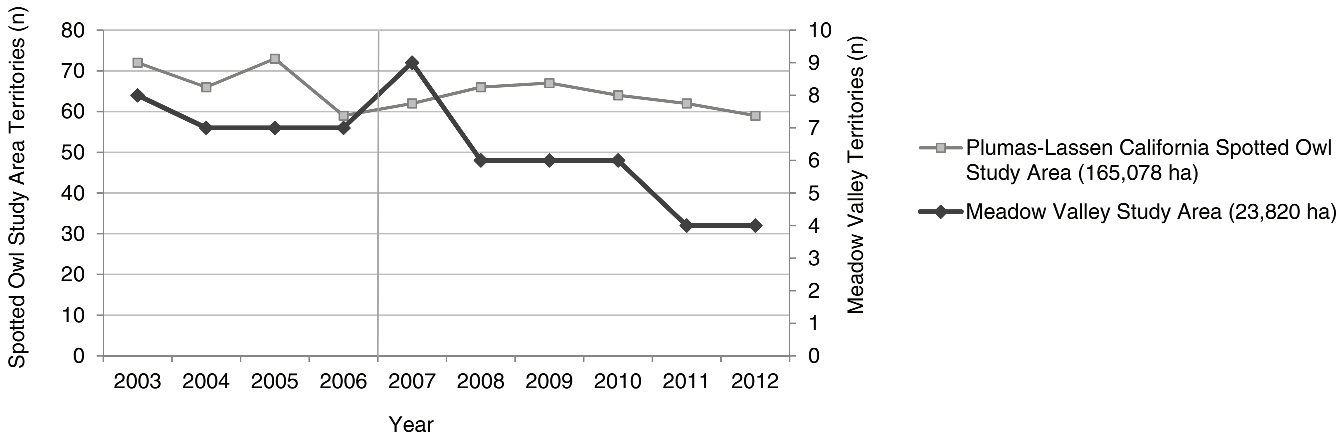

Figure 11. The annual number of territorial California spotted owl sites from 2003 to 2012 within the Meadow Valley study compared with the rest of the Plumas-Lassen study area (in the Plumas and Lassen National Forests). Vertical line represents completion of >80% of treatments.

reduce the population and viability of the CSO, possibly independent of forest structure.

Mechanical treatments can reduce fuels, but, in this study, they also left the largest trees and retained more than 40% canopy cover, two structural characteristics associated with CSO habitat use (Verner et al. 1992). However, although mechanical treatments retain these live features, they often remove snags for operator safety and fuel objectives; reduce tree density and canopy layering; reduce canopy cover to the minimum level (around 40%) considered to function as owl foraging habitat; and simplify the ground structure through a reduction of logs and small trees. Furthermore, DFPZ treatments are often uniformly implemented over large areas along roads, which results in extensive patches of simplified stand structure with regularly spaced trees. Another concern is that treatment size and placement are determined by land-use constraints (gentle slopes, access to roads) and opportunities to affect fire behavior. We have little information about how the location of treatments may affect CSOs' use of areas outside their core nesting locations. Several small mammals appeared to favor sites with steeper slopes (Kelt et al. 2013), possibly reflecting the spatial allocation of treatments in this landscape.

The importance of increasing heterogeneity within stands and across the landscape in mixed-conifer forests is well documented to meet restoration objectives (North et al. 2009, Stephens et al. 2010). Our ability to optimize heterogeneity at large scales may be more effectively achieved with prescribed and managed fires that are allowed to burn under moderate weather conditions. This type of burn often produces variable forest conditions that mimic historic patterns (Collins et al. 2011) to which this fauna, including the CSO, has adapted. Alternatively, mechanical treatments that produce the complex forest structure and composition that more closely mimic the patterns generated under a more active fire regime (North et al. 2009) may provide habitat conditions to support CSOs and a diverse fauna superior to those of the DFPZ and group-selection treatments implemented in Meadow Valley. Although mean stand conditions (e.g., canopy cover) have often been used to infer management impacts on preferred habitat (Tempel et al. in press), the historic heterogeneity of frequent-fire forests suggests we have yet to identify the optimal scales at which to create variable forest conditions.

We encourage further work to examine landscape-level treatments that are intended to emulate the influence of fire in creating spatial heterogeneity in vegetation and fuel conditions. A working hypothesis is that increased habitat heterogeneity, including the retention and development of currently limited but ecologically important forest conditions (areas of large, old trees) and more-open, patchy, early-seral stage conditions, would promote a diverse wildlife community while providing a more fire-resilient landscape. The results from the Meadow Valley study area illustrate the benefits and challenges of working at the landscape scale. Rigorous and controlled experiments are difficult because of the inherent variability across landscapes, sociopolitical constraints, and competing management objectives that can influence planned treatments. However, inferences from these studies can be strengthened by careful replication of management strategies across multiple landscapes.

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Supplemental material

The supplemental material is available online at *http://bioscience.oxfordjournals.org/lookup/suppl/doi:10.1093/biosci/biu137/-/DC1*.

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Frontiers in Ecology and the Environment

The forgotten stage of forest succession: early-successional ecosystems on forest sites

Mark E Swanson, Jerry F Franklin, Robert L Beschta, Charles M Crisafulli, Dominick A DellaSala, Richard L Hutto, David B Lindenmayer, and Frederick J Swanson

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The forgotten stage of forest succession: early-successional ecosystems on forest sites

Mark E Swanson[1*], Jerry F Franklin[2], Robert L Beschta[3], Charles M Crisafulli[4], Dominick A DellaSala[5], Richard L Hutto[6], David B Lindenmayer[7], and Frederick J Swanson[8]

Early-successional forest ecosystems that develop after stand-replacing or partial disturbances are diverse in species, processes, and structure. Post-disturbance ecosystems are also often rich in biological legacies, including surviving organisms and organically derived structures, such as woody debris. These legacies and post-disturbance plant communities provide resources that attract and sustain high species diversity, including numerous early-successional obligates, such as certain woodpeckers and arthropods. Early succession is the only period when tree canopies do not dominate the forest site, and so this stage can be characterized by high productivity of plant species (including herbs and shrubs), complex food webs, large nutrient fluxes, and high structural and spatial complexity. Different disturbances contrast markedly in terms of biological legacies, and this will influence the resultant physical and biological conditions, thus affecting successional pathways. Management activities, such as post-disturbance logging and dense tree planting, can reduce the richness within and the duration of early-successional ecosystems. Where maintenance of biodiversity is an objective, the importance and value of these natural early-successional ecosystems are underappreciated.

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Severe natural disturbances – such as wildfires, windstorms, and insect epidemics – are characteristic of many forest ecosystems and can produce a "stand-replacement" event, by killing all or most of the dominant trees therein (Figure 1). Typically, limited biomass is actually consumed or removed in such events, but many trees and other organisms experience mortality, leaving behind important biological legacies (structures inherited from the pre-disturbance ecosystem; Franklin *et al.* 2000), including standing dead trees and downed boles (tree trunks; Franklin *et al.* 2000). Such legacies provide diverse physical/biological properties and suitable microclimatic conditions for many species. Thereafter, species-diverse plant communities develop because substantial amounts of previously limited resources (light, moisture, and nutrients) become available. These emerging plant communities create additional habitat complexity and provide various energetic resources for terrestrial and aquatic organisms.

In a nutshell:

- Naturally occurring, early-successional ecosystems on forest sites have distinctive characteristics, including high species diversity, as well as complex food webs and ecosystem processes
- This high species diversity is made up of survivors, opportunists, and habitat specialists that require the distinctive conditions present there
- Organic structures, such as live and dead trees, create habitat for surviving and colonizing organisms on many types of recently disturbed sites
- Traditional forestry activities (eg clearcutting or post-disturbance logging) reduce the species richness and key ecological processes associated with early-successional ecosystems; other activities, such as tree planting, can limit the duration (eg by plantation establishment) of this important successional stage

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The ecological importance of early-successional forest ecosystems (ESFEs) has received little attention, except as a transitional phase, before resumption of tree dominance. In forestry, this period is often called the "cohort re-establishment" or "stand initiation" stage, with attention obviously focused on tree regeneration and the re-establishment of closed forest canopies (Franklin *et al.* 2002). Ecological studies have focused primarily on plant-community development and the needs of selected animal (mostly game) species, and not on the diverse ecological roles of ESFEs.

Here, we highlight important features of ESFEs, including their role in sustaining ecosystem processes and biodiversity, so that they may be appropriately considered by resource managers and scientists, and included within management/research programs dedicated to maintaining these functions, particularly at larger spatio-temporal scales. Most published examples focus on sites in western North America, but ESFEs are important elsewhere (Angelstam 1998; DeGraaf *et al.* 2003). We also discuss how traditional forestry practices, such as clearcutting, tree planting, and post-disturbance logging, can affect early-successional communities.

Figure 1. *Stand-replacement disturbance events in forests create large areas free of tree dominance and rich in physical and biological resources, including legacies of the pre-disturbance ecosystem.*

(Franklin *et al.* 2002). Consequently, the duration of ESFEs varies inversely with rapidity of tree regeneration and growth, which, in turn, depend on such variables as tree propagule availability, conditions affecting seedling or sprout establishment, and site productivity. ESFE longevity after natural disturbances is therefore highly variable.

Development of a closed forest canopy may require a century or more in areas with limited seed sources, harsh environmental conditions, severe shrub competition (in some instances), or combinations thereof (Hemstrom and Franklin 1982). For example, tree canopy closure after wildfire in the Douglas fir region of western North America often requires several decades (Poage *et al.* 2009), but can occur much more rapidly when canopy seedbanks are abundant (eg Larson and Franklin 2005). Closed forest canopies may develop quickly in forests dominated by trees with strong sprouting ability (eg many angiosperms) or when windstorms "release" understories of shade-tolerant tree seedling banks by removing all or most of the overstory (Foster *et al.* 1997).

■ Early-successional ecosystems on forest sites

Initial conditions after stand-replacing forest disturbances vary generically, depending on the type of disturbance; this includes the types of physical and biological legacies available. For example, aboveground vegetation may be limited immediately after the disturbance, as in the case of severe wildfires or volcanic eruptions. Conversely, intact understory communities may persist where forests have been blown down by severe windstorms. Spatial heterogeneity in conditions is characteristic, given that disturbances vary greatly in the amount of damage they cause (Turner *et al.* 1998). For instance, severe wildfires frequently include substantial areas of unburned as well as low to medium levels of mortality, creating variability in shade, litterfall, soil moisture, seed distribution, and other factors.

We define ESFEs as those ecosystems that occupy potentially forested sites in time and space between a stand-replacement disturbance and re-establishment of a closed forest canopy. These ecosystems undergo compositional and structural changes (succession) during their occupancy of a site. Changes begin immediately post-disturbance, as a result of the activities of surviving organisms (eg plants, animals, and fungi), including plant growth and seed production. Developmental processes are enriched by colonization of flora and fauna from outside the disturbed area. Successional change is often characterized by progressive dominance of annual and perennial herbs, shrubs, and trees, although all of these species are typically represented throughout the entire sequence of forest stand development (or sere; Halpern 1988).

The ESFE developmental stage ends with re-establishment of tree cover that is sufficiently dense to suppress and often eliminate many smaller shade-intolerant plants

■ Attributes of early-successional ecosystems

After severe disturbances, forest sites are characterized by open, non-tree-dominated environments, but have high levels of structural complexity and spatial heterogeneity and retain legacy materials.

Environmental conditions

Removal of the overstory forest canopy during disturbances dramatically alters the site's microclimate, including light regimes. These changes lead to increased exposure to sunlight, more extreme temperatures (ground and air), higher wind velocities, and lower levels of relative humidity and moisture in litter and surface soil. Shifts in these environmental metrics favor some species, while creating suboptimal or intolerable conditions for others. For example, post-disturbance plant community composition, cover, and physiognomy are altered as shade-tolerant understory herbs are largely displaced by shade-intolerant and drought-tolerant species. New substrates deposited by floods or volcanic eruptions may lack nutrients, provide additional water-holding capacity, or have high albedo, all of which favor shifts in plant communities.

Survivors

Organisms (in a variety of forms) that survive severe disturbances are extremely important for repopulating and

restoring ecosystem functions in the post-disturbance landscape. Even in severely disturbed areas, organisms may survive as individuals (mature or immature) or as reproductive structures (eg spores, seeds, rootstocks, and eggs), which become in situ propagule sources. For example, after the 1980 volcanic eruption of Mount St Helens (Washington State), most pre-eruption flora and many fauna (especially aquatic and burrowing terrestrial species) survived within the blast zone through several different mechanisms (Dale *et al*. 2005).

Surviving organisms are also often vital for the prompt re-establishment of important ecosystem functions, such as conservation of nutrients and stabilization of substrates. For instance, the important role of resprouting vegetation in curbing massive losses of nitrogen was demonstrated by experimentally clearcutting and applying herbicides in a watershed at Hubbard Brook Experimental Forest (Bormann and Likens 1979).

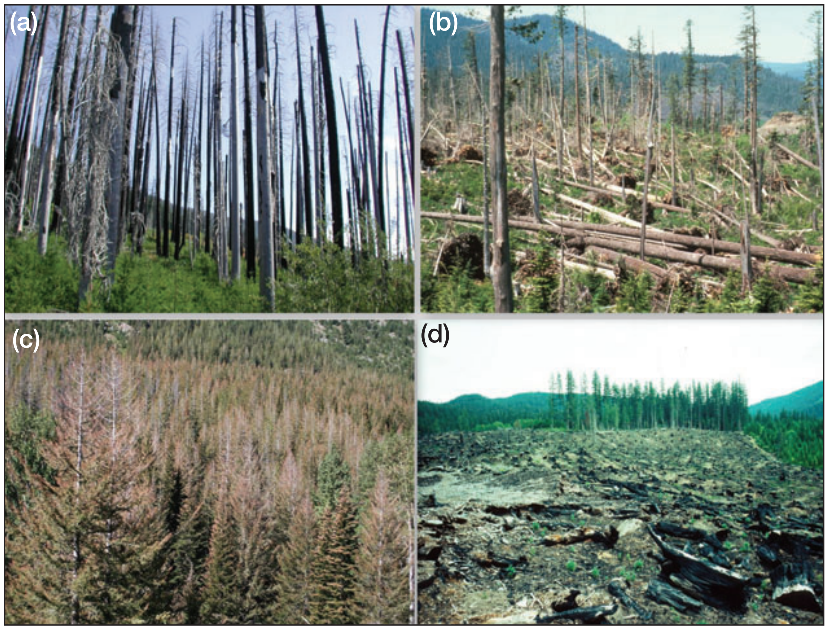

Figure 2. *Different types of disturbances produce different types of biological legacies, including living organisms and structures: (a) standing dead trees (snags) are dominant structural legacies after severe wildfires; (b) downed tree trunks and nearly intact understory communities are characteristic legacies after major windstorms; (c) standing dead trees are also dominant structural legacies after heavy insect infestations; and (d) clearcuts typically eliminate most aboveground structural legacies. Values for each metric are shown in Table 1 and are described in detail in the text.*

Structural complexity

The structural complexity of ESFEs depends initially on legacies, the general nature of which varies with the type of disturbance (Table 1; Figure 2); for example, snags and shrubs originating from belowground perennating (ie resprouting) parts or seeds are dominant legacies after wildfires, whereas downed boles and largely intact understories are typical post-disturbance characteristics of windstorms.

Woody legacies, such as snags and downed boles, play numerous roles in structuring and facilitating the development of the recovering ecosystem – providing habitat for survivors and colonists, moderating the physical environment, enriching aquatic systems in the disturbed area (Jones and Daniels 2008), and providing long-term sources of energy and nutrients (Harmon *et al*. 1986). Although subject to decomposition, these legacies can persist for many decades and sometimes even centuries.

Table 1. Different types of intense disturbances generate different types of biological legacies

	Disturbance				
Biological legacies	*Wildfire*	*Wind*	*Insect*	*Volcano*	*Clearcut*
Live trees	Infrequent	Variable	Variable (depends on stand composition)	Infrequent – confined to margins	Infrequent or absent
Snags	Abundant	Variable	Abundant	Abundant (spatially variable)	Infrequent or absent
Downed woody debris	Variable, but typically abundant	Abundant	Variable, but eventually abundant	Abundant (spatially variable)	Infrequent
Undisturbed understory	Infrequent	Abundant	Abundant	Infrequent – confined to disturbance margins	Infrequent
Spatial heterogeneity of recovery	High	Variable	High	High	Variable – usually low
Time in early-successional condition	Variable	Variable	Long	Variable – usually long	Variable – usually short

Figure 3. *Plant communities with well-developed shrub and perennial herb species are characteristic of early-successional communities on forest sites and provide diverse food resources. Twenty-five years after the Mount St Helens eruption in 1980, this community, which was within the blast zone, includes well-developed shrubs (eg* Sorbus *and* Vaccinium *spp), trees, and perennial herbs (eg* Epilobium angustifolium*).*

Structural complexity is further enhanced by the establishment and development of a variety of plant species, which often include perennial herbs and shrubs characteristic of open environments, as well as individual trees (Figure 3). The diversity of plant morphologies (maximum height, crown width, etc) increases structural richness, so that this associated flora contributes to both horizontal and vertical heterogeneity.

Spatial heterogeneity

Spatial heterogeneity is evident in early-successional ecosystems and has multiple causes: (1) natural variability in the geophysical template (topography and lithology) of the affected landscape; (2) variability in conditions in the pre-disturbance forest ecosystem; (3) variability in the intensity of the disturbance event; and (4) variability in rates and patterns of subsequent developmental processes in the ESFE. The first two sources relate to existing geophysical and biological patterns within the disturbed area. Land formations and patterns of geomorphic processes are certainly key geophysical elements (Swanson *et al.* 1988). The presence of surface water, such as streams and ponds, can be particularly influential in facilitating survival and re-establishment of biota.

Natural disturbances create heterogeneous environments at multiple spatial scales (Heinselmann 1973), because disturbances do not cause damage uniformly. Disturbances such as wildfires and windstorms are variable in intensity (eg "spotting", or initiation of new flame fronts by wind-thrown firebrands, during fire events). Alternatively, geographic variation in environmental conditions and topography (Swanson *et al.* 1988) influences the intensity of the disturbance and results in heterogeneity at multiple scales. Variability in the structure and composition of the pre-disturbance forest also creates spatial and temporal variability (Wardell-Johnson and Horowitz 1996). Some of these patterns may be transient, such as residual snowbanks protecting tree regeneration after the aforementioned Mount St Helens eruption (Dale *et al.* 2005).

Post-disturbance developmental processes also lead to spatial heterogeneity. For example, varying distances to sources of tree seed result in different rates and densities of tree re-establishment (Turner *et al.* 1998). Structural legacies can greatly influence the rates at which wind- or waterborne organic (including propagules) and inorganic materials are deposited. Finally, animal activity can strongly influence patterns of revegetation, as illustrated by the multiple effects that gophers (*Thomomys* spp) can have on post-disturbance landscapes (Crisafulli *et al.* 2005b) or the way ungulate browsing may impede tree regeneration (Hessl and Graumlich 2002).

■ Biological diversity

ESFEs in temperate forest seres show great diversity in the abundance of plant and animal species (Fontaine *et al.* 2009). Species composition may consist of a mix of forest survivors, opportunists, or ruderals (plants that grow on disturbed or poor-quality lands), and habitat specialists that co-exist in the resource-rich ESFE environment (Figure 3). Most forest understory flora can survive disturbances as established plants, perennating rootstocks, or seeds. In one study, in western North America, over 95% of understory species survived the combined disturbance of logging and burning of an old-growth Douglas-fir–western hemlock stand (Halpern 1988). Some important early-successional species (eg *Rubus* spp [blackberry; raspberry], *Ribes* spp [gooseberry], and *Ceanothus* spp [buckbrush]) may persist as long-lived seedbanks.

Opportunistic herbaceous species are often conspicuous dominants early in the development of ESFEs (Figure 4). Many of these weedy species (particularly annuals) decline quickly, although other opportunists will persist as part of the plant community until overtopped by slower growing shrubs or trees. Consequently, diverse plant communities of herbs, shrubs, and young trees emerge in ESFEs; this, combined with the structural legacies from the pre-disturbance ecosystem, often results in high levels of structural richness (Figure 3).

Many animals, including habitat specialists and species typically absent from the eventual tree-dominated com-

munities, thrive under the conditions found in ESFEs. For some species, this is the only successional stage that can provide suitable foraging or nesting habitat. As an example, many butterflies and moths (Lepidoptera) found in forested regions depend on the high diversity and quality of plant forage in ESFEs (eg Miller and Hammond 2007), whereas jewel beetles (Coleoptera: Buprestideae) depend on abundant coarse woody debris. Also, a number of ground-dwelling beetle species occur as habitat specialists in early-successional communities (Heyborne *et al.* 2003).

Figure 4. *Early-successional communities are often dominated by annual herbaceous species for the first few years after disturbance; these are quickly displaced by perennial herbaceous species and shrubs.*

Many vertebrates also respond positively to ESFEs, which may provide the only suitable habitat at a regional scale for some species. Ectothermic animals, such as reptiles (eg Rittenhouse *et al.* 2007), generally respond favorably to sunnier and drier conditions, colonizing early-successional habitat or increasing in abundance if present as survivors. Many amphibians also thrive in ESFEs, provided resources such as water bodies and key structures (eg logs) are available. The diversity and abundance of amphibians in the area affected by the 1980 Mount St Helens eruption is illustrative (Crisafulli *et al.* 2005a); eleven of 15 amphibian species survived the event, and some (eg western toad, *Bufo boreas*) have since had exceptional breeding success.

The broad array of birds using the abundant and varied food sources (eg fruits, nectar, herbivorous insects) and nesting habitat in ESFEs includes many raptors and neotropical migrants, often making bird diversity highest during the ESFE stage of succession (Klaus *et al.* in press). Some species are habitat specialists that directly utilize the legacy of recently killed trees; for instance, black-backed woodpeckers (*Picoides arcticus*) are almost completely restricted to early post-fire conditions (Hutto 2008). Mountain bluebirds (*Sialia currucoides*) and several other woodpecker species also favor structurally rich, early-successional habitats (Figure 5). Observed population declines of many avian species in eastern North America – which, in some cases, have proceeded to a point of conservation concern – are linked to conversion of early-successional habitat to closed forest (Litvaitis 1993).

Small mammal communities in ESFEs typically show high levels of diversity as well, including some obvious habitat specialists. The eastern chestnut mouse (*Pseudomys gracilicaudatus*), for example, inhabits early-successional environments in coastal eastern Australia for 2–5 years after a wildfire, and then declines dramatically until these environments are burned again (Fox 1990). Populations of mesopredators (medium-sized predators, such as raccoons [*Procyon lotor*] and fox species) benefit from the abundance of small vertebrate prey items characteristic of ESFEs. Likewise, some species of large mammals are well known to favor ESFEs (Nyberg and Janz 1990). Utilizing the diverse and luxuriant forage characteristically present in these ecosystems, ungulates, such as members of the Cervidae, in turn serve to benefit large predators (eg wolves [*Canis lupus*]) as well as scavengers, making ESFEs important elements within those species' typically extensive home ranges. Omnivores, such as bears (*Ursus* spp), also rely on the diversity of food sources often present in ESFEs.

Food web diversity

ESFEs are exceptional in the diversity and complexity of food webs they support. Simply stated, a diverse plant community produces many food sources. Food resources for herbivores (grasses, shrubs, forbs) – as well as nectar, seeds, and shrub-borne fruit (eg produced by *Rubus* and *Vaccinium* spp [huckleberry]) – can reach high levels before site dominance by trees. In the temperate Northern Hemisphere, biologically important berry production is maximized in slowly reforesting ESFEs. Resource production in early-successional patches may even augment the richness of adjacent undisturbed forests, as in the case of fluxes of key prey species (Sakai and Noon 1997).

Aquatic biologists have, perhaps, best appreciated the greater complexity of food chains in early-successional versus closed forest environments (Bisson *et al.* 2003). In established forest stands, trees strongly dominate the physical and biological conditions in nearby small streams by controlling light and temperature, stabilizing channels, providing woody debris, and, importantly, offering allochthonous inputs (organic matter originating outside the aquatic ecosystem) – the primary energy and nutrient source for such ecosystems (Vannote *et al.* 1980).

Stand-replacement disturbances remove forest constraints on conditions and processes, and shift streams to an early-

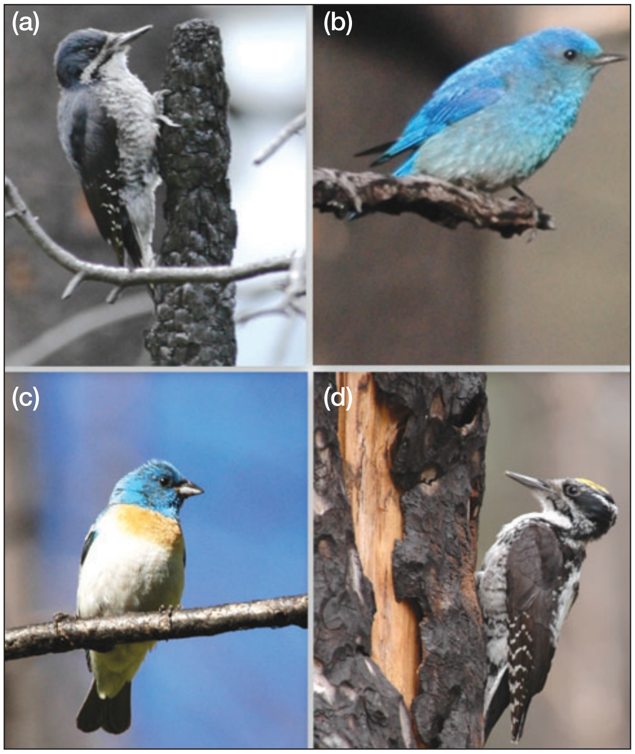

Figure 5. *Bird diversity is typically high in early-successional communities on forest sites and includes many habitat specialists: (a) black-backed woodpeckers (*Picoides arcticus*) are almost entirely restricted to early post-fire habitat; (b) mountain bluebirds (*Sialia currucoides*) favor early-successional ecosystems; (c) lazuli buntings (*Passerina amoena*) and (d) three-toed woodpeckers (*Picoides tridactylus*) have similar requirements.*

successional context (Minshall 2003; Figure 6). This greatly diversifies the types and timing of allochthonous inputs, as well as increases primary productivity. Allochthonous inputs are shifted from primarily tree-derived litter (coniferous-based in many systems) to material from a range of flowering herbs, shrubs, and trees, as well as from conifers. Consequently, litter inputs are highly variable in quality (eg decomposability) and delivery time, as compared with litterfall contributed primarily by evergreen conifer species. Also, inputs to post-disturbance streams often include material with a high nitrogen content, such as litter from the early-successional genera *Alnus* and *Ceanothus* (Hibbs *et al.* 1994).

Greater algal production may increase the diversity and abundance of aquatic invertebrate populations, which, in turn, become prey for fish and other organisms. However, increases in sediment production associated with disturbances can negate some benefits to aquatic processes and organisms (Gregory *et al.* 1987).

■ Processes in ESFEs

Ecosystem processes in ESFEs can be more diverse than those in closed forest systems, where the primary productivity of trees is dominant and organic matter is processed primarily through detrital food webs. Development of more diverse, and perhaps more "balanced", trophic pathways is possible when a disturbance opens a previously closed forest canopy. The contrast is probably greatest in forests dominated by a single tree type, such as evergreen conifers, as opposed to more diverse forests, such as mixed evergreen associations.

Recharging nutrient pools

ESFEs provide major opportunities for recharge of nutrient pools, such as additions to the nitrogen pool by leguminous (eg *Lupinus*) and some non-leguminous early-successional (eg *Alnus* and *Ceanothus*) plant species. These genera are commonly absent from late-successional forests, but are well represented in ESFEs. Nitrogenous additions from these sources are particularly important where the disturbance – eg a wildfire – has volatilized a substantial amount of the existing nitrogen pool.

Mineralization rates of organic material are typically accelerated (sometimes profoundly) after disturbances, as a result of warmer growing season temperatures. Diversified litter inputs in ESFEs, including a greater proportion of easily decomposed litter from herbs and deciduous shrubs, also result in more rapid mineralization. Finally, successional changes in the fungal and microbial communities can also hasten decomposition processes. As noted, these changes will be most profound in forest ecosystems dominated by a single species, including evergreen conifers or hard-leaved, evergreen hardwoods (such as the ash-type eucalypt forests of southeastern Australia).

In aquatic ecosystems that experience fire in adjacent forests, greater post-disturbance light and nutrient availability enhance primary productivity within the water body, causing shifts in food webs from the level of primary producers up through high-level consumers, such as fish (Spencer *et al.* 2003).

Modifying hydrologic and geomorphic regimes

Hydrologic regimes associated with ESFEs contrast greatly with those characterizing closed forest cover. For example, transpiration and interception are dramatically reduced and recover only gradually as forest canopies redevelop. Increases in normally low summer flows and annual water yields may occur immediately after a disturbance, as compared with levels in the dense young forests that may subsequently develop (Jones and Post 2004). The opposite may be true in systems where condensation of cloud or fog on tree crowns is an important component of the hydrologic cycle. ESFEs may also contribute to increased discharge peak runoff flows in hydrologic events of smaller magnitude (Harr 1986), but appear to have little effect on the magnitude of peak flows during large runoff events (Grant *et al.* 2008). From an ecological perspective, this may have a positive outcome, however, because floods restructure and rejuvenate many riparian communities (Gregory *et al.* 1991).

Rates and patterns of geomorphic processes, such as erosion and nutrient leaching losses, are also different between ESFEs and later successional stages. Tree death results in a loss of root strength that is critical for stabilizing soils and deeper rock layers on mountain slopes (Perry *et al.* 2008). Erosion and landslides may occur at higher rates in ESFEs, contributing to the variability of sediment budgets in watersheds (Reeves *et al.* 1995) and creating long-lasting substrates for ruderals. While enhancing erosion processes, ESFEs also provide materials and processes that counteract this effect, such as woody debris, which retain sediments and organic materials, and surviving vegetation, which stabilizes slopes and nutrient stores (eg Bormann and Likens 1979).

Figure 6. *Streams within early-successional forest ecosystems contrast with forest-dominated reaches in many ecosystem attributes, including physical parameters (temperature and insolation), structure, plant and animal composition, and ecosystem processes, such as primary productivity.*

Land management implications

Incorporating ESFE attributes into forest policy and management is highly desirable, given the numerous advantages provided by these ecosystems. Many species and ecological processes are strongly favored by conditions that develop after stand-replacement disturbances. Rapid, artificially accelerated "recovery" of disturbed forest areas (eg via dense planting) to closed forest conditions has serious implications for many species. Clearly the term "recovery" has a different meaning for such early-successional specialists or obligates.

To fulfill their full ecological potential, ESFEs require their full complement of biological legacies (eg dead trees and logs) and sufficient time for early-successional vegetation to mature. Where land managers are interested in conservation of the biota and maintenance of ecological processes associated with such communities, forest policy and practices need to support the maintenance of structurally rich ESFEs in managed landscapes. Natural disturbance events will provide major opportunities for these ecosystems, and managers can build on those opportunities by avoiding actions that (1) eliminate biological legacies, (2) shorten the duration of the ESFEs, and (3) interfere with stand-development processes. Such activities include intensive post-disturbance logging, aggressive reforestation, and elimination of native plants with herbicides.

In particular, post-disturbance logging removes key structural legacies, and damages recolonizing vegetation, soils, and aquatic elements of disturbed areas (Foster and Orwig 2006; Lindenmayer *et al.* 2008). Where socioeconomic considerations necessitate post-disturbance logging, variable retention harvesting (retention of snags, logs, live trees, and other structures through harvest) can maintain structural complexity in logged areas (Eklund *et al.* 2009).

Prompt, dense reforestation can have negative consequences for biodiversity and processes associated with ESFEs, by dramatically shortening their duration. Such efforts reduce spatial and compositional variability characteristic of natural tree-regeneration processes, promote structural uniformity, and initiate intense competitive processes that eliminate elements of biodiversity that might otherwise persist. Artificial reforestation can also reduce genetic diversity by favoring dominance by fewer tree species/genotypes, and may make the system more prone to subsequent, high-severity disturbances (Thompson *et al.* 2007). The elimination of shrubs and broad-leaved trees through herbicide application can alter synergistic relationships, such as the belowground mycorrhizal processes provided by certain shrub species (eg *Arctostaphylos* spp).

Naturally regenerated ESFEs are likely to be better adapted to the present-day climate and may be more adaptable to future climate change. The diverse genotypes in naturally regenerated ESFEs are likely to provide greater resilience to environmental stresses than nursery-grown, planted trees of the same species. Given that climate change is also resulting in altered behavior of pests and pathogens (Dale *et al.* 2001), encouraging greater tree species diversity may also increase ecosystem resilience.

Clearcutting has been proposed as a technique to create ESFEs, but this can provide only highly abridged and simplified ESFE conditions. First, traditional clearcuts leave few biological legacies (eg Lindenmayer and McCarthy 2002), limiting habitat and biodiversity potential. Second, clearcuts are often quickly and densely reforested, and often involve the use of herbicides to limit competition with desired tree species. Clearcuts can provide some early-successional functionality (eg serving as nurseries or post-breeding habitat for many bird species in the southern US; Faaborg 2002), but this service is often truncated by prompt reforestation.

Management plans should provide for the maintenance of areas of naturally developing ESFEs as part of a diverse landscape. This should be in reasonable proportion to *historical* occurrences of different successional stages, as based on region-specific historical ecology. Major disturbance events provide managers with opportunities to incorporate a greater diversity of species and processes in forest landscapes and to enhance landscape heterogeneity. Some aspects of ESFEs can be incorporated into areas managed for production forestry as well, such as through variable retention harvest methods, the incorporation of natural tree regeneration, and extending the duration of herb/shrub communities in some portions of a stand by deliberately maintaining low tree stocking levels.

Finally, we suggest that adjustments in language are needed. Ecologists and managers often refer to "recovery" when discussing post-disturbance ecosystems, inferring that early seral conditions are undesirable and need to be restored to closed canopy conditions as quickly as possible. Emphasizing recovery as the management goal fails to acknowledge the essential ecological roles played by early-successional ecosystems on forest sites. It should also be considered that climate change and other factors may not permit "recovery" to pre-disturbance conditions.

Conclusions

Twentieth-century forest management objectives were centered on wood production and, later, on conservation and development of late-successional forests. Rapid regeneration of dense timber stands was frequently seen as a way to address both of these divergent objectives. Recognizing the ecological value of early-successional ecosystems on forest sites extends the ecological concerns associated with old growth to another "rich" period in a forest sere. This represents an important development in the evolution of holistic management of forest ecosystems, whereby large landscapes are managed for diverse seral stages.

ESFEs provide a distinctive mix of physical, chemical, and biological conditions, are diverse in species and processes, and are poorly represented and undervalued in traditional forest management. Forest policy and practice must give serious attention to sustaining substantial areas of ESFEs and their biological legacies. Similarly, scientists need to initiate research on the structure, composition, and function of ESFEs in different regions and under different disturbance regimes, as well as on the historical extent of these systems, to serve as a reference for conservation planning.

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Comparing selected fire regime condition class (FRCC) and LANDFIRE vegetation model results with tree-ring data

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Abstract. Fire Regime Condition Class (FRCC) has been developed as a nationally consistent interagency method in the US to assess degree of departure between historical and current fire regimes and vegetation structural conditions across differing vegetation types. Historical and existing vegetation map data also are being developed for the nationwide LANDFIRE project to aid in FRCC assessments. Here, we compare selected FRCC and LANDFIRE vegetation characteristics derived from simulation modeling with similar characteristics reconstructed from tree-ring data collected from 11 forested sites in Utah. Reconstructed reference conditions based on trees present in 1880 compared with reference conditions modeled by the Vegetation Dynamics Development Tool for individual Biophysical Settings (BpS) used in FRCC and LANDFIRE assessments showed significance relationships for ponderosa pine, aspen, and mixed-conifer BpS but not for spruce–fir, piñon–juniper, or lodgepole pine BpS. LANDFIRE map data were found to be ~58% accurate for BpS and ~60% accurate for existing vegetation types. Results suggest that limited sampling of age-to-size relationships by different species may be needed to help refine reference condition definitions used in FRCC assessments, and that more empirical data are needed to better parameterize FRCC vegetation models in especially low-frequency fire types.

Additional keywords: reference conditions, successional classes, Vegetation Dynamics Development Tool (VDDT).

Introduction

Altered fire regimes and associated changes in vegetation structure, composition, and fuels pose risks to biodiversity, sustainable ecosystems, and economic and community interests across the United States (USDA/USDI 2000). However, the magnitude of these risks varies between ecosystems as a result of differences in their fire and vegetation histories, successional, compositional, and structural dynamics, and the influence of invasive species (Morgan *et al*. 2001; Schoennagel *et al*. 2004). Fire exclusion over the 20th century has not affected all ecosystems uniformly, and accurate characterization of historical fire regimes and recent vegetation changes is critical to inform management decisions about the need for fuel treatments or ecological restoration across differing plant communities.

Use of historical fire regimes and vegetation conditions to inform fire and fuel management decisions in the US has been refined into the Fire Regime Condition Class (FRCC) concept (Hann and Bunnell 2001; Schmidt *et al*. 2002; Hann and Strohm 2003; Hann *et al*. 2003; Shlisky and Hann 2003). FRCC is an index that compares current with historical fire regimes and vegetation composition and structure to assess degree of departure on a scale from one (least departed) to three (most departed). FRCC is based on an assumption that historical processes and patterns (those present before widespread Euro-American settlement in the mid- to late-1800s) represent longer-term sustainable ecosystem conditions, and that greater departure in current conditions represents a greater risk for uncharacteristic fire behavior and associated ecosystem impacts. Initial coarse-level (1-km^2 resolution) FRCC maps described the degree of departure at a national scale (Schmidt *et al*. 2002). After this initial effort, a set of standard guidebook methods was developed to assess FRCC at landscape to stand scales for local management and planning needs (at time of writing, FRCC Guidebook v1.3; Hann *et al*. 2004). FRCC maps of 30-m^2 resolution are also being developed as part the LANDFIRE project, an effort to provide consistent vegetation, fuels, and fire regime data for the entire US (Rollins and Frame 2006; www.landfire.gov, accessed 19 October 2007). FRCC is now a key variable for defining wildfire risk to ecosystems as a result of its explicit incorporation into the Healthy Forests Restoration Act of 2003 (HFRA 2003). FRCC represents a significant advance in the integration of fire and forest histories and landscape and vegetation ecology to provide an ecologically based method for setting fire-management priorities and objectives across the US (Shlisky and Hann 2003).

Definition of departure indices in FRCC assessments begins with simulation modeling of historical vegetation composition and structure using the Vegetation Dynamics Development Tool (VDDT; Beukema and Kurz 2003). VDDT is used to develop non-spatially defined reference conditions within Biophysical Settings (BpS; formerly referred to as Potential Natural Vegetation Groups (PNVG); Küchler 1964; NRCS 2003). For LANDFIRE, BpS are derived from Nature Serve's ecological

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classification system (Comers *et al*. 2003) and are not directly comparable with those used in FRCC assessments. However, both systems use BpS in a similar manner to represent the vegetation communities that would likely exist under given environmental conditions (climate, soils, and landscape physiography) and historical disturbance regimes. BpS in LANDFIRE are assigned to specific locations in their nationwide mapping efforts, whereas BpS in FRCC assessments are non-spatial and assigned based on individual user needs for specific projects or management requirements. Reference conditions are the proportions of vegetation successional stages (community structure and composition) as affected by varying fire frequencies, severities, and successional pathways within each BpS (Hann *et al*. 2004).

FRCC and LANDFIRE vegetation models (also known as Vegetation Dynamics Models) were defined during regional professional workshops conducted between 2002 and 2009 (2005–09 for LANDFIRE). VDDT model inputs for individual BpS are based on historical fire regime characteristics (frequency and severity) and vegetation data derived from published and unpublished studies and expert opinion developed both at the regional workshops and through subsequent peer reviews (Hann *et al*. 2004). The amount and quality of available historical data for each BpS vary, which can affect the quality and accuracy of the resulting modeled reference conditions. In an FRCC assessment, a field evaluation is conducted of existing vegetation structure, which, in forests, is based on cover type, density of tree stands, tree size, and current successional status. Successional status is determined by visually estimating stand composition, tree density, and average tree age, the latter of which is based on tree diameters. Proportions of current successional classes in a project or management area are estimated during the field assessment and then compared with the proportions of reference conditions derived from VDDT model output. The FRCC departure index (1 to 3) is assigned based at least partially on differences in proportions of successional classes present in the current forest relative to modeled reference conditions in the historical forest.

There is a need to test the process of development of reference conditions by comparing VDDT model output with known fire and vegetation histories. This comparison is critical for assessing consistency and accuracy in the modeling process. Here, we compare VDDT-modeled reference conditions with tree-ring-based reconstructions of reference conditions from 11 forested sites in Utah and eastern Nevada (tree-ring data reported in Heyerdahl *et al*. 2005, and Brown *et al*. 2008*a*). The tree-ring reconstructions span transects aligned along elevation gradients that include multiple forest types. We ask the following questions with this comparison: (1) do FRCC methods of evaluating stand structure based on diameter estimates accurately represent ages of forest vegetation and is there variation based on species and site? (2) Do FRCC and LANDFIRE BpS models adequately capture the range of variation in proportions of reference conditions reconstructed by the tree-ring data? (3) Do LANDFIRE mapped data layers for BpS and Existing Vegetation Types (EVT) match the tree-ring plot data? (4) Can further empirical fire history and tree recruitment data be used to strengthen FRCC evaluation and reference condition modeling outputs? We consider this study to be only an initial test of FRCC and LANDFIRE vegetation modeling methods, but one that may provide an example for future testing needs.

Methods

Study area

Tree-ring sites used for this study extend from the Colorado Plateau of southern Utah, west to the Wah Wah Mountains in the eastern Great Basin of western Utah, and north to the Uinta and Bear River Mountains in northern Utah (Fig. 1, Table 1; Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). The region is a complex of valleys, mesas, canyons, plateaus, and mountains that range in elevation from ~900 to >3600 m. Forest types vary generally across elevation gradients. Piñon (*Pinus edulis* (PIED; four-letter codes are used in tables) and *P. monophylla* (PIMO)) and juniper (*Juniperus scopulorum* (JUSC) and *J. osteosperma* (JUOS)) savannas and woodlands occur at the lowest forest margins above desert shrublands or grasslands. Ponderosa pine (*Pinus ponderosa* (PIPO)) forests occur in montane zones in pure and mixed stands. Douglas-fir (*Pseudotsuga menziesii* (PSME)) often occurs in association with ponderosa pine on north-facing aspects and in relatively mesic sites. Mixed-conifer forests occur at intermediate elevations and include combinations of ponderosa pine, Douglas-fir, piñons, junipers, and firs (*Abies lasiocarpa* (ABLA) or *A. concolor* (ABCO)). Mixed-conifer forests also often occur in association with aspen (*Populus tremuloides* (POTR)). Aspen forms large (>100 ha) pure stands throughout the upper montane and lower subalpine zones across the study area except in the Great Basin. Lodgepole pine (*Pinus contorta* (PICO)) often forms pure stands at mid-elevations (1900 to 2800 m) or occurs in the mixed-conifer zone in northern Utah. Subalpine forests dominated by Engelmann spruce (*Picea engelmannii* (PIEN)) and firs occur at upper elevations (2350 to 3500 m). At the highest forested elevations (generally above 3000 m), pure Engelmann spruce forests occur in mesic sites whereas bristlecone pine (*Pinus longaeva* (PILO)) or limber pine (*P. flexilis* (PIFL)) are typically found in dry or rocky sites.

There was, in general, a gradient in fire frequency across the elevational gradient before fire exclusion that began at all sites in the late 1800s (Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). Fire occurrence was highest in the middle of the elevation range in ponderosa pine and drier mixed-conifer sites. Fire frequency progressively declined both above and below this middle-elevation zone. At upper elevations, generally moist conditions led to high fuel biomass, both living and dead, in many stands, but fewer years in which fuels were dry enough to ignite and spread. At lower elevations in the piñon–juniper woodlands, fuels were often dry enough to burn because of hotter and dryer fire seasons, but because of lower productivity, there were in general less continuous both aerial and surface fuels and fires were not able to spread. In the middle zone, both fuel amounts and moistures were just right (what has come to informally be called the ‘Goldilocks effect’), and able to burn often in wide-spreading fires.

Utah forests underwent a period of intensive grazing and land use beginning in the 1850s as a result of Euro-American settlement. Intensive grazing removed understorey species and began alteration of longer-term historical forest dynamics. Logging also affected forest structure in many areas. The tree-ring study found that cessation of historical patterns of fires began in

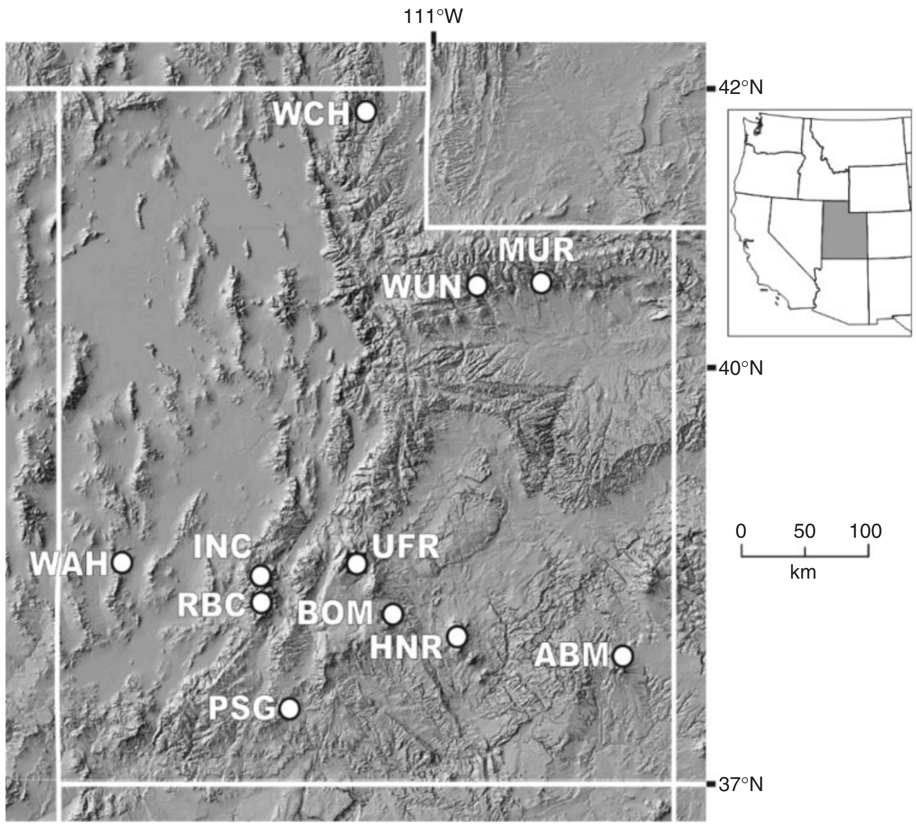

Fig. 1. Locations of tree-ring sites. Three-letter codes correspond to those in Table 1.

Table 1. Tree-ring sites used in the present study arranged from north to south
FRCC (Fire Regime Condition Class) and LANDFIRE BpS (biophysical settings) forest types are listed in Table 4

Site	Minimum elevation (m)	Maximum elevation (m)	Average precipitation (cm)	FRCC and LANDFIRE BpS
Wasatch Mountains (WCH)	2255	2588	100	SPFI, SPDF, CHPI, 10510, 10520, 10500, 10550
Western Uinta (WUN)	2207	3133	60	PPIN, SPDF, SPFI, CHPI, 10540, 10510, 10520, 10500, 10550
Middle Uinta River (MUR)	2308	3250	70	PPIN, SPDF, SPFI, CHPI, 10540, 10510, 10520, 10500, 10550
Wah Wah Mountains (WAH)	2195	2686	40	JUPI, PPIN, SPDF, 10160, 10540, 10500
Upper Fremont River (UFR)	2800	3039	80	SPDF, SPFI, 10510, 10520, 10500
Indian Creek (INC)	2364	2518	65	PPIN, SPDF, 10540, 10500
Beaver Creek (RBC)	2358	3077	90	PPIN, SPDF, SPFI, 10540, 10510, 10520, 10500
Boulder Mountain (BOM)	2405	3377	80	JUPI, PPIN, SPDF, SPFI, 10160, 10540, 10510, 10520, 10500
Henry Mountains (HNR)	2407	3138	60	JUPI, PPIN, SPDF, 10160, 10540, 10500
Abajo Mountains (ABM)	2557	3231	85	JUPI, PPIN, SPDF, SPFI, 10160, 10510, 10520, 10500
Paunsaugunt Plateau (PSG)	2309	2736	45	JUPI, PPIN, SPDF, SPFI, 10160, 10540, 10510, 10520, 10500

the 1860s to 1890s depending on location (Brown *et al*. 2008*a*), similar to patterns seen in forests throughout the western US. Initial reduction in fire frequency was likely the result of grazing that removed grass and herbaceous fuels, followed later by direct fire suppression in the 20th to 21st centuries.

Tree-ring data

The tree-ring study used a systematic sampling design to characterize stand and age structure and fire regimes across forest gradients in each site (Table 1; Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). Similar methods have been used in multiple studies

around the western US and are described in more detail in Heyerdahl *et al*. (2005, 2006), Brown and Wu (2005), Brown (2006), Brown *et al*. (2008*a*, 2008*b*), and Brown and Schoettle (2008). A 500-m grid was established at each site and plots sampled at grid points. Plot centers were located in the field using hand-held global positioning system (GPS) units. An *n*-tree density-adapted sampling method (Jonsson *et al*. 1992) was used to collect data from the nearest ~30 remnant (logs, snags, or stumps) or living trees >20 cm diameter at breast height (DBH) to each plot center. Maximum plot radius was set at 40 m (~0.5 ha) and most plots were ~<0.2 ha in size. For each plot tree, species was recorded and an increment core (on living trees) or cross-section (from logs, snags, and stumps) was collected from ~10 cm height above ground. Sampled cores had to be no more than a field-estimated 10 years from pith to minimize pith offset when assessing pith date. Diameter at sample height (DSH) and DBH were measured on living trees, and DSH was measured or estimated for remnant trees missing bark, sapwood, or heartwood. Distance from plot center and azimuth were measured on all trees for reconstruction of tree basal areas, density, and spatial patterning. To reconstruct surface fire history, cross-sections were cut from any fire-scarred trees found within plots. Additional fire-scarred trees also were sampled within ~80 m of each grid point and between grid points when discovered. GPS coordinates and species of fire-scarred trees outside of plots were recorded.

Standard dendrochronological methods were used to crossdate all samples using locally developed master chronologies (Heyerdahl *et al*. 2005). Pith dates were estimated on cores that did not intersect pith based on the curvature of the innermost rings sampled. The tree recruitment date is considered to be the date of tree pith at 10-cm height. No corrections were made for time to grow from germination to 10 cm sample heights because of the widely varying species and environmental conditions at the sites that were collected for the study. Once crossdating of ring series was completed on all samples, dates for any fire scars seen within the ring series were assigned. Any trees that were not able to be dated were not used in subsequent analyses.

FRCC and LANDFIRE vegetation models

VDDT modeling estimates the relative proportions of non-spatially defined reference conditions that would have occurred under a historical fire regime and an equilibrium (current) climate regime within each BpS (Beukema and Kurz 2003). VDDT input includes average fire frequencies, severities, and other disturbances defined as probabilistic events, and vegetation structural stage development pathways, including changes in species composition and density through a successional sequence. VDDT runs are commonly made for 500 years to allow vegetation conditions to equilibrate over time. VDDT output is proportions of vegetation successional classes – the reference conditions – across a non-spatially referenced landscape at the end of the 500-year model run. Reference conditions for most forest types are summarized into five seral stages that approximate overall developmental characteristics of community age and structure: early-replacement, mid-open, mid-closed, late-open, and late-closed. Each developmental stage represents a successional class defined by average tree age, species composition, structural characteristics, and response to disturbances. LANDFIRE and FRCC assessments use VDDT in a similar manner, but in LANDFIRE, reference condition proportions are then coupled with the spatial model LANDSUM (Keane *et al*. 2002) to map resulting vegetation conditions for each BpS across actual landscapes at a 30-m^2 spatial resolution.

FRCC and LANDFIRE developed their own BpS models using two different vegetation classification systems (Küchler 1964 *v.* Comers *et al*. 2003). Both systems attempt to describe the same historical vegetation using VDDT; however, their models use different probabilities for disturbance, and have somewhat different species distributions and geographic extents (often based on expert opinion; see http://frcc.gov, accessed 19 October 2007; www.landfire.gov for details).

Comparing tree-ring with FRCC and LANDFIRE data

We performed three tests to compare the tree-ring data with FRCC and LANDFIRE vegetation models. First, we compiled age and DBH data derived from the tree-ring study to assess whether FRCC methods of visual estimates of tree diameters accurately represent the age of forest vegetation for defining mid- and late-development classes of reference conditions. FRCC guidebook methods define >23 cm DBH as a visual indicator of a mature tree when conducting field assessments. For this analysis, we assumed that plots with trees averaging ≤23 cm DBH would be considered to be in a mid-development reference condition, and >23 cm would be in late-development. We conducted least-squares linear regressions to estimate fitness of tree age to DBH by species and site. As many of the regression models did not meet the statistical requirements of homoscedasticity, normality, and constant variance in model residuals, a logarithmic transformation was applied to the tree ages before regression. Models that had significant *P* values ($P < 0.05$) were considered to be representative of species growth estimates. We also conducted an analysis of variance (ANOVA) of age and diameter by species and site to both determine the strength of these relationships and how they varied by species and location across the region. All statistical analyses were conducted using the *Statistica* software (StatSoft Inc. 2008). The tree-ring study sampled a total of ~10 000 remnant and living trees; however, we only used data from the living trees for this part of our assessment. Dead trees (stumps, snags, and logs) often were missing bark, sapwood, or portions of the heartwood that reduced confidence in diameter estimates. The DBH-to-age analysis therefore consisted of 5173 living trees from 13 species from the 11 sites.

Our second test was whether VDDT modeled reference conditions captured the range of variation in reference conditions reconstructed by the tree-ring data as of a date of 1880. Dates of initial Euro-American settlement varied across the study area but all sites showed some Euro-American impact by 1880, including cessation of spreading fires in almost all of the sites (Brown *et al*. 2008*a*). As current vegetation may not be representative of past vegetation type, only species present in 1880 and their corresponding ages were used to assign BpS and reference condition to each of a total of 273 plots that were sampled from the 11 sites (Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). Both living and remnant trees were used to estimate the 1880 plot compositions. FRCC and LANDFIRE use key species to

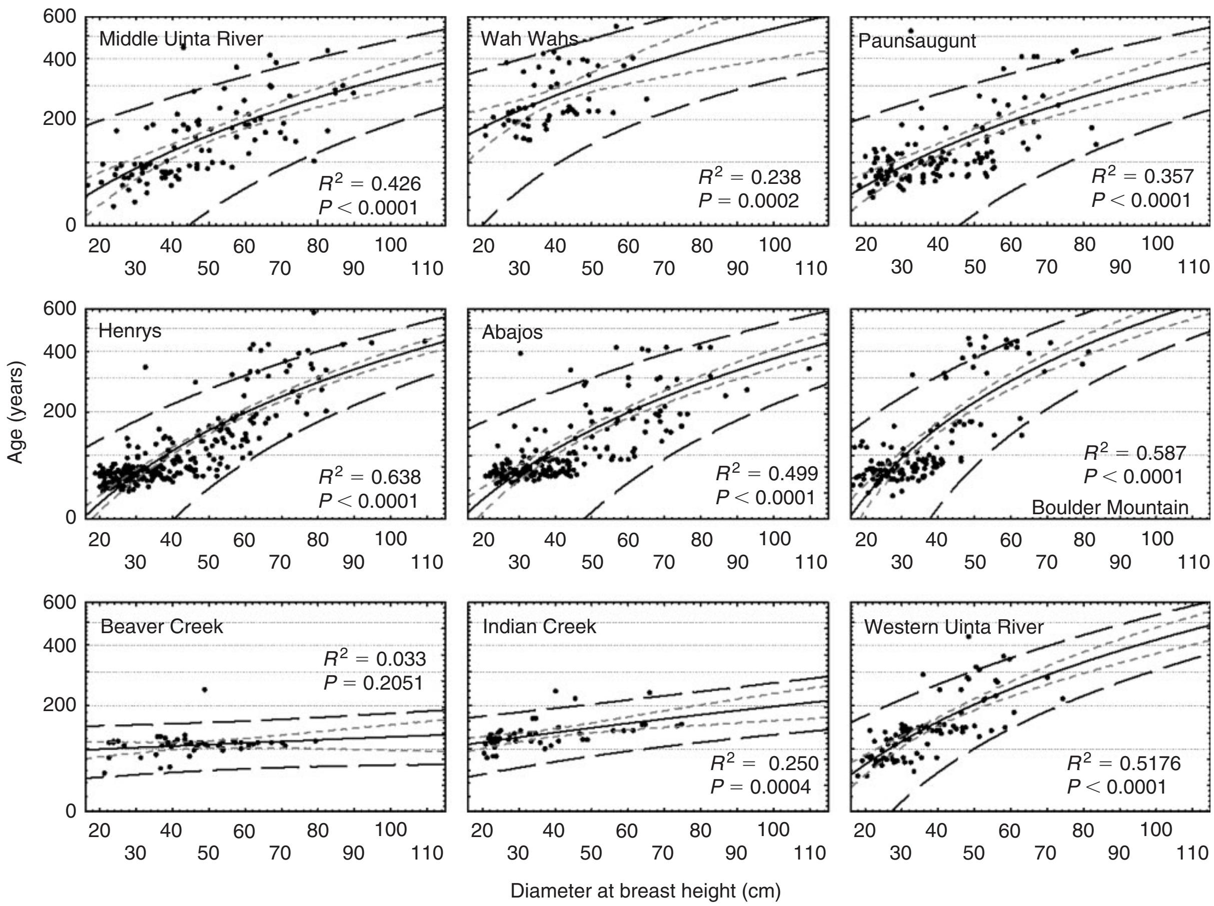

Fig. 2. Diameter at breast height (DBH) and log(age) regressions for ponderosa pine trees by site, with linear fits (solid lines), 95% confidence intervals (gray dashed lines), and 95% prediction intervals (black dashed lines). Overall R^2 for ponderosa pine trees across all sites was 0.44.

define vegetation characteristics when conducting an assessment and we used these species as the basis for assigning BpS and reference condition to each plot.

Historical age class and species composition in 1880 for each plot were compared with FRCC and LANDFIRE reference conditions for selected BpS. FRCC and LANDFIRE BpS descriptions are available on their respective project websites (www.frcc.gov; www.landfire.gov). We did not evaluate the typical five-stage VDDT models because of difficulties in using the tree-ring data to accurately recreate smaller size classes in historical stand densities as a result of probable tree mortality and decay since pre-settlement periods (e.g. Brown and Cook 2006; Brown *et al*. 2008*b*). However, we assume that we are able to define with some confidence mid- and late-development stands based on crossdated ages of trees present in each plot in 1880. The mean age of a 23-cm-DBH live tree varied by species, and we used the tree-ring results to estimate the upper 95% confidence interval for predicted tree size to consider whether a stand was late developmental stage in 1880. We grouped data from open and closed stands together based on age and composition for comparison with succession classes from VDDT output. If any trees in a plot were older than their predicted age-to-size confidence interval, the plot was considered to be in late-development in 1880. If there were no older trees during the historical period, then the plot was considered to have been in mid-development. If there were no trees during the historical period, the plot was considered to have no data and not used in this analysis. Once plots were categorized by BpS and reference condition, they were compared with FRCC and LANDFIRE BpS model proportions of mid- and late-development vegetation based on VDDT output. We used a Chi-square test to determine if the observed tree-ring reference condition proportions were significantly different than the expected based on the VDDT output.

Finally, we compared tree-ring plot data with LANDFIRE BpS and EVT map layers produced by the LANDFIRE project. LANDFIRE data are spatially mapped, which provided a unique opportunity to evaluate vegetation models at a high spatial resolution through comparison with the mapped tree-ring data. Plots were first located through their GPS coordinates relative to LANDFIRE map data. The BpS assignments we made for each plot in 1880 were then compared with LANDFIRE BpS map data. We also compiled the living tree composition in each plot and compared that with the LANDFIRE EVT map data. If key

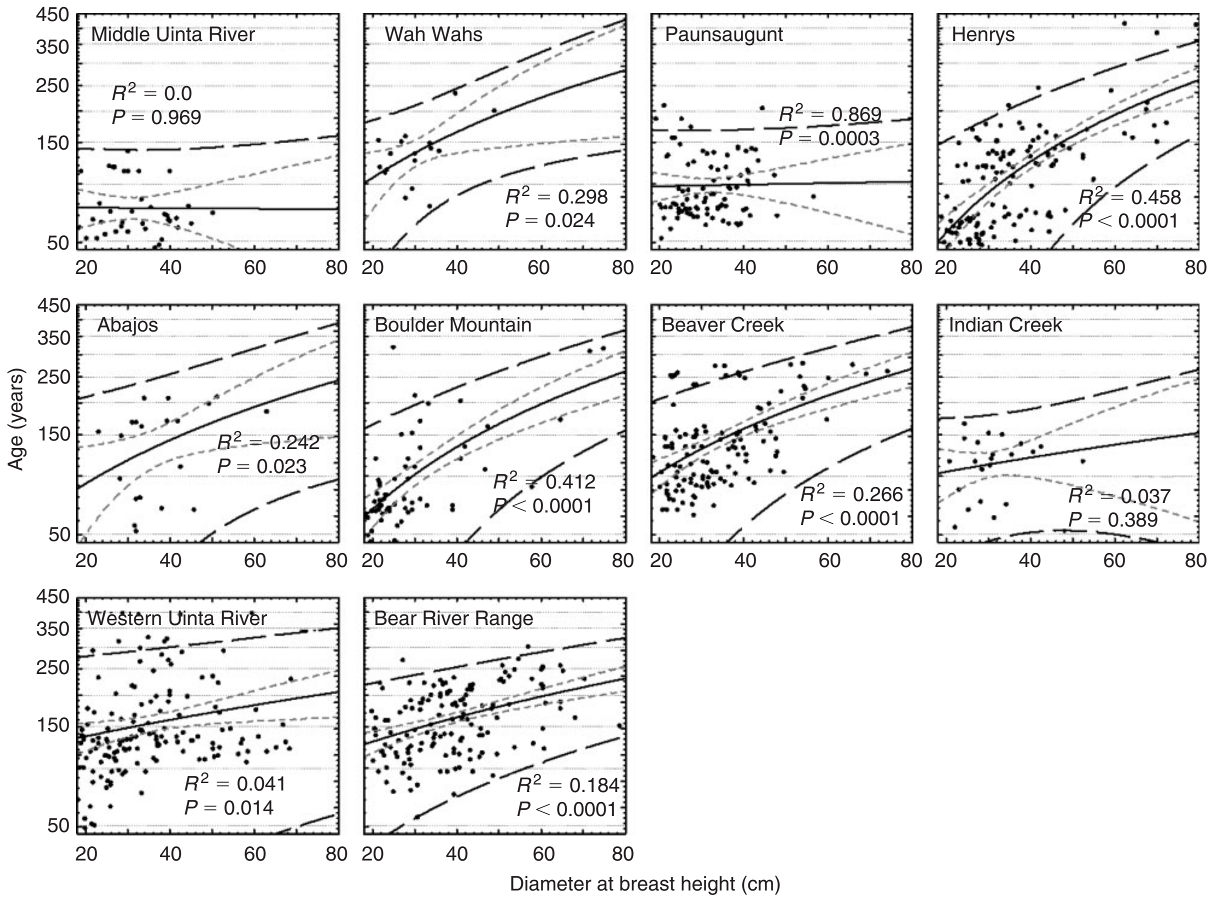

Fig. 3. Diameter at breast height (DBH) and log(age) regressions for Douglas-fir trees by site, with linear fits (solid lines), 95% confidence intervals (gray dashed lines), and 95% prediction intervals (black dashed lines). Overall R^2 for Douglas-fir trees across all sites was 0.21.

species were present in the tree-ring data in comparison with the mapped BpS or EVT, then the grid point was considered to have been accurately mapped in LANDFIRE.

Results

Age–diameter relationships

DBH and tree ages exhibited generally broad relationships, both within species and among sites (Figs 2–4; Tables 2, 3). Ponderosa pine was the only species where age and size were strongly correlated using data from all sites ($R^2 = 0.438$, $P < 0.001$) and were strongly correlated over most of the individual sites (Table 2). There were outliers for most species by DBH and age; however, their deviance did not significantly change the results. Median tree age was predicted for trees at 23 cm using an inverse prediction with 95% confidence interval (Table 3). ANOVA results indicate that species associated with infrequent fire regimes (piñon–juniper, spruce–fir, and bristlecone pine; Heyerdahl *et al.* 2005) were found to have greater average ages than frequent fire species (especially ponderosa pine and Douglas-fir; Fig. 5). Variance of diameters relative to ages for species that contained a large sample n, such as Douglas-fir (PSME), ponderosa pine (PIPO), and Engelmann spruce (PIEN) was small. There was greater variance found in species that had fewer sampled trees and plots, such as bristlecone pine (PILO), Rocky Mountain juniper (JUSC), one-seed juniper (JUOS), limber pine (PIFL), and single leaf piñon (PIMO), but this result is likely an artifact of the smaller number of trees used in each regression. ANOVA indicated that DBH and age estimates for all sites were similar with the exception of WAH (Fig. 5). This may be explained by the large presence of fire-infrequent and older species (bristlecone pine, Rocky Mountain juniper, and one-seed juniper) that were sampled in that site.

FRCC and LANDFIRE BpS models

Median ages of trees >23 cm DBH were used to define the proportions of mid- and late-development reference conditions for trees present in plots in 1880 (Table 3). Reference condition proportions reconstructed from the tree-ring data compared favorably with FRCC BpS models for ponderosa pine (PPIN5), mixed-conifer (SPDF), and lodgepole (CHPI) but not for piñon–juniper (JUPI1, JUPI2), south-western mixed-conifer (MCAN) and spruce–fir (SPFI2, SPFI7; Table 4, Fig. 6). Reference condition proportions reconstructed from the tree-ring data

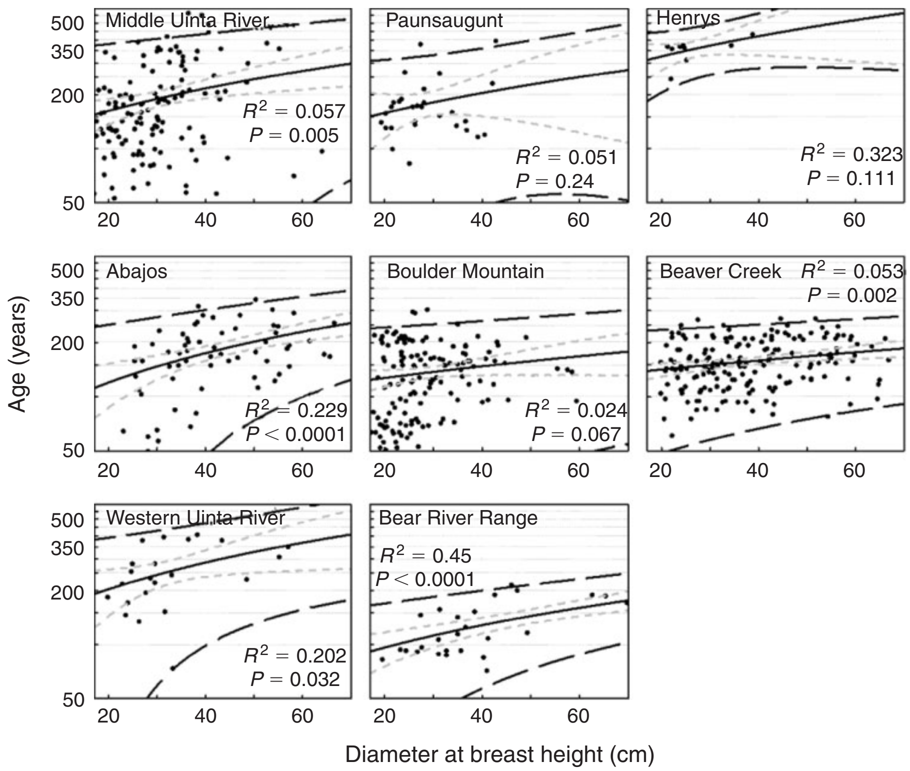

Fig. 4. Diameter at breast height (DBH) and log(age) regressions for Engelmann spruce trees by site, with linear fits (solid lines), 95% confidence intervals (gray dashed lines), and 95% prediction intervals (black dashed lines). Overall R^2 for Engelmann spruce trees across all sites was 0.06.

Table 2. Observed two-sided *P* values for DBH–age regressions for all species at all sites

Bold values represent locations where *P* values are significant at the 95% confidence interval (<0.05) based on sample size (>10 trees)

Species	Site									
	WCH	RBC	ABM	BOM	HNR	PSG	INC	WUN	MUR	WAH
PIPO		0.021	**<0.0001**	**<0.0001**	**<0.0001**	**<0.0001**	**0.0004**	**<0.0001**	**<0.0001**	**0.0002**
PSME	**<0.0001**	**<0.0001**	**0.0234**	**<0.0001**	**<0.0001**	0.88	0.39	**0.01**	0.969	0.024
PIEN	**<0.0001**	**<0.0001**	**<0.0001**	0.066	0.111	0.241		0.03	0.005	
ABLA	**<0.0001**	0.19	**<0.0001**		0.147			0.37		
POTR	**0.01**	0.01	**<0.0001**	0.63	0.107		0.40	0.81	**0.020**	
ABCO		0.22				**<0.0001**	0.22		0.069	0.002
PICO	**<0.0001**				**<0.0001**	**0.0007**				
PIFL	0.28				**<0.0001**	0.090	0.28			
PIED				**<0.0001**	**<0.0001**	0.025				
PIMO										**<0.0001**
JUSC				0.152		**0.111**				0.903
JUOS				**0.0003**		0.677			0.797	**0.0002**
PILO										0.574

compared favorably with LANDFIRE BpS models for Rocky Mountain dry–mesic montane mixed-conifer (10510), aspen and aspen–mixed-conifer low- and high-elevation forests (10110, 10611, 10612), but not for piñon–juniper (10160), ponderosa pine (10540), Rocky Mountain mesic montane mixed-conifer (10520), Rocky Mountain subalpine dry–mesic spruce–fir forest and woodland (10550), and Rocky Mountain lodgepole pine (10500; Table 4, Fig. 6). The JUPI1 BpS model (Table 4) was the most different from the tree-ring data, although the JUPI2 model had a similar trend of a larger proportion of late-successional

stands in comparison with the tree-ring data (Fig. 6). Spruce–fir and lodgepole pine data both showed low correspondence with VDDT model results, including opposite trends of more older than younger stands in the tree-ring data in contrast to the VDDT modeled reference conditions (Fig. 6).

Table 3. Expected median ages of trees >23 cm DBH (diameter at breast height) by species, with lower and upper 95% confidence intervals derived from tree-ring data

NS, age–DBH regression not significant

Species	Age (years) at 23 cm	R^2	*P* value
PIPO	40.9 ± 3.2	0.438	<0.0001
JUOS	114.9 ± 41.9	0.438	<0.0001
PIED	135.3 ± 21.9	0.28	<0.0001
PIFL	66 ± 11.4	0.271	<0.0001
PIMO	176.3 ± 29.8	0.231	<0.0001
PSME	42.9 ± 6	0.213	<0.0001
PICO	54.3 ± 12.6	0.112	<0.0001
POTR	104 ± 9.1	0.095	<0.0001
PIEN	24.7 ± 14.7	0.055	<0.0001
JUSC	NS	0.05	0.0961
ABCO	14.8 ± 14.4	0.023	<0.0001
PILO	NS	0.012	0.6295
ABLA	50.2 ± 10.2	0.01	<0.0001

LANDFIRE map data

LANDFIRE map layers were found to be overall ~58% accurate for BpS and 60% accurate for EVT when compared with the tree-ring data for each plot (Table 5). LANDFIRE maps were 38% accurate for both BpS and EVT, 28% accurate for at least one type (17% EVT accurate and BpS inaccurate, with 11% BpS accurate and EVT inaccurate), and 34% inaccurate. Mixed-conifer and spruce–fir types had the highest accuracies by BpS for LANDFIRE with accuracies ranging from 64 to 82% for BpS and 67 to 79% for EVT. Piñon–juniper was the least accurately mapped BpS and EVT with 13 and 37% accuracy respectively.

Discussion

FRCC and LANDFIRE BpS models

Current stand conditions are determined through visual estimates of stand structure, including tree diameters, in FRCC assessments (Hann *et al*. 2004). FRCC assessments are designed to be a relatively rapid method of characterizing current vegetation and fire regime departures from historical conditions. The expense of collecting field data, such as canopy closure, canopy base height, tree density, stand age structure, and fire and stand histories, make field sampling impractical for FRCC assessments. However, based on the limited findings of this study, it appears that FRCC methods may result in inaccurate measures of plant community departure based on visually estimated

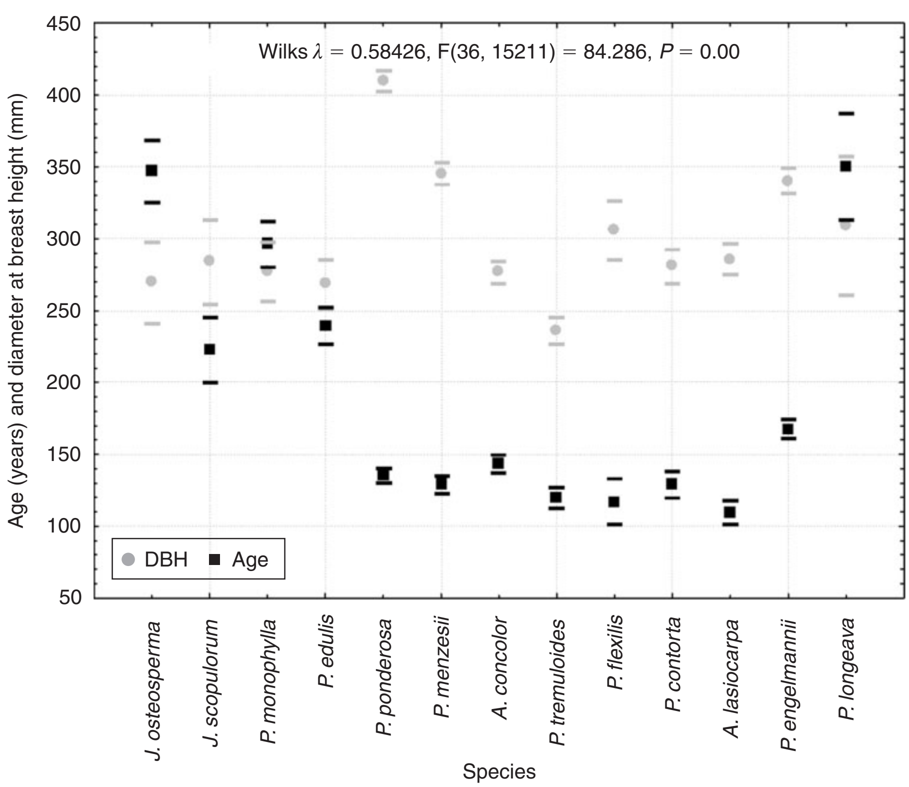

Fig. 5. ANOVA of age and diameter at breast height (DBH) by species and site. Horizontal bars represent 95% confidence intervals.

Table 4. Observed proportions of mid- and late-development reference conditions reconstructed from tree-ring data in plots collected in Utah, compared with FRCC (Fire Regime Condition Class) BpS (biophysical settings) model output for mid- and late-development reference conditions from Hann *et al*. (2004) and LANDFIRE

MFI, mean fire interval (years); *n*, number of plots in the observed data in mid- or late-seral stages. Chi-square fit is for the observed plots *v.* BpS models with 1 degree of freedom, $P = 0.05$ significance for types <3.84. BpS that meet the range of variability in the observed data are highlighted in bold

BpS description *v.* observed data	FRCC code	LANDFIRE code	Mid (%)	Late (%)	MFI	*n*		Chi-square	*P* value
						Mid	Late		
Observed PIED, PIMO, JUSC, JUPI			4	96		1	24		
Piñon–juniper infrequent fire	JUPI2		30	70	435			12.99	0.0003
Piñon–juniper frequent fire	JUPI1		50	50	31			21.16	<0.0001
Colorado Plateau piñon–juniper woodland		10160	55	45	128			26.273	<0.0001
Observed PIPO			26	74		13	37		
Colorado plateau ponderosa	**PPIN5**		25	75	6			**0.027**	0.87
Southern Rocky Mountain ponderosa pine woodland		10540	44	56	15			6.575	0.01
Observed PSME, ABCO, PIPO, PIEN			48	52		35	38		
South-western mixed-conifer	MCAN		35	65	10			5.377	0.02
Rocky Mountain dry–mesic montane mixed-conifer forest and woodland		**10510**	40	60	10			**1.92**	0.166
RM mesic montane mixed-conifer forest and woodland		10520	75	25	33			28.498	<0.0001
Spruce–fir–Douglas-fir[A]	**SPDF**		58	42	19			**0.658**	0.417
Observed PICO			36	64		4	7		
Lodgepole pine	CHPI		65	35	125			3.965	0.046
Rocky Mountain lodgepole pine forest		10500	100	0	124			4.455	0.035
Observed PIEN, ABLA, ABCO			26	74		16	58		
RM subalpine dry–mesic spruce–fir forest and woodland		10550	65	35	212			61.206	<0.0001
Lower subalpine forest	SPFI7		80	20	91			157.622	<0.0001
Upper subalpine forest	SPFI2		70	30	143			82.474	<0.0001
Observed POTR			90	10		33	4		
Deciduous woodland–oak or aspen	DWOA		55	45	100			17.474	<0.0001
Intermountain basins aspen–mixed-conifer forest – low		**10611**	85	15	10			**0.509**	0.475
Intermountain basins aspen–mixed-conifer forest – high		**10612**	95	5	32			**2.63**	0.105
Rocky Mountain aspen forest and woodland		**10110**	80	20	27			**2.146**	0.142
PILO			0	100		0	3		

[A]Includes observed POTR plots.

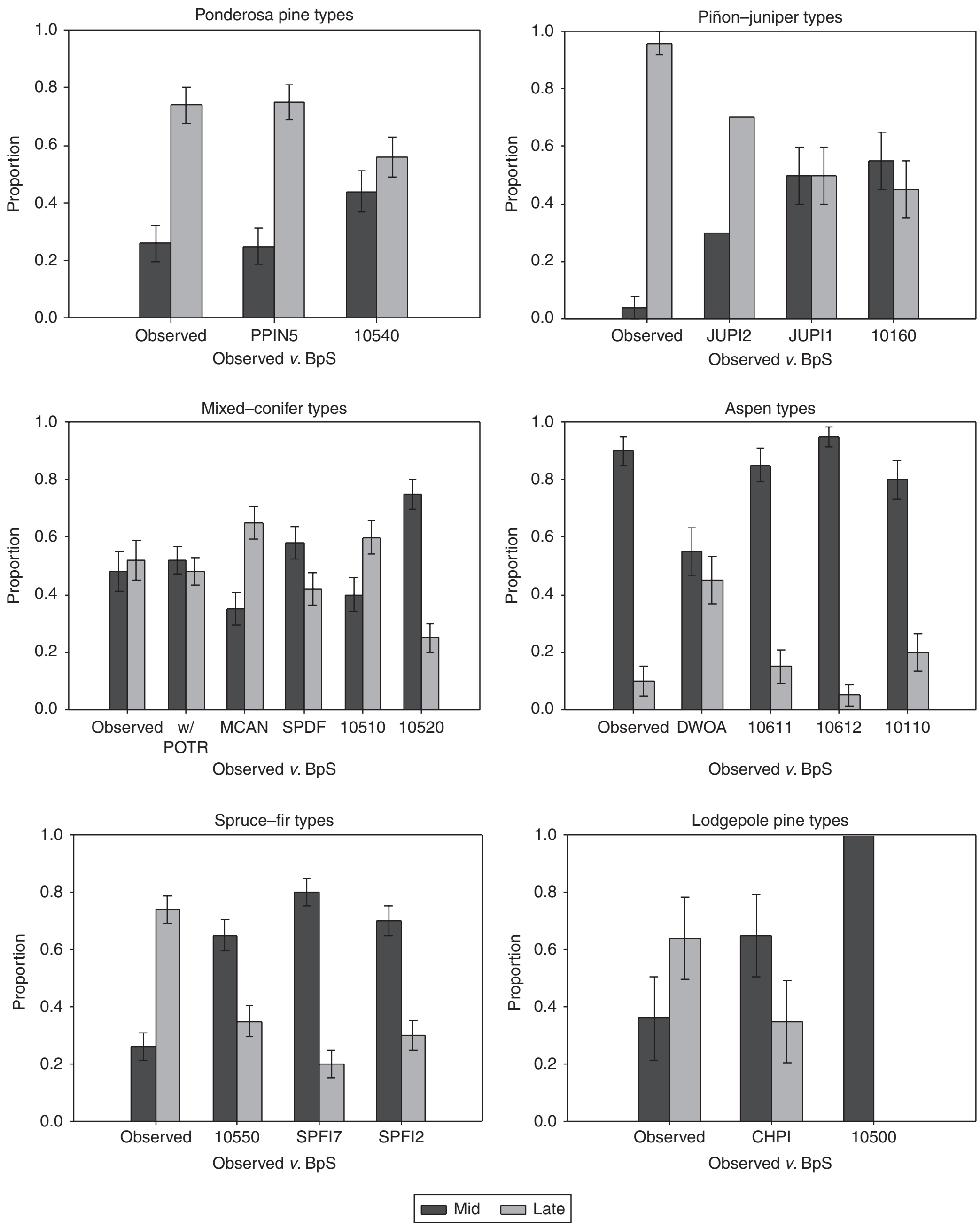

Fig. 6. Proportion of plots observed in the tree-ring data compared to FRCC (Fire Regime Condition Class) and LANDFIRE modeled reference condition proportions. Error bars were generated by calculating the 95% confidence interval from sample variance and standard error of observed points. Tree-ring results are on the left (e.g. observed), FRCC and LANDFIRE models are listed by their four-letter abbreviations on the right (e.g. PPIN5, 10540, etc.).

age–diameter relationships for determining reference condition proportions. Variations in age–size relationships both within species and among sites (Figs 2–5) may limit the ability to accurately gauge departure from estimated historical composition based on VDDT model results. Generally poor relationships between size and age may result in misassignment of current reference condition proportions based only on visual estimates, which may in turn lead to misassignment of the FRCC index.

Better correspondence between the tree-ring data and some BpS models indicates that VDDT models more accurately reflect historical forest structure in frequent-fire forest types such as ponderosa pine, mixed-conifer and aspen, than in

Table 5. LANDFIRE accuracy by BpS (biophysical settings) and EVT (existing vegetation type)
Code is the LANDFIRE map code for BpS or EVT type, *n* is number of plots tested, and % is percentage that were accurately mapped based on tree-ring data at plot scale

	Code	*n*	%
BpS			
Rocky Mountain aspen forest and woodland	10110	31	32
Colorado Plateau piñon–juniper woodland	10160	29	14
Rocky Mountain lodgepole pine forest	10500	7	43
Rocky Mountain dry–mesic montane mixed-conifer forest and woodland	10510	6	33
Rocky Mountain mesic montane mixed-conifer forest and woodland	10520	11	64
Southern Rocky Mountain ponderosa pine woodland	10540	19	53
Rocky Mountain subalpine dry–mesic spruce–fir forest and woodland	10550	82	66
Intermountain basins aspen–mixed-conifer forest – low elevation	10611	22	82
Intermountain basins aspen–mixed-conifer forest – high elevation	10612	31	77
Intermountain basins mountain mahogany woodland and shrubland	10620	6	50
EVT			
Rocky Mountain aspen forest and woodland	2011	26	50
Colorado Plateau piñon–juniper woodland and shrubland	2016	43	37
Rocky Mountain lodgepole pine forest	2050	19	63
Rocky Mountain montane mesic mixed-conifer forest and woodland	2052	9	78
Southern Rocky Mountain ponderosa pine woodland	2054	24	46
Rocky Mountain subalpine dry–mesic spruce–fir forest and woodland	2055	53	79
Intermountain basins aspen–mixed-conifer forest and woodland	2061	64	67
Abies concolor forest alliance	2208	14	71

infrequent-fire types such as spruce–fir and piñon–juniper (Fig. 6). BpS reference condition models were determined by managers and scientists familiar with the local ecology of each region during regional workshops. BpS types that are considered to be representative of each region were identified and described based on available historical and ecological data. Some BpS types, such as ponderosa pine and dry mixed-conifer forests, have extensive fire and forest history data with which to parameterize VDDT model runs. Other BpS types are less well studied and their fire and vegetation histories less certain, especially across the range of environmental and community variation within and between regions. The better correspondence between modeled and reconstructed reference conditions in frequent-fire-type models (ponderosa pine and mixed-conifer forest types; Fig. 6) is likely related to the greater amount of fire and forest history research that has been conducted in these forest types. Conversely, fire-infrequent types (spruce, and piñon–juniper woodland types; Fig. 6) have had less fire history research conducted, with the result that their fire regimes and successional patterns are less well documented for input to VDDT modeling. Furthermore, infrequent-fire types generally have fewer observations of historical fires and forest successional changes available for adequate characterization of fire regime parameters for VDDT modeling (e.g. Brown *et al*. 2008*a*).

Another factor that undoubtedly results in varying model and data results is that individual-site fire histories often have experienced contingent historical events that lead to differences from a ‘typical’ or average fire regime of a particular forest type. Stochastic modeling in FRCC and LANDFIRE generalizes vegetation and its fire regimes into generic types and does not take into account site-specific variability or, more importantly, the history of climate variations or other disturbances that may have affected changes in community structure through time. Variations in site histories undoubtedly contribute to variations in ratios of actual from modeled reference conditions. For example, spruce–fir and lodgepole pine FRCC and LANDFIRE BpS models predict more mid- than late-development stands, but the Utah tree-ring data found the opposite (Fig. 6). This may be due to longer fire intervals in this region than in other areas, leading to generally older stands across landscapes. Many spruce trees found in the tree-ring study were >300 years old at the time of sampling and probably resulted from fires that occurred in the late 1600s, most commonly in 1685 (Heyerdahl *et al*. 2005). However, the current presence of older rather than younger stands does not mean that these forests are outside their historical ranges of variability in either their fire regime or forest structure, but rather that they have not had extensive fires in the intervening period that would have resulted in a larger proportion of mid-successional stands as suggested should be present based on VDDT model results. Without taking into account this history of the forest landscapes, the VDDT models suggest that there is current departure in the landscape proportions of reference conditions in Utah spruce–fir and lodgepole pine forests.

Taking into account differences in fire histories, the trend of model results toward older or younger successional classes in each BpS may be more important to consider in FRCC assessments rather than the absolute proportions of stand structures. This may provide a more realistic perspective for assessing whether a particular BpS should be considered as inside or outside of its historical range of variation. For example, the tree-ring fire data for piñon–juniper (P-J) woodlands show the majority of stands are currently in late-development structural stages (Fig. 6). The FRCC BpS model JUPI2 (Table 4) also predicts more late-development trees than younger, but underpredicts what was found in the tree-ring data. The sensitivity of the VDDT model to fire frequency is critical to the setting of reference conditions. The model inaccuracy may be due to the model’s fire

return interval, currently predicted to be ~450 years. If the interval is increased (~1000 years), the model begins to more closely reflect the tree-ring results. A recent assessment of (P-J) ecosystems in the western US concluded that fire was only a minor disturbance in many less productive stands because of lack of both surface and crown fuels with which to carry fire (Romme *et al*. 2009). We believe that many of the Utah stands sampled probably fell into this category of fire regime historically, which means that if the longer intervals had been used in VDDT modeling, the reference conditions would likely be closer to what was found in the tree-ring data. The error may also be due to the definition of a mid-development stand in terms of the age; the mean ages of sampled piñon and juniper were among the highest in the tree-ring study. The mid-definition could be changed for P-J to an older age class by species to define the mid- from late-successional classes in the reference conditions.

Good correspondence between the tree-ring data and models for ponderosa pine (PPIN5), aspen (10110), and mixed-conifer (SPDF, 10510; Fig. 6) suggests that the reference conditions for these BpS were accurately modeled by VDDT parameters, at least in the Utah study sites. However, results of this study suggest that inaccuracy in piñon–juniper and subalpine types makes any decision based on a VDDT output possibly subject to error. For BpS types in which disturbance may not be the major or only factor in tree recruitment, VDDT models may need further evaluation. Additional empirical disturbance and forest history sampling in piñon–juniper, spruce–fir, and lodgepole pine types should increase the available information about these systems to use in VDDT modeling. However, because of generally longer fire intervals in these forests, any departure from historical to present conditions may be less than in frequent-fire BpS such as ponderosa pine and mixed-conifer forests. A possible result of inaccurate estimations of departure and wrong FRCC classification may be the application of incorrect management actions that could lead to even further departure from historical conditions (see also Romme *et al*. 2009).

The only accurate way to establish the age of a stand is to physically sample the trees for ages. We suggest based on the results of our comparison that at least some limited age sampling is needed for FRCC assessments. This sampling probably should include removing cores from the field and crossdating by trained dendrochronologists to most accurately characterize age and successional status of stands. Additional field-sampled fire history and stand establishment data, especially in the less-well-studied ecosystems, should further increase the accuracy of VDDT models through better dynamic estimations of age structures and relationships with fire regimes. However, we also realize that this type of sampling is expensive and – perhaps more critically to the efficient use of FRCC in forest management decisions – more time-consuming than FRCC visual assessment methods as currently practiced. Nevertheless, we suggest that some sort of compromise solution could be found that would provide both the most accurate as well as timely data possible for FRCC assessment needs.

LANDFIRE maps

Zhu *et al*. (2006) used a cross-validation technique to determine that existing vegetation data layer accuracies are between 60 and 89% in LANDFIRE maps. Our study's comparison of LANDFIRE and tree-ring data falls on the lower end of the estimate of Zhu *et al*. (2006) (Table 4). When broken down by BpS and EVT, some types are more accurately represented in LANDFIRE data than others. EVT mapping in LANDFIRE is most accurate for the mixed-conifer and spruce–fir types. These forests generally have the densest and most continuous canopies, and may have been easiest to identify through remote sensing methods because of their continuous canopies and more distinctive NDVI reflectance in Landsat spectral bands (Zhu *et al*. 2006). Conversely, sparser canopy cover may have led to lower accuracy in other types such as piñon–juniper, which is similar to what Zhu *et al*. (2006) found. It should be noted, however, that piñon–juniper plots sampled for the tree-ring study were generally found in ecotonal areas near lower ends of study sites, and may not be wholly representative of piñon–juniper BpS as defined in the LANDFIRE mapping effort.

Conclusion

Historical forest conditions reconstructed from tree-ring data provide opportunities for comparison with FRCC and LANDFIRE modeled vegetation data across multiple forest types. The tree-ring reconstructions we examined suggest that reference conditions are better modeled in frequent-fire forest types but not necessarily in infrequent-fire forest types, at least in Utah forests. Additional studies in fire-infrequent forest types should increase understanding of historical stand compositions, fire histories, and other disturbances with which to better parameterize VDDT reference condition models. The greatest amount of fire history research has been conducted in ponderosa pine and mixed-conifer forests, which likely contributed to the better correspondence between tree-ring data and VDDT model results that we found in this study. We consider this study as only a first step in comparison of empirical vegetation data with vegetation models used in both FRCC assessments and the nationwide LANDFIRE mapping effort. Tree-ring data provide an opportunity to compare site-specific vegetation patterns and fire regime variations that are often not easily accounted for in modeling efforts. Revised methods for assessing FRCC may need to take into greater account both tree ages and stand histories to more accurately compare with model results. We also suggest that ranges of reference conditions be incorporated into the BpS classifications to better take into account fire and forest histories rather than trying to establish average conditions that must be met for a FRCC index to be assigned.

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Land Use Planning and Wildfire: Development Policies Influence Future Probability of Housing Loss

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Abstract

Increasing numbers of homes are being destroyed by wildfire in the wildland-urban interface. With projections of climate change and housing growth potentially exacerbating the threat of wildfire to homes and property, effective fire-risk reduction alternatives are needed as part of a comprehensive fire management plan. Land use planning represents a shift in traditional thinking from trying to eliminate wildfires, or even increasing resilience to them, toward avoiding exposure to them through the informed placement of new residential structures. For land use planning to be effective, it needs to be based on solid understanding of where and how to locate and arrange new homes. We simulated three scenarios of future residential development and projected landscape-level wildfire risk to residential structures in a rapidly urbanizing, fire-prone region in southern California. We based all future development on an econometric subdivision model, but we varied the emphasis of subdivision decision-making based on three broad and common growth types: infill, expansion, and leapfrog. Simulation results showed that decision-making based on these growth types, when applied locally for subdivision of individual parcels, produced substantial landscape-level differences in pattern, location, and extent of development. These differences in development, in turn, affected the area and proportion of structures at risk from burning in wildfires. Scenarios with lower housing density and larger numbers of small, isolated clusters of development, i.e., resulting from leapfrog development, were generally predicted to have the highest predicted fire risk to the largest proportion of structures in the study area, and infill development was predicted to have the lowest risk. These results suggest that land use planning should be considered an important component to fire risk management and that consistently applied policies based on residential pattern may provide substantial benefits for future risk reduction.

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Introduction

The recognition that homes are vulnerable to wildfire in the wildland-urban interface (WUI) has been established for decades [e.g., 1,2]; but with a recent surge in structures burning, this issue is now receiving widespread attention in policy, the media, and the scientific literature. Single fire events, like those in Greece, Australia, southern California, and Colorado have resulted in scores of lost lives, thousands of structures burned, and billions of dollars in expenditures [3–6]. With the potential for increasingly severe fire conditions under climate change [7] and projections of continued housing development [8], it is becoming clear that more effective fire-risk reduction solutions are needed. "Fire risk" here refers to the probability of a structure burning in a wildfire within a given time period.

Traditional fire-risk reduction focuses heavily on fire suppression and manipulation of wildland vegetation to reduce hazardous fuels [9]. Enormous resources are invested in vegetation management [10], but as increasing numbers of homes burn down despite this massive investment, the "business-as-usual" approach to fire management is undergoing reevaluation. One issue is that fuel treatments may not be located in the most strategic positions, i.e., in the wildland-urban interface [11]. Yet, even if treatments surrounded all communities, scattered development patterns are difficult for firefighters to reach [12–14], and fuel treatments do little to protect homes without firefighter access [15–16]. Fuel treatments may also be ineffective against embers or flaming materials that blow ahead of the fire front [17].

One alternative to traditional fire management that is receiving widespread attention is to prepare communities through the use of fire-safe building materials or creating defensible space around structures [17–18]. These actions represent an important shift in emphasis from trying to prevent wildfires in fire-prone areas to better anticipating fires that are ultimately inevitable. Nevertheless, the cost of building and retrofitting homes to be fire-safe can be prohibitive, and these actions do not guarantee immunity from fire [19].

Land use planning is an alternative that represents a further shift in thinking, beyond the preparation of communities to withstand an inevitable fire, to preventing new residential structures from being exposed to fire in the first place. The reason homes are vulnerable to fires at the wildland-urban interface is a function of its very definition: "where homes meet or intermingle with wildland vegetation" [20]. In other words, the location and

pattern of homes influence their fire risk, and past land-use decision-making has allowed homes to be constructed in highly flammable areas [21]. Land use planning for fire safety is beginning to receive some attention in the literature [22–23], and there is growing recognition of the potential benefits of directing development outside of the most hazardous locations [8,19,24].

Despite recent attention in the literature, land use planning for wildfire has yet to gain traction in practice, particularly in the United States. However, fire history has been used to help define land zoning for fire planning in Italy [22], and bushfire hazard maps are integrated into planning policy in Victoria, Australia [25]. Although some inertia inevitably arises from complications with existing policy and plans, a primary impediment to the design and implementation of fire-smart land use planning is lack of guidance about specific locations, patterns of development, or appropriate methodology to direct the placement of new development. Without a solid knowledge base to draw from, planners will be misinformed about which planning decisions may result in the greatest overall reduction of residential landscape risk. Even worse, poor science could result in placement of homes in areas that actually have high fire hazard.

Research on how planning decisions contributed to structures burning in the past provides some guidance about what actions may work in the future. Analysis of hundreds of homes that burned in southern California the last decade showed that housing arrangement and location strongly influence fire risk, particularly through housing density and spacing, location along the perimeter of development, slope, and fire history [26]. Although high-density structure-to-structure loss can occur [27–28], structures in areas with low- to intermediate- housing density were most likely to burn, potentially due to intermingling with wildland vegetation or difficulty of firefighter access. Fire frequency also tends to be highest at low to intermediate housing density, at least in regions where humans are the primary cause of ignitions [29–30].

These results suggest, for example, that placing new residential development within the boundaries of existing high-density developments or in areas of low relief may reduce fire risk. However, it is difficult to know whether broad-scale planning policies would actually result in the intended housing arrangement and pattern at the landscape scale, and whether those patterns would result in lower fire risk. Our objective here was to simulate three scenarios of future residential development, and to project wildfire risk, in a rapidly urbanizing and fire-prone region where we have studied past structure loss [25]. We based all future development on an econometric subdivision model, but we varied the emphasis of subdivision decision-making based on three broad and common growth types.

Although cities vary in extent, fragmentation, and residential density [31–32], urban form typically adheres to a set of common patterns [33–34], and we based our development scenarios on the three primary means by which residential development typically occurs: infill, expansion, or leapfrog [34]. Infill is characterized by development of vacant land surrounded by existing development, typically in built-up areas where public facilities already exist. [35–36], and should result in higher structure density rather than increased urban extent. Expansion growth occurs along the edge of existing development, extends the size of the urban patch to which it is adjacent, and may have variable influence on structure density. Leapfrog growth occurs when development occurs beyond existing urban areas such that the new structure is surrounded by undeveloped land. This type of growth would expand the urban extent and initially result in lower structure density; but these areas may eventually become centers of growth from which infill or expansion can occur. We asked:

1) Do residential development policies reflecting broad growth types affect the resulting pattern and footprint of development across the landscape?
2) Do differences in extent, location, and pattern of residential development translate into differences in wildfire risk, based on the current configuration of structures?
3) Which development process, infill, expansion, or leapfrog, results in the lowest projected fire risk across the landscape?

Methods

Study Area

The study area included all land within the South Coast Ecoregion of San Diego County, California, US, encompassing an area of 8312 km^2. The region is topographically diverse with high levels of biodiversity, and urban development has been the primary cause of natural habitat loss and species extinction [37]. Owing to the Mediterranean climate, with mild, wet winters and long summer droughts, the native shrublands dominating the landscape are extremely fire-prone. San Diego County was the site of major wildfire losses in 2003 and 2007 [38], although large wildfire events have occurred in the county since record-keeping began, and are expected to continue, as fire frequency has steadily increased in recent decades [29,39]. The county is home to more than three million residents, and approximately one million more people are expected by 2030 [40]. Although most residential development has been concentrated along the coast, expansion of housing is expected in the eastern, unincorporated part of the county.

Econometric Subdivision Model

A host of alternative modeling approaches exist to simulate future land use scenarios [41], including a cellular automaton model that we previously applied to the study area [42]. We chose to use an econometric modelling approach for this study because we wanted to capture fine-scale, structure-level patterns and processes that are correlated with housing loss to wildfire [26]; and econometric models may perform better at the scale of individual parcels [43].

Although we based the three development scenarios on generalized planning policies, we also wanted to ensure that the residential projections were realistic and adhered to current planning regulations. The objective of the econometric modeling was to estimate the likelihood that residential parcels will subdivide in the future. Therefore, we used a probit model to estimate the transition probability of each parcel based on a range of potential explanatory variables typically associated with parcel subdivision and housing development [44–45].

To develop the model of subdivision probability, we acquired GIS data of the county's parcel boundaries in years 2005 and 2009 from the San Diego Association of Governments (SANDAG). The dependent variable was equal to 1 if a parcel subdivided between 2005 and 2009, and zero otherwise. Using these data layers we first determined which parcels were legally able to subdivide given current land use regulations. Minimum lot size restrictions are typically considered the most import restriction for determining future land use. We deemed a parcel eligible for subdivision if the current lot size was greater than twice the minimum legal size given the land class. To determine which parcels subdivided between 2005 and 2009, we queried parcel IDs where the total

area was reduced by at least the minimum lot size between the two time periods. Finally, we were able to generate a suite of variables that determine the likelihood of a parcel developing in the future (Table S1).

We overlaid the parcel boundaries over a range of GIS layers representing our explanatory variables. These data are available to download at (http://www.sandag.org/index.asp?subclassid = 100&fuseaction = home.subclasshome). Our explanatory variables included: parcel size, parcel size squared, six dummy variables which capture non-linear effects of parcel size, distance to the coast, distance to the coast squared; distance to city center and its square, current zoning, slope, land use, roads, if the parcel is in a protected area, if the parcel is in a development area, if the parcel is in the redevelopment area (Table 1).

Spatial Model of Future Development under Planning Alternatives

The outcome of the land use change econometric model is the subdivision probability for each parcel for a five-year time step. Based on these probabilities, we developed a GIS spatial simulation model of future land use under three distinct planning scenarios: infill (development in open or low density parcels within already developed areas), expansion (development on the fringe of developed areas), and leapfrog (development in open areas). The model runs in four 5-year time steps from 2010 to 2030, and generates the spatial locations of new housing units in the county.

Although development decisions could feasibly depend on fire risk, we did not model that here. There is no evidence that fire has influenced past regional planning decisions, so it was not used as an explanatory variable in the econometric model. Although we could have evaluated the potential for future development decisions to be based in part on fire risk, this would have required simulation of feedbacks between fires and probability of development. Because our objective in this study was to isolate the effects of the three distinct growth types, we modeled fire risk only as a function of development pattern and not vice versa.

We constructed a complete spatial database of existing residential structures in the study area [26]. These structures and their corresponding parcel boundaries served as the initial conditions for all three scenarios of the spatial simulation model. The current and projected future GIS layers of structures were also subsequently used in the fire risk model (see below). The

Table 1. Variables and results from the probit regression model of parcel subdivision in San Diego County.

Subdivided (1 = yes,0 = no)	Coefficient	Std. Err.	z	P>|z|	[95% Conf. Interval]	
Acres of lot	0.0026342	0.00075	3.51	0	0.001164	0.004105
Acres of lot ^{2}	−3.02E-06	1.29E-06	−2.34	0.019	−5.55E-06	−4.93E-07
Distance to ocean	−7.42E-06	1.33E-06	−5.59	0	−0.00001	−4.82E-06
Distance to ocean ^{2}	2.33E-11	8.28E-12	2.82	0.005	7.11E-12	3.96E-11
Distance to major road	2.17E-07	2.74E-06	0.08	0.937	−5.16E-06	5.59E-06
Distance to major road ^{2}	−1.94E-11	1.70E-11	−1.14	0.252	−5.27E-11	1.38E-11
Distance to nearest city center	−0.0000115	1.70E-06	−6.76	0	−1.5E-05	−8.16E-06
Distance to nearest city center ^{2}	2.89E-11	9.70E-12	2.98	0.003	9.91E-12	4.79E-11
Slope between 0–5%	0.6211289	0.211761	2.93	0.003	0.206085	1.036173
Slope between 5–10%	0.3911427	0.210684	1.86	0.063	−0.02179	0.804076
Slope between 10–25%	0.0716669	0.212725	0.34	0.736	−0.34527	0.4886
Rural Residential	−0.3563149	0.071512	−4.98	0	−0.49648	−0.21615
Single Family	0.1361149	0.068678	1.98	0.047	0.001509	0.270721
Multi-Family	−0.2505093	0.151486	−1.65	0.098	−0.54742	0.046397
Road	0.015329	0.086094	0.18	0.859	−0.15341	0.184069
Open Space	−0.7440933	0.099145	−7.51	0	−0.93841	−0.54977
Orchard/Vineyard	−0.5813305	0.097867	−5.94	0	−0.77315	−0.38951
Agriculture	−0.9785208	0.132734	−7.37	0	−1.23867	−0.71837
Vacant Land	−0.5222501	0.074586	−7	0	−0.66844	−0.37606
Zoned protected	0.253769	0.076881	3.3	0.001	0.103086	0.404452
Area marked for redevelopment	−0.2680261	0.14069	−1.91	0.057	−0.54377	0.007722
Area marked for development	0.5780101	0.064103	9.02	0	0.452371	0.703649
Parcel between 10–20 acres	−0.3379532	0.065899	−5.13	0	−0.46711	−0.20879
Parcel between 5–10 acres	−0.6119036	0.067012	−9.13	0	−0.74325	−0.48056
Parcel between 2–5 acres	−1.16297	0.07062	−16.47	0	−1.30138	−1.02456
Parcel between 1–2 acres	−1.563956	0.090286	−17.32	0	−1.74091	−1.387
Parcel between.5–1 acres	−1.999939	0.099893	−20.02	0	−2.19573	−1.80415
Parcel between.25–.5 acres	−2.178273	0.117101	−18.6	0	−2.40779	−1.94876
Constant	−1.397931	0.227467	−6.15	0	−1.84376	−0.9521

Sample size 113 001, LR Chi^2 1535.23, pro>chi 0, pseudo R^2 0.22. Further description of the variables is provided in Table S1.
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dataset of existing housing includes locations of 687,869 structures, of which 4% were located within the perimeter of one of 40 fires that burned since 2001. During these fires, 4315 structures were completely destroyed, and another 935 were damaged.

For future development scenarios, we wanted to allocate an equal number of new structures to the landscape. This was to ensure that any predicted difference in fire risk was a function of the arrangement and location of structures, not the total number of structures. Nevertheless, differences in the total number of structures were simulated with each of the 5-year time steps. We determined the number of housing units to add during the simulations based on projections made by San Diego County [46]. Using factors such as development proposals, general plan densities, and information from jurisdictions, the county estimated that between 331,378 units and 486,336 units could be supported within the developable residential land by 2030. Because the eastern, desert portion of the county was not included in our study area, we used a conservative approach and simulated the addition of 331,378 new dwelling units. We divided this number by four to define the number of new dwelling units to add at each time step, assuming a linear growth rate.

One output of the econometric model was the prediction of the maximum number of new dwelling units that could be added to each parcel. However, dwelling units may consist of apartments as well as single family homes. The mix of single and multifamily units in the region has remained relatively constant over time, and the overall trend has been a mix of roughly 1/3 multifamily and 2/3 single family units. Because the fire risk model is based on points representing structure locations across the landscape, regardless of the number of dwelling units per structure, we needed to generate a conversion factor from dwelling units to structures. We therefore defined a minimum lot size of 0.25 acre on which no more than a single structure could be built, regardless of the number of dwelling units in it (i.e., a single family home or apartment complex). Then, once a parcel was selected for development by the model (see details below), we divided its total area by the maximum number of dwelling units to be added, according to the econometric model. If the result was larger than 0.25, we subdivided parcels according to the result. If not, we quantified how many 0.25 acre parcels fit into the original parcel, and generated the new parcel boundaries accordingly.

Using the initial map of parcels (year 2010), we classified each parcel that was defined as eligible for development (in the previous stage) as suitable for one of the three planning scenarios described above, according to the number of developed parcels in its immediate neighborhood (i.e., those parcels that share a boundary with the focal parcel). We defined 'developed parcels' as ones that had more than one house per 20 acres (8.09 ha). Therefore, according to these density thresholds, we allowed some parcels with nonzero housing density to be considered as 'undeveloped' because these large, rural parcels might contain a single or a handful of houses but they exist within a large open area. In other words, the overall land cover of these parcels was effectively undeveloped, and we therefore assumed that development in adjacent parcels would be akin to development in open areas.

We defined infill parcels as those that were completely surrounded by developed parcels. Expansion parcels had at least one neighboring parcel that was undeveloped; and leapfrog parcels were those with no developed parcels in their immediate surroundings. We reclassified the type of each available parcel in the same manner after each time step, to account for changing dynamics in the development map of the county.

We conducted three simulations, one for each development scenario (infill, expansion, and leapfrog). In each simulation, all parcels were eligible to subdivide, regardless of their class. Therefore, to build a simulation for a specific scenario, we increased the development probability of parcels of the selected scenario by 20%, to favor their development compared to the other types of parcels, without prohibiting development in the other parcel types. This approach was necessary because the projected number of dwelling units was much larger than it would be possible to fit in infill and leapfrog class parcels solely. For example, as the spatial coverage of developed parcel expands, there is less contiguous area that is undevelopable and suitable for leapfrog development. Therefore, the scenarios are not exclusive, but rather a mixture of the three development types. Yet, in each scenario, there is one main type of development, and smaller amounts of development events of the other two types.

Due to the immense computational demand of the simulations, we adopted a deterministic, rather than a stochastic approach to decide on which parcels were subdivided. After enhancing the transition probability according to the corresponding scenario, we ranked and then sorted all parcels according to their probability of subdivision. We then sequentially selected parcels, while simultaneously tallying the number of dwelling units in them, until the development target in that time step (one fourth of the total number of dwelling units to be added: 82,795) was reached. Once the development target was reached, we moved to the next time step. After each time step, the remaining parcels that were still eligible for development were re-classified to development types according to the new spatial configuration of the landscape.

Once a parcel was selected for subdivision, and the number of new parcels to develop in it was calculated (as detailed above), an equal-area spatial splitting model was employed to split the parent parcel to the predefined number of equal-area child parcels. We developed a simple splitting model which is based on iterative splitting of larger parcels into two smaller parcels using a straight line splitting boundary. Once the parcel was fully split into the needed number of sub-parcels, we allocated a new structure inside each new parcel by generating a point at its centroid (center of gravity). The point datasets of all structure locations per time step per scenario were passed over to the fire risk model, which is described below.

Fire Risk Modeling and Analysis

To project the distribution of fire risk under alternative scenarios, we used MaxEnt [47–48], a map-based modeling software used primarily for species distribution modeling [48], but we have used it successfully for ignition modeling [50] and for projecting current fire risk in the study area [26]. For this study, we slightly modified the model from Syphard et al. [26]. The dependent variable was the location of structures destroyed by fire between 2001 and 2010. Although inclusion of damaged structures in the data set does not significantly affect results [26], we only included completely destroyed structures to avoid the introduction of any uncertainty.

The MaxEnt software uses a machine-learning algorithm that iteratively evaluates contrasts among values of predictor values at locations where structures burned versus values distributed across the entire study area. The model assumes that the best approximation of an unknown distribution (i.e., structure destruction) is the one with maximum entropy. The output is an exponential function that assigns a probability to every cell of a map. Thus, the resulting continuous maps of fire risk represented the probability of a structure being destroyed by fire. In these output maps, areas of predicted high fire risk that did not have structures on them represented environmental conditions similar to those in which structures have actually burned.

We based the explanatory variables on those that were significantly related to burned structures in Syphard et al. [26], including maps depicting housing arrangement and pattern, housing location, and biophysical factors. Housing pattern variables reflected individual structure locations as well as the arrangement of structures within housing clusters. We calculated housing clusters, defined as groups of structures located within a maximum of 100 m from each other, by creating 100 m buffers around all structures and dissolving the overlapping boundaries [51].

Because burned structures were significantly related to small housing clusters [26], we calculated the area of every cluster as an attribute, and then created raster grids based on that attribute. Low-to intermediate housing density and distance to the edge of the cluster were also significant explanatory variables relative to housing pattern and location [26], so we also created raster grids for those. GIS buffer measures at 1-km have been found to explain approximately 90% of the variation in rural residential density [52], so we developed density grids using simple density interpolation based on a 1-km search radius, with area determined through square map units. To create grids representing distance to the edge of clusters, we first collapsed the cluster polygons into vector polyline files, and then created grids of interpolated Euclidean Distance to the edge within each cluster.

Because the MaxEnt model randomly selects background samples in the map to compare with locations of destroyed structures, we used a mask to restrict sampling to the developed environment within cluster boundaries; the distance to the edge of the cluster would represent a different relationship inside a cluster boundary versus outside in the wildland. We also modified the grids to ensure that any random sample located within the 100m buffer zone would receive a value of 100m; thus, all points within the buffer were considered "the edge of the development".

After creating the grids representing housing pattern and arrangement of the current configuration of structures, we applied the same algorithms to the maps of simulated future structure locations. We thus generated grids representing future housing pattern and arrangement under alternative development scenarios. The other explanatory variables, including fire history, slope, fuel type, southwest aspect, and distance to coast [26] remained constant through time for current and future scenarios. Although historic fire frequency and fuel type typically change through time, we did not simulate their dynamics here because we wanted to isolate the effect of planning decisions on housing pattern and arrangement while holding everything else constant.

We conditioned the MaxEnt model on present distributions of housing using ten thousand random background points and destroyed structures located no closer than 500-m to minimize any effect of spatial autocorrelation. We used 80% (260 records) of these data for model training, and 20% [66 records) for testing. We repeated the process using cross-validation with five replicates and used the average of these five models for analyses. For smoother functions of the explanatory variables, we used hinge features, linear, and quadratic with an increase in regularization of beta set at 2.5, based on Elith et al. [48]. The smoother response curves minimize over fitting of the model. We conducted jackknife tests of explanatory variable importance.

We first developed the model using mapped explanatory variables derived from the current configuration of structures. To project fire risk under the different time steps of the alternative development scenarios, projected the model conditioned upon current conditions onto maps representing future conditions by substituting the grids representing future housing pattern and arrangement. This is similar to how potential future distributions of species are projected under climate change scenarios [49].

To quantify differences among current and future alternative scenarios, we calculated metrics representing housing density, pattern, and footprint to determine the extent to which the planning policies produced differences in housing pattern and location. We compared the modeled structure fire risk of the scenarios by overlaying all maps of structure locations with their respective mapped output grids from the MaxEnt models and calculating probability of burning for every structure point. We also calculated total area of risk by selecting three threshold criteria [53]. These criteria, at 0.05, 0.25, and 0.5 represented three different degrees of risk, and we calculated the proportion of structures that were located in risk areas for every time step in all scenarios.

Results

The probit econometric model, run on 113 001 observations, showed that larger parcels were most likely to subdivide, although the relationship between parcel size and subdivision probability was non-linear (Table 1). Parcels closer to existing roads, the ocean, those with lower slopes, and those designated as fit for development were all most likely to develop. Parcels designated in redevelopment areas were less likely to develop. Overall, the model had a pseudo r –squared of 0.22.

The land use simulation model, based on a combination of the econometric subdivision model and three different growth policies, resulted in substantial differences in the extent and pattern of housing of the three scenarios. The total area of housing development, or the housing footprint, was largest for simulations where leapfrog growth dominated, followed by expansion-type development, and then infill (Figure 1a). The differences in the housing footprint became larger among the scenarios over time, but the largest difference was between infill and the other two development types. As the housing footprint expanded in the three scenarios, the corresponding housing density declined, so that leapfrog growth resulted in the lowest housing density per 1-km, followed by expansion and then infill (Figure 2b). Despite the near inverse of this relationship, there was generally a larger separation among scenarios with regard to housing density. With larger housing footprints and lower housing density, the number of separate housing clusters increased while their size decreased (Figure 2c).

In the first two time steps of the model (2015 and 2020), the simulated development pattern closely followed the desired pattern in the scenario, although some of the growth in the infill scenario ended up becoming expansion or leapfrog (Table 2). In the last two time steps (2025 and 2030), there were not enough infill parcels left, and thus, the majority of growth in these simulations became expansion, followed by infill, and then leapfrog. In the last time step, there were not enough isolated parcels in the leapfrog scenario and thus, the majority of development became expansion. Thus in general, as more development occurred in the simulations by the year 2030, the majority took the form of expansion.

The area under the curve (AUC) of receiver operating characteristic (ROC) plots, indicating the ability of the MaxEnt model to discriminate between burned and unburned structures, averaged across five cross-validated replicate runs was 0.91. The AUC represents the probability that, for a randomly selected set of observations, the model prediction was higher for a burned structure than for an unburned structure [49].The two most important variables in the model according to the internal jackknife tests in MaxEnt [47] were related to housing pattern:

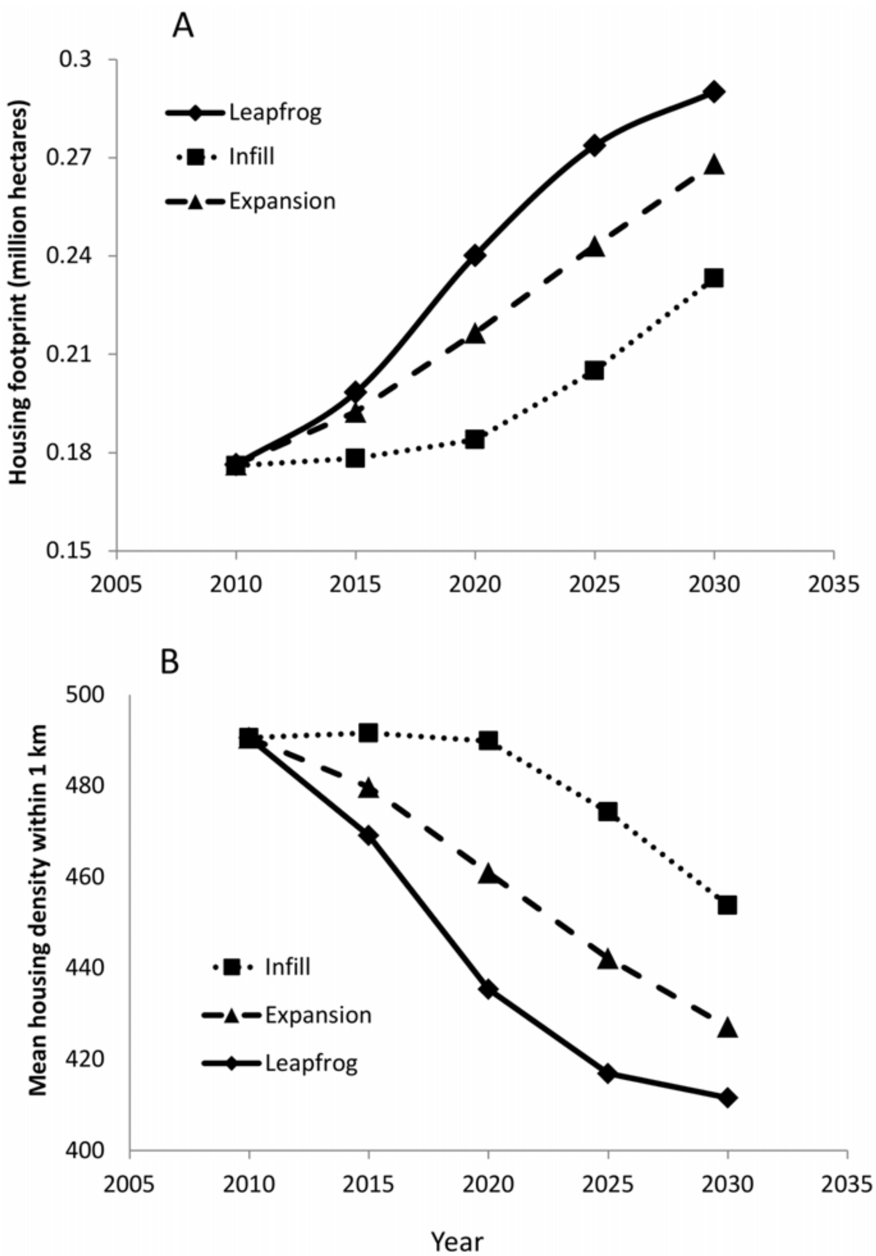

Figure 1. Trends of development extent and pattern for three planning policy simulations from 2010–2030, including A) total housing footprint representing the area of land within all housing clusters, and B) mean housing density averaged across all housing clusters.
doi:10.1371/journal.pone.0071708.g001

low to intermediate housing density and small cluster size and housing density (Figure 3). The distance to the edge of housing cluster was a less important contribution.

Maps showing the probability of a structure being destroyed in a wildfire, displayed as a gradient from low to high risk, show broad agreement relative to the general areas of the landscape that are riskiest, with correlation coefficients ranging from 0.85–0.91 (Figure 4). Nevertheless, subtle differences are apparent in the three development-scenario maps by year 2030, with the highest-risk areas in the expansion scenario located farther east than infill, and the highest-risk areas in leapfrog occupying a wider extent than either of the other two scenarios.

Differences among current housing and the three development scenarios are clearly illustrated through the mean landscape risk, or total probability of all structures burning (Figure 5). All three development scenarios were predicted to experience an increase in mean landscape risk over the duration of the simulations, except for infill at year 2015. The highest landscape risk to structures was predicted for the leapfrog scenario, followed by expansion, and then infill. The increase in risk over time is more gradual for the infill scenario than the other two scenarios.

The ranking of scenarios varied according to the proportion of structures located within different levels of risk defined through binary thresholding (Figure 6). When the continuous risk maps were thresholded at the lowest number of 0.05, a large proportion

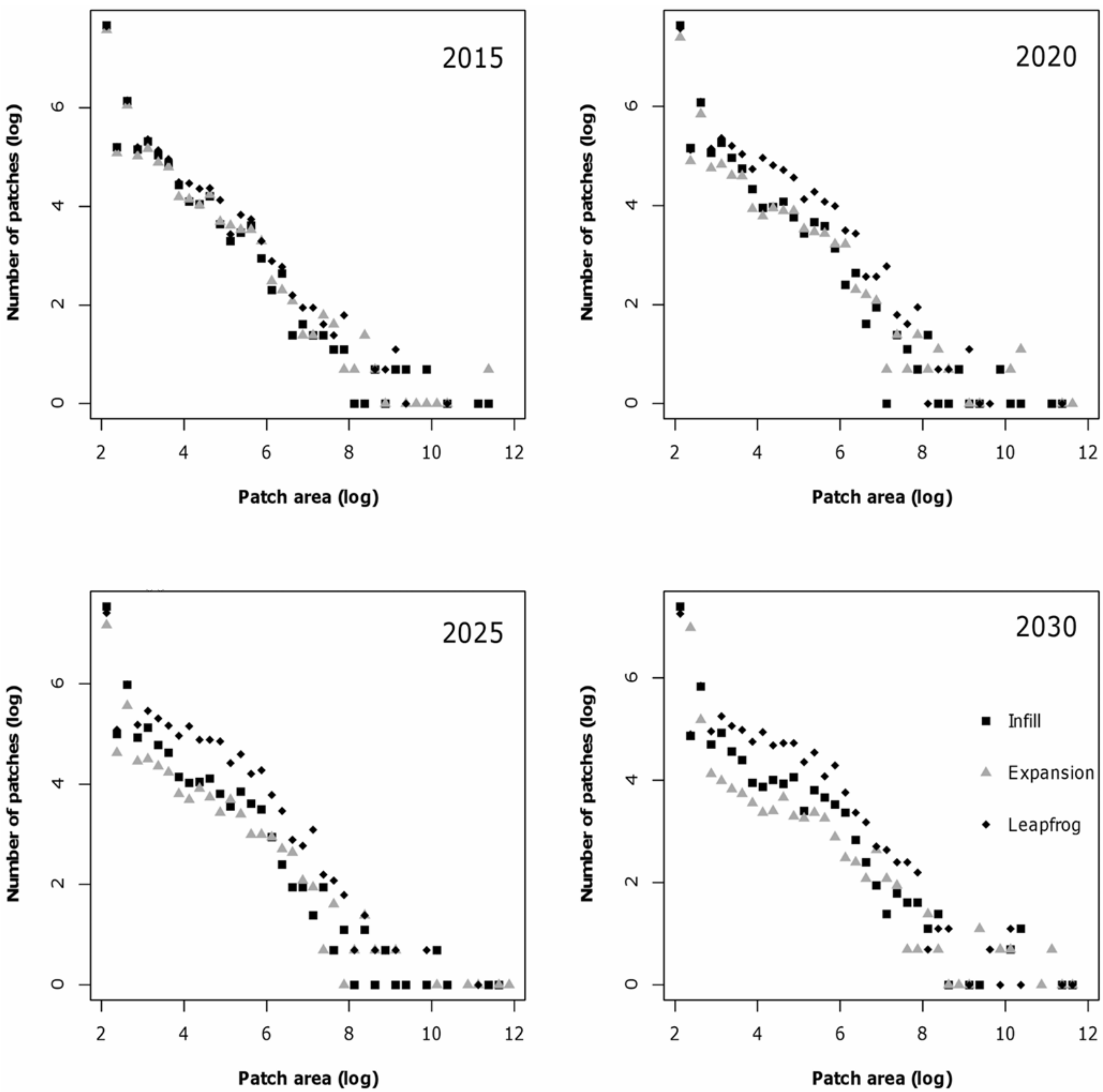

Figure 2. Trends in number of patches and patch area for three planning policy simulations from 2010–2030. Numbers were log-transformed for better visual representation of the scenarios.
doi:10.1371/journal.pone.0071708.g002

of structures in all scenarios fell within areas defined as risky according to this criterion. At this threshold, the proportion of structures in high-risk areas increased linearly for the expansion and leapfrog development scenarios while the proportion of infill homes increased more gradually. When risk was defined more conservatively at 0.25, temporal trends for the leapfrog and infill scenarios were similar to the 0.05 threshold. However, the proportion of structures at risk in the expansion scenario initially increased to 2020, but this proportion leveled off and declined by 2030. When the threshold was highest at 0.50, a very low proportion of structures in any scenario were located in areas at risk. But in these high-risk areas, the expansion scenario switched places with infill to have the lowest proportion of structures at risk in all time steps. Leapfrog had the largest proportion of homes at risk. This proportion of homes located in areas at risk with a threshold at 0.5 declined over time for all three scenarios.

Discussion

Our simulations of residential development showed that planning policies based on different growth types, applied locally for subdivision of individual parcels, will likely produce substantial and cumulative landscape-level differences in pattern, location, and extent of development. These differences in development pattern, in turn, will likely affect the area and proportion of

Table 2. Pattern of simulated development under infill, expansion, and leapfrog growth policies.

		Actual development		
Development scenario	**year**	***Infill***	***Expansion***	***Leapfrog***
Infill	2015	9450	18	6
	2020	11787	153	29
	2025	236	624	144
	2030	325	890	179
Expansion	2015	0	772	0
	2020	0	1243	2
	2025	0	1871	1
	2030	0	2662	0
Leapfrog	2015	0	10	408
	2020	0	5	1132
	2025	1	83	3563
	2030	34	917	0

The numbers in the table denote the numbers of patches of a given development type.
doi:10.1371/journal.pone.0071708.t002

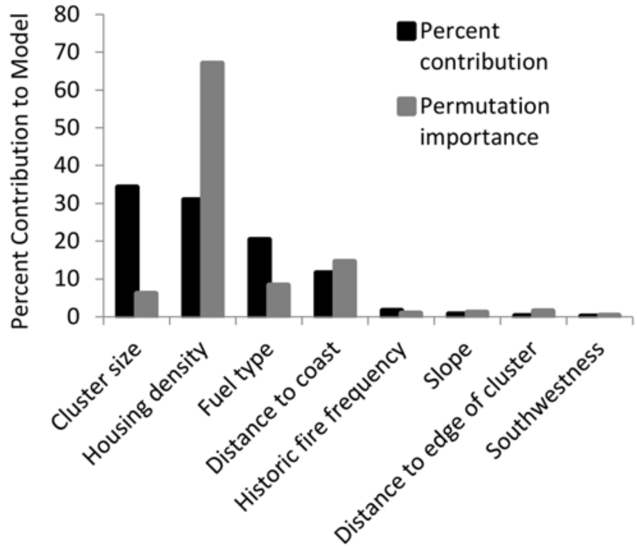

Figure 3. The importance of explanatory variables averaged across five cross-validated replications in the MaxEnt fire risk model. Percent contribution is determined as a function of the information gain from each environmental variable throughout the MaxEnt model iterations. Permutation importance reflects the drop in model accuracy that results from random permutations of each environmental variable, normalized to percentages.
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structures at risk from burning in wildfires. In particular, the scenarios with lower housing density and larger numbers of small, isolated clusters of development, i.e., leapfrog followed by expansion and infill, were generally predicted to have the highest predicted fire risk to the largest proportion of structures in the study area. Nevertheless, rankings of scenarios were affected by the definition of risk.

Theoretically, it makes sense that leapfrog development produced fragmented development with larger numbers of small patches, lower housing density, and a larger housing footprint; and that infill resulted in the opposite, with expansion in the middle. By definition, leapfrog development requires open space around all sides of the newly developed parcel, whereas infill requires development on all sides, and expansion requires development on one side and open space on another. Implementing these planning policies on real landscapes, however, can be complex if there are more houses to build than there are parcels that meet the definitions of the three planning rules, and thus not all development conforms strictly to the policy [54]. In our simulations, parcels meeting the definition of each growth type had a higher probability of subdividing; yet, as we were simulating a real landscape, many newly developed parcels did not meet the scenario criteria. That the three scenarios nevertheless produced substantial differences in landscape-level development patterns shows that decision-making at the individual level can lead to meaningful broad-scale effects.

The objective of the econometric model was to provide a baseline probability to predict which parcels were most likely to subdivide; thus, the econometric model itself provides no explanation of how a given policy affects likelihood of subdivision, although it does indicate the correlation between the policy and the outcome. In our setting, which areas are protected, marked for redevelopment, or marked for development may be endogenous to the land owner decision to subdivide. In the case of these variables especially, our results should not be interpreted as causal predictors. Likewise, we use data only from 2005–2009 to predict changes to 2030. If major changes in the land market take place over this time horizon our model will not be able to take this into account.

Although some differences in predicted fire risk among the three scenarios likely stemmed from location of new structures relative to variables such as distance to coast, fuel type, or slope, the most important variables in the fire risk model were housing density and cluster size, with most structure loss historically occurring in areas with low housing density and in small, isolated housing clusters. Thus, leapfrog development was generally the riskiest scenario and infill the least risky. The most surprising result was the variation in predicted risk for the expansion scenario over time and at different thresholds. While leapfrog and infill showed similar trajectories across thresholds, expansion went from being the highest-risk scenario at the low threshold to being the lowest-risk scenario at the highest threshold. Because the threshold is merely a way to group structures into a binary classification, this means that, while the average risk calculated across all homes shows expansion to rank in the middle of infill and leapfrog throughout the simulation (Figure 5), the other two scenarios have a relatively larger proportion of homes that are modeled to be at a very high risk (i.e., 0.25 or 0.5), particularly by the end of the simulations. Because the total number of structures with a risk greater than 0.25 or 0.5 is relatively low in all scenarios, this difference in distribution of homes at the highest risk is not reflected in the mean. Another reason for the shift in rank of expansion over time is that, as more development occupied the landscape, there were fewer parcels remaining to accomplish infill or leapfrog type growth in the other scenarios. Thus, by the end of the simulations in year 2030, the majority of growth in all scenarios was expansion, and there was some convergence between scenarios. Finally, the change in risk of expansion growth over time may reflect that, despite the relatively low importance of distance to edge of cluster as an explanatory variable, expansion growth is characterized as having an initially fragmented landscape pattern that eventually merges into large patches with low edge.

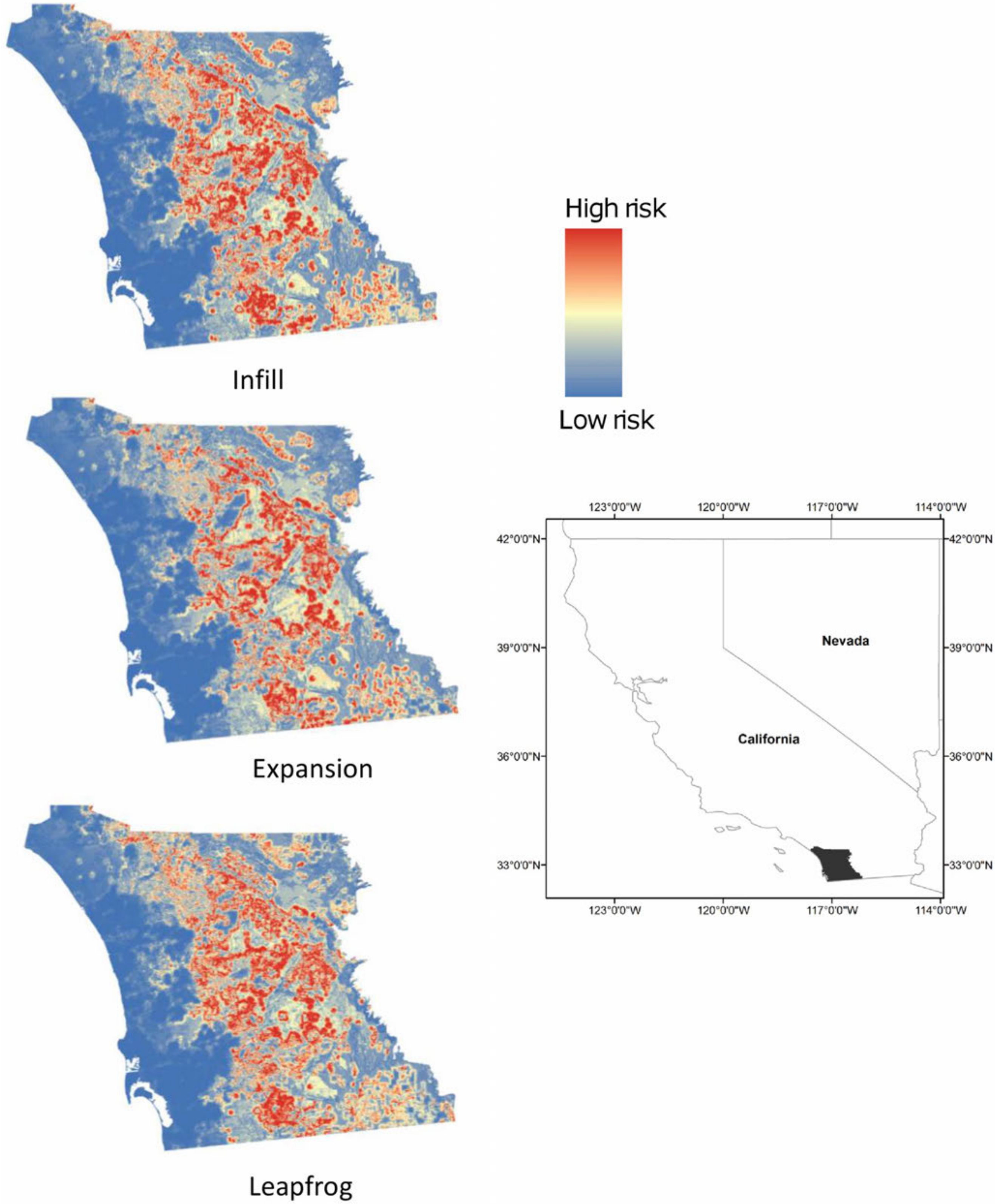

Figure 4. Maps of the study area showing projected wildfire risk at year 2030 for simulations of residential development under policies emphasizing infill, expansion, or leapfrog growth.
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Although leapfrog development clearly ranked highest in terms of fire risk, the interpretation of which planning policy is best may depend on fire management objectives and resources, as well as other considerations such as biodiversity or ecological impacts.

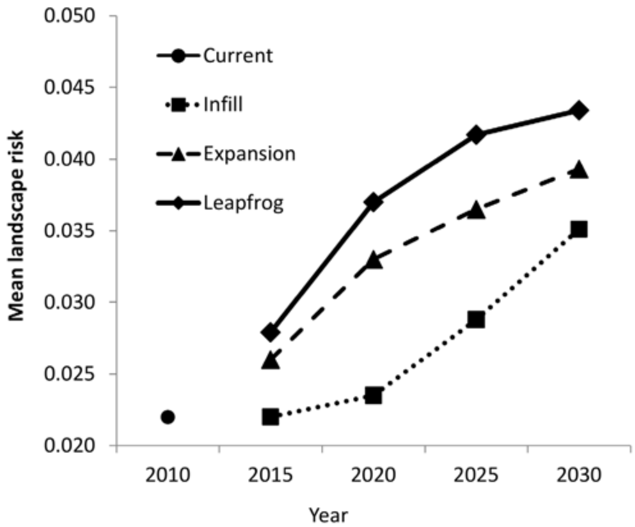

Figure 5. Projected landscape fire risk, reflecting the probability of burning in a wildfire averaged across all residential structures on the current landscape and in three development scenarios of infill, expansion, and leapfrog for year 2030.
doi:10.1371/journal.pone.0071708.g005

The spatial pattern of development affects multiple ecological functions and services [55], with potentially varying conservation implications; both leapfrog and expansion development consumed more land than infill, which would likely lead to more ecological degradation [56]; nevertheless, higher-density clustered development may be dominated by more invasive species [57]. Trade-offs between fire protection and conservation are common, but techniques are available for identifying mutually beneficial solutions [58].

Different perceptions of the fire risk results could also potentially translate into different planning priorities for management. For example, if the priority is to plan for the lowest overall risk to structures, then the mean landscape risk clearly delineates the rankings of options, with infill being the winner. However, if the objective is to reduce the number of structures at the highest risk threshold, i.e., $>=0.5$, then expansion is the best option, at least by 2030. An important consideration for fire management is the total area that needs to be protected, as well as the length of wildland-urban interface [8,13]. Therefore, despite the lower number of structures at the highest risk thresholds, expansion creates more edge than infill and may translate into greater challenges for firefighter protection.

Although we did not create separate scenarios for high or low growth, the results at different time steps can be substituted to envision the potential outcome of developing more or fewer houses. In the short term, the total fire risk is projected to increase proportionately as more land is developed. However, given the inverse relationship between housing density and fire risk, it is possible that this trend could reverse if housing growth eventually resulted in expansive high-density development.

Land use planning is one of a range of options available for reducing fire risk, and the best outcome will likely be achieved through a combination of strategies that include homeowner actions, improvements in fire-safe building codes, and advanced fire suppression tactics. Although we isolated the effect of land use planning policy in the three development scenarios, the fire risk model nevertheless showed that the pattern and location of structures in this study area were the most important out of a suite of factors influencing structure loss. We used a correlative approach that did not incorporate mechanisms or feedbacks, but our models clearly illustrated differences in the cumulative effects of individual planning decisions. The relationship between spatial pattern of development and fire risk is likely related to the intermixing of development and wildland vegetation [29,59]; thus, these results likely apply to a wide range of fire-prone ecosystems with large proportions of human-caused ignitions. Nevertheless, because fire risk is highly variable over space and time, and due to a range of human and biophysical variables [60], we recommend planners develop their own models for the best understanding of where the most fire-prone areas are in their region [19].

With projections of substantial global change in climate and human development, we recommend that land use planning should be considered as an important component to fire risk management, potentially to become as successful as the prevention of building on flood plains [61]. History has shown us that preventing fires is impossible in areas where large wildfires are a natural ecological process [4,9]. As Roger Kennedy put it, "the

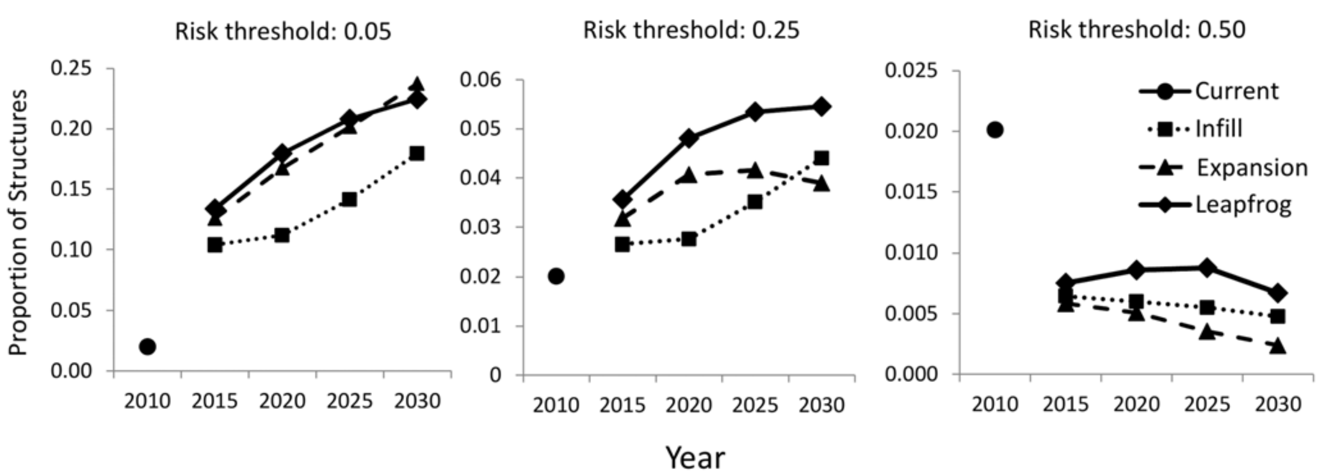

Figure 6. Proportion of residential structures that are located in areas of high fire risk defined using thresholds from the fire risk model of 0.05, 0.25, and 0.5 for current structures and for structures simulated under infill, expansion, and leapfrog growth policies.
doi:10.1371/journal.pone.0071708.g006

problem isn't fires; the problem is people in the wrong places [62]."

Supporting Information

Table S1 Definitions and summary statistics for variables used in the probit model.
(DOCX)

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Author Contributions

Conceived and designed the experiments: ADS JEK ABM VB. Performed the experiments: ADS ABM VB. Analyzed the data: ADS ABM VB. Wrote the paper: ADS JEK ABM VB.

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The role of defensible space for residential structure protection during wildfires

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Abstract. With the potential for worsening fire conditions, discussion is escalating over how to best reduce effects on urban communities. A widely supported strategy is the creation of defensible space immediately surrounding homes and other structures. Although state and local governments publish specific guidelines and requirements, there is little empirical evidence to suggest how much vegetation modification is needed to provide significant benefits. We analysed the role of defensible space by mapping and measuring a suite of variables on modern pre-fire aerial photography for 1000 destroyed and 1000 surviving structures for all fires where homes burned from 2001 to 2010 in San Diego County, CA, USA. Structures were more likely to survive a fire with defensible space immediately adjacent to them. The most effective treatment distance varied between 5 and 20 m (16–58 ft) from the structure, but distances larger than 30 m (100 ft) did not provide additional protection, even for structures located on steep slopes. The most effective actions were reducing woody cover up to 40% immediately adjacent to structures and ensuring that vegetation does not overhang or touch the structure. Multiple-regression models showed landscape-scale factors, including low housing density and distances to major roads, were more important in explaining structure destruction. The best long-term solution will involve a suite of prevention measures that include defensible space as well as building design approach, community education and proactive land use planning that limits exposure to fire.

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Introduction

Across the globe and over recent decades, homes have been destroyed in wildfires at an unprecedented rate. In the last decade, large wildfires across Australia, southern Europe, Russia, the US and Canada have resulted in tens of thousands of properties destroyed, in addition to lost lives and enormous social, economic and ecological effects (Filmon 2004; Boschetti *et al.* 2008; Keeley *et al.* 2009; Blanchi *et al.* 2010; Vasquez 2011). The potential for climate change to worsen fire conditions (Hessl 2011), and the projection of continued housing growth in fire-prone wildlands (Gude *et al.* 2008) suggest that many more communities will face the threat of catastrophic wildfire in the future.

Concern over increasing fire threat has escalated discussion over how to best prepare for wildfires and reduce their effects. Although ideas such as greater focus on fire hazard in land use planning, using fire-resistant building materials and reducing human-caused ignitions (e.g. Cary *et al.* 2009; Quarles *et al.* 2010; Syphard *et al.* 2012) are gaining traction, the traditional strategy of fuels management continues to receive the most attention. Fuels management in the form of prescribed fires or mechanical treatments has historically occurred in remote, wildland locations (Schoennagel *et al.* 2009), but recent studies suggest that treatments located closer to homes and communities may provide greater protection (Witter and Taylor 2005; Stockmann *et al.* 2010; Gibbons *et al.* 2012). In fact, one of the most commonly recommended strategies in terms of fuels and fire protection is to create defensible space immediately around structures (Cohen 2000; Winter *et al.* 2009). Defensible space is an area around a structure where vegetation has been modified, or 'cleared,' to increase the chance of the structure surviving a wildfire. The idea is to mitigate home loss by minimising direct contact with fire, reducing radiative heating, lowering the probability of ignitions from embers and providing a safer place for fire fighters to defend a structure against fire (Gill and Stephens 2009; Cheney *et al.* 2001). Many jurisdictions provide specific guidelines and practices for creating defensible space, including minimum distances that are required among trees and shrubs as well as minimum total distances from the structure. These distances may be enforced through local ordinances or state-wide laws. In California, for example, a state law in 2005 increased the required total distance from 9 m (30 ft) to 30 m (100 ft).

Despite these specific guidelines on how to create defensible space, there is little scientific evidence to support the amount and location of vegetation modification that is actually effective

at providing significant benefits. Most spacing guidelines and laws are based on 'expert opinion' or recommendations from older publications that lack scientific reference or rationale (e.g. Maire 1979; Smith and Adams 1991; Gilmer 1994). However, one study has provided scientific support for, and forms the basis of, most guidelines, policy and laws requiring a minimum of 30 m (100 ft) of defensible space (Cohen 1999, 2000). The modelling and experimental research in that study showed that flames from forest fires located 10–40 m (33–131 ft) away would not scorch or ignite a wooden home; and case studies showed 90% of homes with non-flammable roofs and vegetation clearance of 10–20 m (33–66 ft) could survive wildfires (Cohen 2000). However, the models and experimental research in that study focussed on crown fires in spruce or jack pine forests, and the primary material of home construction was wood. Therefore, it is unknown how well this guideline applies to regions dominated by other forest types, grasslands, or nonforested woody shrublands and in regions where wooden houses are not the norm.

Some older case studies showed that most homes with non-flammable roofs and 10–18 m (33–ft) of defensible space survived the 1961 Bel Air fire in California (Howard *et al.* 1973); most homes with non-flammable roofs and more than 10 m (33 ft) of defensible space also survived the 1990 Painted Cave fire (Foote and Gilless 1996). Also, several fire-behaviour modelling studies have been conducted in chaparral shrublands. One study showed that reducing vegetative cover to 50% at 9–30 m (30–ft) from structures effectively reduced fireline intensity and flame lengths, and that removal of 80% cover would result in unintended consequences such as exotic grass invasion, loss of habitat and increase in highly flammable flashy fuels (A. Fege and D. Pumphrey, unpubl. data). Another showed that separation distances adequate to protect firefighters varied according to fuel model and that wind speeds greater than 23 km h^{-1} negated the effect of slope, and wind speed above 48 km h^{-1} negated any protective effect of defensible space (F. Bilz, E. McCormick and R. Unkovich, unpubl. data, 2009). Results obtained through modelling equations of thermal radiation also found safety distances to vary as a function of fuel type, type of fire, home construction material and protective garments worn by firefighters (Zárate *et al.* 2008).

Although there is no empirical evidence to support the need for more than 30 m (100 ft) of defensible space, there has been a concerted effort in some areas to increase this distance, particularly on steep slopes. In California, a senate bill was introduced in 2008 (SB 1618) to encourage property owners to clear 91 m (300 ft) through the reduction of environmental regulations and permitting needed at that distance. Although this bill was defeated in committee, many local ordinances do require homeowners to clear 91 m (300 ft) or more, and there are reports that some people are unable to get fire insurance without 91 m (300 ft) of defensible space (F. Sproul, pers. comm.). In contrast, homeowner acceptance of and compliance with defensible space policies can be challenging (Winter *et al.* 2009; Absher and Vaske 2011), and in many cases homeowners do not create any defensible space.

It is critically important to develop empirical research that quantifies the amount, location and distance of defensible space that provides significant fire protection benefits so that guidelines and policies are developed with scientific support. Data that are directly applicable to southern California are especially important, as this region experiences the highest annual rate of wildfire-destroyed homes in the US. Not having sufficient defensible space is obviously undesirable because of the hazard to homeowners. However, there are clear trade-offs involved when vegetation reduction is excessive, as it results in the loss of native habitats, potential for increased erosion and invasive species establishment, and it potentially even increases fire risk because of the high flammability of weedy grasslands (Spittler 1995; Keeley *et al.* 2005; Syphard *et al.* 2006).

It is also important to understand the role of defensible space in residential structure protection relative to other factors that explain why some homes are destroyed in fires and some are not. Recent research shows that landscape-scale factors, such as housing arrangement and location, as well as biophysical variables characterising properties and neighbourhoods such as slope and fuel type, were important in explaining which homes burned in two southern California study areas (Syphard *et al.* 2012; 2013). Understanding the relative importance of different variables at different scales may help to identify which combinations of factors are most critical to consider for fire safety.

Our objective was to provide an empirical analysis of the role of defensible space in protecting structures during wildfires in southern California shrublands. Using recent pre-fire aerial photography, we mapped and measured a suite of variables describing defensible space for burned and unburned structures within the perimeters of major fires from 2001 to 2010 in San Diego County to ask the following questions:

1. How much defensible space is needed to provide significant protection to homes during wildfires, and is it beneficial to have more than the legally required 30 m (100 ft)?
2. Does the amount of defensible space needed for protection depend on slope inclination?
3. What is the role of defensible space relative to other factors that influence structure loss, such as terrain, fuel type and housing density?

Methods

Study area

The properties and structures analysed were located in San Diego County, California, USA (Fig. 1) – a topographically diverse region with a Mediterranean climate characterised by cool, wet winters and long summer droughts. Fire typically is a direct threat to structures adjacent to wildland areas. Native shrublands in southern California are extremely flammable during the late summer and fall (autumn) and when ignited, burn in high-intensity, stand-replacing crown fires. Although 500 homes on average have been lost annually since the mid-1900s (Calfire 2000), that rate has doubled since 2000. Most of these homes have burned during extreme fire weather conditions that accompany the autumn Santa Ana winds. The wildland–urban interface here includes more than 5 million homes, covering more than 28 000 km^2 (Hammer *et al.* 2007).

Property data

The data for properties to analyse came from a complete spatial database of existing residential structures and their

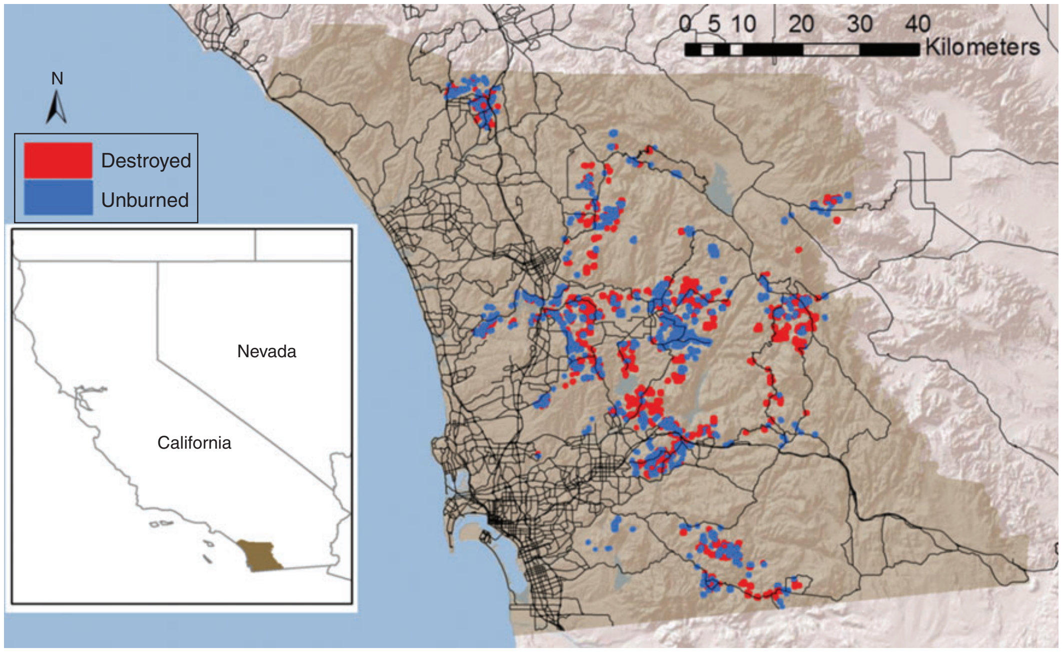

Fig. 1. Location of destroyed and unburned structures within the South Coast ecoregion of San Diego County, California, USA.

corresponding property boundaries developed for San Diego County (Syphard *et al.* 2012). This dataset included 687 869 structures, of which 4315 were completely destroyed by one of 40 major fires that occurred from 2001 to 2010. Our goal was to compare homes that were exposed to wildfire and survived with those that were exposed and destroyed. To determine exposure to fire, we only considered structures located both within a GIS layer of fire perimeters and within areas mapped as having burned at a minimum of low severity through thematic Monitoring Trends in Burn Severity produced by the USA Geological Survey and USDA Forest Service. From these data, we used a random sample algorithm in GIS software to select 1000 destroyed and 1000 unburned homes that were not adjacent to each other, to minimise any potential for spatial autocorrelation. Our final property dataset included structures that burned across eight different fires. More than 97% of these structures burned in Santa Ana wind-driven fire events (Fig. 1).

Calculating defensible space and additional explanatory variables

To estimate defensible space, we developed and explored a suite of variables relative to the distance and amount of defensible space surrounding structures, as well as the proximity of woody vegetation to the structure (Table 1). We measured these variables based on interpretation of Google Earth aerial imagery. We based our measurements on the most recent imagery before the date of the fire. In almost all cases, imagery was available for less than 1 year before the fire.

Our definition of defensible space followed the guidelines published by the California Department of Forestry and Fire Protection (Calfire 2006). 'Clearance' included all areas that were not covered by woody vegetation, including paved areas or grass. Although Google Earth prevents the identification of understorey vegetation, woody trees and shrubs were easily distinguished from grass, and our objective was to measure horizontal distances as required by Calfire rather than assess the relative flammability of different vegetation types. Trees or shrubs were allowed to be within the defensible space zone as long as they were separated by the minimum horizontal required distance, which was 3 m (10 ft) from the edge of one tree canopy to the edge of the next (Fig. 2). Although greater distances between trees or shrubs are recommended on steeper slopes, we followed the same guidelines for all properties. For all structures, we started the distance measurements by drawing lines from the centre of the four orthogonal sides of the structure that ended when they intersected anything that no longer met the requirements in the guidelines. A fair number of structures are not four sided; thus, the start of the centre point was placed at a location that approximated the farthest extent of the structure along each of four orthogonal sides.

We developed two sets of measurements of the distance of defensible space based on what is feasible for homeowners within their properties *v.* the total effective distance of defensible space. We made these two measurements because homeowners are only required to create defensible space within their own property, and this would reflect the effect of individual homeowner compliance. Therefore, even if cleared vegetation extended beyond the property line, the first set of distance measurements ended at the property boundary. The second set of measurements ignored the property boundaries and accounted for the total potential effect of treatment. For all measurements, we recorded the cover types (e.g. structure >3 m (10 ft) long, property boundary, or vegetation type) at which the distance measurements stopped (Table 1). Because property

Table 1. Defensible space variables measured for every structure

Urban veg, landscaping vegetation that was not in compliance with regulations within urban matrix; wildland veg, wildland vegetation that was not in compliance with regulations; orchard, shrub to tree-sized vegetation in rows; urban to wildland, landscaping vegetation that leads into wildland vegetation; structure, any building longer than 3 m (10 ft)

Variable	Definition
Distance defensible space within property	Measure of clearance from side of structure to property boundary calculated for four orthogonal directions from structure and averaged
Total distance defensible space	Measure of clearance from side of structure to end of clearance calculated for four orthogonal directions from structure and averaged
Cover type at end of defensible space	Type of cover encountered at end of measurement (urban veg, wildland veg, orchard, urban to wildland, structure)
Percentage clearance	Percentage of clearance calculated across the entire property
Neighbours' vegetation	Binary indicator of whether neighbours' uncleared vegetation was located within 30 m (100 ft) of the main structure
Vegetation touching structure	Number of sides on which woody vegetation touches main structure (1–4) Structure with more than 4 sides were viewed as a box and given a number between 1 and 4
Vegetation overhanging roof	Was vegetation overhanging the roof? (yes or no)

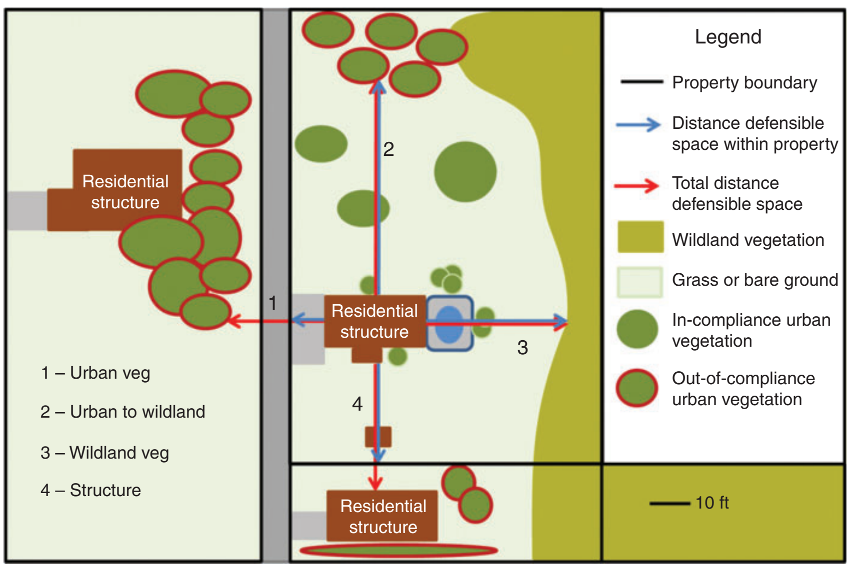

Fig. 2. Illustration of defensible space measurements. See Table 1 for full definition of terms.

owners usually can only clear vegetation on their own land, it is possible that the effectiveness of defensible space partly depends upon the actions of neighbouring homeowners. Therefore, we also recorded whether or not any neighbours' un-cleared vegetation was located within 30 m (100 ft) of the structure.

To assess the total amount of woody vegetation that can safely remain on a property and still receive significant benefits of defensible space, we calculated the total percentage of cleared land, woody vegetation and structure area across every property. This was accomplished by overlaying a grid on each property and determining the proportion of squares falling into each class. Preliminary results showed these three measurements to be highly correlated, so we only retained percentage clearance for further analysis. To evaluate the relative effect of woody vegetation directly adjacent to structures, we also calculated the number of sides of the structure with vegetation touching and recorded whether any trees were overhanging structures' roofs.

In addition to defensible space measurements, we evaluated other factors known to influence the likelihood of housing loss to fire in the region (Syphard *et al.* 2012, 2013). Using the same data as in Syphard *et al.* (2012, 2013), we extracted spatial information from continuous grids of explanatory variables for the locations of all structures in our analysis. Variables included interpolated housing density based on a 1-km search radius; percentage slope derived from a 30-m digital elevation model (DEM); Euclidean distance to nearest major and minor road and fuel type, which was based on a simple classification of US Forest Service data (Syphard *et al.* 2012), including urban, grass, shrubland and forest & woodland.

Analysis

We performed several analyses to determine whether relative differences in home protection are provided by different distances and amounts of defensible space, particularly beyond the legally required 30 m (100 ft), and to identify the effective treatment distance for homes on low and steep slopes.

Categorical analysis

For the first analysis, we divided our data into several groups to identify potential differences among specific categories of defensible space distance around structures located on shallow and steep slopes. We first sorted the full dataset of 2000 structures by slope and then split the data in the middle to create groups of homes with shallow slope and steep slope. We divided the data in half to keep the number of structures even within both groups and to avoid specifying an arbitrary number to define what constitutes shallow or steep slope. The two equal-sized subsets of data ranged from 0 to 9%, with a mean of 8% for shallow slope, and from 9 to 40%, with a mean of 27% for steep slope. Within these data subsets, we next created groups reflecting different mean distances of defensible space around structures. We also performed separate analyses based on whether defensible space measurements were calculated within the property boundary or whether measurements accounted for the total distance of defensible space.

Within all groups, we calculated the proportion of homes that were destroyed by wildfire. We performed Pearson's Chi-square tests of independence to determine whether or not the proportion of destroyed structures within groups was significantly different (Agresti 2007). We based one test on four equal-interval groups within the legally required distance of 30 m (100 ft): 0–7 m (0–25 ft), 8–15 m (26–50 ft), 16–23 m (51–75 ft) and 24–30 m (76–100 ft). A second test was based on three groups (24–30 m (75–100 ft), 31–90 m (101–300 ft) and >90 m (>300 ft) or >60 m (>200 ft)) to evaluate whether groups with mean defensible space distances >30 m (>100 ft) were significantly different from groups with <30 m (<100 ft). When defensible space distances were only measured to the property boundary, few structures had mean defensible space >90 m (>300 ft). Therefore, we used a cut-off of 60 m (200 ft) to increase the sample size in the Chi-square analysis. In addition to the Chi-square analysis, we calculated the relative risk among every successive pair of categories (Sheskin 2004). The relative risk was calculated as the ratio of proportions of burned homes within two groups of homes that had different defensible space distances.

Effective treatment analysis

In addition to comparing the relative effect of defensible space among different groups of mean distances, as described above, we also considered that the protective effect of defensible space for structures exposed to wildfire is conceptually similar to the effect of medication in producing a therapeutic response in people who are sick. In addition to pharmacological applications, treatment–response relationships have been used for radiation, herbicide, drought tolerance and ecotoxicological studies (e.g. Streibig *et al.* 1993; Cedergreen *et al.* 2005; Knezevic *et al.* 2007; Kursar *et al.* 2009). The effect produced by a drug or treatment typically varies according to the concentration or amount, often up to a point at which further increase provides no additional response. The effective treatment (ET50), therefore, is a specific concentration or exposure that produces a therapeutic response or desired effect. Here we considered the treatment to be the distance or amount of defensible space.

Using the software package DRC in R (Knezevic *et al.* 2007; Ritz and Streibig 2013), we evaluated the treatment–response relationship of defensible space in survival of structures during wildfire. To calculate the effective treatment, we fit a log-logistic model with logistic regression because we had a binary dependent variable (burned or unburned). We specified a 2-parameter model where the lower limit was fixed at 0 and the upper limit was fixed at 1. We again performed separate analyses for data subsets reflecting shallow and steep slope, as well as from measurements of defensible space taken within, or regardless of, property boundaries. We also performed analyses to find the effective treatment of percentage clearance of trees and shrubs within the property.

Multiple regression analysis

To evaluate the role of defensible space relative to other variables, we developed multiple generalised linear regression models (GLMs) (Venables and Ripley 1994). We again had a binary dependent variable (burned versus unburned), so we specified a logit link and binomial response. Although the proportion of 0s and 1s in the response may be important to consider for true prediction (King and Zeng 2001; Syphard *et al.* 2008), our objective here was solely to evaluate variable importance. We developed multiple regression models for all possible combinations of the predictor variables and used the corrected Akaike's Information Criterion (AICc) to rank models and select the best ones for each region using package MuMIn in R (R Development Core Team 2012; Burnham and Anderson 2002). We recorded all top-ranked models that had an AICc value within 2 of that of the model with lowest AICc to identify all models with empirical support. To assess variable importance, we calculated the sum of Akaike weights for all models that contained each variable. On a scale of 0–1, this metric represents the weight of evidence that models containing the variable in question are the best model (Burnham and Anderson 2002). The distance of defensible space measured within property boundaries was highly correlated with the distance of defensible space measured beyond property boundaries ($r = 0.82$), so we developed two separate analyses – one using variables measured only within the property boundary and the other using variables that accounted for defensible space outside of the property boundary as well as the potential effect of neighbours having uncleared vegetation within 30 m (100 ft) of the structure. A test to avoid multicollinearity showed all other variables within each multiple regression analysis to be uncorrelated ($r < 0.5$).

Surrounding matrix

To assess whether the proportion of destroyed structures varied according to their surrounding matrix, we summarised the most common cover type at the end of defensible space measurements (descriptions in Table 1) for all structures. These summaries

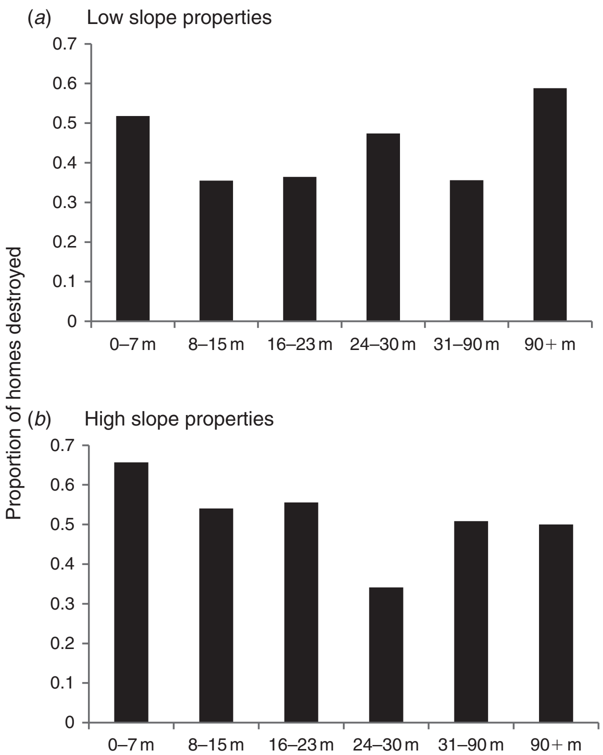

Fig. 3. Proportion of destroyed homes grouped by distances of defensible space based upon total distance of clearance within property boundary, for structures on (*a*) shallow slopes (mean 8%) and (*b*) steep slopes (mean 27%).

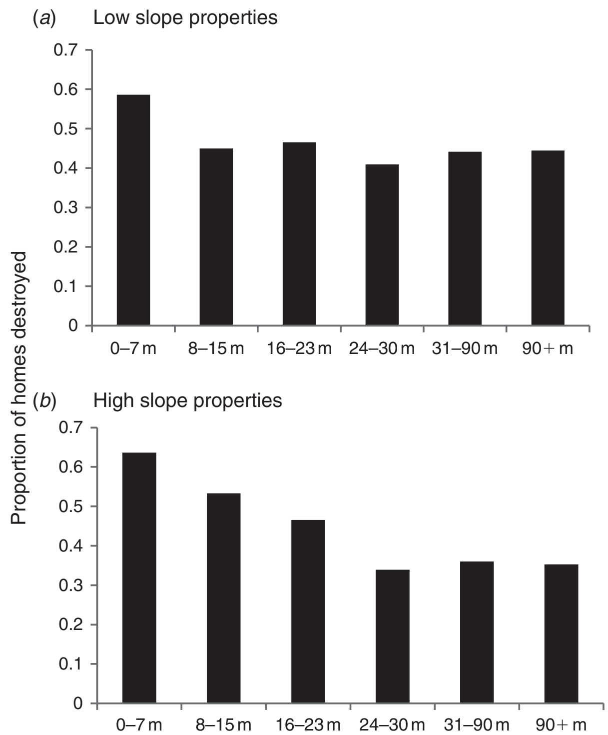

Fig. 4. Proportion of destroyed homes grouped by distances of defensible space based upon total distance of clearance regardless of property boundary, for structures on (*a*) shallow slopes (mean 8%) and (*b*) steep slopes (mean 27%).

were based on the majority surrounding cover type from the four orthogonal sides of the structure. We also noted cases in which there was a tie (e.g. two sides were urban vegetation and two sides were structures).

Results

Categorical analysis

When the distance of defensible space was measured both 'only within property boundaries' (Fig. 3) and 'regardless of property boundaries' (Fig. 4), the Chi-square test showed a significant difference ($P < 0.001$) in the proportion of destroyed structures among the four equal-interval groups of distance ranging from 0 to 30 m (0–100 ft). This relationship was consistent on both shallow-slope and steep-slope properties, although the relative risk analysis showed considerable variation among classes (Table 2) There was a steadily decreasing proportion of destroyed structures at greater distances of defensible space up to 30 m (100 ft) on the steep-slope structures with defensible space measured regardless of property boundaries (Fig. 4*b*). Otherwise, the biggest difference in proportion of destroyed structures occurred between 0 and 7 m (0–25 ft) and 8–15 m (26–50 ft) (Figs 3*a–b*, 4*a*).

When the distance of defensible space was measured in intervals from 24 m (75 ft) and beyond, the Chi-square test showed no significant difference among groups ($P = 0.96$ for shallow-slope properties and $P = 0.74$ for steep-slope properties) (Figs 3, 4), although again, the relative risk analysis showed considerable variation (Table 2).There was a slight increase in the proportion of homes destroyed at longer distance intervals when the defensible space was measured only to the property boundaries (Fig. 3*a–b*). This slight increase is less apparent when distances were measured regardless of boundaries (Fig. 4*a–b*).

The relative risk calculations showed that the ratio of proportions was generally more variable among successive pairs when the distances were measured within property boundaries (Table 2). For these calculations, the risk of a structure being destroyed was significantly lower when the defensible space distance was 8–15 m (25–50 ft) compared to 0–7 m (0–25 ft) on both shallow- and steep-slope properties. On the steep-slope properties, there was an additional reduction of risk when comparing 24–30 m (75–100 ft) to 16–23 m (50–75 ft). However, the risk of a home being destroyed was slightly significantly higher when there was 31–90 m (101–225 ft) compared to 16–23 m (50–75 ft). For distances that were measured regardless of property boundary (total clearance), the only significant differences in risk of burning were a reduction in risk for 8–15 m (25–50 ft) compared to 0–7 m (0–25 ft).

Table 2. Number of burned and unburned structures within defensible space distance categories (m), their relative risk and significance
A relative risk of 1 indicates no difference; <1 means the chance of a structure burning is less than the other group; >1 means the chance is higher than the other group. The relative risk is calculated for pairs that include the existing row and the row above. Confidence intervals are in parentheses

	Distance within property				Total distance			
	Burned	Unburned	Relative risk	*P*	Burned	Unburned	Relative risk	*P*
Shallow slope								
0–7	200	186			162	114		
8–15	109	198	0.69 (0.12)	<0.001	108	132	0.77	0.002
16–23	51	89	1.03 (0.30)	0.850	78	90	1.03	0.770
24–30	36	40	1.30 (0.39)	0.110	50	70	0.90	0.430
31–90	28	47	0.79 (0.24)	0.220	79	99	1.06	0.640
60 or 90+	10	6	1.67 (0.63)	0.040	8	9	1.01	0.830
Steep slope								
0–7	245	128			224	128		
8–15	174	148	0.82 (0.10)	0.001	158	139	0.84	0.008
16–23	85	68	1.03 (0.16)	0.750	73	83	0.87	0.210
24–30	29	56	0.61 (0.17)	0.004	26	50	0.73	0.080
31–	29	28	1.49 (0.48)	0.050	39	68	1.06	0.760
60 or 90+	5	5	0.98 (0.47)	0.950	4	8	0.91	0.830

Table 3. Effective treatment results reflecting the distance (in metres, with feet in parentheses) and percentage clearance within properties that provided significant improvement in structure survival during wildfires
The property mean is the average distance of defensible space or percentage clearance that was calculated on the properties before the wildfires and provides a means to compare the effective treatment result to the actual amount on the properties

	All parcels effective treatment ($n = 2000$)	Parcel mean	Shallow slope (mean 8%) effective treatment ($n = 1000$)	Parcel mean	Steep slope (mean 27%) effective treatment ($n = 1000$)	Parcel mean
Defensible space within parcel	10 (33)	13 (44)	4 (13)	14 (45)	25 (82)	11 (35)
Total distance defensible space	10 (32)	19 (63)	5 (16)	20 (67)	20 (65)	18 (58)
Mean percentage clearance on property	36	48	31	51	37	35

Effective treatment analysis

Analysis of the treatment–response relationships among defensible space and structures that survived wildfire showed that, when all structures are considered together, the mean actual defensible space that existed around structures before the fires was longer than the calculated effective treatment (Table 3). Regardless of whether the defensible space was measured within or beyond property boundaries, the estimated effective treatment of defensible space was nearly the same at 10 m (32–33 ft).

The effective treatment distance was much shorter for structures on shallow slopes (4–5 m (13–16 ft)) than for structures on steep slopes (20–25 m (65–82 ft)), but in all cases was <30 m (<100 ft). Although longer distances of defensible space were calculated as effective on steeper slopes, these structures actually had shorter mean distances of defensible space around their properties than structures on low slopes (Table 3).

The calculated effective treatment of the mean percentage clearance on properties was 36% for all properties, 31% for structures on shallow slopes and 37% for structures on steep slopes (Table 3). In total, the properties all had higher actual percentage clearance on their property than was calculated to be effective. However, this mainly reflects the shallow-slope properties, as those structures on steep slopes had less clearance than the effective treatment.

Multiple regression analysis

When defensible space was measured only to the property boundaries, it was not included in the best model, according to the all-subsets multiple regression analysis (Table 4). However, it was included in the best model when factoring in the distance of defensible space measured beyond property boundaries (Table 5). In both multiple regression analyses, low housing density and shorter distances to major roads were ranked as the most important variables according to their Akaike weights. Slope and surrounding fuel type were also in both of the best models as well as other measures of defensible space, including the percentage clearance on property and whether vegetation was overhanging the structure's roof. The number of sides in which vegetation was touching the structure was included in the best model when defensible space was only measured to the property boundary. The total explained deviance for the multiple regression models was low (12–13%) for both analyses.

Table 4. Results of multiple regression models of destroyed homes using all possible variable combinations and corrected Akaike's Information Criterion (AICc)

Includes variables measured within property boundary only. Top-ranked models include all those ($n = 12$) with AICc within 2 of the model with the lowest AICc. Relative variable importance is the sum of 'Akaike weights' over all models including the explanatory variable

Variable in order of importance	Relative variable importance	Model-averaged coefficient	Number inclusions in top-ranked models
Housing density	1	−0.003	12
Distance to major road	1	−0.0005	12
Percentage clearance	1	−0.02	12
Slope	1	0.03	12
Vegetation overhang roof	1	0.5	12
Fuel type	0.67	Factor	9
Vegetation touch structure	0.49	0.07	6
Distance defensible space within property	0.45	−0.0002	5
South-westness	0.36	−0.0007	3
Distance to minor road	0.28	−0.0002	1
D^2 of top-ranked model			0.123

Table 5. Results of multiple regression models of destroyed homes using all possible variable combinations and corrected Akaike's Information Criterion (AICc)

Includes variables measured beyond property boundary. Top-ranked models include all those ($n = 6$) with AICc within 2 of the model with the lowest AICc. Relative variable importance is the sum of 'Akaike weights' over all models including the explanatory variable

Variable in order of importance	Relative variable importance	Model-averaged coefficient	Number inclusions in top-ranked models
Housing density	1	−0.003	6
Distance to major road	1	−0.0005	6
Total distance defensible space	1	−0.004	6
Percentage clearance	1	−0.01	6
Vegetation overhang roof	0.99	0.4	6
Slope	0.99	0.03	6
Fuel type	0.86	Factor	4
South-westness	0.42	−0.0009	2
Distance to minor road	0.36	−0.0009	2
Neighbours' vegetation	0.27	0.08	1
Vegetation touch structure	0.27	0.18	1
D^2 of top-ranked model			0.125

Surrounding matrix

The cover type that most frequently surrounded the structures at the end of the defensible space measurements was urban vegetation, followed by urban vegetation leading into wildland vegetation, and wildland vegetation (Fig. 5). Many structures were equally surrounded by different cover types. There were no significant differences in the proportion of structures destroyed depending on the surrounding cover type. However, a disproportionately large proportion of structures burned (28 *v.* 9% unburned) when they were surrounded by urban vegetation that extended straight into wildland vegetation.

Discussion

For homes that burned in southern Californian urban areas adjacent to non-forested ecosystems, most burned in high-intensity Santa Ana wind-driven wildfires and defensible space increased the likelihood of structure survival during wildfire. The most effective treatment distance varied between 5 and 20 m (16–58 ft), depending on slope and how the defensible space was measured, but distances longer than 30 m (100 ft) provided no significant additional benefit. Structures on steeper slopes benefited from more defensible space than structures on shallow slopes, but the effective treatment was still less than 30 m (100 ft). The steepest overall decline in destroyed structures occurred when mean defensible space increased from 0–7 m (0–25 ft) to 8–15 m (26–50 ft). That, along with the multiple regression results showing the significance of vegetation touching or overhanging the structure, suggests it is most critical to modify vegetation immediately adjacent to the house, and to move outward from there. Similarly, vegetation overhanging the structure was also strongly correlated with structure loss in Australia (Leonard *et al.* 2009).

In terms of fuel modification, the multiple regression models also showed that the percentage of clearance was just as, or more important than, the linear distance of defensible space.

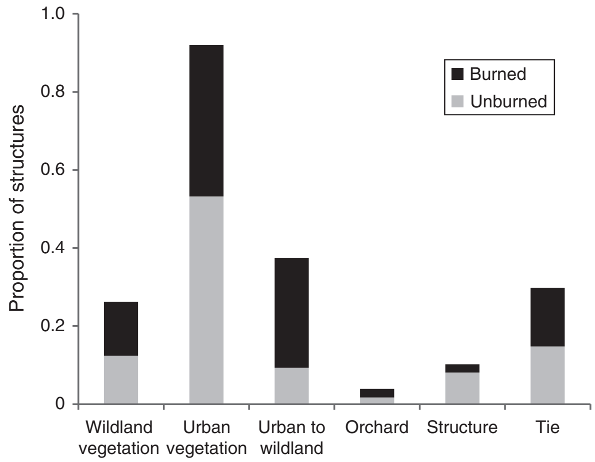

Fig. 5. Proportion of destroyed and unburned structures based on the primary surrounding cover type at the end of defensible space measurements. There were no significant differences in the proportion of burned and unburned structures within cover types ($P = 0.14$). Cover types are defined in Table 1.

However, as with defensible space, percentage clearance did not need to be draconian to be effective. Even on steep slopes, the effective percentage clearance needed on the property was <40%, with no significant advantage beyond that. Although these steep-slope structures benefited more from clearance, they tended to have less clearance than the effective amount, which may be why slope was such an important variable in the multiple regression models. Shallow-slope structures, in contrast, had more clearance on average than was calculated to be effective, suggesting these property owners do not need to modify their behaviours as much relative to people living on steep slopes.

Although the term 'clearance' is often used interchangeably with defensible space, this term is incorrect when misinterpreted to mean clearing all vegetation, and our results underline this difference. The idea behind defensible space is to reduce the continuity of fuels through maintenance of certain distances among trees and shrubs. Although we could not identify the vertical profile of fuels through Google Earth imagery, the fact that at least 60% of the horizontal woody vegetative cover can remain on the property with significant protective effects demonstrates the importance of distinguishing defensible space from complete vegetation removal. Thus, we suggest the term 'clearance' be replaced with 'fuel treatment' as a better way of communicating fire hazard reduction needs to home owners.

The percentage cover of woody shrubs and trees was not evenly distributed across properties, and we did not collect data describing how the cover was distributed. Considering the importance of defensible space and vegetation modification immediately adjacent to the structure, it should follow that actions to reduce cover should also be focussed in close proximity to the structure. The hazard of vegetation near the structure has apparently been recognised for some time (Foote *et al.* 1991; Ramsey and McArthur 1994), but it is not stressed enough, and rarely falls within the scope of defensible space guidelines or ordinances.

In addition to the importance of vegetation overhanging or touching the structure, it is important to understand that ornamental vegetation may be just as, if not more, dangerous than native vegetation in southern California. Although the results showed no significant differences in the cover types in the surrounding matrix, there was a disproportionately large number of structures destroyed (28% burned *v.* 9% unburned) when ornamental vegetation on the property led directly into the wildland. Ornamental vegetation may produce highly flammable litter (Ganteaume *et al.* 2013) or may be particularly dangerous after a drought when it is dry, or has not been maintained, and species of conifer, juniper, cypress, eucalypt, *Acacia* and palm have been present in the properties of many structures that have been destroyed (Franklin 1996). Nevertheless, ornamental vegetation is allowed to be included as defensible space in many codes and ordinances (Haines *et al.* 2008).

One reason that longer defensible space distances did not significantly increase structure protection may be that most homes are not destroyed by the direct ignition of the fire front but rather due to ember-ignited spot fires, sometimes from fire brands carried as far as several km away. Although embers decay with distance, the difference between 30 and 90 m (100 and 300 ft) may be small relative to the distance embers travel under the severe wind conditions that were present at the time of the fires. The ignitability of whatever the embers land on, particularly adjacent to the house, is therefore most critical for propagating the fire within the property or igniting the home (Cohen 1999; Maranghides and Mell 2009).

Aside from roofing or home construction materials and vegetation immediately adjacent to structures (Quarles *et al.* 2010; Keeley *et al.* 2013), the flammability of the vegetation in the property may also play a role. Large, cleared swaths of land are likely occupied at least in part by exotic annual grasses that are highly ignitable for much of the year. Conversion of woody shrubs with higher moisture content into low-fuel-volume grasslands could potentially increase fire risk in some situations by increasing the ignitability of the fuel; and if the vegetation between a structure and a fire is not readily combustible, it could protect the structure by absorbing heat flux and filtering fire brands (Wilson and Ferguson 1986).

The slight increase in proportion of structures destroyed with longer distances of defensible space within parcel boundaries was surprising. However, that increase was not significant in the Chi-square analysis, although there were some significant differences in the pairwise relative risk analysis. Nevertheless, the largest significant effect of defensible space was between the categories of 0–7 m (0–25 ft) to 8–15 m (26–50 ft), and it may be that differences in categories beyond these distances are not highly meaningful or reflect an artefact of the definition of distance categories. These relationships at longer distances are likely also weak compared to the effect of other variables operating at a landscape scale. Although the categorical analysis allowed us to answer questions relative to legal requirements and specific distances, the effective treatment analysis was important for identifying thresholds in the continuous variable.

The multiple regression models showed that landscape factors such as low housing density and longer distances to major roads were more important than distance of defensible space for explaining structure destruction, and the importance of

these variables is consistent with previous studies (Syphard *et al.* 2012, 2013), despite the smaller spatial extent studied here. Whereas this study used an unburned control group exposed to the same fires as the destroyed structures, previous studies accounted for structures across entire landscapes. The likelihood of a fire destroying a home is actually a result of two major components: the first is the likelihood that there will be a fire, and the second is the likelihood that a structure will burn in that fire. In this study, we only focussed on structure loss given the presence of a fire, and the total explained variation for the multiple regression models was quite low at ~12%. However, when the entire landscape was accounted for in the total likelihood of structure destruction, the explained variation of housing density alone was >30% (Syphard *et al.* 2012). One reason for the relationship between low housing density and structure destruction is that structures are embedded within a matrix of wildland fuel that leads to greater overall exposure, which is consistent with Australian research that showed a linear decrease of structure loss with increased distance to forest (Chen and McAneney 2004). That research, however, only focussed on distance to wildland boundaries and did not quantify variability in defensible space or ornamental vegetation immediately surrounding structures. Thus, fire safety is important to consider at multiple scales and for multiple variables, which will ultimately require the cooperation of multiple stakeholders.

Conclusions

Structure loss to wildfire is clearly a complicated function of many biophysical, human and spatial factors (Keeley *et al.* 2009; Syphard *et al.* 2012). For such a large sample size, we were unable to account for home construction materials, but this is also well understood to be a major factor, with older homes and wooden roofs being most vulnerable (Franklin 1996; Cohen 1999, 2000). In terms of actionable measures to reduce fire risk, this study shows a clear role for defensible space up to 30 m (100 ft). Although the effective distances were on average much shorter than 30 m (100 ft), we recognise that additional distance may be necessary to provide sufficient protection to firefighters, which we did not address in this study (Cheney *et al.* 2001). In contrast, the data in this study do not support defensible space beyond 30 m (100 ft), even for structures on steep slopes. In addition to the fact that longer distances did not contribute significant additional benefit, excessive vegetation clearance presents a clear detriment to natural habitat and ecological resources. Results here suggest the best actions a homeowner can take are to reduce percentage cover up to 40% immediately adjacent to the structure and to ensure that vegetation does not overhang or touch the structure.

In addition to defensible space, this study also underlines the potential importance of land use planning to develop communities that are fire safe in the long term, in particular through their reduction to exposure to wildfire in the first place. Localised subdivision decisions emphasising infill-type development patterns may significantly reduce fire risk in the future, in addition to minimising habitat loss and fragmentation (Syphard *et al.* 2013). This study was conducted in southern California, which has some of the worst fire weather in the world and many properties surrounded by large, flammable exotic trees. Therefore, recommendations here should apply to other non-forested ecosystems as well as many forested regions.

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Importance of Roadless Areas in Biodiversity Conservation in Forested Ecosystems: Case Study of the Klamath-Siskiyou Ecoregion of the United States

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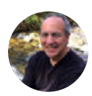

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Importance of Roadless Areas in Biodiversity Conservation in Forested Ecosystems: Case Study of the Klamath-Siskiyou Ecoregion of the United States

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Abstract: *Although many roadless areas on federal lands have been mapped in the United States since the 1970s, there has been little specific research on how and to what degree roadless areas contribute to biodiversity conservation. We examined the ecological attributes of mapped roadless areas for the Klamath-Siskiyou ecoregion of northwestern California and southwestern Oregon (U.S.A.). Attributes examined include special elements (such as natural heritage, serpentine geology, late-seral forests, Port Orford cedar* [Chamaecyparis lawsoniana]*), and key watersheds; elevation and habitat representation; and overall landscape connectivity. We compared designated wilderness to roadless areas, giving special attention to the relative importance of small roadless areas (405–2024 ha). We mapped nearly 500 roadless areas of ≥405 ha. Roadless areas occupied more than twice the land area of wilderness (approximately 27% of the entire ecoregion) and contained approximately 36% of the known occurrences of heritage elements, 37% of the mapped serpentine habitats, 36% of the remaining late-seral forests, 60% of Port Orford cedar strongholds, and 42% of key watersheds for aquatic biodiversity. In addition, roadless areas were composed of significant amounts of low- and mid-elevation sites and a substantial number of the 214 mapped physical-biological habitat types with strong complementarity with designated wilderness. Fragmentation analyses showed that roadless areas contributed to regional connectivity in important ways. Also, small roadless areas were an important component of the roadless-areas conservation assessment. For the Klamath-Siskiyou ecoregion, roadless areas and designated wilderness provide an important foundation upon which to develop a comprehensive regional conservation strategy.*

Importancia de Áreas sin Caminos para la Conservación de la Biodiversidad en Ecosistemas Forestales: Estudio de Caso de la Ecoregión Klamath-Siskiyou, E.U.A.

Resumen: *Aunque se ha mapeado una gran cantidad de superficie sin caminos en tierras federales de los E.U.A. desde la década de 1970, se ha investigado poco sobre como y hasta qué grado las áreas sin caminos contribuyen a la conservación de la biodiversidad. Examinamos los atributos ecológicos de áreas sin caminos en la ecoregión de Klamath-Siskiyou en el noroeste de California y suroeste de Oregon (E.U.A.). Los atributos examinados incluyeron elementos especiales (patrimonio natural, geología de serpentina, bosques serales recientes, cedro [*Chamaecyparis lawsoniana*] y cuencas clave), elevación y representación de hábitat y la conectividad general del paisaje. Comparamos áreas designadas para vida silvestre con áreas sin caminos, poniendo especial atención a la importancia relativa de áreas pequeñas sin caminos (405-2024 ha). Hicimos mapas de cerca de 500 áreas sin caminos ≥ (405 ha). Las áreas sin caminos ocuparon más del doble de la superficie de las áreas de vida silvestre (aproximadamente el 27% de toda la ecoregión). Las áreas sin caminos y contenían aproximadamente el 36% de elementos ancestrales conocidos, el 37% de los habitats serpentinos mapeados, el 36% de los bosques serales recientes, 60% de la población de cedro y 42% de las cuencas*

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clave para la biodiversidad acuática. Adicionalmente, las áreas sin caminos estaban compuestas de cantidades significativas de sitios bajos y de elevación media y cantidades sustanciales de 214 tipos de hábitats fisicobiológicos con una fuerte complementariedad con áreas designadas para vida silvestre. Los análisis de fragmentación mostraron que las áreas sin caminos contribuyeron a la conectividad regional de manera importante. También, las áreas sin caminos pequeñas fueron un componente importante de la evaluación de conservación de las áreas sin caminos. Para la ecoregión Klamath-Siskiyou, las áreas sin caminos y las designadas para vida silvestre son un fundamento importante sobre el que se puede desarrollar una estrategia integral de conservación regional.

Introduction

Natural habitat destruction and fragmentation are the leading causes of the decline and loss of species worldwide and have been a topic of considerable research and review (Harris 1984; Wilcove et al. 1986; Usher 1987; Saunders et al. 1991; Wilson 1992). Fragmentation of forested landscapes has been under particular scrutiny and has been shown to contribute to population declines in many species (Matthiae & Stearns 1981; Harris 1984). The process of deforestation and fragmentation can be complex, having both spatial and temporal components (Zipperer 1993). Roads allow access to pristine areas and fragment native ecosystems into smaller and smaller patches of various sizes and shapes (Dickman 1987; Atkinson & Cairns 1992). Nearly all native ecosystems are destined to resemble ever smaller and more isolated habitat islands as humans continue to encroach on remaining natural habitats (Wilcox 1980).

According to the National Research Council (1997), there are approximately 4 million miles of roadway in the United States. That covers about 1% of the conterminous United States, but the negative ecological effects of the "road-effect zone" are often much greater—18–22% (Forman 2000). Roads constructed to gain access to resources on public lands have been substantial, and in some cases extremely heavy, over the last 50 years. It is difficult to determine the number of all roads currently on public lands, but agency estimates exist. The U.S. Forest Service (USFS) maintains approximately 440,000 miles of roads, nearly 10 times the total length of the interstate highway system.

Roads and the maintenance of roads affect natural terrestrial and aquatic environments in many ways. Increased erosion, air, and water pollution, spread of invasive exotics, road mortality and avoidance, and habitat fragmentation all accompany roads (reviewed by Andrews 1990; Spellerberg 1998; Jones et al. 2000; Trombulak & Frissell 2000). Roads directly fragment natural ecosystems (Reed et al. 1996), but—more importantly—they also provide access to areas, which leads to subsequent human disturbances from activities such as logging, mining, grazing, agriculture, and urban development. These disturbances result in substantial declines in native species and an overall degradation of ecosystem integrity. Roads, deforestation, and fragmentation are intimately related.

On USFS land, two lengthy roadless-area reviews and evaluations (RARE I and RARE II) were performed in the 1970s. The objective of the reviews was to inventory the roadless areas of the national forest system to determine which ones should be considered for wilderness designation as a result of the Wilderness Act of 1964 (Crowell & Cutler 1983). This substantial mapping activity set a minimum size of 2024 ha for designation as "wilderness;" smaller areas would be ineligible for such designation. During the 96th Congress, over half of the 6 million ha of roadless areas recommended by RARE II were declared wilderness. In the early 1980s, with the Reagan administration and new leadership in the 97th Congress, all actions on roadless areas were suspended (Crowell & Cutler 1983). Since that time, roadless areas have been mapped by a wide range of conservationists using a variety of techniques. As large unroaded lands disappear, a minimum size of 405 ha is now being examined for wilderness designation. Geographic information systems (GIS) have been employed extensively in recent years to address this issue. The nongovernmental conservation community has widely promoted roadless areas as important conservation lands (Foreman & Wolke 1992; Noss & Cooperrider 1994), but few studies have evaluated their actual ecological benefits.

On 13 October 1999, President Clinton directed the USFS to provide strong and lasting protection for roadless areas in the national forest system. The President's directive initiated a rule-making process on roadless areas by the USFS that lead to over 517,000 comments from the public (USFS 2000). On 10 May 2000, the USFS released a proposed rule and draft environmental impact statement for which the public provided over 1.1 million additional responses (USFS 2000). The rule-making process for roadless areas marked the biggest public commenting process ever undertaken by the USFS. Given the significance of the impending roadless-area policy, we sought to evaluate the ecological attributes of existing roadless areas in a forest ecoregion, the Klamath-Siskiyou. The ecological attributes we examined included five special elements of conservation concern, two representation evaluations, and regional landscape connectivity.

Methods

Study Area

The Klamath-Siskiyou ecoregion of northwestern California and southwestern Oregon has long been recognized for its outstanding biodiversity (Whittaker 1960; Kruckeberg 1984; DellaSala et al. 1999). The World Conservation Union (IUCN) considers the Klamath-Siskiyou an area of global botanical significance (Wagner 1997), and the World Wildlife Fund chose it as a global 200 ecoregion, meaning that it is of high biodiversity value and under considerable threat (Ricketts et al. 1999). The ecoregion, as we define it, covers over 43,000 km^2, of which approximately 63% is in public ownership (83% of this by the USFS). Nearly 13% of the ecoregion is considered strictly protected, primarily through a number of relatively large, scattered wilderness areas (Fig. 1).

Roadless Area Mapping and Evaluation

As part of a regional conservation assessment (Strittholt et al. 1999), we assembled road data and mapped roadless areas (≥405 ha). We used 1:100,000 road data for the entire ecoregion and 1:24,000 for the public-lands portion. Using the 1:100,000 scaled data, we calculated road density with a 5 × 5 km moving-window operation on density results from a 1 × 1 km fixed grid, which provided a context for the subsequent mapping and evaluation of roadless areas. We mapped roadless areas on existing public lands using a newly developed technique that combines raster and vector GIS operations on 1:24,000–scaled roads data (Strittholt et al. 1999). We included all public lands in mapping and assessing ecological values regardless of agency jurisdiction. In both road density and roadless-area mapping, we excluded trails from the analysis. We used Arc/Info GIS to conduct the computer-mapping analyses.

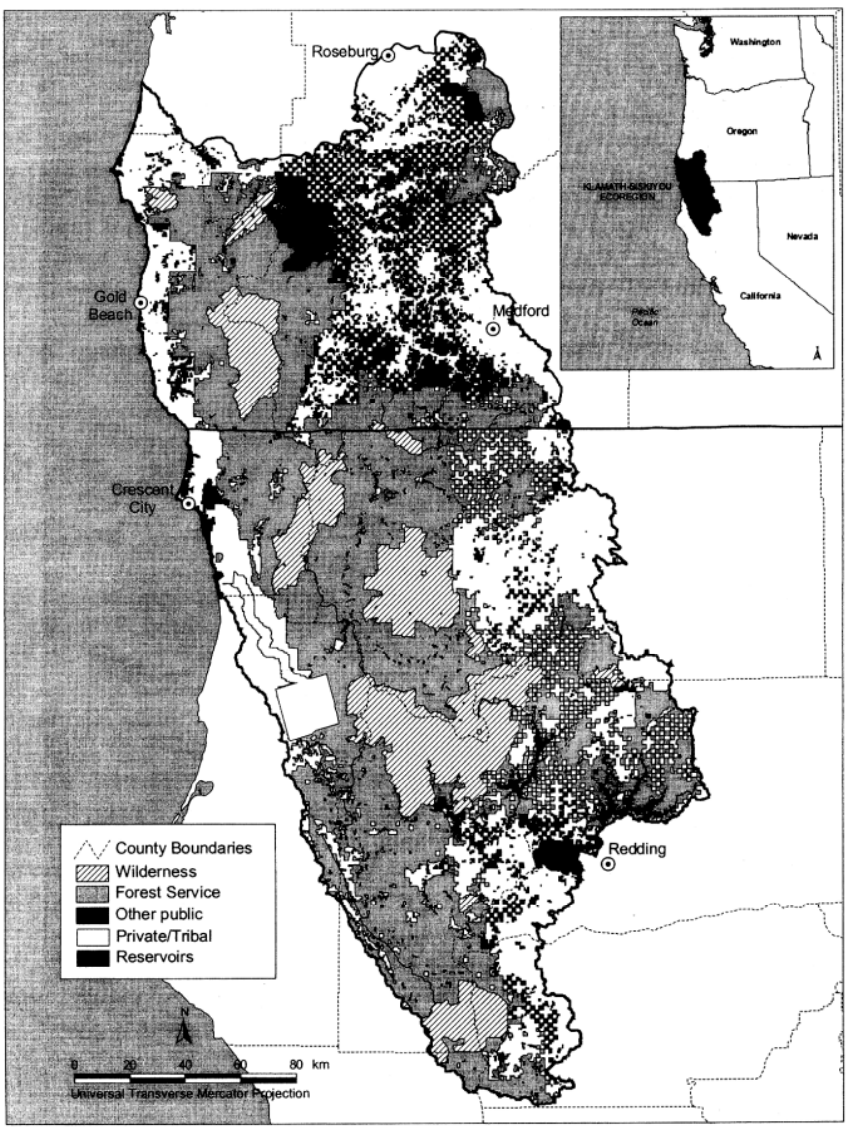

Figure 1. Klamath-Siskiyou primary ownership and existing wilderness areas.

Using all available GIS data layers, we focused on evaluating the ecological or conservation attributes of the remaining roadless areas in the Klamath-Siskiyou ecoregion, and examined large (>2024 ha) and small (405–2024 ha) roadless areas separately. We also subjected the designated wilderness areas to the same data queries and summarized the conservation benefits of adding the roadless areas to the current wilderness system for the ecoregion.

Special Elements

We evaluated the mapped roadless areas in terms of five special-element criteria in the region, including the presence or absence of natural-heritage elements, serpentine geology (of special importance to this region), late-seral forest, high abundances of Port Orford cedar (*Chamaecyparis lawsoniana*) with few or no signs of infestation by a lethal root-rot fungus, and watersheds of aquatic biodiversity importance (or key watersheds) as identified by Forest Ecosystem Management Assessment Team (1993).

Occurrences of natural-heritage elements are known point locations for plant and animal species of special conservation interest, including rare and endangered species. Computer databases are collected and maintained on these species by natural-heritage programs organized by individual U.S. states. For this study, heritage data were obtained from the Oregon Natural Heritage Program and California Natural Diversity Database, and all records since 1970 were included in the analysis. We examined 8793 heritage-element occurrence records, which fell within the Klamath-Siskiyou ecoregion boundary. We treated each record (or point location) equally; rarer species were not weighted. We subdivided all point locations into six categories, primarily by taxonomic group, and overlaid them against data layers for mapped roadless areas and designated wilderness areas.

The Klamath-Siskiyou ecoregion is noted for its abundance of serpentine bedrock geology (13%, or 575,550 ha of the ecoregion). Serpentine is a metamorphic rock upon which ultramafic soils are built. Ultramafic soils are unique in many of their physical and chemical properties: they are low in exchangeable calcium and high in magnesium, and they tend to be deficient in many soil nutrients. Many of these soils contain high levels of heavy metals, such as nickel, chromium, and cobalt, that impede normal plant growth and development (Coleman & Kruckeberg 1999). For these reasons, serpentine geology is one of several known factors that have contributed to species endemism in the Klamath-Siskiyou (DellaSala et al. 1999; Strittholt et al. 1999). Protection of areas of serpentine geology therefore becomes an important regional conservation objective. From existing U.S. Geological Survey state geology maps (1:500,000 map scale) for Oregon and California, we digitized serpentine geology. Using overlay operations, we then calculated the area of serpentine geology in designated wilderness and mapped roadless areas.

Based on interpretation of mid-1990s satellite images, approximately 22% of the Klamath-Siskiyou ecoregion contained late-seral forest (928,356 ha), and 80% of it was on public land (Strittholt et al. 1999). Late-seral forest, which once dominated much of the Pacific Northwest, has been in significant decline since the end of World War II. Many species and natural processes depend upon older forests, and they are of special conservation concern in the Klamath-Siskiyou. We calculated the area of late-seral forest in designated wilderness and mapped roadless areas. Spatial resolution of the forest data was 30 × 30 m and based on two different datasets, one for the Oregon portion of the ecoregion (Cohen et al. 1995) and the other for the California portion (Legacy, unpublished data). Late seral was defined as any forest type more than 100 years of age. Late-seral condition is not equally important among the various forest types found in the Klamath-Siskiyou. For example, some globally imperiled forest types are found in the ecoregion, including white fir (*Abies concolor*), Port Orford cedar, Brewer spruce (*Picea breweriana*), and huckleberry oak (*Quercus vaccinifolia*), and these are of particular concern (DellaSala et al. 1999), but we did not have the data necessary to evaluate this criterion in greater depth. In addition, only a subset of this area could be regarded as old-growth forest, but, again, such detailed information on forest age was not available for the ecoregion.

Port Orford cedar is an important southwestern Oregon–northwest California endemic tree species (Lang 1999) that grows primarily in riparian areas, where it provides channel stabilization, shade for waterways, and microhabitat for numerous aquatic species (Jimerson & Creasy 1997). In an area where migratory species—most notably salmon—make up the bulk of the region's fish fauna, streamside integrity is of paramount importance, and Port Orford cedar is one of the dominant riparian tree species throughout the western sections of the Klamath-Siskiyou ecoregion. These cedars are at risk because of their value in Asian markets (there is no major domestic market for Port Orford cedar) and, more important, from an imported root-rot fungus (*Phytophthora lateralis*) (Lang 1999). This fungus is water-borne and is usually associated with the building and use of roads for logging, mining, and recreation (Jimerson & Creasy 1997). Spores are easily picked up from infected sites and transported to uninfected ones on the tires of vehicles. Infestation usually results in mortality (Zobel et al. 1985). Because of their limited distribution and current threats from management and the exotic root-rot fungus, Port Orford cedar plant communities were classified as G2 (or globally imperiled) by The Nature Conservancy (Grossman et al. 1994).

We mapped Port Orford cedar distribution and root-rot fungus infestation by sixth-field subwatershed, and we evaluated both cedar density and infestation rate against designated wilderness and mapped roadless areas. Watersheds (or catchments) include all lands enclosed by a continuous hydrologic surface-drainage divide and lying upslope from a specified point on a stream (Maxwell et al. 1995). Watersheds are hierarchical and can be modeled from digital elevation models (Maidment & Djokic 2000) or delineated from hydrology and contour maps. The U.S. Geological Survey developed its own hierarchical watershed-delineation system based on a hydrologic unit code-naming convention. In this system, every two digits of a hydrologic unit code correspond to each level in the hydrologic unit system: A 12-digit code identifies six nested subwatersheds, the sixth pair of digits representing the finest subwatershed.

Of the 1135 sixth-field subwatersheds mapped for the Klamath-Sisikyou ecoregion, 215 contained cedar data, all on USFS public lands. We assigned ordinal scores (1–5, with 5 representing the highest cedar density and lowest level of infestation) to cedar density and degree of infestation separately, added them together to obtain a composite score, and assigned the score to each subwatershed (Strittholt et al. 1999). Of the 903,559 ha (n = 215 subwatersheds) containing cedar and infestation data, 19% (179,430 ha, n = 40 subwatersheds) scored low, meaning they had a combination of low cedar density and high infestation. Approximately 22% of the subwatersheds (199,389 ha, n = 48) scored high (i.e., high cedar density and low infestation). We classified the remaining subwatersheds (58% or 524,740 ha, n = 127) as medium (i.e., moderate cedar density and moderate infestation). We then overlaid the designated wilderness and mapped roadless areas on these composite-score categories and summarized the results.

Of the 1135 sixth-field subwatershed basins mapped for the Klamath-Siskiyou ecoregion (Strittholt et al. 1999), 333 were previously identified as key watersheds by fisheries biologists of the Forest Ecosystem Management Assessment Team (1993). Key watersheds are those believed to be of special importance for aquatic biodiversity, with particular emphasis on the salmonid species and stocks at risk throughout the Pacific Northwest. We determined the number and area of key watersheds in designated wilderness and mapped roadless areas.

Representation

We measured elevation bands and natural habitat types to evaluate the contribution roadless areas make to ecosystem representation. Elevations in the Klamath-Siskiyou ecoregion range from sea level to 2700 m. To simplify the assessment, we pooled the elevation values from the U.S. Geological Survey digital elevation models (DEMs) into 305-m intervals, which resulted in nine elevation bands (or classes). We then calculated the area of each elevation class occupied by designated wilderness and roadless areas. We further simplified the results into classes of low (sea level–915 m), medium (915–1525 m), and high (>1525 m) elevation.

We also examined representation by merging physical and biological habitat types. We combined 19 physical habitat types (Vance-Borland 1999) with 26 natural vegetation types, resulting in 214 distinct natural habitat classes (Strittholt et al. 1999). We calculated degree of representation for each of the 214 natural habitat types according to designated wilderness and mapped roadless areas, and we assigned each to one of four possible representation categories: (1) >50% represented, (2) 25–50% represented, (3) <25% represented, and (4) none represented.

Regional Landscape Connectivity

We examined landscape connectivity of the ecoregion with Arc/Info and FRAGSTATS, a versatile fragmentation software. As with the other analyses, we considered the designated wilderness, and mapped large and small roadless areas separately. FRAGSTATS calculates a large number of fragmentation metrics (or indices) on three distinct levels, patch, class, and landscape (McGarigal & Marks 1995). We focused the assessment on six class-level metrics, including (1) class area, (2) core percent of landscape, (3) mean core area, (4) total core area index, (5) mean nearest-neighbor distance, and (6) edge density (see Appendix for definitions).

We applied the fragmentation metrics in two ways. First, we ran FRAGSTATS on a merged raster file containing designated wilderness and the two roadless-area size categories, treating each as a distinct landscape class. Second, we ran FRAGSTATS on a series of map layers to ascertain the level of connectivity each provided. One map contained only designated wilderness, a second contained designated wilderness and large roadless areas (>2024 ha), and a third contained designated wilderness and all roadless areas (≥405 ha). In each case, we assigned all areas of interest as the same class. By analyzing these three cases, we tested the level of landscape connectivity for the current condition (existing protected areas only), a landscape with the existing protected areas plus all large roadless areas, and existing protected areas plus all large and small roadless areas. We put the edge buffer distance at 90 m for all trials, based on the mean edge-effect distance determined for changes in the physical environment at the edges of Pacific Northwest forests (Chen et al. 1992).

Results

Roads and Roadless Areas

The total road length of all surface types for the Klamath-Siskiyou mapped at 1:100,000 scale was 44,522 km. The road data available at 1:24,000 scale (approximately 75% of the ecoregion), added 42% more road length than the data for coarser roads over the same area. Road-density results calculated from the coarser dataset showed sizeable, somewhat isolated regions of low road density. Designated wilderness was at the center of these larger areas.

Designated wilderness in the ecoregion covered 533,700 ha in several large, scattered patches. We mapped nearly 500 roadless areas of ≥405 ha on public lands of the Klamath-Siskiyou ecoregion area, which totaled 1,186,422 ha, approximately 27% of the ecoregion and more than twice the area of designated wilderness (Fig. 2). We mapped 131 large roadless areas covering 871,815 ha and 367 smaller roadless areas covering 314,607 ha, or 26% of the total roadless area. The USFS lands accounted for most (92%) of the roadless area mapped, followed by Bureau of Land Management (7.6%) and other public lands (e.g., National Park Service, 0.4%).

Special Elements

Roadless areas contained nearly four times more heritage elements than designated wilderness areas; the largest gains occurred in the plant and vertebrate categories (Table 1). In general, roadless areas captured approximately 36% of all known heritage elements. When added to those within designated wilderness, the total number increased to 3816 records, or 43% of all known records (Table 1). Small roadless areas accounted for 931 element records, or 29% of the total roadless-area records, adding substantially more plant, vertebrate, and invertebrate occurrences.

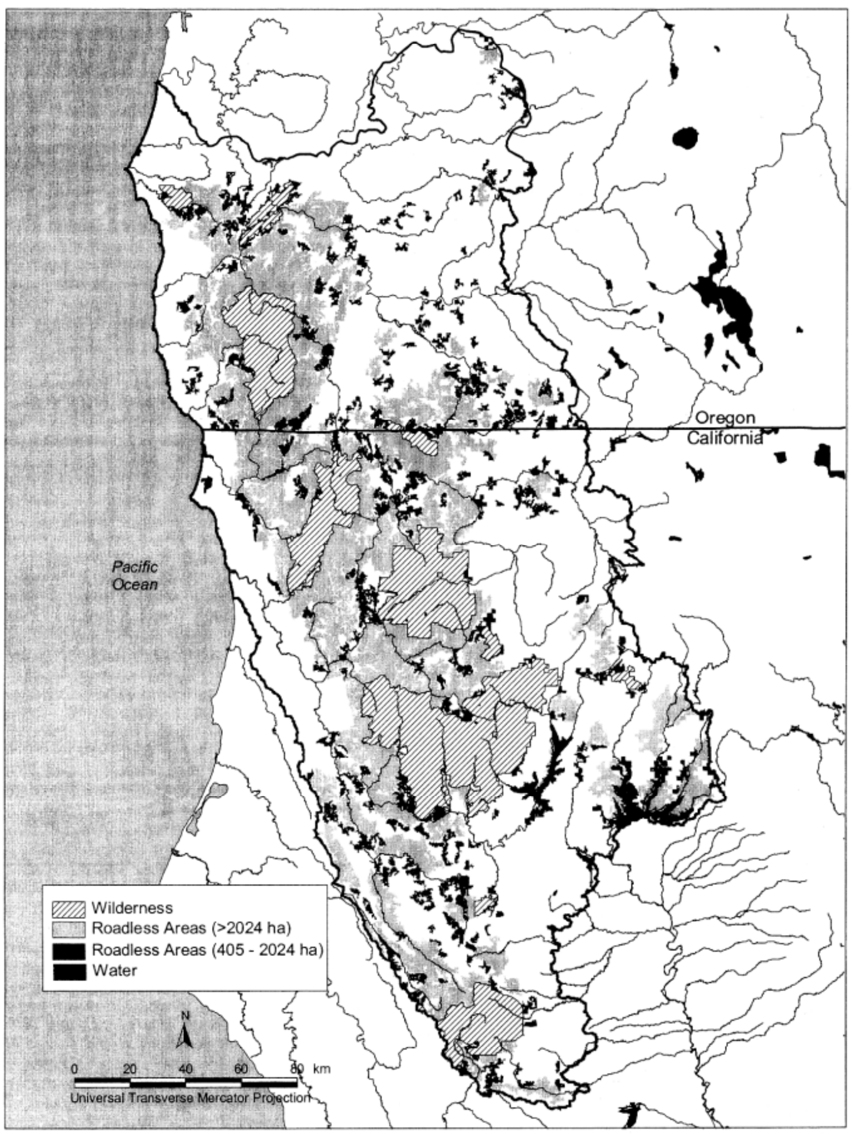

Figure 2. Large (>2024 ha) and small (405–2024 ha) roadless areas within the Klamath-Siskiyou ecoregion.

Table 1. Frequency of heritage-element occurrence records for existing wilderness and mapped roadless areas within the Klamath-Siskiyou ecoregion.

Element category	*No. of records*	*No. in wilderness*	*No. in roadless areas (≥405 ha)*	*No. in roadless areas (405–2,024 ha)*	*Wilderness (%)*	*Roadless areas (%)*	*Combined (%)*
Plants	3837	389	1306	341	10.1	34.0	44.1
Vertebrates	4652	212	1749	562	4.6	37.6	41.2
Invertebrates	132	2	80	26	1.5	60.6	62.1
Community	8	0	6	2	0.0	75.0	75.0
Aquatic	6	0	0	0	0.0	0.0	0.0
Special feature	158	36	36	0	22.8	22.8	45.6
Total	8793	639	3177	931	7.3	36.1	43.4

We found 209,051 ha (36%) of the existing serpentine geology in the Klamath-Siskiyou ecoregion in roadless areas. The contribution from the smaller roadless-area class was 47,090 ha, 22% of the roadless-area total. Designated wilderness areas captured 100,170 ha (17%) of serpentine geology in the ecoregion. Adding all remaining roadless areas to designated wilderness would more than triple the area of serpentine currently protected, bringing the ecoregion protection total to approximately 54%.

Of the 928,356 ha of late-seral forest mapped for the Klamath-Siskiyou, 337,180 ha (36%) were in roadless areas. Small roadless areas accounted for 93,508 ha, or 28% of the roadless-area total. In comparison, designated wilderness areas contained only 16% (149,386 ha) of the remaining late-seral forest. We found no difference in the average density of late-seral forest between wilderness and roadless areas: both contained approximately 28% late-seral forest. The smaller roadless-area class showed a slightly higher percentage of late-seral cover as a group (30%). Of course, individual wilderness areas and roadless areas showed higher and lower percentages of late-seral forest cover. Adding roadless areas into protection brought the total area of late-seral forest protected in the ecoregion to 52% (486,566 ha).

In the Port Orford cedar assessment, roadless areas contained far more subwatershed areas classified as having either low or medium cedar composite scores than did designated wilderness areas (Table 2). These areas require special management, such as road closure and removal, to stop the spread of the root-rot fungus. The area identified as having a high cedar composite score—high cedar density and low fungus infestation—was included largely in designated wilderness or roadless areas and is the remaining stronghold on the public-land portion of the region for protecting cedar and the many ecosystem services this species provides.

Key watersheds covered 1,157,812 ha, or 26% of the Klamath-Siskiyou ecoregion. Over 42% of the key watershed area (491,954 ha) was also roadless. The contribution from smaller roadless areas was 89,506 ha, or 18% of the roadless area total. Of the 333 key watersheds, 190 (57%) were 80% contained (54 were completely contained) within wilderness and roadless areas. Only 13 (4%) contained no roadless area. Key watershed in designated wilderness covered 303,054 ha (26%). The combined key watershed total for both wilderness and roadless areas was 795,008 ha, 68% of all key watershed area.

Representation

Wilderness and roadless areas showed a marked difference in elevation representation (Fig. 3). When compared to wilderness, roadless areas captured much more

Table 2. Scored results of Port Orford cedar density and infestation by root-rot fungus, organized by subwatershed (n = 215) compared with wilderness and roadless areas within the Klamath-Siskiyou ecoregion.

*Cedar composite score categories**	*Total area (ha)*	*Wilderness (ha)*	*Roadless (ha)*	*Wilderness (%)*	*Roadless (%)*	*Combined (%)*
Low (n = 40)	179,430	8,948	79,948	5.0	44.5	49.5
Medium (n = 127)	524,740	88,166	228,352	16.8	43.5	60.3
High (n = 48)	199,389	60,927	113,933	30.5	57.1	87.6
Total (n = 215)	903,559	158,041	422,233	17.5	46.7	64.2

**For details on cedar composite scoring method resulting in categories of low (low cedar density and high infestaion), medium, and high (high cedar density and low infestation), see Strittholt et al. (1999).*

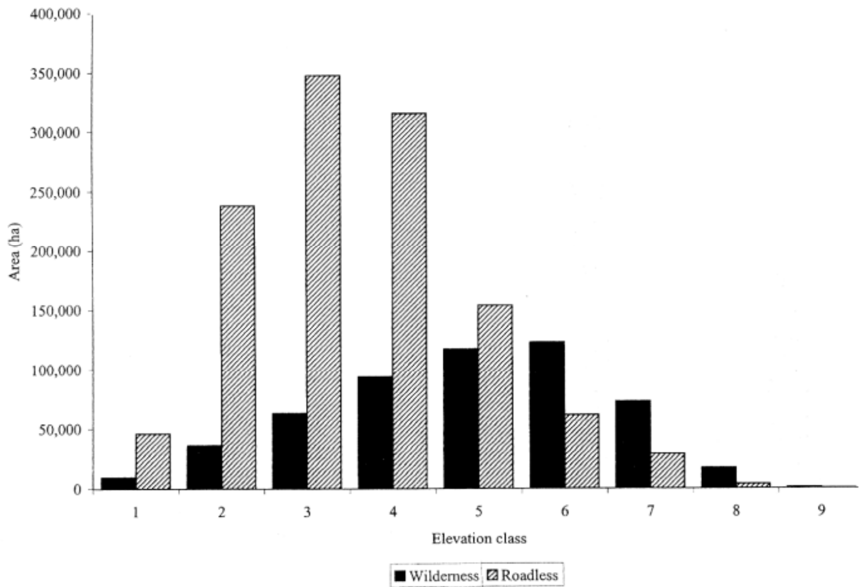

Figure 3. Representation of elevation classes by wilderness and mapped roadless areas for the Klamath-Siskiyou ecoregion. Elevation classes are reported in approximately 305-m intervals from mean sea level to 305 m (elevation class 1).

of the low- and medium-elevation classes (<1525 m) (Table 3). The small roadless areas made up about the same proportion (approximately 26%) of each elevation class, providing additional representation of low and medium elevations. Roadless areas did proportionally better than designated wilderness at representing low and medium elevations. Most of the existing high-elevation sites in the Klamath-Siskiyou were in designated wilderness.

Of the 214 combined physical-biological habitat types described and mapped for the Klamath-Siskiyou, roadless areas contained a wider array of habitat types than designated wilderness. Roadless areas also contained many different habitat types than designated wilderness, with 96 new types (45%) represented at the ≥25% level (Fig. 4). When combined, designated wilderness and roadless areas complimented each other well, with 64% (138/214) of the classes represented at the ≥25% representation level. Although not visible in Fig. 4, smaller roadless areas made important contributions to 148 different habitat types, including 24 not found in any designated wilderness or large roadless areas. In addition, 54 habitat types were highly concentrated in smaller road-

Table 3. Representation of elevation classes by wilderness and mapped roadless areas for the Klamath-Siskiyou ecoregion.

Elevation class[a]	*Total area (ha)*	*Wilderness (ha)*	*Roadless (ha)*	*Combined (ha)*	*Wilderness (%)*	*Roadless (%)*	*Combined (%)*[b]	*Roadless contribution (%)*
Low								
1	351,765	9,500	46,182	55,682	2.7	13.1	15.8	82.9
2	934,857	36,000	238,407	274,407	3.8	25.5	29.3	86.9
3	1,145,382	62,900	347,664	410,564	5.5	30.3	35.8	84.7
subtotal	2,432,004	108,400	632,253	740,653	4.4	26.0	30.4	85.4
Medium								
4	900,862	93,800	315,876	409,676	10.4	35.1	45.5	77.1
5	523,358	117,100	153,840	270,940	22.4	29.4	51.8	56.8
subtotal	1,424,220	210,900	469,716	680,616	14.8	33.0	47.8	69.0
High								
6	283,128	122,900	61,476	184,376	43.4	21.7	65.1	33.3
7	124,405	72,900	28,989	101,889	58.6	23.3	81.9	28.4
8	21,449	17,500	3,949	21,449	81.6	18.4	100.0	18.4
9	1,716	1,100	400	1,500	64.1	23.3	87.4	26.7
subtotal	430,698	214,400	94,814	309,214	49.8	22.0	71.8	30.7
Total	4,286,922	533,700	1,196,783	1,730,483				

[a] *Elevation classes are reported in approximately 305-m intervals from mean sea level, to 305 m (elevation class 1), to the highest elevation band of 2440 m (elevation class 9).*
[b] *Wilderness plus roadless totals.*

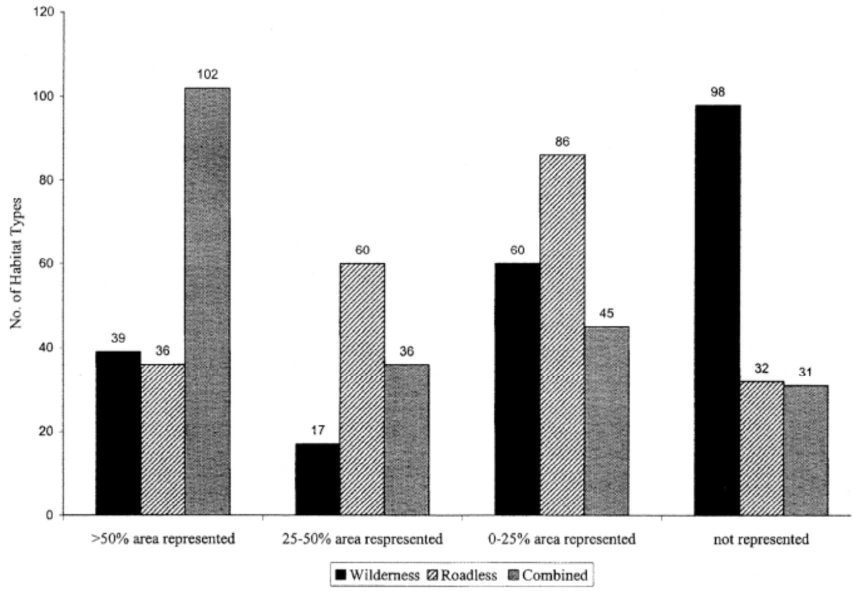

Figure 4. Physical-biological habitat representation (n = 214) for wilderness and mapped roadless areas for the Klamath-Siskiyou ecoregion.

less areas. More than 50% of the total area of these habitat types was located in this roadless-area size class alone.

Regional Landscape Connectivity

Treating each component of a single landscape—wilderness, larger roadless areas, and smaller roadless areas—as a separate class produces a number of interesting but not unexpected results (Table 4). Designated wilderness areas in the Klamath-Siskiyou are large and scattered, centering on the major high-elevation portions of the region. Based on this description, the wilderness areas showed some expected fragmentation results: large values for total core area, mean core area, and mean nearest-neighbor distance. Edge density was also predictably low. Compared to wilderness areas, the larger roadless areas showed a substantial decrease in the mean core-area measurement (26,736 ha) and in the mean nearest-neighbor distance (5356 m). Core percent of landscape and edge density were higher in larger roadless areas than in any other class. The higher edge density signifies that many of the patches, while large, are relatively convoluted. Total core area index was lower than for wilderness, but still high.

Mean core area for the smaller roadless areas was predictably low, but this group still showed a relatively high total core-area index value. Based on a lower edge-density value, smaller roadless areas were less convoluted in shape than the larger size class. This class accounted for nearly 15% of all core area among the three classes examined.

The fragmentation results for each of the three landscape scenarios—wilderness only, wilderness plus large roadless areas, and wilderness plus all roadless areas of ≥405 ha—showed predictable increases in class area and edge density as more land parcels were given theoretical protection status (Table 5). Because only one class type was used for each landscape scenario, the values for core percentage of landscape and total core-area index were identical. Going from the least-protected to the most-protected scenarios, declines were observed for core percentage of landscape; total core-area index, mean core area, and mean nearest neighbor, and increases were observed in class area and edge density. The substantial declines in mean nearest-neighbor values

Table 4. Fragmentation results for the Klamath-Siskiyou ecoregion calculated for a single landscape containing wilderness, small roadless areas (405–2024 ha), and large roadless areas (>2024 ha) treated as separate classes.

Fragmentation index	*Wilderness*	*Roadless areas (>2024 ha)*	*Roadless areas (405-2024 ha)*
Class area (ha)	529,619	872,013	315,297
Core percentage of landscape	30	45	15
Total core area index (%)	97	88	80
Mean core area (ha)	32,178	5,442	665
Mean nearest-neighbor distance (m)	6,459	1,103	864
Edge density (m/ha)	3.88	9.13	5.42

Table 5. Fragmentation results for the Klamath-Siskiyou ecoregion, calculated for three separate landscapes: wilderness only, wilderness + large roadless areas, and wilderness + all roadless areas.

Fragmentation index	*Wilderness only*	*Wilderness + roadless areas (>2024 ha)*	*Widlerness + roadless areas (≥405 ha)*
Class area (ha)	529,619	1,401,632	1,716,929
Core percentage of landscape	97	92	90
Total core area index (%)	97	92	90
Mean core area (ha)	32,178	12,087	3,554
Mean nearest-neighbor distance (m)	6,459	1,370	225
Edge density (m/ha)	3.88	11.50	14.46

provide the best evidence out of these indices that adding roadless areas to the current distribution of protected areas would provide improved regional landscape connectivity. Including the smaller roadless areas added significantly to regional landscape connectivity.

Discussion

The extensive literature on the importance of intact natural habitats makes a compelling case for the potential role of roadless areas as refugia for native biodiversity and as areas crucial to forest integrity and function. Equally impressive is the mounting body of evidence showing the ecological costs of roads. Although some species may benefit from the physical changes and fragmentation caused by roads, their overall and cumulative effect in natural forest landscapes is negative, often seriously so. The suggestion that research on the effects of roads on natural ecosystems is inconclusive (e.g., Heinz Center 1999) is largely unsupported by the literature. Thus, the exclusion of roads from fragmentation assessments presents an incomplete picture of the effects of one of the most predominate anthropogenically induced changes to forested ecosystems in North America. For a variety of reasons, some of which are ecological, roadless areas are once again on the political agenda.

On numerous scientific grounds, our analyses strongly support protecting roadless areas in the Klamath-Siskiyou ecoregion. Roadless areas contained many known locations of species of concern, including rare and endangered species, many more than could be explained by additional land area alone, and the contribution by small roadless areas was noteworthy. That is not to say that roadless areas contained the majority of the known locations of rare species or even all of the known biological hotspots in the ecoregion. For example, there are a few well-known biological hotspots largely within the roaded public and private lands of the Klamath-Siskiyou (Noss et al. 1999; Strittholt et al. 1999). If one conservation goal is to protect most or all of the known rare species, roadless areas contribute only partially to this objective. Additional conservation measures and approaches are needed to achieve this goal in the Klamath-Siskiyou. In addition, our assumption is that rare species located in designated wilderness and roadless areas would be better protected than if they existed under other management practices. This assumption would not always be substantiated by the natural history of a particular species and the individual site conditions under which it lives; a species-specific needs assessment is required to make an accurate determination. We also suspect that many more rare species are present in roadless areas than are presently known because of the remoteness of roadless areas, a lack of organized biological surveys, and a current sampling bias toward areas with road access. The actual importance of roadless areas with regard to this criterion may actually be higher than the current data analysis indicates.

The amount of late-seral forest in roadless areas is considerable, but the percentage is about that found in wilderness areas (both are around 28% late seral). What is important is that these forests are part of relatively large, intact blocks of habitat representing important ecological values. Larger patches of forest that can support a wider range of species, including those requiring large home ranges, are more secure from human-induced effects and are large enough to allow natural processes such as fire to operate without human interference. As with heritage elements, although roadless areas offer some protection to late-seral forests, not all important late-seral stands are included in roadless areas; This includes particular stands of late-seral forest within the Klamath-Siskiyou.

The contribution roadless areas made to Port Orford cedar conservation was particularly significant. The roadless area and designated wilderness components of the best-condition category were high; when combined, they accounted for nearly 88% of the best remaining areas of uninfected cedar. Because root-rot fungus is easily spread by the tires of vehicles moving on roads from infected watersheds to uninfected ones, maintaining large roadless areas is one important strategy for stopping the spread of the disease and therefore protecting this important tree species. At this level of analysis, smaller roadless areas do not seem to be as important for cedar as larger ones; but as in the case of key watersheds, the position of these roadless areas may be as important as the total area they protect.

Our representation analyses suggest that an important role of roadless areas is adding low and medium elevations and many combined physical-biological habitat classes to the reserve network. Other researchers have shown that many of the existing protected areas (especially wilderness areas) are concentrated on higher elevations (Harris 1984; Scott et al. 1993; Strittholt & Frost 1997), and the assumption could be made that most of the remaining roadless areas are much the same. Based on our assessment, this was not the case in the Klamath-Siskiyou. Roadless areas contributed significantly to low and medium elevations. Lower elevations contained most of the region's biological richness (DellaSala et al. 1999), and roadless areas were well represented at these elevations. If full representation of existing habitat types is a regional conservation goal, roadless areas contribute significantly to this objective, but conservation actions on roaded public and private lands need to be taken as well (Strittholt et al. 1999).

The way wilderness and roadless areas complimented one another in representing the various physical-biological classes defined for the Klamath-Siskiyou ecoregion is an important finding for regional conservation. As one would expect from the elevation results, wilderness areas contained large portions of the physical-biological habitat types found at the higher elevations. Wilderness areas are concentrated on most of the forested and non-forested ecosystems at high elevations, including most of the red fir (*Abies magnifica*) and white fir (*A. concolor*) forests and much of the higher Jeffrey pine (*Pinus jeffreyi*), ponderosa pine (*P. ponderosa*), montane-hardwood conifer, and Klamath mixed-conifer forest types. Roadless areas added additional area to some of these habitat types, and—more important—picked up different physical zones for the same plant community types as well as totally new habitat types, including forests of Douglas-fir, montane hardwood, and Sierra mixed conifers growing under various physical zones defined and mapped for the region (Strittholt et al. 1999; Vance-Borland 1999). Smaller roadless areas were particularly important in contributing to 54 different habitat types and were exclusively responsible for capturing 24 habitat types not encountered at all in wilderness or the larger roadless areas.

Our fragmentation results demonstrated the importance of roadless areas to regional connectivity. The larger roadless areas contributed the most, but protecting the smaller roadless areas would contribute to the protection of the overall landscape connectivity in this ecoregion, and it would do so while maintaining a high level of core interior habitat. In the Klamath-Siskiyou, wilderness and roadless-area patches are positioned in such a way that, when they are protected, once-isolated habitat islands become linked throughout the heart of the ecoregion. In the future, the issue of inter-regional linkage will have to be addressed, and roadless areas could provide the nuclei around which future corridors targeting individual species or ecosystem processes could be designed.

Small roadless areas in the Klamath-Siskiyou did not equally address all conservation issues examined, but they address many of them significantly, especially heritage elements, late-seral forests, elevation representation, habitat-type representation, and overall landscape connectivity. Based on the results, small roadless areas warrant inclusion in future assessment and planning of roadless areas. If small roadless areas are important in the Klamath-Siskiyou, where a high proportion of larger roadless areas exist, then they are likely to be even more important in regions where the majority of roadless areas are small, such as the southern Appalachians.

We considered the importance of roadless areas as a whole to the conservation of regional biodiversity. The next logical step in the roadless-areas assessment for the Klamath-Siskiyou and elsewhere will be to evaluate each roadless area individually to measure its relative ecological attributes. To some extent, this has already been done for the Klamath-Siskiyou (Strittholt et al. 1999), and an expansion of this assessment is currently underway that will include the national forests of western Washington, Oregon, and northern California. The better the ecological attributes of each roadless area are quantified, the better regional conservation can be planned. That is not to say that the ecological criteria we examined (or ecological criteria in general) are the only values roadless areas possess, and ecological reasons are not the only criteria for evaluation. Other values humans place on wild places, such as recreational, aesthetic, educational, economic, and scientific values, are all important reasons for establishing greater environmental protection of roadless areas.

The opportunity to protect some of the largest remaining forest habitat in the United States may be a fleeting one. Time is of the essence. Based on agency inventories in 1979, the USFS had about 32 million ha (42%) of roadless areas. Since the 1979 RARE II process, the estimated rate of loss of roadless areas has been about 400,000 ha each year nationally (Foreman & Wolke 1992). Under the new roadless-area rule making (USFS 2000), an average of 95 km of new roads each year are planned for roadless country through 2004.

Providing protection of large natural areas is an indispensable component of an overall conservation strategy (Noss & Cooperrider 1994), but a protected-area network cannot be compoed primarily of poor-quality conservation lands (Hall 1988). Protected-area networks must be carefully designed to maintain regional biodiversity, ecosystem processes, and overall ecosystem integrity. The existing roadless areas on federal lands in the United States are the remaining pieces of the natural landscape that once covered the nation. Most biologically productive lands are already developed, and opportunities are limited to design effective protected-area networks without the necessity for substantial restoration. Roadless areas provide the remaining building blocks toward this end, so it is important to understand the contribution these areas make to an overall conservation strategy. Without this information, there is little hope for effective conservation or informed land-use planning.

The roadless areas in the Klamath-Siskiyou, if protected, would provide a wide range of ecological attributes important in this ecoregion, which is why roadless areas became the cornerstone of a recent reserve design for the ecoregion (Noss et al. 1999; Strittholt et al. 1999). Our results underscore the significance of roadless areas for regional conservation, but roadless areas alone cannot provide all the ecological elements needed to maintain regional biodiversity. Thus, it will be important to manage roadless areas responsibly and restore them where necessary (DellaSala et al. 1999; Strittholt et al. 1999). We do not know if roadless areas in other regions will yield similar results, particularly since, according to the USFS (2000), 60% of roadless areas on national forests are at elevations above 1524 m. But we believe that our methodologies may be applied elsewhere, particularly as the inventory of small roadless areas takes on greater significance in federal-lands policy.

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Appendix

Descriptions of the six fragmentation metrics used in the Klamath-Siskiyou landscape connectivity analysis.*

Metric	*Description*
Class area (ha)	sum of the areas of all patches of the corresponding patch type (range: ≥0, without limit)
Core percentage of landscape	sum of the core areas of each patch of the corresponding patch type, divided by total landscape area (range: ≥0, ≤ 100)
Mean core area (ha)	sum of the core areas of each patch of the corresponding patch type, divided by the number of patches of the same type (range: ≥0, without limit)
Total core-area index (%)	sum of the core areas of each patch of the corresponding patch type, divided by the sum of the areas of each patch of the same type, multiplied by 100 (range: ≥ 0, ≤ 100)
Mean nearest-neighbor distance (m)	sum of the distance to the nearest neighboring type of the same type, based on nearest edge-to-edge distance for each patch of the corresponding patch type, divided by the number of patches of the same type (range: ≥0, without limit)
Edge density (m/ha)	sum of the lengths of all edge segments involving the corresponding patch type, divided by the total landscape area (range: ≥0, without limit)

* *Adapted from McGarigal and Marks (1995).*

Frontiers in Ecology and the Environment

The forgotten stage of forest succession: early-successional ecosystems on forest sites

Mark E Swanson, Jerry F Franklin, Robert L Beschta, Charles M Crisafulli, Dominick A DellaSala, Richard L Hutto, David B Lindenmayer, and Frederick J Swanson

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The forgotten stage of forest succession: early-successional ecosystems on forest sites

Mark E Swanson[1*], Jerry F Franklin[2], Robert L Beschta[3], Charles M Crisafulli[4], Dominick A DellaSala[5], Richard L Hutto[6], David B Lindenmayer[7], and Frederick J Swanson[8]

Early-successional forest ecosystems that develop after stand-replacing or partial disturbances are diverse in species, processes, and structure. Post-disturbance ecosystems are also often rich in biological legacies, including surviving organisms and organically derived structures, such as woody debris. These legacies and post-disturbance plant communities provide resources that attract and sustain high species diversity, including numerous early-successional obligates, such as certain woodpeckers and arthropods. Early succession is the only period when tree canopies do not dominate the forest site, and so this stage can be characterized by high productivity of plant species (including herbs and shrubs), complex food webs, large nutrient fluxes, and high structural and spatial complexity. Different disturbances contrast markedly in terms of biological legacies, and this will influence the resultant physical and biological conditions, thus affecting successional pathways. Management activities, such as post-disturbance logging and dense tree planting, can reduce the richness within and the duration of early-successional ecosystems. Where maintenance of biodiversity is an objective, the importance and value of these natural early-successional ecosystems are underappreciated.

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Severe natural disturbances – such as wildfires, windstorms, and insect epidemics – are characteristic of many forest ecosystems and can produce a "stand-replacement" event, by killing all or most of the dominant trees therein (Figure 1). Typically, limited biomass is actually consumed or removed in such events, but many trees and other organisms experience mortality, leaving behind important biological legacies (structures inherited from the pre-disturbance ecosystem; Franklin *et al*. 2000), including standing dead trees and downed boles (tree trunks; Franklin *et al*. 2000). Such legacies provide diverse physical/biological properties and suitable microclimatic conditions for many species. Thereafter, species-diverse plant communities develop because substantial amounts of previously limited resources (light, moisture, and nutrients) become available. These emerging plant communities create additional habitat complexity and provide various energetic resources for terrestrial and aquatic organisms.

In a nutshell:

- Naturally occurring, early-successional ecosystems on forest sites have distinctive characteristics, including high species diversity, as well as complex food webs and ecosystem processes
- This high species diversity is made up of survivors, opportunists, and habitat specialists that require the distinctive conditions present there
- Organic structures, such as live and dead trees, create habitat for surviving and colonizing organisms on many types of recently disturbed sites
- Traditional forestry activities (eg clearcutting or post-disturbance logging) reduce the species richness and key ecological processes associated with early-successional ecosystems; other activities, such as tree planting, can limit the duration (eg by plantation establishment) of this important successional stage

The ecological importance of early-successional forest ecosystems (ESFEs) has received little attention, except as a transitional phase, before resumption of tree dominance. In forestry, this period is often called the "cohort re-establishment" or "stand initiation" stage, with attention obviously focused on tree regeneration and the re-establishment of closed forest canopies (Franklin *et al*. 2002). Ecological studies have focused primarily on plant-community development and the needs of selected animal (mostly game) species, and not on the diverse ecological roles of ESFEs.

Here, we highlight important features of ESFEs, including their role in sustaining ecosystem processes and biodiversity, so that they may be appropriately considered by resource managers and scientists, and included within management/research programs dedicated to maintaining these functions, particularly at larger spatio-temporal scales. Most published examples focus on sites in western North America, but ESFEs are important elsewhere (Angelstam 1998; DeGraaf *et al*. 2003). We also discuss how traditional forestry practices, such as clearcutting, tree planting, and post-disturbance logging, can affect early-successional communities.

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Figure 1. *Stand-replacement disturbance events in forests create large areas free of tree dominance and rich in physical and biological resources, including legacies of the pre-disturbance ecosystem.*

(Franklin *et al.* 2002). Consequently, the duration of ESFEs varies inversely with rapidity of tree regeneration and growth, which, in turn, depend on such variables as tree propagule availability, conditions affecting seedling or sprout establishment, and site productivity. ESFE longevity after natural disturbances is therefore highly variable.

Development of a closed forest canopy may require a century or more in areas with limited seed sources, harsh environmental conditions, severe shrub competition (in some instances), or combinations thereof (Hemstrom and Franklin 1982). For example, tree canopy closure after wildfire in the Douglas fir region of western North America often requires several decades (Poage *et al.* 2009), but can occur much more rapidly when canopy seedbanks are abundant (eg Larson and Franklin 2005). Closed forest canopies may develop quickly in forests dominated by trees with strong sprouting ability (eg many angiosperms) or when windstorms "release" understories of shade-tolerant tree seedling banks by removing all or most of the overstory (Foster *et al.* 1997).

■ Early-successional ecosystems on forest sites

Initial conditions after stand-replacing forest disturbances vary generically, depending on the type of disturbance; this includes the types of physical and biological legacies available. For example, aboveground vegetation may be limited immediately after the disturbance, as in the case of severe wildfires or volcanic eruptions. Conversely, intact understory communities may persist where forests have been blown down by severe windstorms. Spatial heterogeneity in conditions is characteristic, given that disturbances vary greatly in the amount of damage they cause (Turner *et al.* 1998). For instance, severe wildfires frequently include substantial areas of unburned as well as low to medium levels of mortality, creating variability in shade, litterfall, soil moisture, seed distribution, and other factors.

We define ESFEs as those ecosystems that occupy potentially forested sites in time and space between a stand-replacement disturbance and re-establishment of a closed forest canopy. These ecosystems undergo compositional and structural changes (succession) during their occupancy of a site. Changes begin immediately post-disturbance, as a result of the activities of surviving organisms (eg plants, animals, and fungi), including plant growth and seed production. Developmental processes are enriched by colonization of flora and fauna from outside the disturbed area. Successional change is often characterized by progressive dominance of annual and perennial herbs, shrubs, and trees, although all of these species are typically represented throughout the entire sequence of forest stand development (or sere; Halpern 1988).

The ESFE developmental stage ends with re-establishment of tree cover that is sufficiently dense to suppress and often eliminate many smaller shade-intolerant plants

■ Attributes of early-successional ecosystems

After severe disturbances, forest sites are characterized by open, non-tree-dominated environments, but have high levels of structural complexity and spatial heterogeneity and retain legacy materials.

Environmental conditions

Removal of the overstory forest canopy during disturbances dramatically alters the site's microclimate, including light regimes. These changes lead to increased exposure to sunlight, more extreme temperatures (ground and air), higher wind velocities, and lower levels of relative humidity and moisture in litter and surface soil. Shifts in these environmental metrics favor some species, while creating suboptimal or intolerable conditions for others. For example, post-disturbance plant community composition, cover, and physiognomy are altered as shade-tolerant understory herbs are largely displaced by shade-intolerant and drought-tolerant species. New substrates deposited by floods or volcanic eruptions may lack nutrients, provide additional water-holding capacity, or have high albedo, all of which favor shifts in plant communities.

Survivors

Organisms (in a variety of forms) that survive severe disturbances are extremely important for repopulating and

restoring ecosystem functions in the post-disturbance landscape. Even in severely disturbed areas, organisms may survive as individuals (mature or immature) or as reproductive structures (eg spores, seeds, rootstocks, and eggs), which become in situ propagule sources. For example, after the 1980 volcanic eruption of Mount St Helens (Washington State), most pre-eruption flora and many fauna (especially aquatic and burrowing terrestrial species) survived within the blast zone through several different mechanisms (Dale *et al*. 2005).

Surviving organisms are also often vital for the prompt re-establishment of important ecosystem functions, such as conservation of nutrients and stabilization of substrates. For instance, the important role of resprouting vegetation in curbing massive losses of nitrogen was demonstrated by experimentally clearcutting and applying herbicides in a watershed at Hubbard Brook Experimental Forest (Bormann and Likens 1979).

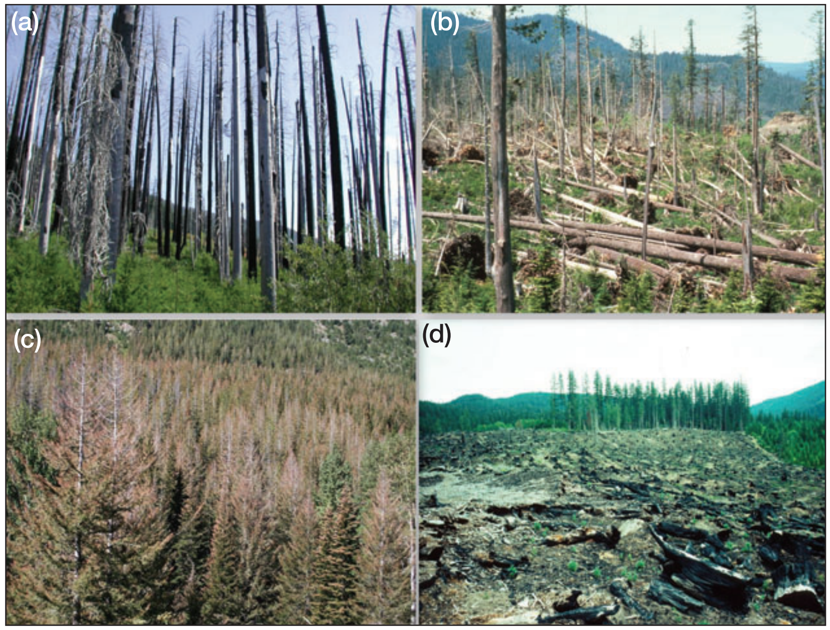

Figure 2. *Different types of disturbances produce different types of biological legacies, including living organisms and structures: (a) standing dead trees (snags) are dominant structural legacies after severe wildfires; (b) downed tree trunks and nearly intact understory communities are characteristic legacies after major windstorms; (c) standing dead trees are also dominant structural legacies after heavy insect infestations; and (d) clearcuts typically eliminate most aboveground structural legacies. Values for each metric are shown in Table 1 and are described in detail in the text.*

Structural complexity

The structural complexity of ESFEs depends initially on legacies, the general nature of which varies with the type of disturbance (Table 1; Figure 2); for example, snags and shrubs originating from belowground perennating (ie resprouting) parts or seeds are dominant legacies after wildfires, whereas downed boles and largely intact understories are typical post-disturbance characteristics of windstorms.

Woody legacies, such as snags and downed boles, play numerous roles in structuring and facilitating the development of the recovering ecosystem – providing habitat for survivors and colonists, moderating the physical environment, enriching aquatic systems in the disturbed area (Jones and Daniels 2008), and providing long-term sources of energy and nutrients (Harmon *et al*. 1986). Although subject to decomposition, these legacies can persist for many decades and sometimes even centuries.

Table 1. Different types of intense disturbances generate different types of biological legacies

	Disturbance				
Biological legacies	*Wildfire*	*Wind*	*Insect*	*Volcano*	*Clearcut*
Live trees	Infrequent	Variable	Variable (depends on stand composition)	Infrequent – confined to margins	Infrequent or absent
Snags	Abundant	Variable	Abundant	Abundant (spatially variable)	Infrequent or absent
Downed woody debris	Variable, but typically abundant	Abundant	Variable, but eventually abundant	Abundant (spatially variable)	Infrequent
Undisturbed understory	Infrequent	Abundant	Abundant	Infrequent – confined to disturbance margins	Infrequent
Spatial heterogeneity of recovery	High	Variable	High	High	Variable – usually low
Time in early-successional condition	Variable	Variable	Long	Variable – usually long	Variable – usually short

Figure 3. *Plant communities with well-developed shrub and perennial herb species are characteristic of early-successional communities on forest sites and provide diverse food resources. Twenty-five years after the Mount St Helens eruption in 1980, this community, which was within the blast zone, includes well-developed shrubs (eg* Sorbus *and* Vaccinium *spp), trees, and perennial herbs (eg* Epilobium angustifolium*).*

Structural complexity is further enhanced by the establishment and development of a variety of plant species, which often include perennial herbs and shrubs characteristic of open environments, as well as individual trees (Figure 3). The diversity of plant morphologies (maximum height, crown width, etc) increases structural richness, so that this associated flora contributes to both horizontal and vertical heterogeneity.

Spatial heterogeneity

Spatial heterogeneity is evident in early-successional ecosystems and has multiple causes: (1) natural variability in the geophysical template (topography and lithology) of the affected landscape; (2) variability in conditions in the pre-disturbance forest ecosystem; (3) variability in the intensity of the disturbance event; and (4) variability in rates and patterns of subsequent developmental processes in the ESFE. The first two sources relate to existing geophysical and biological patterns within the disturbed area. Land formations and patterns of geomorphic processes are certainly key geophysical elements (Swanson *et al.* 1988). The presence of surface water, such as streams and ponds, can be particularly influential in facilitating survival and re-establishment of biota.

Natural disturbances create heterogeneous environments at multiple spatial scales (Heinselmann 1973), because disturbances do not cause damage uniformly. Disturbances such as wildfires and windstorms are variable in intensity (eg "spotting", or initiation of new flame fronts by wind-thrown firebrands, during fire events). Alternatively, geographic variation in environmental conditions and topography (Swanson *et al.* 1988) influences the intensity of the disturbance and results in heterogeneity at multiple scales. Variability in the structure and composition of the pre-disturbance forest also creates spatial and temporal variability (Wardell-Johnson and Horowitz 1996). Some of these patterns may be transient, such as residual snowbanks protecting tree regeneration after the aforementioned Mount St Helens eruption (Dale *et al.* 2005).

Post-disturbance developmental processes also lead to spatial heterogeneity. For example, varying distances to sources of tree seed result in different rates and densities of tree re-establishment (Turner *et al.* 1998). Structural legacies can greatly influence the rates at which wind- or waterborne organic (including propagules) and inorganic materials are deposited. Finally, animal activity can strongly influence patterns of revegetation, as illustrated by the multiple effects that gophers (*Thomomys* spp) can have on post-disturbance landscapes (Crisafulli *et al.* 2005b) or the way ungulate browsing may impede tree regeneration (Hessl and Graumlich 2002).

Biological diversity

ESFEs in temperate forest seres show great diversity in the abundance of plant and animal species (Fontaine *et al.* 2009). Species composition may consist of a mix of forest survivors, opportunists, or ruderals (plants that grow on disturbed or poor-quality lands), and habitat specialists that co-exist in the resource-rich ESFE environment (Figure 3). Most forest understory flora can survive disturbances as established plants, perennating rootstocks, or seeds. In one study, in western North America, over 95% of understory species survived the combined disturbance of logging and burning of an old-growth Douglas-fir–western hemlock stand (Halpern 1988). Some important early-successional species (eg *Rubus* spp [blackberry; raspberry], *Ribes* spp [gooseberry], and *Ceanothus* spp [buckbrush]) may persist as long-lived seedbanks.

Opportunistic herbaceous species are often conspicuous dominants early in the development of ESFEs (Figure 4). Many of these weedy species (particularly annuals) decline quickly, although other opportunists will persist as part of the plant community until overtopped by slower growing shrubs or trees. Consequently, diverse plant communities of herbs, shrubs, and young trees emerge in ESFEs; this, combined with the structural legacies from the pre-disturbance ecosystem, often results in high levels of structural richness (Figure 3).

Many animals, including habitat specialists and species typically absent from the eventual tree-dominated com-

munities, thrive under the conditions found in ESFEs. For some species, this is the only successional stage that can provide suitable foraging or nesting habitat. As an example, many butterflies and moths (Lepidoptera) found in forested regions depend on the high diversity and quality of plant forage in ESFEs (eg Miller and Hammond 2007), whereas jewel beetles (Coleoptera: Buprestideae) depend on abundant coarse woody debris. Also, a number of ground-dwelling beetle species occur as habitat specialists in early-successional communities (Heyborne *et al.* 2003).

Figure 4. *Early-successional communities are often dominated by annual herbaceous species for the first few years after disturbance; these are quickly displaced by perennial herbaceous species and shrubs.*

Many vertebrates also respond positively to ESFEs, which may provide the only suitable habitat at a regional scale for some species. Ectothermic animals, such as reptiles (eg Rittenhouse *et al.* 2007), generally respond favorably to sunnier and drier conditions, colonizing early-successional habitat or increasing in abundance if present as survivors. Many amphibians also thrive in ESFEs, provided resources such as water bodies and key structures (eg logs) are available. The diversity and abundance of amphibians in the area affected by the 1980 Mount St Helens eruption is illustrative (Crisafulli *et al.* 2005a); eleven of 15 amphibian species survived the event, and some (eg western toad, *Bufo boreas*) have since had exceptional breeding success.

The broad array of birds using the abundant and varied food sources (eg fruits, nectar, herbivorous insects) and nesting habitat in ESFEs includes many raptors and neotropical migrants, often making bird diversity highest during the ESFE stage of succession (Klaus *et al.* in press). Some species are habitat specialists that directly utilize the legacy of recently killed trees; for instance, black-backed woodpeckers (*Picoides arcticus*) are almost completely restricted to early post-fire conditions (Hutto 2008). Mountain bluebirds (*Sialia currucoides*) and several other woodpecker species also favor structurally rich, early-successional habitats (Figure 5). Observed population declines of many avian species in eastern North America – which, in some cases, have proceeded to a point of conservation concern – are linked to conversion of early-successional habitat to closed forest (Litvaitis 1993).

Small mammal communities in ESFEs typically show high levels of diversity as well, including some obvious habitat specialists. The eastern chestnut mouse (*Pseudomys gracilicaudatus*), for example, inhabits early-successional environments in coastal eastern Australia for 2–5 years after a wildfire, and then declines dramatically until these environments are burned again (Fox 1990). Populations of mesopredators (medium-sized predators, such as raccoons [*Procyon lotor*] and fox species) benefit from the abundance of small vertebrate prey items characteristic of ESFEs. Likewise, some species of large mammals are well known to favor ESFEs (Nyberg and Janz 1990). Utilizing the diverse and luxuriant forage characteristically present in these ecosystems, ungulates, such as members of the Cervidae, in turn serve to benefit large predators (eg wolves [*Canis lupus*]) as well as scavengers, making ESFEs important elements within those species' typically extensive home ranges. Omnivores, such as bears (*Ursus* spp), also rely on the diversity of food sources often present in ESFEs.

■ Food web diversity

ESFEs are exceptional in the diversity and complexity of food webs they support. Simply stated, a diverse plant community produces many food sources. Food resources for herbivores (grasses, shrubs, forbs) – as well as nectar, seeds, and shrub-borne fruit (eg produced by *Rubus* and *Vaccinium* spp [huckleberry]) – can reach high levels before site dominance by trees. In the temperate Northern Hemisphere, biologically important berry production is maximized in slowly reforesting ESFEs. Resource production in early-successional patches may even augment the richness of adjacent undisturbed forests, as in the case of fluxes of key prey species (Sakai and Noon 1997).

Aquatic biologists have, perhaps, best appreciated the greater complexity of food chains in early-successional versus closed forest environments (Bisson *et al.* 2003). In established forest stands, trees strongly dominate the physical and biological conditions in nearby small streams by controlling light and temperature, stabilizing channels, providing woody debris, and, importantly, offering allochthonous inputs (organic matter originating outside the aquatic ecosystem) – the primary energy and nutrient source for such ecosystems (Vannote *et al.* 1980).

Stand-replacement disturbances remove forest constraints on conditions and processes, and shift streams to an early-

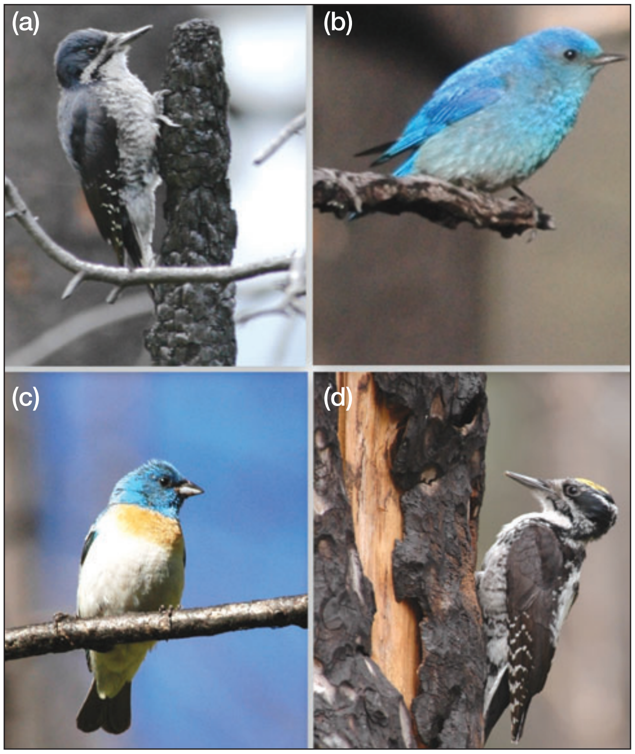

Figure 5. *Bird diversity is typically high in early-successional communities on forest sites and includes many habitat specialists: (a) black-backed woodpeckers (*Picoides arcticus*) are almost entirely restricted to early post-fire habitat; (b) mountain bluebirds (*Sialia currucoides*) favor early-successional ecosystems; (c) lazuli buntings (*Passerina amoena*) and (d) three-toed woodpeckers (*Picoides tridactylus*) have similar requirements.*

successional context (Minshall 2003; Figure 6). This greatly diversifies the types and timing of allochthonous inputs, as well as increases primary productivity. Allochthonous inputs are shifted from primarily tree-derived litter (coniferous-based in many systems) to material from a range of flowering herbs, shrubs, and trees, as well as from conifers. Consequently, litter inputs are highly variable in quality (eg decomposability) and delivery time, as compared with litterfall contributed primarily by evergreen conifer species. Also, inputs to post-disturbance streams often include material with a high nitrogen content, such as litter from the early-successional genera *Alnus* and *Ceanothus* (Hibbs *et al.* 1994).

Greater algal production may increase the diversity and abundance of aquatic invertebrate populations, which, in turn, become prey for fish and other organisms. However, increases in sediment production associated with disturbances can negate some benefits to aquatic processes and organisms (Gregory *et al.* 1987).

■ Processes in ESFEs

Ecosystem processes in ESFEs can be more diverse than those in closed forest systems, where the primary productivity of trees is dominant and organic matter is processed primarily through detrital food webs. Development of more diverse, and perhaps more "balanced", trophic pathways is possible when a disturbance opens a previously closed forest canopy. The contrast is probably greatest in forests dominated by a single tree type, such as evergreen conifers, as opposed to more diverse forests, such as mixed evergreen associations.

Recharging nutrient pools

ESFEs provide major opportunities for recharge of nutrient pools, such as additions to the nitrogen pool by leguminous (eg *Lupinus*) and some non-leguminous early-successional (eg *Alnus* and *Ceanothus*) plant species. These genera are commonly absent from late-successional forests, but are well represented in ESFEs. Nitrogenous additions from these sources are particularly important where the disturbance – eg a wildfire – has volatilized a substantial amount of the existing nitrogen pool.

Mineralization rates of organic material are typically accelerated (sometimes profoundly) after disturbances, as a result of warmer growing season temperatures. Diversified litter inputs in ESFEs, including a greater proportion of easily decomposed litter from herbs and deciduous shrubs, also result in more rapid mineralization. Finally, successional changes in the fungal and microbial communities can also hasten decomposition processes. As noted, these changes will be most profound in forest ecosystems dominated by a single species, including evergreen conifers or hard-leaved, evergreen hardwoods (such as the ash-type eucalypt forests of southeastern Australia).

In aquatic ecosystems that experience fire in adjacent forests, greater post-disturbance light and nutrient availability enhance primary productivity within the water body, causing shifts in food webs from the level of primary producers up through high-level consumers, such as fish (Spencer *et al.* 2003).

Modifying hydrologic and geomorphic regimes

Hydrologic regimes associated with ESFEs contrast greatly with those characterizing closed forest cover. For example, transpiration and interception are dramatically reduced and recover only gradually as forest canopies redevelop. Increases in normally low summer flows and annual water yields may occur immediately after a disturbance, as compared with levels in the dense young forests that may subsequently develop (Jones and Post 2004). The opposite may be true in systems where condensation of cloud or fog on tree crowns is an important component of the hydrologic cycle. ESFEs may also contribute to increased discharge peak runoff flows in hydrologic events of smaller magnitude (Harr 1986), but appear to have little effect on the magnitude of peak flows during large runoff events (Grant *et al.* 2008). From an ecological perspective, this may have a positive outcome, however, because floods restructure and rejuvenate many riparian communities (Gregory *et al.* 1991).

Rates and patterns of geomorphic processes, such as erosion and nutrient leaching losses, are also different between ESFEs and later successional stages. Tree death results in a loss of root strength that is critical for stabilizing soils and deeper rock layers on mountain slopes (Perry *et al.* 2008). Erosion and landslides may occur at higher rates in ESFEs, contributing to the variability of sediment budgets in watersheds (Reeves *et al.* 1995) and creating long-lasting substrates for ruderals. While enhancing erosion processes, ESFEs also provide materials and processes that counteract this effect, such as woody debris, which retain sediments and organic materials, and surviving vegetation, which stabilizes slopes and nutrient stores (eg Bormann and Likens 1979).


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Figure 6. *Streams within early-successional forest ecosystems contrast with forest-dominated reaches in many ecosystem attributes, including physical parameters (temperature and insolation), structure, plant and animal composition, and ecosystem processes, such as primary productivity.*

Land management implications

Incorporating ESFE attributes into forest policy and management is highly desirable, given the numerous advantages provided by these ecosystems. Many species and ecological processes are strongly favored by conditions that develop after stand-replacement disturbances. Rapid, artificially accelerated "recovery" of disturbed forest areas (eg via dense planting) to closed forest conditions has serious implications for many species. Clearly the term "recovery" has a different meaning for such early-successional specialists or obligates.

To fulfill their full ecological potential, ESFEs require their full complement of biological legacies (eg dead trees and logs) and sufficient time for early-successional vegetation to mature. Where land managers are interested in conservation of the biota and maintenance of ecological processes associated with such communities, forest policy and practices need to support the maintenance of structurally rich ESFEs in managed landscapes. Natural disturbance events will provide major opportunities for these ecosystems, and managers can build on those opportunities by avoiding actions that (1) eliminate biological legacies, (2) shorten the duration of the ESFEs, and (3) interfere with stand-development processes. Such activities include intensive post-disturbance logging, aggressive reforestation, and elimination of native plants with herbicides.

In particular, post-disturbance logging removes key structural legacies, and damages recolonizing vegetation, soils, and aquatic elements of disturbed areas (Foster and Orwig 2006; Lindenmayer *et al.* 2008). Where socioeconomic considerations necessitate post-disturbance logging, variable retention harvesting (retention of snags, logs, live trees, and other structures through harvest) can maintain structural complexity in logged areas (Eklund *et al.* 2009).

Prompt, dense reforestation can have negative consequences for biodiversity and processes associated with ESFEs, by dramatically shortening their duration. Such efforts reduce spatial and compositional variability characteristic of natural tree-regeneration processes, promote structural uniformity, and initiate intense competitive processes that eliminate elements of biodiversity that might otherwise persist. Artificial reforestation can also reduce genetic diversity by favoring dominance by fewer tree species/genotypes, and may make the system more prone to subsequent, high-severity disturbances (Thompson *et al.* 2007). The elimination of shrubs and broad-leaved trees through herbicide application can alter synergistic relationships, such as the belowground mycorrhizal processes provided by certain shrub species (eg *Arctostaphylos* spp).

Naturally regenerated ESFEs are likely to be better adapted to the present-day climate and may be more adaptable to future climate change. The diverse genotypes in naturally regenerated ESFEs are likely to provide greater resilience to environmental stresses than nursery-grown, planted trees of the same species. Given that climate change is also resulting in altered behavior of pests and pathogens (Dale *et al.* 2001), encouraging greater tree species diversity may also increase ecosystem resilience.

Clearcutting has been proposed as a technique to create ESFEs, but this can provide only highly abridged and simplified ESFE conditions. First, traditional clearcuts leave few biological legacies (eg Lindenmayer and McCarthy 2002), limiting habitat and biodiversity potential. Second, clearcuts are often quickly and densely reforested, and often involve the use of herbicides to limit competition with desired tree species. Clearcuts can provide some early-successional functionality (eg serving as nurseries or post-breeding habitat for many bird species in the southern US; Faaborg 2002), but this service is often truncated by prompt reforestation.

Management plans should provide for the maintenance of areas of naturally developing ESFEs as part of a diverse landscape. This should be in reasonable proportion to *historical* occurrences of different successional stages, as based on region-specific historical ecology. Major disturbance events provide managers with opportunities to incorporate a greater diversity of species and processes in forest landscapes and to enhance landscape heterogeneity. Some aspects of ESFEs can be incorporated into areas managed for production forestry as well, such as through variable retention harvest methods, the incorporation of natural tree regeneration, and extending the duration of herb/shrub communities in some portions of a stand by deliberately maintaining low tree stocking levels.

Finally, we suggest that adjustments in language are needed. Ecologists and managers often refer to "recovery" when discussing post-disturbance ecosystems, inferring that early seral conditions are undesirable and need to be restored to closed canopy conditions as quickly as possible. Emphasizing recovery as the management goal fails to acknowledge the essential ecological roles played by early-successional ecosystems on forest sites. It should also be considered that climate change and other factors may not permit "recovery" to pre-disturbance conditions.

Conclusions

Twentieth-century forest management objectives were centered on wood production and, later, on conservation and development of late-successional forests. Rapid regeneration of dense timber stands was frequently seen as a way to address both of these divergent objectives. Recognizing the ecological value of early-successional ecosystems on forest sites extends the ecological concerns associated with old growth to another "rich" period in a forest sere. This represents an important development in the evolution of holistic management of forest ecosystems, whereby large landscapes are managed for diverse seral stages.

ESFEs provide a distinctive mix of physical, chemical, and biological conditions, are diverse in species and processes, and are poorly represented and undervalued in traditional forest management. Forest policy and practice must give serious attention to sustaining substantial areas of ESFEs and their biological legacies. Similarly, scientists need to initiate research on the structure, composition, and function of ESFEs in different regions and under different disturbance regimes, as well as on the historical extent of these systems, to serve as a reference for conservation planning.

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Comparing selected fire regime condition class (FRCC) and LANDFIRE vegetation model results with tree-ring data

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Abstract. Fire Regime Condition Class (FRCC) has been developed as a nationally consistent interagency method in the US to assess degree of departure between historical and current fire regimes and vegetation structural conditions across differing vegetation types. Historical and existing vegetation map data also are being developed for the nationwide LANDFIRE project to aid in FRCC assessments. Here, we compare selected FRCC and LANDFIRE vegetation characteristics derived from simulation modeling with similar characteristics reconstructed from tree-ring data collected from 11 forested sites in Utah. Reconstructed reference conditions based on trees present in 1880 compared with reference conditions modeled by the Vegetation Dynamics Development Tool for individual Biophysical Settings (BpS) used in FRCC and LANDFIRE assessments showed significance relationships for ponderosa pine, aspen, and mixed-conifer BpS but not for spruce–fir, piñon–juniper, or lodgepole pine BpS. LANDFIRE map data were found to be ~58% accurate for BpS and ~60% accurate for existing vegetation types. Results suggest that limited sampling of age-to-size relationships by different species may be needed to help refine reference condition definitions used in FRCC assessments, and that more empirical data are needed to better parameterize FRCC vegetation models in especially low-frequency fire types.

Additional keywords: reference conditions, successional classes, Vegetation Dynamics Development Tool (VDDT).

Introduction

Altered fire regimes and associated changes in vegetation structure, composition, and fuels pose risks to biodiversity, sustainable ecosystems, and economic and community interests across the United States (USDA/USDI 2000). However, the magnitude of these risks varies between ecosystems as a result of differences in their fire and vegetation histories, successional, compositional, and structural dynamics, and the influence of invasive species (Morgan *et al*. 2001; Schoennagel *et al*. 2004). Fire exclusion over the 20th century has not affected all ecosystems uniformly, and accurate characterization of historical fire regimes and recent vegetation changes is critical to inform management decisions about the need for fuel treatments or ecological restoration across differing plant communities.

Use of historical fire regimes and vegetation conditions to inform fire and fuel management decisions in the US has been refined into the Fire Regime Condition Class (FRCC) concept (Hann and Bunnell 2001; Schmidt *et al*. 2002; Hann and Strohm 2003; Hann *et al*. 2003; Shlisky and Hann 2003). FRCC is an index that compares current with historical fire regimes and vegetation composition and structure to assess degree of departure on a scale from one (least departed) to three (most departed). FRCC is based on an assumption that historical processes and patterns (those present before widespread Euro-American settlement in the mid- to late-1800s) represent longer-term sustainable ecosystem conditions, and that greater departure in current conditions represents a greater risk for uncharacteristic fire behavior and associated ecosystem impacts. Initial coarse-level (1-km^2 resolution) FRCC maps described the degree of departure at a national scale (Schmidt *et al*. 2002). After this initial effort, a set of standard guidebook methods was developed to assess FRCC at landscape to stand scales for local management and planning needs (at time of writing, FRCC Guidebook v1.3; Hann *et al*. 2004). FRCC maps of 30-m^2 resolution are also being developed as part the LANDFIRE project, an effort to provide consistent vegetation, fuels, and fire regime data for the entire US (Rollins and Frame 2006; www.landfire.gov, accessed 19 October 2007). FRCC is now a key variable for defining wildfire risk to ecosystems as a result of its explicit incorporation into the Healthy Forests Restoration Act of 2003 (HFRA 2003). FRCC represents a significant advance in the integration of fire and forest histories and landscape and vegetation ecology to provide an ecologically based method for setting fire-management priorities and objectives across the US (Shlisky and Hann 2003).

Definition of departure indices in FRCC assessments begins with simulation modeling of historical vegetation composition and structure using the Vegetation Dynamics Development Tool (VDDT; Beukema and Kurz 2003). VDDT is used to develop non-spatially defined reference conditions within Biophysical Settings (BpS; formerly referred to as Potential Natural Vegetation Groups (PNVG); Küchler 1964; NRCS 2003). For LANDFIRE, BpS are derived from Nature Serve's ecological

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classification system (Comers *et al*. 2003) and are not directly comparable with those used in FRCC assessments. However, both systems use BpS in a similar manner to represent the vegetation communities that would likely exist under given environmental conditions (climate, soils, and landscape physiography) and historical disturbance regimes. BpS in LANDFIRE are assigned to specific locations in their nationwide mapping efforts, whereas BpS in FRCC assessments are non-spatial and assigned based on individual user needs for specific projects or management requirements. Reference conditions are the proportions of vegetation successional stages (community structure and composition) as affected by varying fire frequencies, severities, and successional pathways within each BpS (Hann *et al*. 2004).

FRCC and LANDFIRE vegetation models (also known as Vegetation Dynamics Models) were defined during regional professional workshops conducted between 2002 and 2009 (2005–09 for LANDFIRE). VDDT model inputs for individual BpS are based on historical fire regime characteristics (frequency and severity) and vegetation data derived from published and unpublished studies and expert opinion developed both at the regional workshops and through subsequent peer reviews (Hann *et al*. 2004). The amount and quality of available historical data for each BpS vary, which can affect the quality and accuracy of the resulting modeled reference conditions. In an FRCC assessment, a field evaluation is conducted of existing vegetation structure, which, in forests, is based on cover type, density of tree stands, tree size, and current successional status. Successional status is determined by visually estimating stand composition, tree density, and average tree age, the latter of which is based on tree diameters. Proportions of current successional classes in a project or management area are estimated during the field assessment and then compared with the proportions of reference conditions derived from VDDT model output. The FRCC departure index (1 to 3) is assigned based at least partially on differences in proportions of successional classes present in the current forest relative to modeled reference conditions in the historical forest.

There is a need to test the process of development of reference conditions by comparing VDDT model output with known fire and vegetation histories. This comparison is critical for assessing consistency and accuracy in the modeling process. Here, we compare VDDT-modeled reference conditions with tree-ring-based reconstructions of reference conditions from 11 forested sites in Utah and eastern Nevada (tree-ring data reported in Heyerdahl *et al*. 2005, and Brown *et al*. 2008*a*). The tree-ring reconstructions span transects aligned along elevation gradients that include multiple forest types. We ask the following questions with this comparison: (1) do FRCC methods of evaluating stand structure based on diameter estimates accurately represent ages of forest vegetation and is there variation based on species and site? (2) Do FRCC and LANDFIRE BpS models adequately capture the range of variation in proportions of reference conditions reconstructed by the tree-ring data? (3) Do LANDFIRE mapped data layers for BpS and Existing Vegetation Types (EVT) match the tree-ring plot data? (4) Can further empirical fire history and tree recruitment data be used to strengthen FRCC evaluation and reference condition modeling outputs? We consider this study to be only an initial test of FRCC and LANDFIRE vegetation modeling methods, but one that may provide an example for future testing needs.

Methods

Study area

Tree-ring sites used for this study extend from the Colorado Plateau of southern Utah, west to the Wah Wah Mountains in the eastern Great Basin of western Utah, and north to the Uinta and Bear River Mountains in northern Utah (Fig. 1, Table 1; Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). The region is a complex of valleys, mesas, canyons, plateaus, and mountains that range in elevation from ~900 to >3600 m. Forest types vary generally across elevation gradients. Piñon (*Pinus edulis* (PIED; four-letter codes are used in tables) and *P. monophylla* (PIMO)) and juniper (*Juniperus scopulorum* (JUSC) and *J. osteosperma* (JUOS)) savannas and woodlands occur at the lowest forest margins above desert shrublands or grasslands. Ponderosa pine (*Pinus ponderosa* (PIPO)) forests occur in montane zones in pure and mixed stands. Douglas-fir (*Pseudotsuga menziesii* (PSME)) often occurs in association with ponderosa pine on north-facing aspects and in relatively mesic sites. Mixed-conifer forests occur at intermediate elevations and include combinations of ponderosa pine, Douglas-fir, piñons, junipers, and firs (*Abies lasiocarpa* (ABLA) or *A. concolor* (ABCO)). Mixed-conifer forests also often occur in association with aspen (*Populus tremuloides* (POTR)). Aspen forms large (>100 ha) pure stands throughout the upper montane and lower subalpine zones across the study area except in the Great Basin. Lodgepole pine (*Pinus contorta* (PICO)) often forms pure stands at mid-elevations (1900 to 2800 m) or occurs in the mixed-conifer zone in northern Utah. Subalpine forests dominated by Engelmann spruce (*Picea engelmannii* (PIEN)) and firs occur at upper elevations (2350 to 3500 m). At the highest forested elevations (generally above 3000 m), pure Engelmann spruce forests occur in mesic sites whereas bristlecone pine (*Pinus longaeva* (PILO)) or limber pine (*P. flexilis* (PIFL)) are typically found in dry or rocky sites.

There was, in general, a gradient in fire frequency across the elevational gradient before fire exclusion that began at all sites in the late 1800s (Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). Fire occurrence was highest in the middle of the elevation range in ponderosa pine and drier mixed-conifer sites. Fire frequency progressively declined both above and below this middle-elevation zone. At upper elevations, generally moist conditions led to high fuel biomass, both living and dead, in many stands, but fewer years in which fuels were dry enough to ignite and spread. At lower elevations in the piñon–juniper woodlands, fuels were often dry enough to burn because of hotter and dryer fire seasons, but because of lower productivity, there were in general less continuous both aerial and surface fuels and fires were not able to spread. In the middle zone, both fuel amounts and moistures were just right (what has come to informally be called the 'Goldilocks effect'), and able to burn often in wide-spreading fires.

Utah forests underwent a period of intensive grazing and land use beginning in the 1850s as a result of Euro-American settlement. Intensive grazing removed understorey species and began alteration of longer-term historical forest dynamics. Logging also affected forest structure in many areas. The tree-ring study found that cessation of historical patterns of fires began in

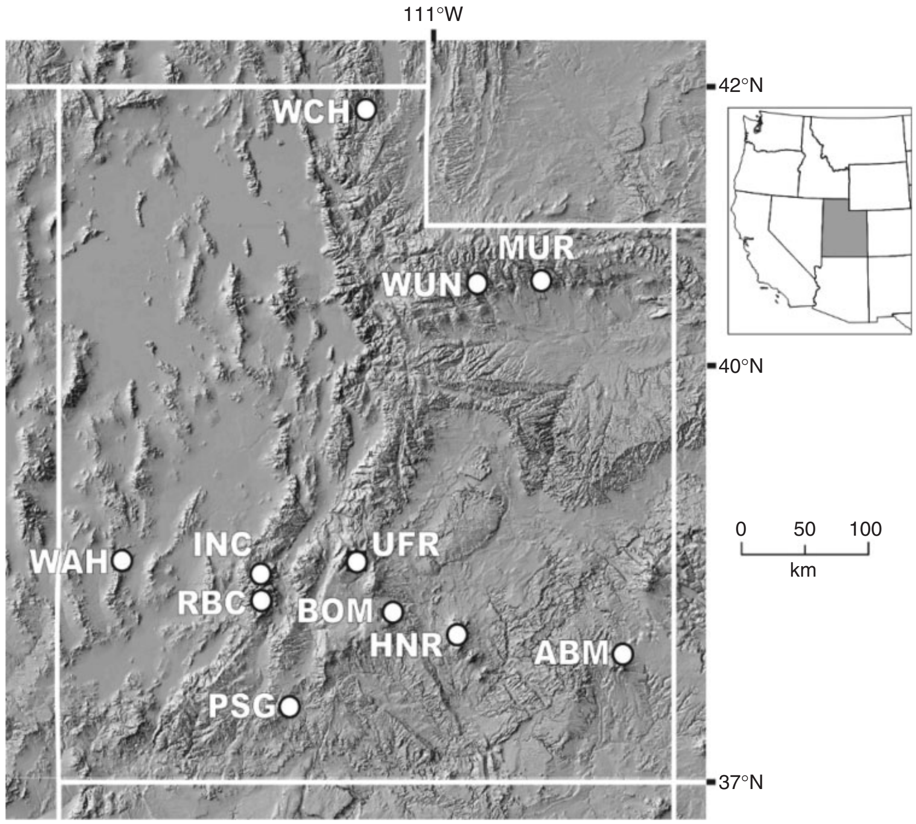

Fig. 1. Locations of tree-ring sites. Three-letter codes correspond to those in Table 1.

Table 1. Tree-ring sites used in the present study arranged from north to south
FRCC (Fire Regime Condition Class) and LANDFIRE BpS (biophysical settings) forest types are listed in Table 4

Site	Minimum elevation (m)	Maximum elevation (m)	Average precipitation (cm)	FRCC and LANDFIRE BpS
Wasatch Mountains (WCH)	2255	2588	100	SPFI, SPDF, CHPI, 10510, 10520, 10500, 10550
Western Uinta (WUN)	2207	3133	60	PPIN, SPDF, SPFI, CHPI, 10540, 10510, 10520, 10500, 10550
Middle Uinta River (MUR)	2308	3250	70	PPIN, SPDF, SPFI, CHPI, 10540, 10510, 10520, 10500, 10550
Wah Wah Mountains (WAH)	2195	2686	40	JUPI, PPIN, SPDF, 10160, 10540, 10500
Upper Fremont River (UFR)	2800	3039	80	SPDF, SPFI, 10510, 10520, 10500
Indian Creek (INC)	2364	2518	65	PPIN, SPDF, 10540, 10500
Beaver Creek (RBC)	2358	3077	90	PPIN, SPDF, SPFI, 10540, 10510, 10520, 10500
Boulder Mountain (BOM)	2405	3377	80	JUPI, PPIN, SPDF, SPFI, 10160, 10540, 10510, 10520, 10500
Henry Mountains (HNR)	2407	3138	60	JUPI, PPIN, SPDF, 10160, 10540, 10500
Abajo Mountains (ABM)	2557	3231	85	JUPI, PPIN, SPDF, SPFI, 10160, 10510, 10520, 10500
Paunsaugunt Plateau (PSG)	2309	2736	45	JUPI, PPIN, SPDF, SPFI, 10160, 10540, 10510, 10520, 10500

the 1860s to 1890s depending on location (Brown *et al*. 2008*a*), similar to patterns seen in forests throughout the western US. Initial reduction in fire frequency was likely the result of grazing that removed grass and herbaceous fuels, followed later by direct fire suppression in the 20th to 21st centuries.

Tree-ring data

The tree-ring study used a systematic sampling design to characterize stand and age structure and fire regimes across forest gradients in each site (Table 1; Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). Similar methods have been used in multiple studies

around the western US and are described in more detail in Heyerdahl *et al*. (2005, 2006), Brown and Wu (2005), Brown (2006), Brown *et al*. (2008*a*, 2008*b*), and Brown and Schoettle (2008). A 500-m grid was established at each site and plots sampled at grid points. Plot centers were located in the field using hand-held global positioning system (GPS) units. An *n*-tree density-adapted sampling method (Jonsson *et al*. 1992) was used to collect data from the nearest ~30 remnant (logs, snags, or stumps) or living trees >20 cm diameter at breast height (DBH) to each plot center. Maximum plot radius was set at 40 m (~0.5 ha) and most plots were ~<0.2 ha in size. For each plot tree, species was recorded and an increment core (on living trees) or cross-section (from logs, snags, and stumps) was collected from ~10 cm height above ground. Sampled cores had to be no more than a field-estimated 10 years from pith to minimize pith offset when assessing pith date. Diameter at sample height (DSH) and DBH were measured on living trees, and DSH was measured or estimated for remnant trees missing bark, sapwood, or heartwood. Distance from plot center and azimuth were measured on all trees for reconstruction of tree basal areas, density, and spatial patterning. To reconstruct surface fire history, cross-sections were cut from any fire-scarred trees found within plots. Additional fire-scarred trees also were sampled within ~80 m of each grid point and between grid points when discovered. GPS coordinates and species of fire-scarred trees outside of plots were recorded.

Standard dendrochronological methods were used to crossdate all samples using locally developed master chronologies (Heyerdahl *et al*. 2005). Pith dates were estimated on cores that did not intersect pith based on the curvature of the innermost rings sampled. The tree recruitment date is considered to be the date of tree pith at 10-cm height. No corrections were made for time to grow from germination to 10 cm sample heights because of the widely varying species and environmental conditions at the sites that were collected for the study. Once crossdating of ring series was completed on all samples, dates for any fire scars seen within the ring series were assigned. Any trees that were not able to be dated were not used in subsequent analyses.

FRCC and LANDFIRE vegetation models

VDDT modeling estimates the relative proportions of non-spatially defined reference conditions that would have occurred under a historical fire regime and an equilibrium (current) climate regime within each BpS (Beukema and Kurz 2003). VDDT input includes average fire frequencies, severities, and other disturbances defined as probabilistic events, and vegetation structural stage development pathways, including changes in species composition and density through a successional sequence. VDDT runs are commonly made for 500 years to allow vegetation conditions to equilibrate over time. VDDT output is proportions of vegetation successional classes – the reference conditions – across a non-spatially referenced landscape at the end of the 500-year model run. Reference conditions for most forest types are summarized into five seral stages that approximate overall developmental characteristics of community age and structure: early-replacement, mid-open, mid-closed, late-open, and late-closed. Each developmental stage represents a successional class defined by average tree age, species composition, structural characteristics, and response to disturbances. LANDFIRE and FRCC assessments use VDDT in a similar manner, but in LANDFIRE, reference condition proportions are then coupled with the spatial model LANDSUM (Keane *et al*. 2002) to map resulting vegetation conditions for each BpS across actual landscapes at a 30-m^2 spatial resolution.

FRCC and LANDFIRE developed their own BpS models using two different vegetation classification systems (Küchler 1964 *v.* Comers *et al*. 2003). Both systems attempt to describe the same historical vegetation using VDDT; however, their models use different probabilities for disturbance, and have somewhat different species distributions and geographic extents (often based on expert opinion; see http://frcc.gov, accessed 19 October 2007; www.landfire.gov for details).

Comparing tree-ring with FRCC and LANDFIRE data

We performed three tests to compare the tree-ring data with FRCC and LANDFIRE vegetation models. First, we compiled age and DBH data derived from the tree-ring study to assess whether FRCC methods of visual estimates of tree diameters accurately represent the age of forest vegetation for defining mid- and late-development classes of reference conditions. FRCC guidebook methods define >23 cm DBH as a visual indicator of a mature tree when conducting field assessments. For this analysis, we assumed that plots with trees averaging ≤23 cm DBH would be considered to be in a mid-development reference condition, and >23 cm would be in late-development. We conducted least-squares linear regressions to estimate fitness of tree age to DBH by species and site. As many of the regression models did not meet the statistical requirements of homoscedasticity, normality, and constant variance in model residuals, a logarithmic transformation was applied to the tree ages before regression. Models that had significant P values ($P < 0.05$) were considered to be representative of species growth estimates. We also conducted an analysis of variance (ANOVA) of age and diameter by species and site to both determine the strength of these relationships and how they varied by species and location across the region. All statistical analyses were conducted using the *Statistica* software (StatSoft Inc. 2008). The tree-ring study sampled a total of ~10 000 remnant and living trees; however, we only used data from the living trees for this part of our assessment. Dead trees (stumps, snags, and logs) often were missing bark, sapwood, or portions of the heartwood that reduced confidence in diameter estimates. The DBH-to-age analysis therefore consisted of 5173 living trees from 13 species from the 11 sites.

Our second test was whether VDDT modeled reference conditions captured the range of variation in reference conditions reconstructed by the tree-ring data as of a date of 1880. Dates of initial Euro-American settlement varied across the study area but all sites showed some Euro-American impact by 1880, including cessation of spreading fires in almost all of the sites (Brown *et al*. 2008*a*). As current vegetation may not be representative of past vegetation type, only species present in 1880 and their corresponding ages were used to assign BpS and reference condition to each of a total of 273 plots that were sampled from the 11 sites (Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). Both living and remnant trees were used to estimate the 1880 plot compositions. FRCC and LANDFIRE use key species to

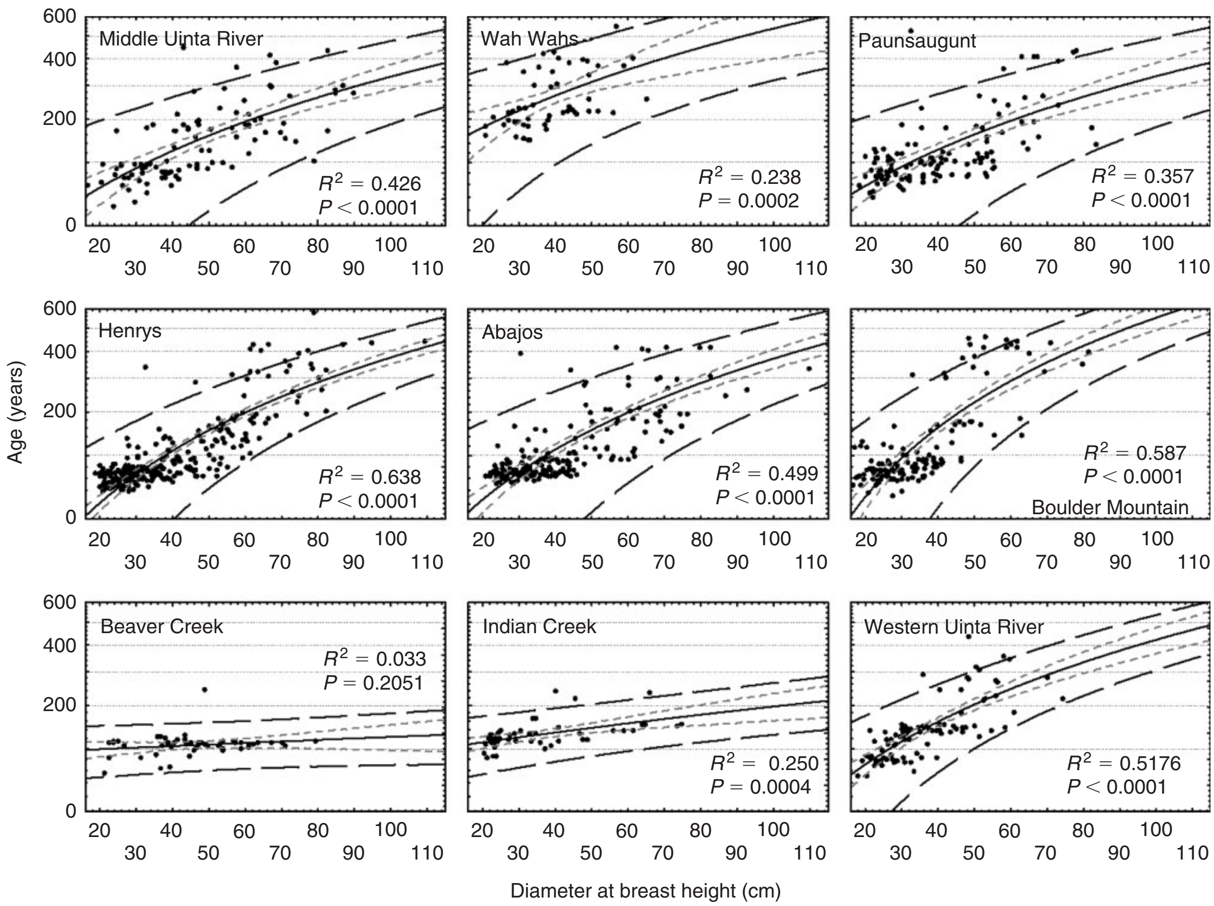

Fig. 2. Diameter at breast height (DBH) and log(age) regressions for ponderosa pine trees by site, with linear fits (solid lines), 95% confidence intervals (gray dashed lines), and 95% prediction intervals (black dashed lines). Overall R^2 for ponderosa pine trees across all sites was 0.44.

define vegetation characteristics when conducting an assessment and we used these species as the basis for assigning BpS and reference condition to each plot.

Historical age class and species composition in 1880 for each plot were compared with FRCC and LANDFIRE reference conditions for selected BpS. FRCC and LANDFIRE BpS descriptions are available on their respective project websites (www.frcc.gov; www.landfire.gov). We did not evaluate the typical five-stage VDDT models because of difficulties in using the tree-ring data to accurately recreate smaller size classes in historical stand densities as a result of probable tree mortality and decay since pre-settlement periods (e.g. Brown and Cook 2006; Brown *et al*. 2008*b*). However, we assume that we are able to define with some confidence mid- and late-development stands based on crossdated ages of trees present in each plot in 1880. The mean age of a 23-cm-DBH live tree varied by species, and we used the tree-ring results to estimate the upper 95% confidence interval for predicted tree size to consider whether a stand was late developmental stage in 1880. We grouped data from open and closed stands together based on age and composition for comparison with succession classes from VDDT output. If any trees in a plot were older than their predicted age-to-size confidence interval, the plot was considered to be in late-development in 1880. If there were no older trees during the historical period, then the plot was considered to have been in mid-development. If there were no trees during the historical period, the plot was considered to have no data and not used in this analysis. Once plots were categorized by BpS and reference condition, they were compared with FRCC and LANDFIRE BpS model proportions of mid- and late-development vegetation based on VDDT output. We used a Chi-square test to determine if the observed tree-ring reference condition proportions were significantly different than the expected based on the VDDT output.

Finally, we compared tree-ring plot data with LANDFIRE BpS and EVT map layers produced by the LANDFIRE project. LANDFIRE data are spatially mapped, which provided a unique opportunity to evaluate vegetation models at a high spatial resolution through comparison with the mapped tree-ring data. Plots were first located through their GPS coordinates relative to LANDFIRE map data. The BpS assignments we made for each plot in 1880 were then compared with LANDFIRE BpS map data. We also compiled the living tree composition in each plot and compared that with the LANDFIRE EVT map data. If key

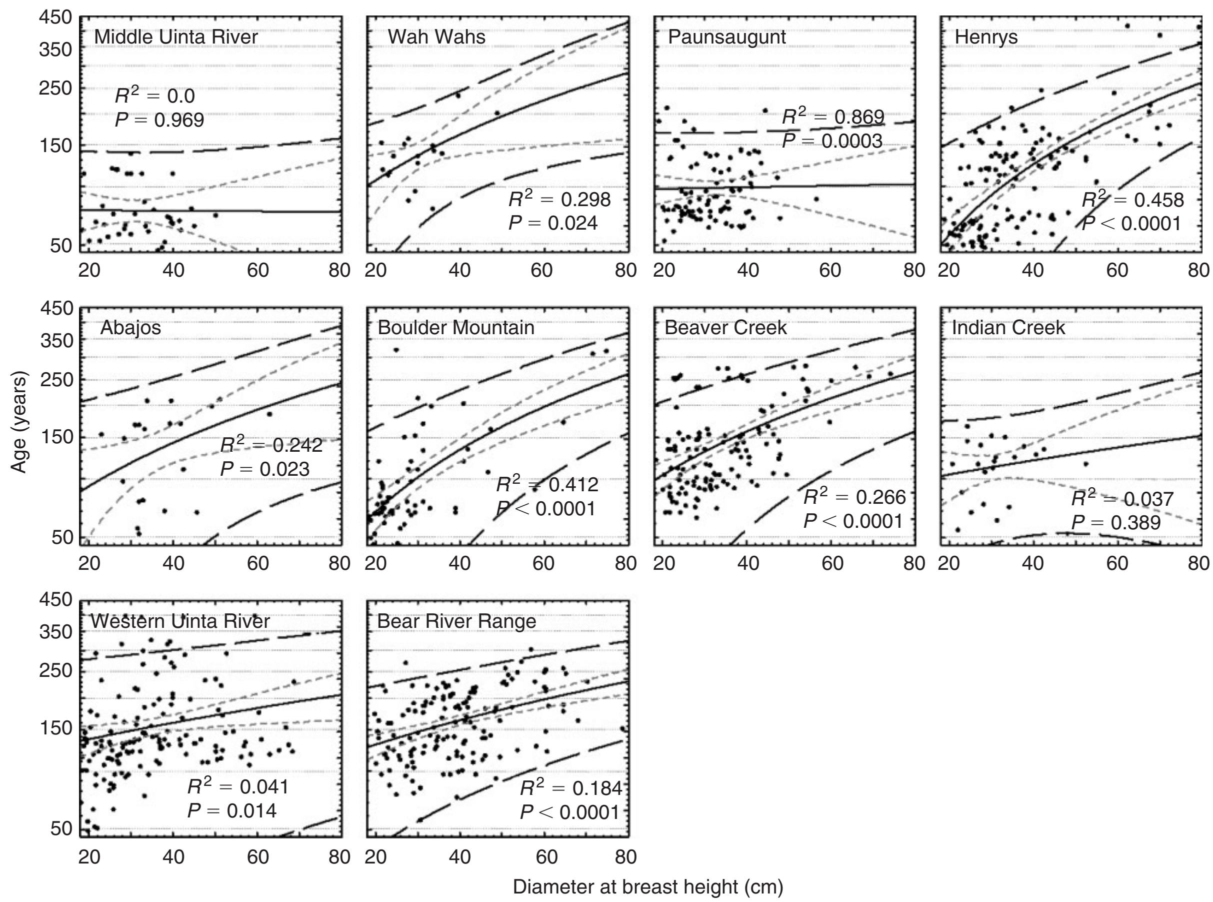

Fig. 3. Diameter at breast height (DBH) and log(age) regressions for Douglas-fir trees by site, with linear fits (solid lines), 95% confidence intervals (gray dashed lines), and 95% prediction intervals (black dashed lines). Overall R^2 for Douglas-fir trees across all sites was 0.21.

species were present in the tree-ring data in comparison with the mapped BpS or EVT, then the grid point was considered to have been accurately mapped in LANDFIRE.

Results

Age–diameter relationships

DBH and tree ages exhibited generally broad relationships, both within species and among sites (Figs 2–4; Tables 2, 3). Ponderosa pine was the only species where age and size were strongly correlated using data from all sites ($R^2 = 0.438$, $P < 0.001$) and were strongly correlated over most of the individual sites (Table 2). There were outliers for most species by DBH and age; however, their deviance did not significantly change the results. Median tree age was predicted for trees at 23 cm using an inverse prediction with 95% confidence interval (Table 3). ANOVA results indicate that species associated with infrequent fire regimes (piñon–juniper, spruce–fir, and bristlecone pine; Heyerdahl *et al.* 2005) were found to have greater average ages than frequent fire species (especially ponderosa pine and Douglas-fir; Fig. 5). Variance of diameters relative to ages for species that contained a large sample *n*, such as Douglas-fir (PSME), ponderosa pine (PIPO), and Engelmann spruce (PIEN) was small. There was greater variance found in species that had fewer sampled trees and plots, such as bristlecone pine (PILO), Rocky Mountain juniper (JUSC), one-seed juniper (JUOS), limber pine (PIFL), and single leaf piñon (PIMO), but this result is likely an artifact of the smaller number of trees used in each regression. ANOVA indicated that DBH and age estimates for all sites were similar with the exception of WAH (Fig. 5). This may be explained by the large presence of fire-infrequent and older species (bristlecone pine, Rocky Mountain juniper, and one-seed juniper) that were sampled in that site.

FRCC and LANDFIRE BpS models

Median ages of trees >23 cm DBH were used to define the proportions of mid- and late-development reference conditions for trees present in plots in 1880 (Table 3). Reference condition proportions reconstructed from the tree-ring data compared favorably with FRCC BpS models for ponderosa pine (PPIN5), mixed-conifer (SPDF), and lodgepole (CHPI) but not for piñon–juniper (JUPI1, JUPI2), south-western mixed-conifer (MCAN) and spruce–fir (SPFI2, SPFI7; Table 4, Fig. 6). Reference condition proportions reconstructed from the tree-ring data

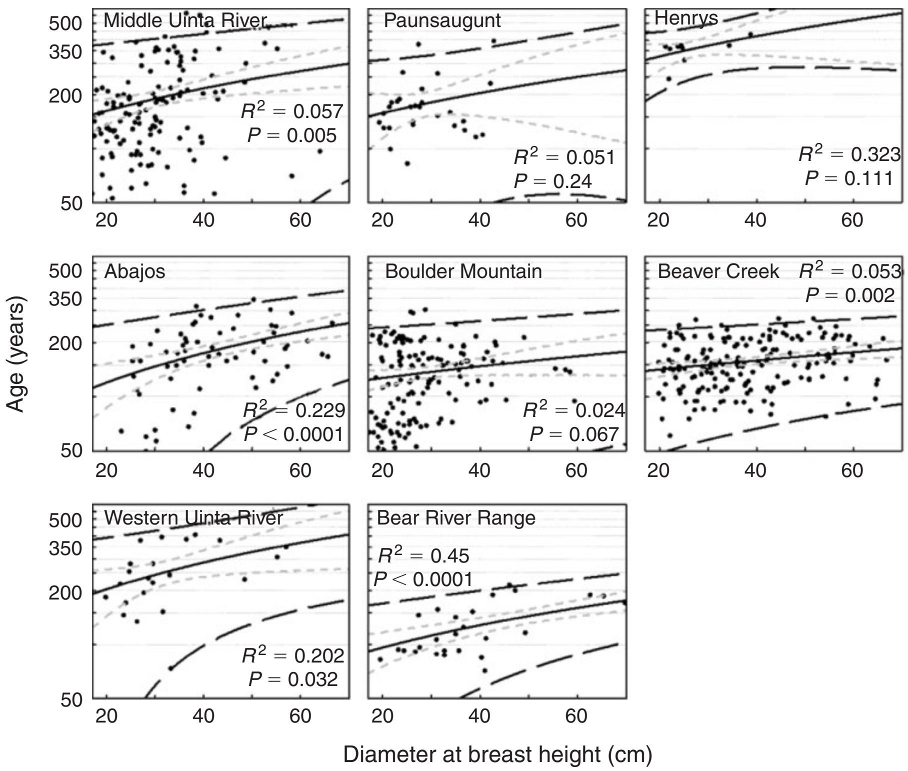

Fig. 4. Diameter at breast height (DBH) and log(age) regressions for Engelmann spruce trees by site, with linear fits (solid lines), 95% confidence intervals (gray dashed lines), and 95% prediction intervals (black dashed lines). Overall R^2 for Engelmann spruce trees across all sites was 0.06.

Table 2. Observed two-sided *P* values for DBH–age regressions for all species at all sites

Bold values represent locations where *P* values are significant at the 95% confidence interval (<0.05) based on sample size (>10 trees)

Species	Site									
	WCH	RBC	ABM	BOM	HNR	PSG	INC	WUN	MUR	WAH
PIPO		0.021	**<0.0001**	**<0.0001**	**<0.0001**	**<0.0001**	**0.0004**	**<0.0001**	**<0.0001**	**0.0002**
PSME	**<0.0001**	**<0.0001**	**0.0234**	**<0.0001**	**<0.0001**	0.88	0.39	**0.01**	0.969	0.024
PIEN	**<0.0001**	**<0.0001**	**<0.0001**	0.066	0.111	0.241		0.03	0.005	
ABLA	**<0.0001**	0.19	**<0.0001**		0.147			0.37		
POTR	**0.01**	0.01	**<0.0001**	0.63	0.107		0.40	0.81	**0.020**	
ABCO		0.22				**<0.0001**	0.22		0.069	0.002
PICO	**<0.0001**				**<0.0001**	**0.0007**				
PIFL	0.28				**<0.0001**	0.090	0.28			
PIED				**<0.0001**	**<0.0001**	0.025				
PIMO										**<0.0001**
JUSC				0.152		**0.111**				0.903
JUOS				**0.0003**		0.677			0.797	**0.0002**
PILO										0.574

compared favorably with LANDFIRE BpS models for Rocky Mountain dry–mesic montane mixed-conifer (10510), aspen and aspen–mixed-conifer low- and high-elevation forests (10110, 10611, 10612), but not for piñon–juniper (10160), ponderosa pine (10540), Rocky Mountain mesic montane mixed-conifer (10520), Rocky Mountain subalpine dry–mesic spruce–fir forest and woodland (10550), and Rocky Mountain lodgepole pine (10500; Table 4, Fig. 6). The JUPI1 BpS model (Table 4) was the most different from the tree-ring data, although the JUPI2 model had a similar trend of a larger proportion of late-successional

stands in comparison with the tree-ring data (Fig. 6). Spruce–fir and lodgepole pine data both showed low correspondence with VDDT model results, including opposite trends of more older than younger stands in the tree-ring data in contrast to the VDDT modeled reference conditions (Fig. 6).

Table 3. Expected median ages of trees >23 cm DBH (diameter at breast height) by species, with lower and upper 95% confidence intervals derived from tree-ring data

NS, age–DBH regression not significant

Species	Age (years) at 23 cm	R^2	*P* value
PIPO	40.9 ± 3.2	0.438	<0.0001
JUOS	114.9 ± 41.9	0.438	<0.0001
PIED	135.3 ± 21.9	0.28	<0.0001
PIFL	66 ± 11.4	0.271	<0.0001
PIMO	176.3 ± 29.8	0.231	<0.0001
PSME	42.9 ± 6	0.213	<0.0001
PICO	54.3 ± 12.6	0.112	<0.0001
POTR	104 ± 9.1	0.095	<0.0001
PIEN	24.7 ± 14.7	0.055	<0.0001
JUSC	NS	0.05	0.0961
ABCO	14.8 ± 14.4	0.023	<0.0001
PILO	NS	0.012	0.6295
ABLA	50.2 ± 10.2	0.01	<0.0001

LANDFIRE map data

LANDFIRE map layers were found to be overall ~58% accurate for BpS and 60% accurate for EVT when compared with the tree-ring data for each plot (Table 5). LANDFIRE maps were 38% accurate for both BpS and EVT, 28% accurate for at least one type (17% EVT accurate and BpS inaccurate, with 11% BpS accurate and EVT inaccurate), and 34% inaccurate. Mixed-conifer and spruce–fir types had the highest accuracies by BpS for LANDFIRE with accuracies ranging from 64 to 82% for BpS and 67 to 79% for EVT. Piñon–juniper was the least accurately mapped BpS and EVT with 13 and 37% accuracy respectively.

Discussion

FRCC and LANDFIRE BpS models

Current stand conditions are determined through visual estimates of stand structure, including tree diameters, in FRCC assessments (Hann *et al*. 2004). FRCC assessments are designed to be a relatively rapid method of characterizing current vegetation and fire regime departures from historical conditions. The expense of collecting field data, such as canopy closure, canopy base height, tree density, stand age structure, and fire and stand histories, make field sampling impractical for FRCC assessments. However, based on the limited findings of this study, it appears that FRCC methods may result in inaccurate measures of plant community departure based on visually estimated

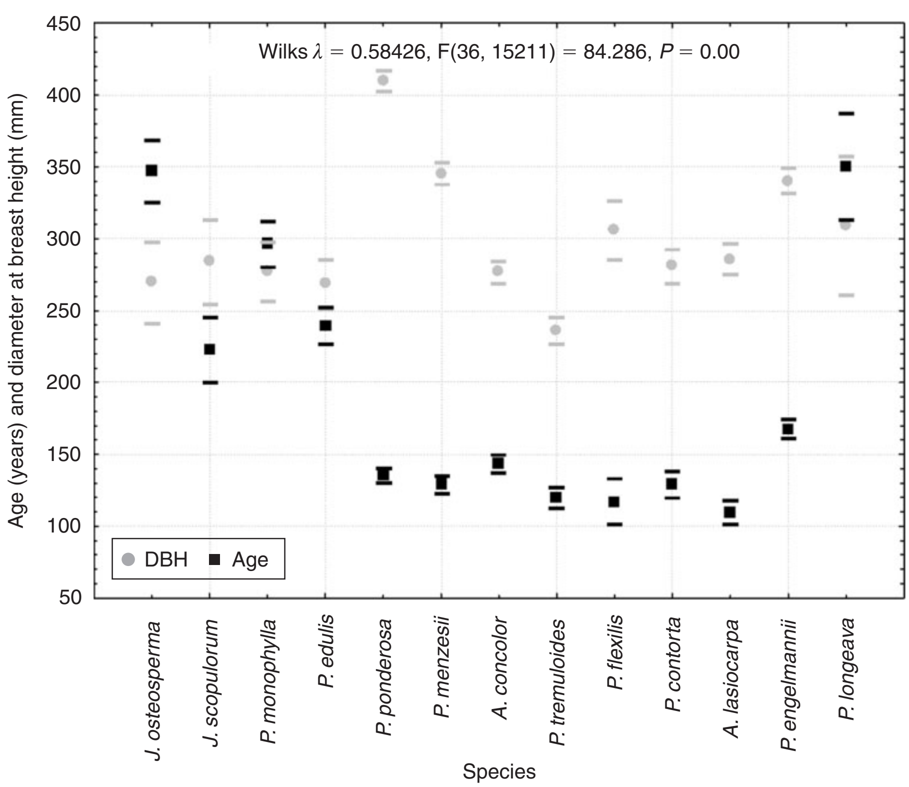

Fig. 5. ANOVA of age and diameter at breast height (DBH) by species and site. Horizontal bars represent 95% confidence intervals.

Table 4. Observed proportions of mid- and late-development reference conditions reconstructed from tree-ring data in plots collected in Utah, compared with FRCC (Fire Regime Condition Class) BpS (biophysical settings) model output for mid- and late-development reference conditions from Hann *et al.* (2004) and LANDFIRE

MFI, mean fire interval (years); *n*, number of plots in the observed data in mid- or late-seral stages. Chi-square fit is for the observed plots *v.* BpS models with 1 degree of freedom, $P = 0.05$ significance for types <3.84. BpS that meet the range of variability in the observed data are highlighted in bold

BpS description *v.* observed data	FRCC code	LANDFIRE code	Mid (%)	Late (%)	MFI	*n*		Chi-square	*P* value
						Mid	Late		
Observed PIED, PIMO, JUSC, JUPI			4	96		1	24		
Piñon–juniper infrequent fire	JUPI2		30	70	435			12.99	0.0003
Piñon–juniper frequent fire	JUPI1		50	50	31			21.16	<0.0001
Colorado Plateau piñon–juniper woodland		10160	55	45	128			26.273	<0.0001
Observed PIPO			26	74		13	37		
Colorado plateau ponderosa	**PPIN5**		25	75	6			**0.027**	0.87
Southern Rocky Mountain ponderosa pine woodland		10540	44	56	15			6.575	0.01
Observed PSME, ABCO, PIPO, PIEN			48	52		35	38		
South-western mixed-conifer	MCAN		35	65	10			5.377	0.02
Rocky Mountain dry–mesic montane mixed-conifer forest and woodland		**10510**	40	60	10			**1.92**	0.166
RM mesic montane mixed-conifer forest and woodland		10520	75	25	33			28.498	<0.0001
Spruce–fir–Douglas-fir[A]	**SPDF**		58	42	19			**0.658**	0.417
Observed PICO			36	64		4	7		
Lodgepole pine	CHPI		65	35	125			3.965	0.046
Rocky Mountain lodgepole pine forest		10500	100	0	124			4.455	0.035
Observed PIEN, ABLA, ABCO			26	74		16	58		
RM subalpine dry–mesic spruce–fir forest and woodland		10550	65	35	212			61.206	<0.0001
Lower subalpine forest	SPFI7		80	20	91			157.622	<0.0001
Upper subalpine forest	SPFI2		70	30	143			82.474	<0.0001
Observed POTR			90	10		33	4		
Deciduous woodland–oak or aspen	DWOA		55	45	100			17.474	<0.0001
Intermountain basins aspen–mixed-conifer forest – low		**10611**	85	15	10			**0.509**	0.475
Intermountain basins aspen–mixed-conifer forest – high		**10612**	95	5	32			**2.63**	0.105
Rocky Mountain aspen forest and woodland		**10110**	80	20	27			**2.146**	0.142
PILO			0	100		0	3		

[A] Includes observed POTR plots.

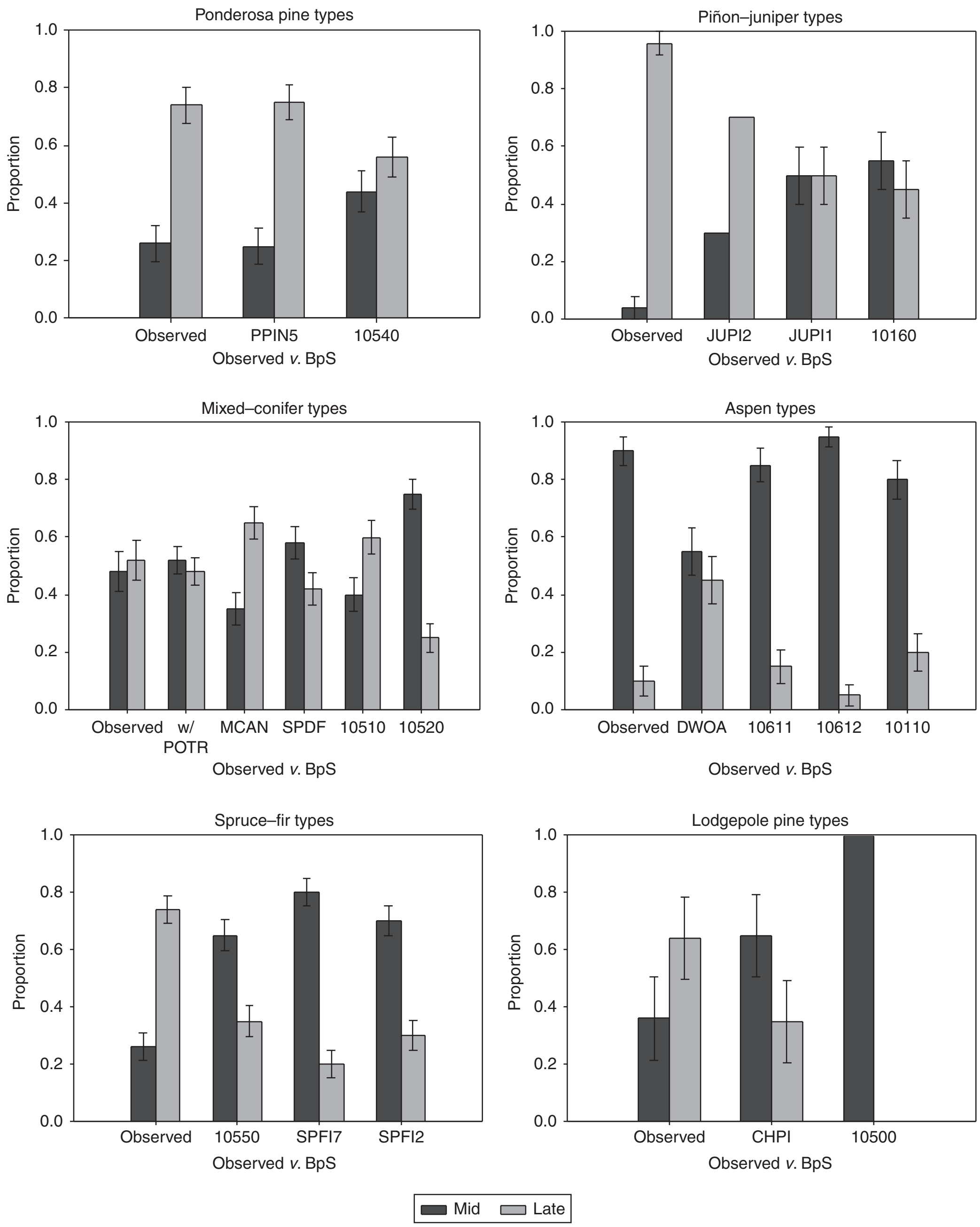

Fig. 6. Proportion of plots observed in the tree-ring data compared to FRCC (Fire Regime Condition Class) and LANDFIRE modeled reference condition proportions. Error bars were generated by calculating the 95% confidence interval from sample variance and standard error of observed points. Tree-ring results are on the left (e.g. observed), FRCC and LANDFIRE models are listed by their four-letter abbreviations on the right (e.g. PPIN5, 10540, etc.).

age–diameter relationships for determining reference condition proportions. Variations in age–size relationships both within species and among sites (Figs 2–5) may limit the ability to accurately gauge departure from estimated historical composition based on VDDT model results. Generally poor relationships between size and age may result in misassignment of current reference condition proportions based only on visual estimates, which may in turn lead to misassignment of the FRCC index.

Better correspondence between the tree-ring data and some BpS models indicates that VDDT models more accurately reflect historical forest structure in frequent-fire forest types such as ponderosa pine, mixed-conifer and aspen, than in

Table 5. LANDFIRE accuracy by BpS (biophysical settings) and EVT (existing vegetation type)
Code is the LANDFIRE map code for BpS or EVT type, *n* is number of plots tested, and % is percentage that were accurately mapped based on tree-ring data at plot scale

	Code	*n*	%
BpS			
Rocky Mountain aspen forest and woodland	10110	31	32
Colorado Plateau piñon–juniper woodland	10160	29	14
Rocky Mountain lodgepole pine forest	10500	7	43
Rocky Mountain dry–mesic montane mixed-conifer forest and woodland	10510	6	33
Rocky Mountain mesic montane mixed-conifer forest and woodland	10520	11	64
Southern Rocky Mountain ponderosa pine woodland	10540	19	53
Rocky Mountain subalpine dry–mesic spruce–fir forest and woodland	10550	82	66
Intermountain basins aspen–mixed-conifer forest – low elevation	10611	22	82
Intermountain basins aspen–mixed-conifer forest – high elevation	10612	31	77
Intermountain basins mountain mahogany woodland and shrubland	10620	6	50
EVT			
Rocky Mountain aspen forest and woodland	2011	26	50
Colorado Plateau piñon–juniper woodland and shrubland	2016	43	37
Rocky Mountain lodgepole pine forest	2050	19	63
Rocky Mountain montane mesic mixed-conifer forest and woodland	2052	9	78
Southern Rocky Mountain ponderosa pine woodland	2054	24	46
Rocky Mountain subalpine dry–mesic spruce–fir forest and woodland	2055	53	79
Intermountain basins aspen–mixed-conifer forest and woodland	2061	64	67
Abies concolor forest alliance	2208	14	71

infrequent-fire types such as spruce–fir and piñon–juniper (Fig. 6). BpS reference condition models were determined by managers and scientists familiar with the local ecology of each region during regional workshops. BpS types that are considered to be representative of each region were identified and described based on available historical and ecological data. Some BpS types, such as ponderosa pine and dry mixed-conifer forests, have extensive fire and forest history data with which to parameterize VDDT model runs. Other BpS types are less well studied and their fire and vegetation histories less certain, especially across the range of environmental and community variation within and between regions. The better correspondence between modeled and reconstructed reference conditions in frequent-fire-type models (ponderosa pine and mixed-conifer forest types; Fig. 6) is likely related to the greater amount of fire and forest history research that has been conducted in these forest types. Conversely, fire-infrequent types (spruce, and piñon–juniper woodland types; Fig. 6) have had less fire history research conducted, with the result that their fire regimes and successional patterns are less well documented for input to VDDT modeling. Furthermore, infrequent-fire types generally have fewer observations of historical fires and forest successional changes available for adequate characterization of fire regime parameters for VDDT modeling (e.g. Brown *et al*. 2008*a*).

Another factor that undoubtedly results in varying model and data results is that individual-site fire histories often have experienced contingent historical events that lead to differences from a ‘typical’ or average fire regime of a particular forest type. Stochastic modeling in FRCC and LANDFIRE generalizes vegetation and its fire regimes into generic types and does not take into account site-specific variability or, more importantly, the history of climate variations or other disturbances that may have affected changes in community structure through time. Variations in site histories undoubtedly contribute to variations in ratios of actual from modeled reference conditions. For example, spruce–fir and lodgepole pine FRCC and LANDFIRE BpS models predict more mid- than late-development stands, but the Utah tree-ring data found the opposite (Fig. 6). This may be due to longer fire intervals in this region than in other areas, leading to generally older stands across landscapes. Many spruce trees found in the tree-ring study were >300 years old at the time of sampling and probably resulted from fires that occurred in the late 1600s, most commonly in 1685 (Heyerdahl *et al*. 2005). However, the current presence of older rather than younger stands does not mean that these forests are outside their historical ranges of variability in either their fire regime or forest structure, but rather that they have not had extensive fires in the intervening period that would have resulted in a larger proportion of mid-successional stands as suggested should be present based on VDDT model results. Without taking into account this history of the forest landscapes, the VDDT models suggest that there is current departure in the landscape proportions of reference conditions in Utah spruce–fir and lodgepole pine forests.

Taking into account differences in fire histories, the trend of model results toward older or younger successional classes in each BpS may be more important to consider in FRCC assessments rather than the absolute proportions of stand structures. This may provide a more realistic perspective for assessing whether a particular BpS should be considered as inside or outside of its historical range of variation. For example, the tree-ring fire data for piñon–juniper (P-J) woodlands show the majority of stands are currently in late-development structural stages (Fig. 6). The FRCC BpS model JUPI2 (Table 4) also predicts more late-development trees than younger, but underpredicts what was found in the tree-ring data. The sensitivity of the VDDT model to fire frequency is critical to the setting of reference conditions. The model inaccuracy may be due to the model’s fire

return interval, currently predicted to be ~450 years. If the interval is increased (~1000 years), the model begins to more closely reflect the tree-ring results. A recent assessment of (P-J) ecosystems in the western US concluded that fire was only a minor disturbance in many less productive stands because of lack of both surface and crown fuels with which to carry fire (Romme *et al*. 2009). We believe that many of the Utah stands sampled probably fell into this category of fire regime historically, which means that if the longer intervals had been used in VDDT modeling, the reference conditions would likely be closer to what was found in the tree-ring data. The error may also be due to the definition of a mid-development stand in terms of the age; the mean ages of sampled piñon and juniper were among the highest in the tree-ring study. The mid-definition could be changed for P-J to an older age class by species to define the mid- from late-successional classes in the reference conditions.

Good correspondence between the tree-ring data and models for ponderosa pine (PPIN5), aspen (10110), and mixed-conifer (SPDF, 10510; Fig. 6) suggests that the reference conditions for these BpS were accurately modeled by VDDT parameters, at least in the Utah study sites. However, results of this study suggest that inaccuracy in piñon–juniper and subalpine types makes any decision based on a VDDT output possibly subject to error. For BpS types in which disturbance may not be the major or only factor in tree recruitment, VDDT models may need further evaluation. Additional empirical disturbance and forest history sampling in piñon–juniper, spruce–fir, and lodgepole pine types should increase the available information about these systems to use in VDDT modeling. However, because of generally longer fire intervals in these forests, any departure from historical to present conditions may be less than in frequent-fire BpS such as ponderosa pine and mixed-conifer forests. A possible result of inaccurate estimations of departure and wrong FRCC classification may be the application of incorrect management actions that could lead to even further departure from historical conditions (see also Romme *et al*. 2009).

The only accurate way to establish the age of a stand is to physically sample the trees for ages. We suggest based on the results of our comparison that at least some limited age sampling is needed for FRCC assessments. This sampling probably should include removing cores from the field and crossdating by trained dendrochronologists to most accurately characterize age and successional status of stands. Additional field-sampled fire history and stand establishment data, especially in the less-well-studied ecosystems, should further increase the accuracy of VDDT models through better dynamic estimations of age structures and relationships with fire regimes. However, we also realize that this type of sampling is expensive and – perhaps more critically to the efficient use of FRCC in forest management decisions – more time-consuming than FRCC visual assessment methods as currently practiced. Nevertheless, we suggest that some sort of compromise solution could be found that would provide both the most accurate as well as timely data possible for FRCC assessment needs.

LANDFIRE maps

Zhu *et al*. (2006) used a cross-validation technique to determine that existing vegetation data layer accuracies are between 60 and 89% in LANDFIRE maps. Our study's comparison of LANDFIRE and tree-ring data falls on the lower end of the estimate of Zhu *et al*. (2006) (Table 4). When broken down by BpS and EVT, some types are more accurately represented in LANDFIRE data than others. EVT mapping in LANDFIRE is most accurate for the mixed-conifer and spruce–fir types. These forests generally have the densest and most continuous canopies, and may have been easiest to identify through remote sensing methods because of their continuous canopies and more distinctive NDVI reflectance in Landsat spectral bands (Zhu *et al*. 2006). Conversely, sparser canopy cover may have led to lower accuracy in other types such as piñon–juniper, which is similar to what Zhu *et al*. (2006) found. It should be noted, however, that piñon–juniper plots sampled for the tree-ring study were generally found in ecotonal areas near lower ends of study sites, and may not be wholly representative of piñon–juniper BpS as defined in the LANDFIRE mapping effort.

Conclusion

Historical forest conditions reconstructed from tree-ring data provide opportunities for comparison with FRCC and LANDFIRE modeled vegetation data across multiple forest types. The tree-ring reconstructions we examined suggest that reference conditions are better modeled in frequent-fire forest types but not necessarily in infrequent-fire forest types, at least in Utah forests. Additional studies in fire-infrequent forest types should increase understanding of historical stand compositions, fire histories, and other disturbances with which to better parameterize VDDT reference condition models. The greatest amount of fire history research has been conducted in ponderosa pine and mixed-conifer forests, which likely contributed to the better correspondence between tree-ring data and VDDT model results that we found in this study. We consider this study as only a first step in comparison of empirical vegetation data with vegetation models used in both FRCC assessments and the nationwide LANDFIRE mapping effort. Tree-ring data provide an opportunity to compare site-specific vegetation patterns and fire regime variations that are often not easily accounted for in modeling efforts. Revised methods for assessing FRCC may need to take into greater account both tree ages and stand histories to more accurately compare with model results. We also suggest that ranges of reference conditions be incorporated into the BpS classifications to better take into account fire and forest histories rather than trying to establish average conditions that must be met for a FRCC index to be assigned.

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Land Use Planning and Wildfire: Development Policies Influence Future Probability of Housing Loss

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Abstract

Increasing numbers of homes are being destroyed by wildfire in the wildland-urban interface. With projections of climate change and housing growth potentially exacerbating the threat of wildfire to homes and property, effective fire-risk reduction alternatives are needed as part of a comprehensive fire management plan. Land use planning represents a shift in traditional thinking from trying to eliminate wildfires, or even increasing resilience to them, toward avoiding exposure to them through the informed placement of new residential structures. For land use planning to be effective, it needs to be based on solid understanding of where and how to locate and arrange new homes. We simulated three scenarios of future residential development and projected landscape-level wildfire risk to residential structures in a rapidly urbanizing, fire-prone region in southern California. We based all future development on an econometric subdivision model, but we varied the emphasis of subdivision decision-making based on three broad and common growth types: infill, expansion, and leapfrog. Simulation results showed that decision-making based on these growth types, when applied locally for subdivision of individual parcels, produced substantial landscape-level differences in pattern, location, and extent of development. These differences in development, in turn, affected the area and proportion of structures at risk from burning in wildfires. Scenarios with lower housing density and larger numbers of small, isolated clusters of development, i.e., resulting from leapfrog development, were generally predicted to have the highest predicted fire risk to the largest proportion of structures in the study area, and infill development was predicted to have the lowest risk. These results suggest that land use planning should be considered an important component to fire risk management and that consistently applied policies based on residential pattern may provide substantial benefits for future risk reduction.

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Introduction

The recognition that homes are vulnerable to wildfire in the wildland-urban interface (WUI) has been established for decades [e.g., 1,2]; but with a recent surge in structures burning, this issue is now receiving widespread attention in policy, the media, and the scientific literature. Single fire events, like those in Greece, Australia, southern California, and Colorado have resulted in scores of lost lives, thousands of structures burned, and billions of dollars in expenditures [3–6]. With the potential for increasingly severe fire conditions under climate change [7] and projections of continued housing development [8], it is becoming clear that more effective fire-risk reduction solutions are needed. "Fire risk" here refers to the probability of a structure burning in a wildfire within a given time period.

Traditional fire-risk reduction focuses heavily on fire suppression and manipulation of wildland vegetation to reduce hazardous fuels [9]. Enormous resources are invested in vegetation management [10], but as increasing numbers of homes burn down despite this massive investment, the "business-as-usual" approach to fire management is undergoing reevaluation. One issue is that fuel treatments may not be located in the most strategic positions, i.e., in the wildland-urban interface [11]. Yet, even if treatments surrounded all communities, scattered development patterns are difficult for firefighters to reach [12–14], and fuel treatments do little to protect homes without firefighter access [15–16]. Fuel treatments may also be ineffective against embers or flaming materials that blow ahead of the fire front [17].

One alternative to traditional fire management that is receiving widespread attention is to prepare communities through the use of fire-safe building materials or creating defensible space around structures [17–18]. These actions represent an important shift in emphasis from trying to prevent wildfires in fire-prone areas to better anticipating fires that are ultimately inevitable. Nevertheless, the cost of building and retrofitting homes to be fire-safe can be prohibitive, and these actions do not guarantee immunity from fire [19].

Land use planning is an alternative that represents a further shift in thinking, beyond the preparation of communities to withstand an inevitable fire, to preventing new residential structures from being exposed to fire in the first place. The reason homes are vulnerable to fires at the wildland-urban interface is a function of its very definition: "where homes meet or intermingle with wildland vegetation" [20]. In other words, the location and

pattern of homes influence their fire risk, and past land-use decision-making has allowed homes to be constructed in highly flammable areas [21]. Land use planning for fire safety is beginning to receive some attention in the literature [22–23], and there is growing recognition of the potential benefits of directing development outside of the most hazardous locations [8,19,24].

Despite recent attention in the literature, land use planning for wildfire has yet to gain traction in practice, particularly in the United States. However, fire history has been used to help define land zoning for fire planning in Italy [22], and bushfire hazard maps are integrated into planning policy in Victoria, Australia [25]. Although some inertia inevitably arises from complications with existing policy and plans, a primary impediment to the design and implementation of fire-smart land use planning is lack of guidance about specific locations, patterns of development, or appropriate methodology to direct the placement of new development. Without a solid knowledge base to draw from, planners will be misinformed about which planning decisions may result in the greatest overall reduction of residential landscape risk. Even worse, poor science could result in placement of homes in areas that actually have high fire hazard.

Research on how planning decisions contributed to structures burning in the past provides some guidance about what actions may work in the future. Analysis of hundreds of homes that burned in southern California the last decade showed that housing arrangement and location strongly influence fire risk, particularly through housing density and spacing, location along the perimeter of development, slope, and fire history [26]. Although high-density structure-to-structure loss can occur [27–28], structures in areas with low- to intermediate- housing density were most likely to burn, potentially due to intermingling with wildland vegetation or difficulty of firefighter access. Fire frequency also tends to be highest at low to intermediate housing density, at least in regions where humans are the primary cause of ignitions [29–30].

These results suggest, for example, that placing new residential development within the boundaries of existing high-density developments or in areas of low relief may reduce fire risk. However, it is difficult to know whether broad-scale planning policies would actually result in the intended housing arrangement and pattern at the landscape scale, and whether those patterns would result in lower fire risk. Our objective here was to simulate three scenarios of future residential development, and to project wildfire risk, in a rapidly urbanizing and fire-prone region where we have studied past structure loss [25]. We based all future development on an econometric subdivision model, but we varied the emphasis of subdivision decision-making based on three broad and common growth types.

Although cities vary in extent, fragmentation, and residential density [31–32], urban form typically adheres to a set of common patterns [33–34], and we based our development scenarios on the three primary means by which residential development typically occurs: infill, expansion, or leapfrog [34]. Infill is characterized by development of vacant land surrounded by existing development, typically in built-up areas where public facilities already exist. [35–36], and should result in higher structure density rather than increased urban extent. Expansion growth occurs along the edge of existing development, extends the size of the urban patch to which it is adjacent, and may have variable influence on structure density. Leapfrog growth occurs when development occurs beyond existing urban areas such that the new structure is surrounded by undeveloped land. This type of growth would expand the urban extent and initially result in lower structure density; but these areas may eventually become centers of growth from which infill or expansion can occur. We asked:

1) Do residential development policies reflecting broad growth types affect the resulting pattern and footprint of development across the landscape?
2) Do differences in extent, location, and pattern of residential development translate into differences in wildfire risk, based on the current configuration of structures?
3) Which development process, infill, expansion, or leapfrog, results in the lowest projected fire risk across the landscape?

Methods

Study Area

The study area included all land within the South Coast Ecoregion of San Diego County, California, US, encompassing an area of 8312 km^2. The region is topographically diverse with high levels of biodiversity, and urban development has been the primary cause of natural habitat loss and species extinction [37]. Owing to the Mediterranean climate, with mild, wet winters and long summer droughts, the native shrublands dominating the landscape are extremely fire-prone. San Diego County was the site of major wildfire losses in 2003 and 2007 [38], although large wildfire events have occurred in the county since record-keeping began, and are expected to continue, as fire frequency has steadily increased in recent decades [29,39]. The county is home to more than three million residents, and approximately one million more people are expected by 2030 [40]. Although most residential development has been concentrated along the coast, expansion of housing is expected in the eastern, unincorporated part of the county.

Econometric Subdivision Model

A host of alternative modeling approaches exist to simulate future land use scenarios [41], including a cellular automaton model that we previously applied to the study area [42]. We chose to use an econometric modelling approach for this study because we wanted to capture fine-scale, structure-level patterns and processes that are correlated with housing loss to wildfire [26]; and econometric models may perform better at the scale of individual parcels [43].

Although we based the three development scenarios on generalized planning policies, we also wanted to ensure that the residential projections were realistic and adhered to current planning regulations. The objective of the econometric modeling was to estimate the likelihood that residential parcels will subdivide in the future. Therefore, we used a probit model to estimate the transition probability of each parcel based on a range of potential explanatory variables typically associated with parcel subdivision and housing development [44–45].

To develop the model of subdivision probability, we acquired GIS data of the county's parcel boundaries in years 2005 and 2009 from the San Diego Association of Governments (SANDAG). The dependent variable was equal to 1 if a parcel subdivided between 2005 and 2009, and zero otherwise. Using these data layers we first determined which parcels were legally able to subdivide given current land use regulations. Minimum lot size restrictions are typically considered the most import restriction for determining future land use. We deemed a parcel eligible for subdivision if the current lot size was greater than twice the minimum legal size given the land class. To determine which parcels subdivided between 2005 and 2009, we queried parcel IDs where the total

area was reduced by at least the minimum lot size between the two time periods. Finally, we were able to generate a suite of variables that determine the likelihood of a parcel developing in the future (Table S1).

We overlaid the parcel boundaries over a range of GIS layers representing our explanatory variables. These data are available to download at (http://www.sandag.org/index.asp?subclassid = 100&fuseaction = home.subclasshome). Our explanatory variables included: parcel size, parcel size squared, six dummy variables which capture non-linear effects of parcel size, distance to the coast, distance to the coast squared; distance to city center and its square, current zoning, slope, land use, roads, if the parcel is in a protected area, if the parcel is in a development area, if the parcel is in the redevelopment area (Table 1).

Spatial Model of Future Development under Planning Alternatives

The outcome of the land use change econometric model is the subdivision probability for each parcel for a five-year time step. Based on these probabilities, we developed a GIS spatial simulation model of future land use under three distinct planning scenarios: infill (development in open or low density parcels within already developed areas), expansion (development on the fringe of developed areas), and leapfrog (development in open areas). The model runs in four 5-year time steps from 2010 to 2030, and generates the spatial locations of new housing units in the county.

Although development decisions could feasibly depend on fire risk, we did not model that here. There is no evidence that fire has influenced past regional planning decisions, so it was not used as an explanatory variable in the econometric model. Although we could have evaluated the potential for future development decisions to be based in part on fire risk, this would have required simulation of feedbacks between fires and probability of development. Because our objective in this study was to isolate the effects of the three distinct growth types, we modeled fire risk only as a function of development pattern and not vice versa.

We constructed a complete spatial database of existing residential structures in the study area [26]. These structures and their corresponding parcel boundaries served as the initial conditions for all three scenarios of the spatial simulation model. The current and projected future GIS layers of structures were also subsequently used in the fire risk model (see below). The

Table 1. Variables and results from the probit regression model of parcel subdivision in San Diego County.

Subdivided (1 = yes,0 = no)	Coefficient	Std. Err.	z	P>|z|	[95% Conf. Interval]	
Acres of lot	0.0026342	0.00075	3.51	0	0.001164	0.004105
Acres of lot ^{2}	−3.02E-06	1.29E-06	−2.34	0.019	−5.55E-06	−4.93E-07
Distance to ocean	−7.42E-06	1.33E-06	−5.59	0	−0.00001	−4.82E-06
Distance to ocean ^{2}	2.33E-11	8.28E-12	2.82	0.005	7.11E-12	3.96E-11
Distance to major road	2.17E-07	2.74E-06	0.08	0.937	−5.16E-06	5.59E-06
Distance to major road ^{2}	−1.94E-11	1.70E-11	−1.14	0.252	−5.27E-11	1.38E-11
Distance to nearest city center	−0.0000115	1.70E-06	−6.76	0	−1.5E-05	−8.16E-06
Distance to nearest city center ^{2}	2.89E-11	9.70E-12	2.98	0.003	9.91E-12	4.79E-11
Slope between 0–5%	0.6211289	0.211761	2.93	0.003	0.206085	1.036173
Slope between 5–10%	0.3911427	0.210684	1.86	0.063	−0.02179	0.804076
Slope between 10–25%	0.0716669	0.212725	0.34	0.736	−0.34527	0.4886
Rural Residential	−0.3563149	0.071512	−4.98	0	−0.49648	−0.21615
Single Family	0.1361149	0.068678	1.98	0.047	0.001509	0.270721
Multi-Family	−0.2505093	0.151486	−1.65	0.098	−0.54742	0.046397
Road	0.015329	0.086094	0.18	0.859	−0.15341	0.184069
Open Space	−0.7440933	0.099145	−7.51	0	−0.93841	−0.54977
Orchard/Vineyard	−0.5813305	0.097867	−5.94	0	−0.77315	−0.38951
Agriculture	−0.9785208	0.132734	−7.37	0	−1.23867	−0.71837
Vacant Land	−0.5222501	0.074586	−7	0	−0.66844	−0.37606
Zoned protected	0.253769	0.076881	3.3	0.001	0.103086	0.404452
Area marked for redevelopment	−0.2680261	0.14069	−1.91	0.057	−0.54377	0.007722
Area marked for development	0.5780101	0.064103	9.02	0	0.452371	0.703649
Parcel between 10–20 acres	−0.3379532	0.065899	−5.13	0	−0.46711	−0.20879
Parcel between 5–10 acres	−0.6119036	0.067012	−9.13	0	−0.74325	−0.48056
Parcel between 2–5 acres	−1.16297	0.07062	−16.47	0	−1.30138	−1.02456
Parcel between 1–2 acres	−1.563956	0.090286	−17.32	0	−1.74091	−1.387
Parcel between.5–1 acres	−1.999939	0.099893	−20.02	0	−2.19573	−1.80415
Parcel between.25–.5 acres	−2.178273	0.117101	−18.6	0	−2.40779	−1.94876
Constant	−1.397931	0.227467	−6.15	0	−1.84376	−0.9521

Sample size 113 001, LR Chi^2 1535.23, pro>chi 0, pseudo R^2 0.22. Further description of the variables is provided in Table S1.
doi:10.1371/journal.pone.0071708.t001

dataset of existing housing includes locations of 687,869 structures, of which 4% were located within the perimeter of one of 40 fires that burned since 2001. During these fires, 4315 structures were completely destroyed, and another 935 were damaged.

For future development scenarios, we wanted to allocate an equal number of new structures to the landscape. This was to ensure that any predicted difference in fire risk was a function of the arrangement and location of structures, not the total number of structures. Nevertheless, differences in the total number of structures were simulated with each of the 5-year time steps. We determined the number of housing units to add during the simulations based on projections made by San Diego County [46]. Using factors such as development proposals, general plan densities, and information from jurisdictions, the county estimated that between 331,378 units and 486,336 units could be supported within the developable residential land by 2030. Because the eastern, desert portion of the county was not included in our study area, we used a conservative approach and simulated the addition of 331,378 new dwelling units. We divided this number by four to define the number of new dwelling units to add at each time step, assuming a linear growth rate.

One output of the econometric model was the prediction of the maximum number of new dwelling units that could be added to each parcel. However, dwelling units may consist of apartments as well as single family homes. The mix of single and multifamily units in the region has remained relatively constant over time, and the overall trend has been a mix of roughly 1/3 multifamily and 2/3 single family units. Because the fire risk model is based on points representing structure locations across the landscape, regardless of the number of dwelling units per structure, we needed to generate a conversion factor from dwelling units to structures. We therefore defined a minimum lot size of 0.25 acre on which no more than a single structure could be built, regardless of the number of dwelling units in it (i.e., a single family home or apartment complex). Then, once a parcel was selected for development by the model (see details below), we divided its total area by the maximum number of dwelling units to be added, according to the econometric model. If the result was larger than 0.25, we subdivided parcels according to the result. If not, we quantified how many 0.25 acre parcels fit into the original parcel, and generated the new parcel boundaries accordingly.

Using the initial map of parcels (year 2010), we classified each parcel that was defined as eligible for development (in the previous stage) as suitable for one of the three planning scenarios described above, according to the number of developed parcels in its immediate neighborhood (i.e., those parcels that share a boundary with the focal parcel). We defined 'developed parcels' as ones that had more than one house per 20 acres (8.09 ha). Therefore, according to these density thresholds, we allowed some parcels with nonzero housing density to be considered as 'undeveloped' because these large, rural parcels might contain a single or a handful of houses but they exist within a large open area. In other words, the overall land cover of these parcels was effectively undeveloped, and we therefore assumed that development in adjacent parcels would be akin to development in open areas.

We defined infill parcels as those that were completely surrounded by developed parcels. Expansion parcels had at least one neighboring parcel that was undeveloped; and leapfrog parcels were those with no developed parcels in their immediate surroundings. We reclassified the type of each available parcel in the same manner after each time step, to account for changing dynamics in the development map of the county.

We conducted three simulations, one for each development scenario (infill, expansion, and leapfrog). In each simulation, all parcels were eligible to subdivide, regardless of their class. Therefore, to build a simulation for a specific scenario, we increased the development probability of parcels of the selected scenario by 20%, to favor their development compared to the other types of parcels, without prohibiting development in the other parcel types. This approach was necessary because the projected number of dwelling units was much larger than it would be possible to fit in infill and leapfrog class parcels solely. For example, as the spatial coverage of developed parcel expands, there is less contiguous area that is undevelopable and suitable for leapfrog development. Therefore, the scenarios are not exclusive, but rather a mixture of the three development types. Yet, in each scenario, there is one main type of development, and smaller amounts of development events of the other two types.

Due to the immense computational demand of the simulations, we adopted a deterministic, rather than a stochastic approach to decide on which parcels were subdivided. After enhancing the transition probability according to the corresponding scenario, we ranked and then sorted all parcels according to their probability of subdivision. We then sequentially selected parcels, while simultaneously tallying the number of dwelling units in them, until the development target in that time step (one fourth of the total number of dwelling units to be added: 82,795) was reached. Once the development target was reached, we moved to the next time step. After each time step, the remaining parcels that were still eligible for development were re-classified to development types according to the new spatial configuration of the landscape.

Once a parcel was selected for subdivision, and the number of new parcels to develop in it was calculated (as detailed above), an equal-area spatial splitting model was employed to split the parent parcel to the predefined number of equal-area child parcels. We developed a simple splitting model which is based on iterative splitting of larger parcels into two smaller parcels using a straight line splitting boundary. Once the parcel was fully split into the needed number of sub-parcels, we allocated a new structure inside each new parcel by generating a point at its centroid (center of gravity). The point datasets of all structure locations per time step per scenario were passed over to the fire risk model, which is described below.

Fire Risk Modeling and Analysis

To project the distribution of fire risk under alternative scenarios, we used MaxEnt [47–48], a map-based modeling software used primarily for species distribution modeling [48], but we have used it successfully for ignition modeling [50] and for projecting current fire risk in the study area [26]. For this study, we slightly modified the model from Syphard et al. [26]. The dependent variable was the location of structures destroyed by fire between 2001 and 2010. Although inclusion of damaged structures in the data set does not significantly affect results [26], we only included completely destroyed structures to avoid the introduction of any uncertainty.

The MaxEnt software uses a machine-learning algorithm that iteratively evaluates contrasts among values of predictor values at locations where structures burned versus values distributed across the entire study area. The model assumes that the best approximation of an unknown distribution (i.e., structure destruction) is the one with maximum entropy. The output is an exponential function that assigns a probability to every cell of a map. Thus, the resulting continuous maps of fire risk represented the probability of a structure being destroyed by fire. In these output maps, areas of predicted high fire risk that did not have structures on them represented environmental conditions similar to those in which structures have actually burned.

We based the explanatory variables on those that were significantly related to burned structures in Syphard et al. [26], including maps depicting housing arrangement and pattern, housing location, and biophysical factors. Housing pattern variables reflected individual structure locations as well as the arrangement of structures within housing clusters. We calculated housing clusters, defined as groups of structures located within a maximum of 100 m from each other, by creating 100 m buffers around all structures and dissolving the overlapping boundaries [51].

Because burned structures were significantly related to small housing clusters [26], we calculated the area of every cluster as an attribute, and then created raster grids based on that attribute. Low-to intermediate housing density and distance to the edge of the cluster were also significant explanatory variables relative to housing pattern and location [26], so we also created raster grids for those. GIS buffer measures at 1-km have been found to explain approximately 90% of the variation in rural residential density [52], so we developed density grids using simple density interpolation based on a 1-km search radius, with area determined through square map units. To create grids representing distance to the edge of clusters, we first collapsed the cluster polygons into vector polyline files, and then created grids of interpolated Euclidean Distance to the edge within each cluster.

Because the MaxEnt model randomly selects background samples in the map to compare with locations of destroyed structures, we used a mask to restrict sampling to the developed environment within cluster boundaries; the distance to the edge of the cluster would represent a different relationship inside a cluster boundary versus outside in the wildland. We also modified the grids to ensure that any random sample located within the 100m buffer zone would receive a value of 100m; thus, all points within the buffer were considered "the edge of the development".

After creating the grids representing housing pattern and arrangement of the current configuration of structures, we applied the same algorithms to the maps of simulated future structure locations. We thus generated grids representing future housing pattern and arrangement under alternative development scenarios. The other explanatory variables, including fire history, slope, fuel type, southwest aspect, and distance to coast [26] remained constant through time for current and future scenarios. Although historic fire frequency and fuel type typically change through time, we did not simulate their dynamics here because we wanted to isolate the effect of planning decisions on housing pattern and arrangement while holding everything else constant.

We conditioned the MaxEnt model on present distributions of housing using ten thousand random background points and destroyed structures located no closer than 500-m to minimize any effect of spatial autocorrelation. We used 80% (260 records) of these data for model training, and 20% [66 records) for testing. We repeated the process using cross-validation with five replicates and used the average of these five models for analyses. For smoother functions of the explanatory variables, we used hinge features, linear, and quadratic with an increase in regularization of beta set at 2.5, based on Elith et al. [48]. The smoother response curves minimize over fitting of the model. We conducted jackknife tests of explanatory variable importance.

We first developed the model using mapped explanatory variables derived from the current configuration of structures. To project fire risk under the different time steps of the alternative development scenarios, projected the model conditioned upon current conditions onto maps representing future conditions by substituting the grids representing future housing pattern and arrangement. This is similar to how potential future distributions of species are projected under climate change scenarios [49].

To quantify differences among current and future alternative scenarios, we calculated metrics representing housing density, pattern, and footprint to determine the extent to which the planning policies produced differences in housing pattern and location. We compared the modeled structure fire risk of the scenarios by overlaying all maps of structure locations with their respective mapped output grids from the MaxEnt models and calculating probability of burning for every structure point. We also calculated total area of risk by selecting three threshold criteria [53]. These criteria, at 0.05, 0.25, and 0.5 represented three different degrees of risk, and we calculated the proportion of structures that were located in risk areas for every time step in all scenarios.

Results

The probit econometric model, run on 113 001 observations, showed that larger parcels were most likely to subdivide, although the relationship between parcel size and subdivision probability was non-linear (Table 1). Parcels closer to existing roads, the ocean, those with lower slopes, and those designated as fit for development were all most likely to develop. Parcels designated in redevelopment areas were less likely to develop. Overall, the model had a pseudo r –squared of 0.22.

The land use simulation model, based on a combination of the econometric subdivision model and three different growth policies, resulted in substantial differences in the extent and pattern of housing of the three scenarios. The total area of housing development, or the housing footprint, was largest for simulations where leapfrog growth dominated, followed by expansion-type development, and then infill (Figure 1a). The differences in the housing footprint became larger among the scenarios over time, but the largest difference was between infill and the other two development types. As the housing footprint expanded in the three scenarios, the corresponding housing density declined, so that leapfrog growth resulted in the lowest housing density per 1-km, followed by expansion and then infill (Figure 2b). Despite the near inverse of this relationship, there was generally a larger separation among scenarios with regard to housing density. With larger housing footprints and lower housing density, the number of separate housing clusters increased while their size decreased (Figure 2c).

In the first two time steps of the model (2015 and 2020), the simulated development pattern closely followed the desired pattern in the scenario, although some of the growth in the infill scenario ended up becoming expansion or leapfrog (Table 2). In the last two time steps (2025 and 2030), there were not enough infill parcels left, and thus, the majority of growth in these simulations became expansion, followed by infill, and then leapfrog. In the last time step, there were not enough isolated parcels in the leapfrog scenario and thus, the majority of development became expansion. Thus in general, as more development occurred in the simulations by the year 2030, the majority took the form of expansion.

The area under the curve (AUC) of receiver operating characteristic (ROC) plots, indicating the ability of the MaxEnt model to discriminate between burned and unburned structures, averaged across five cross-validated replicate runs was 0.91. The AUC represents the probability that, for a randomly selected set of observations, the model prediction was higher for a burned structure than for an unburned structure [49].The two most important variables in the model according to the internal jackknife tests in MaxEnt [47] were related to housing pattern:

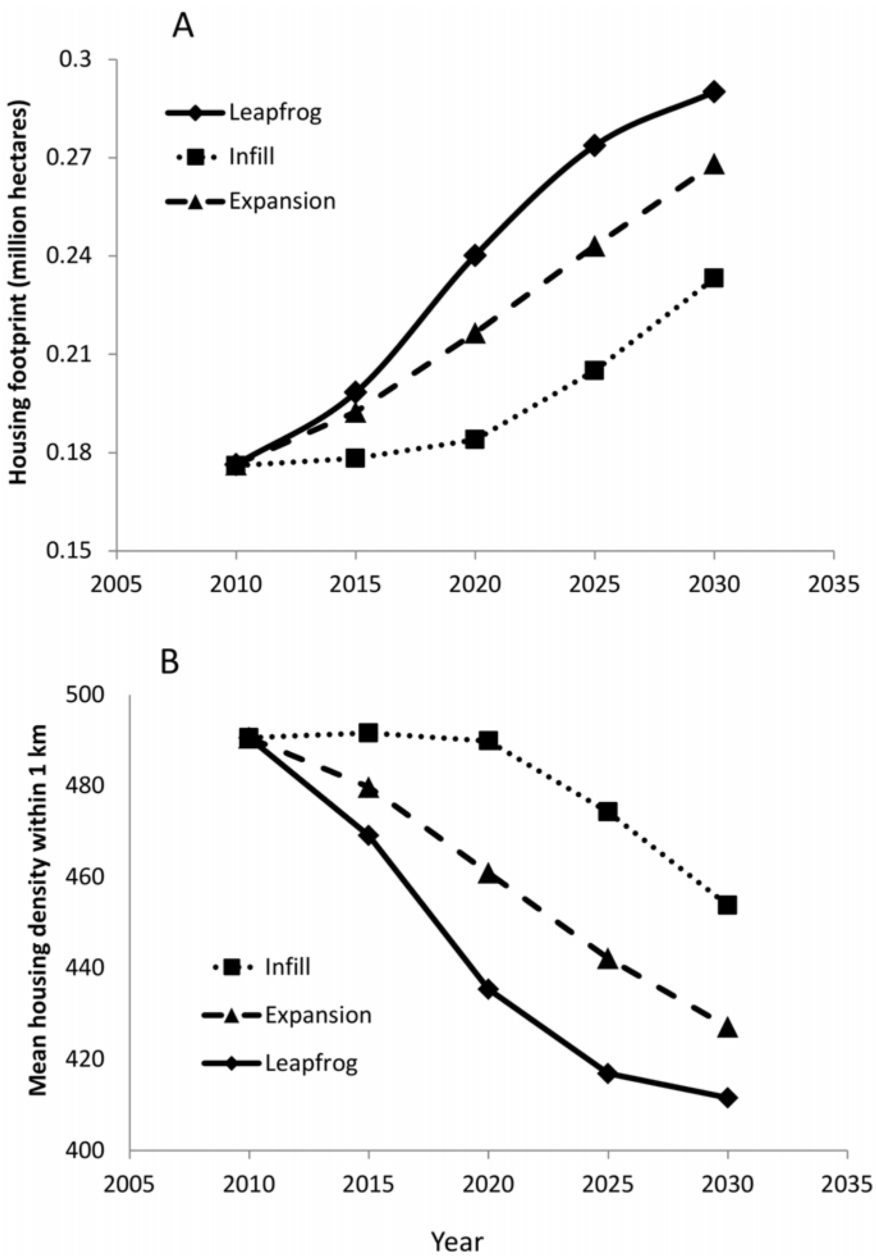

Figure 1. Trends of development extent and pattern for three planning policy simulations from 2010–2030, including A) total housing footprint representing the area of land within all housing clusters, and B) mean housing density averaged across all housing clusters.
doi:10.1371/journal.pone.0071708.g001

low to intermediate housing density and small cluster size and housing density (Figure 3). The distance to the edge of housing cluster was a less important contribution.

Maps showing the probability of a structure being destroyed in a wildfire, displayed as a gradient from low to high risk, show broad agreement relative to the general areas of the landscape that are riskiest, with correlation coefficients ranging from 0.85–0.91 (Figure 4). Nevertheless, subtle differences are apparent in the three development-scenario maps by year 2030, with the highest-risk areas in the expansion scenario located farther east than infill, and the highest-risk areas in leapfrog occupying a wider extent than either of the other two scenarios.

Differences among current housing and the three development scenarios are clearly illustrated through the mean landscape risk, or total probability of all structures burning (Figure 5). All three development scenarios were predicted to experience an increase in mean landscape risk over the duration of the simulations, except for infill at year 2015. The highest landscape risk to structures was predicted for the leapfrog scenario, followed by expansion, and then infill. The increase in risk over time is more gradual for the infill scenario than the other two scenarios.

The ranking of scenarios varied according to the proportion of structures located within different levels of risk defined through binary thresholding (Figure 6). When the continuous risk maps were thresholded at the lowest number of 0.05, a large proportion

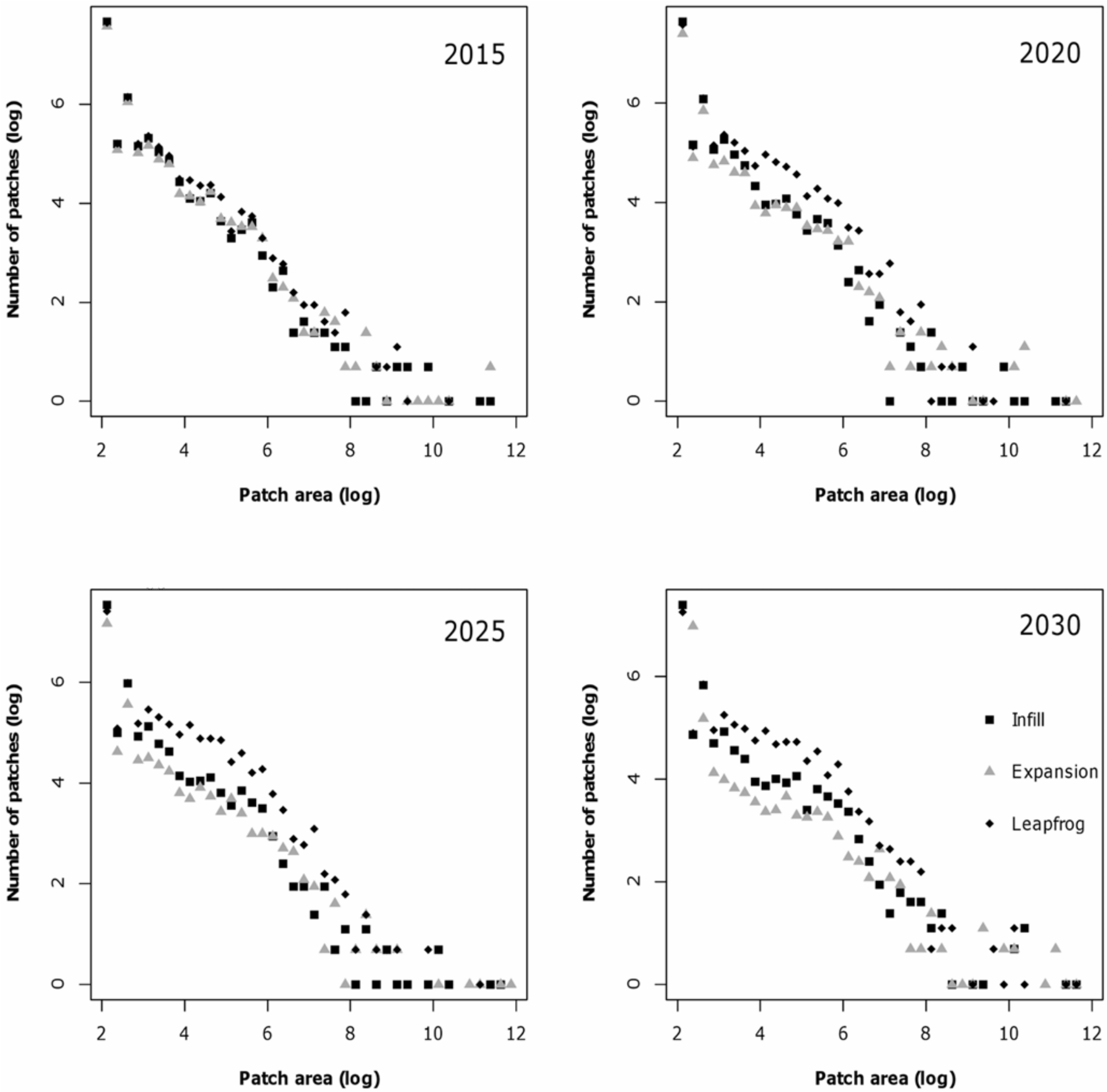

Figure 2. Trends in number of patches and patch area for three planning policy simulations from 2010–2030. Numbers were log-transformed for better visual representation of the scenarios.
doi:10.1371/journal.pone.0071708.g002

of structures in all scenarios fell within areas defined as risky according to this criterion. At this threshold, the proportion of structures in high-risk areas increased linearly for the expansion and leapfrog development scenarios while the proportion of infill homes increased more gradually. When risk was defined more conservatively at 0.25, temporal trends for the leapfrog and infill scenarios were similar to the 0.05 threshold. However, the proportion of structures at risk in the expansion scenario initially increased to 2020, but this proportion leveled off and declined by 2030. When the threshold was highest at 0.50, a very low proportion of structures in any scenario were located in areas at risk. But in these high-risk areas, the expansion scenario switched places with infill to have the lowest proportion of structures at risk in all time steps. Leapfrog had the largest proportion of homes at risk. This proportion of homes located in areas at risk with a threshold at 0.5 declined over time for all three scenarios.

Discussion

Our simulations of residential development showed that planning policies based on different growth types, applied locally for subdivision of individual parcels, will likely produce substantial and cumulative landscape-level differences in pattern, location, and extent of development. These differences in development pattern, in turn, will likely affect the area and proportion of

Table 2. Pattern of simulated development under infill, expansion, and leapfrog growth policies.

		Actual development		
Development scenario	**year**	*Infill*	*Expansion*	*Leapfrog*
Infill	2015	9450	18	6
	2020	11787	153	29
	2025	236	624	144
	2030	325	890	179
Expansion	2015	0	772	0
	2020	0	1243	2
	2025	0	1871	1
	2030	0	2662	0
Leapfrog	2015	0	10	408
	2020	0	5	1132
	2025	1	83	3563
	2030	34	917	0

The numbers in the table denote the numbers of patches of a given development type.
doi:10.1371/journal.pone.0071708.t002

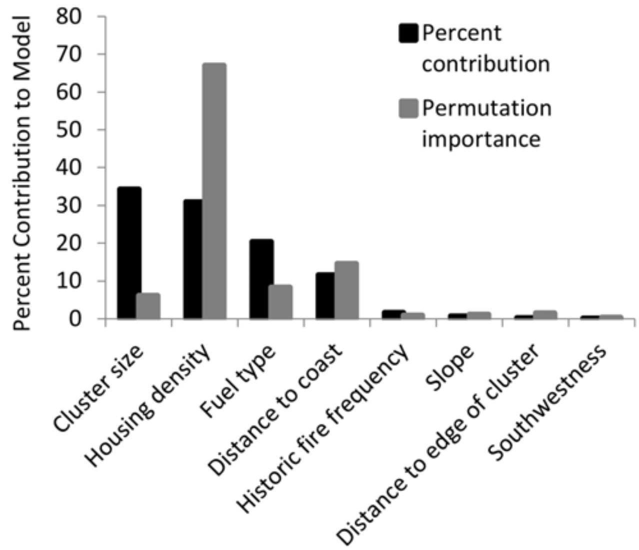

Figure 3. The importance of explanatory variables averaged across five cross-validated replications in the MaxEnt fire risk model. Percent contribution is determined as a function of the information gain from each environmental variable throughout the MaxEnt model iterations. Permutation importance reflects the drop in model accuracy that results from random permutations of each environmental variable, normalized to percentages.
doi:10.1371/journal.pone.0071708.g003

structures at risk from burning in wildfires. In particular, the scenarios with lower housing density and larger numbers of small, isolated clusters of development, i.e., leapfrog followed by expansion and infill, were generally predicted to have the highest predicted fire risk to the largest proportion of structures in the study area. Nevertheless, rankings of scenarios were affected by the definition of risk.

Theoretically, it makes sense that leapfrog development produced fragmented development with larger numbers of small patches, lower housing density, and a larger housing footprint; and that infill resulted in the opposite, with expansion in the middle. By definition, leapfrog development requires open space around all sides of the newly developed parcel, whereas infill requires development on all sides, and expansion requires development on one side and open space on another. Implementing these planning policies on real landscapes, however, can be complex if there are more houses to build than there are parcels that meet the definitions of the three planning rules, and thus not all development conforms strictly to the policy [54]. In our simulations, parcels meeting the definition of each growth type had a higher probability of subdividing; yet, as we were simulating a real landscape, many newly developed parcels did not meet the scenario criteria. That the three scenarios nevertheless produced substantial differences in landscape-level development patterns shows that decision-making at the individual level can lead to meaningful broad-scale effects.

The objective of the econometric model was to provide a baseline probability to predict which parcels were most likely to subdivide; thus, the econometric model itself provides no explanation of how a given policy affects likelihood of subdivision, although it does indicate the correlation between the policy and the outcome. In our setting, which areas are protected, marked for redevelopment, or marked for development may be endogenous to the land owner decision to subdivide. In the case of these variables especially, our results should not be interpreted as causal predictors. Likewise, we use data only from 2005–2009 to predict changes to 2030. If major changes in the land market take place over this time horizon our model will not be able to take this into account.

Although some differences in predicted fire risk among the three scenarios likely stemmed from location of new structures relative to variables such as distance to coast, fuel type, or slope, the most important variables in the fire risk model were housing density and cluster size, with most structure loss historically occurring in areas with low housing density and in small, isolated housing clusters. Thus, leapfrog development was generally the riskiest scenario and infill the least risky. The most surprising result was the variation in predicted risk for the expansion scenario over time and at different thresholds. While leapfrog and infill showed similar trajectories across thresholds, expansion went from being the highest-risk scenario at the low threshold to being the lowest-risk scenario at the highest threshold. Because the threshold is merely a way to group structures into a binary classification, this means that, while the average risk calculated across all homes shows expansion to rank in the middle of infill and leapfrog throughout the simulation (Figure 5), the other two scenarios have a relatively larger proportion of homes that are modeled to be at a very high risk (i.e., 0.25 or 0.5), particularly by the end of the simulations. Because the total number of structures with a risk greater than 0.25 or 0.5 is relatively low in all scenarios, this difference in distribution of homes at the highest risk is not reflected in the mean. Another reason for the shift in rank of expansion over time is that, as more development occupied the landscape, there were fewer parcels remaining to accomplish infill or leapfrog type growth in the other scenarios. Thus, by the end of the simulations in year 2030, the majority of growth in all scenarios was expansion, and there was some convergence between scenarios. Finally, the change in risk of expansion growth over time may reflect that, despite the relatively low importance of distance to edge of cluster as an explanatory variable, expansion growth is characterized as having an initially fragmented landscape pattern that eventually merges into large patches with low edge.

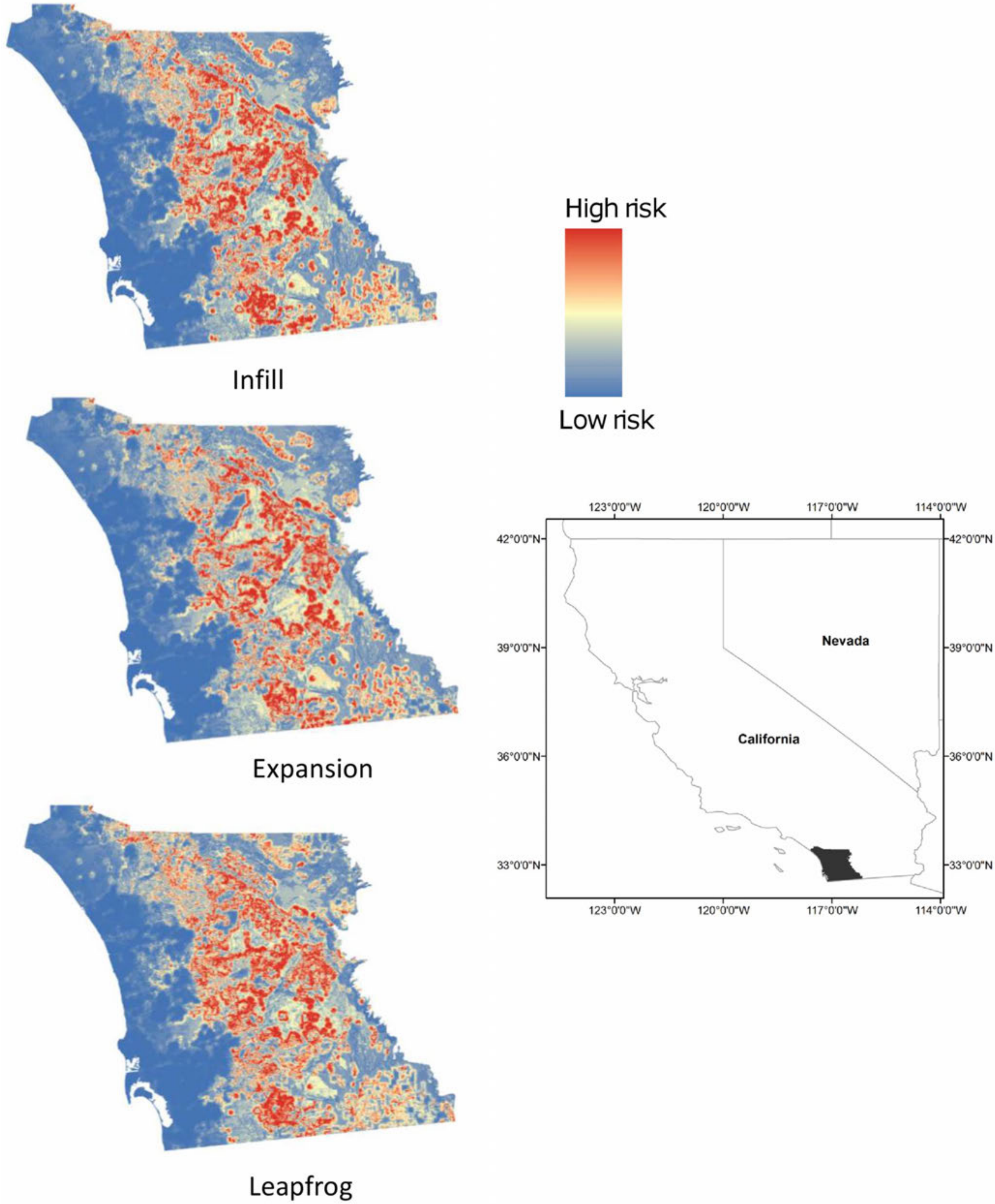

Figure 4. Maps of the study area showing projected wildfire risk at year 2030 for simulations of residential development under policies emphasizing infill, expansion, or leapfrog growth.
doi:10.1371/journal.pone.0071708.g004

Although leapfrog development clearly ranked highest in terms of fire risk, the interpretation of which planning policy is best may depend on fire management objectives and resources, as well as other considerations such as biodiversity or ecological impacts.

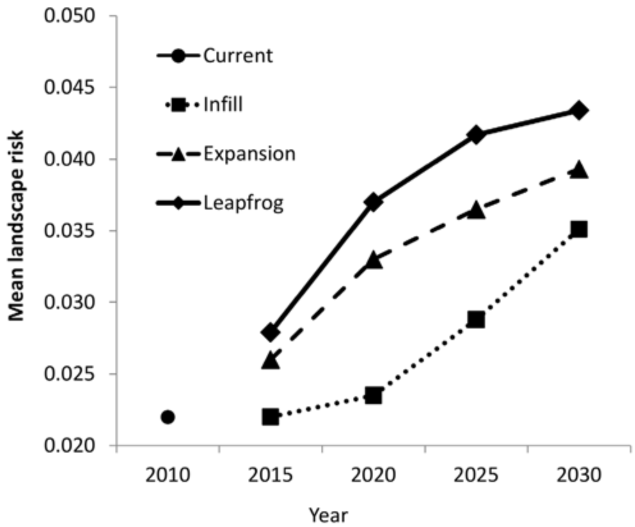

Figure 5. Projected landscape fire risk, reflecting the probability of burning in a wildfire averaged across all residential structures on the current landscape and in three development scenarios of infill, expansion, and leapfrog for year 2030.
doi:10.1371/journal.pone.0071708.g005

The spatial pattern of development affects multiple ecological functions and services [55], with potentially varying conservation implications; both leapfrog and expansion development consumed more land than infill, which would likely lead to more ecological degradation [56]; nevertheless, higher-density clustered development may be dominated by more invasive species [57]. Trade-offs between fire protection and conservation are common, but techniques are available for identifying mutually beneficial solutions [58].

Different perceptions of the fire risk results could also potentially translate into different planning priorities for management. For example, if the priority is to plan for the lowest overall risk to structures, then the mean landscape risk clearly delineates the rankings of options, with infill being the winner. However, if the objective is to reduce the number of structures at the highest risk threshold, i.e., $>=0.5$, then expansion is the best option, at least by 2030. An important consideration for fire management is the total area that needs to be protected, as well as the length of wildland-urban interface [8,13]. Therefore, despite the lower number of structures at the highest risk thresholds, expansion creates more edge than infill and may translate into greater challenges for firefighter protection.

Although we did not create separate scenarios for high or low growth, the results at different time steps can be substituted to envision the potential outcome of developing more or fewer houses. In the short term, the total fire risk is projected to increase proportionately as more land is developed. However, given the inverse relationship between housing density and fire risk, it is possible that this trend could reverse if housing growth eventually resulted in expansive high-density development.

Land use planning is one of a range of options available for reducing fire risk, and the best outcome will likely be achieved through a combination of strategies that include homeowner actions, improvements in fire-safe building codes, and advanced fire suppression tactics. Although we isolated the effect of land use planning policy in the three development scenarios, the fire risk model nevertheless showed that the pattern and location of structures in this study area were the most important out of a suite of factors influencing structure loss. We used a correlative approach that did not incorporate mechanisms or feedbacks, but our models clearly illustrated differences in the cumulative effects of individual planning decisions. The relationship between spatial pattern of development and fire risk is likely related to the intermixing of development and wildland vegetation [29,59]; thus, these results likely apply to a wide range of fire-prone ecosystems with large proportions of human-caused ignitions. Nevertheless, because fire risk is highly variable over space and time, and due to a range of human and biophysical variables [60], we recommend planners develop their own models for the best understanding of where the most fire-prone areas are in their region [19].

With projections of substantial global change in climate and human development, we recommend that land use planning should be considered as an important component to fire risk management, potentially to become as successful as the prevention of building on flood plains [61]. History has shown us that preventing fires is impossible in areas where large wildfires are a natural ecological process [4,9]. As Roger Kennedy put it, "the

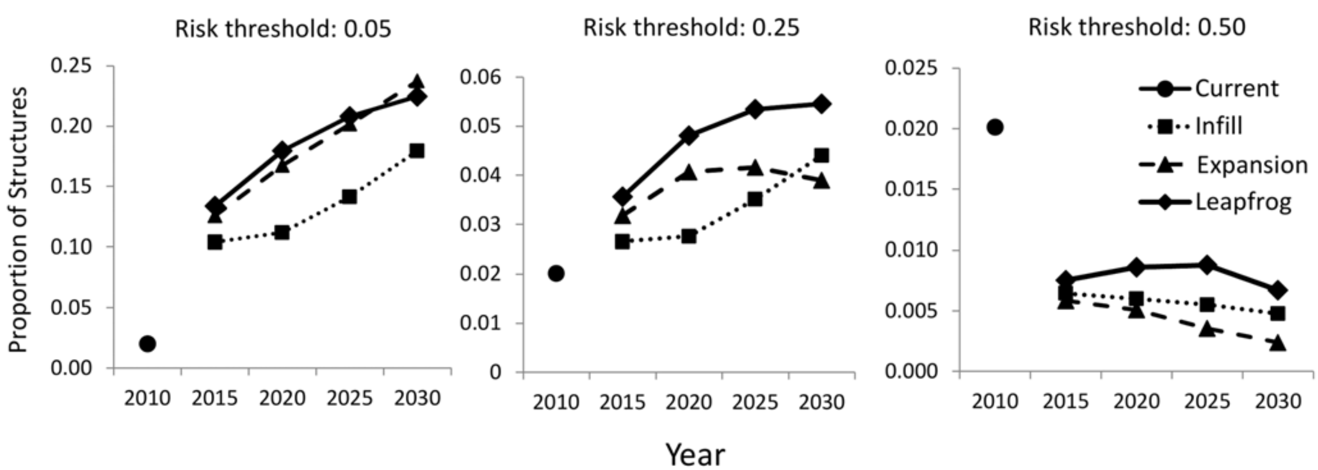

Figure 6. Proportion of residential structures that are located in areas of high fire risk defined using thresholds from the fire risk model of 0.05, 0.25, and 0.5 for current structures and for structures simulated under infill, expansion, and leapfrog growth policies.
doi:10.1371/journal.pone.0071708.g006

problem isn't fires; the problem is people in the wrong places [62]."

Supporting Information

Table S1 Definitions and summary statistics for variables used in the probit model.
(DOCX)

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Author Contributions

Conceived and designed the experiments: ADS JEK ABM VB. Performed the experiments: ADS ABM VB. Analyzed the data: ADS ABM VB. Wrote the paper: ADS JEK ABM VB.

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The role of defensible space for residential structure protection during wildfires

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Abstract. With the potential for worsening fire conditions, discussion is escalating over how to best reduce effects on urban communities. A widely supported strategy is the creation of defensible space immediately surrounding homes and other structures. Although state and local governments publish specific guidelines and requirements, there is little empirical evidence to suggest how much vegetation modification is needed to provide significant benefits. We analysed the role of defensible space by mapping and measuring a suite of variables on modern pre-fire aerial photography for 1000 destroyed and 1000 surviving structures for all fires where homes burned from 2001 to 2010 in San Diego County, CA, USA. Structures were more likely to survive a fire with defensible space immediately adjacent to them. The most effective treatment distance varied between 5 and 20 m (16–58 ft) from the structure, but distances larger than 30 m (100 ft) did not provide additional protection, even for structures located on steep slopes. The most effective actions were reducing woody cover up to 40% immediately adjacent to structures and ensuring that vegetation does not overhang or touch the structure. Multiple-regression models showed landscape-scale factors, including low housing density and distances to major roads, were more important in explaining structure destruction. The best long-term solution will involve a suite of prevention measures that include defensible space as well as building design approach, community education and proactive land use planning that limits exposure to fire.

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Introduction

Across the globe and over recent decades, homes have been destroyed in wildfires at an unprecedented rate. In the last decade, large wildfires across Australia, southern Europe, Russia, the US and Canada have resulted in tens of thousands of properties destroyed, in addition to lost lives and enormous social, economic and ecological effects (Filmon 2004; Boschetti *et al.* 2008; Keeley *et al.* 2009; Blanchi *et al.* 2010; Vasquez 2011). The potential for climate change to worsen fire conditions (Hessl 2011), and the projection of continued housing growth in fire-prone wildlands (Gude *et al.* 2008) suggest that many more communities will face the threat of catastrophic wildfire in the future.

Concern over increasing fire threat has escalated discussion over how to best prepare for wildfires and reduce their effects. Although ideas such as greater focus on fire hazard in land use planning, using fire-resistant building materials and reducing human-caused ignitions (e.g. Cary *et al.* 2009; Quarles *et al.* 2010; Syphard *et al.* 2012) are gaining traction, the traditional strategy of fuels management continues to receive the most attention. Fuels management in the form of prescribed fires or mechanical treatments has historically occurred in remote, wildland locations (Schoennagel *et al.* 2009), but recent studies suggest that treatments located closer to homes and communities may provide greater protection (Witter and Taylor 2005; Stockmann *et al.* 2010; Gibbons *et al.* 2012). In fact, one of the most commonly recommended strategies in terms of fuels and fire protection is to create defensible space immediately around structures (Cohen 2000; Winter *et al.* 2009). Defensible space is an area around a structure where vegetation has been modified, or 'cleared,' to increase the chance of the structure surviving a wildfire. The idea is to mitigate home loss by minimising direct contact with fire, reducing radiative heating, lowering the probability of ignitions from embers and providing a safer place for fire fighters to defend a structure against fire (Gill and Stephens 2009; Cheney *et al.* 2001). Many jurisdictions provide specific guidelines and practices for creating defensible space, including minimum distances that are required among trees and shrubs as well as minimum total distances from the structure. These distances may be enforced through local ordinances or state-wide laws. In California, for example, a state law in 2005 increased the required total distance from 9 m (30 ft) to 30 m (100 ft).

Despite these specific guidelines on how to create defensible space, there is little scientific evidence to support the amount and location of vegetation modification that is actually effective

at providing significant benefits. Most spacing guidelines and laws are based on 'expert opinion' or recommendations from older publications that lack scientific reference or rationale (e.g. Maire 1979; Smith and Adams 1991; Gilmer 1994). However, one study has provided scientific support for, and forms the basis of, most guidelines, policy and laws requiring a minimum of 30 m (100 ft) of defensible space (Cohen 1999, 2000). The modelling and experimental research in that study showed that flames from forest fires located 10–40 m (33–131 ft) away would not scorch or ignite a wooden home; and case studies showed 90% of homes with non-flammable roofs and vegetation clearance of 10–20 m (33–66 ft) could survive wildfires (Cohen 2000). However, the models and experimental research in that study focussed on crown fires in spruce or jack pine forests, and the primary material of home construction was wood. Therefore, it is unknown how well this guideline applies to regions dominated by other forest types, grasslands, or nonforested woody shrublands and in regions where wooden houses are not the norm.

Some older case studies showed that most homes with non-flammable roofs and 10–18 m (33–ft) of defensible space survived the 1961 Bel Air fire in California (Howard *et al.* 1973); most homes with non-flammable roofs and more than 10 m (33 ft) of defensible space also survived the 1990 Painted Cave fire (Foote and Gilless 1996). Also, several fire-behaviour modelling studies have been conducted in chaparral shrublands. One study showed that reducing vegetative cover to 50% at 9–30 m (30–ft) from structures effectively reduced fireline intensity and flame lengths, and that removal of 80% cover would result in unintended consequences such as exotic grass invasion, loss of habitat and increase in highly flammable flashy fuels (A. Fege and D. Pumphrey, unpubl. data). Another showed that separation distances adequate to protect firefighters varied according to fuel model and that wind speeds greater than 23 km h^{-1} negated the effect of slope, and wind speed above 48 km h^{-1} negated any protective effect of defensible space (F. Bilz, E. McCormick and R. Unkovich, unpubl. data, 2009). Results obtained through modelling equations of thermal radiation also found safety distances to vary as a function of fuel type, type of fire, home construction material and protective garments worn by firefighters (Zárate *et al.* 2008).

Although there is no empirical evidence to support the need for more than 30 m (100 ft) of defensible space, there has been a concerted effort in some areas to increase this distance, particularly on steep slopes. In California, a senate bill was introduced in 2008 (SB 1618) to encourage property owners to clear 91 m (300 ft) through the reduction of environmental regulations and permitting needed at that distance. Although this bill was defeated in committee, many local ordinances do require homeowners to clear 91 m (300 ft) or more, and there are reports that some people are unable to get fire insurance without 91 m (300 ft) of defensible space (F. Sproul, pers. comm.). In contrast, homeowner acceptance of and compliance with defensible space policies can be challenging (Winter *et al.* 2009; Absher and Vaske 2011), and in many cases homeowners do not create any defensible space.

It is critically important to develop empirical research that quantifies the amount, location and distance of defensible space that provides significant fire protection benefits so that guidelines and policies are developed with scientific support. Data that are directly applicable to southern California are especially important, as this region experiences the highest annual rate of wildfire-destroyed homes in the US. Not having sufficient defensible space is obviously undesirable because of the hazard to homeowners. However, there are clear trade-offs involved when vegetation reduction is excessive, as it results in the loss of native habitats, potential for increased erosion and invasive species establishment, and it potentially even increases fire risk because of the high flammability of weedy grasslands (Spittler 1995; Keeley *et al.* 2005; Syphard *et al.* 2006).

It is also important to understand the role of defensible space in residential structure protection relative to other factors that explain why some homes are destroyed in fires and some are not. Recent research shows that landscape-scale factors, such as housing arrangement and location, as well as biophysical variables characterising properties and neighbourhoods such as slope and fuel type, were important in explaining which homes burned in two southern California study areas (Syphard *et al.* 2012; 2013). Understanding the relative importance of different variables at different scales may help to identify which combinations of factors are most critical to consider for fire safety.

Our objective was to provide an empirical analysis of the role of defensible space in protecting structures during wildfires in southern California shrublands. Using recent pre-fire aerial photography, we mapped and measured a suite of variables describing defensible space for burned and unburned structures within the perimeters of major fires from 2001 to 2010 in San Diego County to ask the following questions:

1. How much defensible space is needed to provide significant protection to homes during wildfires, and is it beneficial to have more than the legally required 30 m (100 ft)?
2. Does the amount of defensible space needed for protection depend on slope inclination?
3. What is the role of defensible space relative to other factors that influence structure loss, such as terrain, fuel type and housing density?

Methods

Study area

The properties and structures analysed were located in San Diego County, California, USA (Fig. 1) – a topographically diverse region with a Mediterranean climate characterised by cool, wet winters and long summer droughts. Fire typically is a direct threat to structures adjacent to wildland areas. Native shrublands in southern California are extremely flammable during the late summer and fall (autumn) and when ignited, burn in high-intensity, stand-replacing crown fires. Although 500 homes on average have been lost annually since the mid-1900s (Calfire 2000), that rate has doubled since 2000. Most of these homes have burned during extreme fire weather conditions that accompany the autumn Santa Ana winds. The wildland–urban interface here includes more than 5 million homes, covering more than 28 000 km^2 (Hammer *et al.* 2007).

Property data

The data for properties to analyse came from a complete spatial database of existing residential structures and their

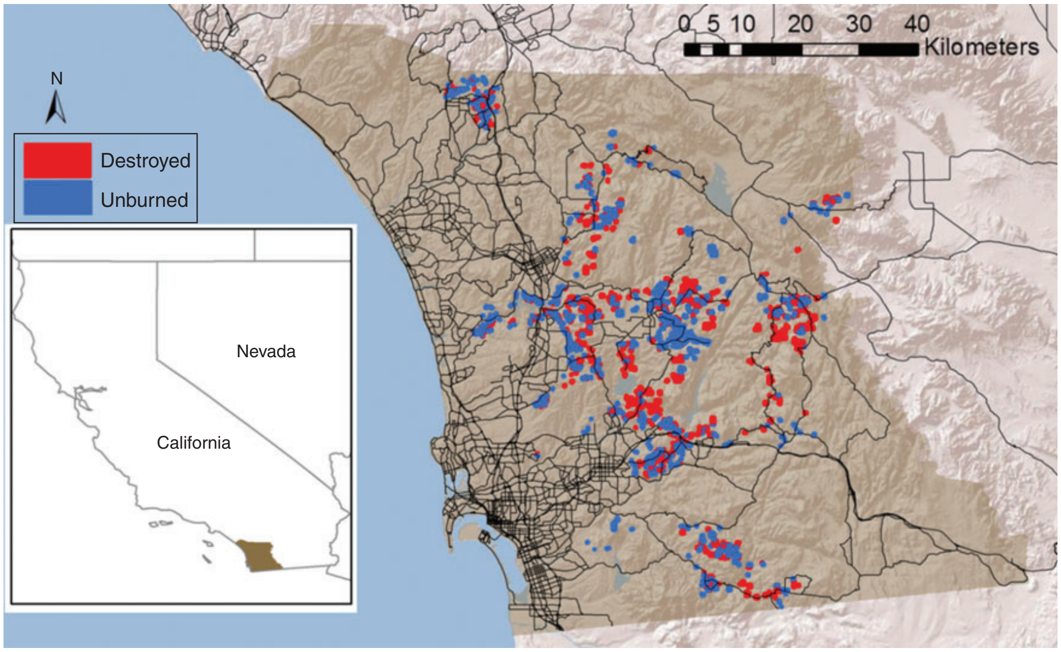

Fig. 1. Location of destroyed and unburned structures within the South Coast ecoregion of San Diego County, California, USA.

corresponding property boundaries developed for San Diego County (Syphard *et al.* 2012). This dataset included 687 869 structures, of which 4315 were completely destroyed by one of 40 major fires that occurred from 2001 to 2010. Our goal was to compare homes that were exposed to wildfire and survived with those that were exposed and destroyed. To determine exposure to fire, we only considered structures located both within a GIS layer of fire perimeters and within areas mapped as having burned at a minimum of low severity through thematic Monitoring Trends in Burn Severity produced by the USA Geological Survey and USDA Forest Service. From these data, we used a random sample algorithm in GIS software to select 1000 destroyed and 1000 unburned homes that were not adjacent to each other, to minimise any potential for spatial autocorrelation. Our final property dataset included structures that burned across eight different fires. More than 97% of these structures burned in Santa Ana wind-driven fire events (Fig. 1).

Calculating defensible space and additional explanatory variables

To estimate defensible space, we developed and explored a suite of variables relative to the distance and amount of defensible space surrounding structures, as well as the proximity of woody vegetation to the structure (Table 1). We measured these variables based on interpretation of Google Earth aerial imagery. We based our measurements on the most recent imagery before the date of the fire. In almost all cases, imagery was available for less than 1 year before the fire.

Our definition of defensible space followed the guidelines published by the California Department of Forestry and Fire Protection (Calfire 2006). 'Clearance' included all areas that were not covered by woody vegetation, including paved areas or grass. Although Google Earth prevents the identification of understorey vegetation, woody trees and shrubs were easily distinguished from grass, and our objective was to measure horizontal distances as required by Calfire rather than assess the relative flammability of different vegetation types. Trees or shrubs were allowed to be within the defensible space zone as long as they were separated by the minimum horizontal required distance, which was 3 m (10 ft) from the edge of one tree canopy to the edge of the next (Fig. 2). Although greater distances between trees or shrubs are recommended on steeper slopes, we followed the same guidelines for all properties. For all structures, we started the distance measurements by drawing lines from the centre of the four orthogonal sides of the structure that ended when they intersected anything that no longer met the requirements in the guidelines. A fair number of structures are not four sided; thus, the start of the centre point was placed at a location that approximated the farthest extent of the structure along each of four orthogonal sides.

We developed two sets of measurements of the distance of defensible space based on what is feasible for homeowners within their properties *v.* the total effective distance of defensible space. We made these two measurements because homeowners are only required to create defensible space within their own property, and this would reflect the effect of individual homeowner compliance. Therefore, even if cleared vegetation extended beyond the property line, the first set of distance measurements ended at the property boundary. The second set of measurements ignored the property boundaries and accounted for the total potential effect of treatment. For all measurements, we recorded the cover types (e.g. structure >3 m (10 ft) long, property boundary, or vegetation type) at which the distance measurements stopped (Table 1). Because property

Table 1. Defensible space variables measured for every structure

Urban veg, landscaping vegetation that was not in compliance with regulations within urban matrix; wildland veg, wildland vegetation that was not in compliance with regulations; orchard, shrub to tree-sized vegetation in rows; urban to wildland, landscaping vegetation that leads into wildland vegetation; structure, any building longer than 3 m (10 ft)

Variable	Definition
Distance defensible space within property	Measure of clearance from side of structure to property boundary calculated for four orthogonal directions from structure and averaged
Total distance defensible space	Measure of clearance from side of structure to end of clearance calculated for four orthogonal directions from structure and averaged
Cover type at end of defensible space	Type of cover encountered at end of measurement (urban veg, wildland veg, orchard, urban to wildland, structure)
Percentage clearance	Percentage of clearance calculated across the entire property
Neighbours' vegetation	Binary indicator of whether neighbours' uncleared vegetation was located within 30 m (100 ft) of the main structure
Vegetation touching structure	Number of sides on which woody vegetation touches main structure (1–4) Structure with more than 4 sides were viewed as a box and given a number between 1 and 4
Vegetation overhanging roof	Was vegetation overhanging the roof? (yes or no)

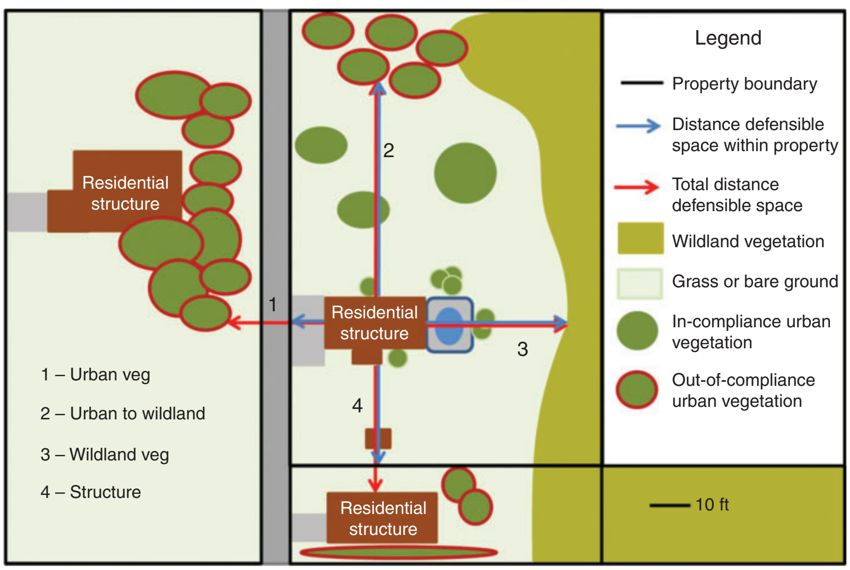

Fig. 2. Illustration of defensible space measurements. See Table 1 for full definition of terms.

owners usually can only clear vegetation on their own land, it is possible that the effectiveness of defensible space partly depends upon the actions of neighbouring homeowners. Therefore, we also recorded whether or not any neighbours' un-cleared vegetation was located within 30 m (100 ft) of the structure.

To assess the total amount of woody vegetation that can safely remain on a property and still receive significant benefits of defensible space, we calculated the total percentage of cleared land, woody vegetation and structure area across every property. This was accomplished by overlaying a grid on each property and determining the proportion of squares falling into each class. Preliminary results showed these three measurements to be highly correlated, so we only retained percentage clearance for further analysis. To evaluate the relative effect of woody vegetation directly adjacent to structures, we also calculated the number of sides of the structure with vegetation touching and recorded whether any trees were overhanging structures' roofs.

In addition to defensible space measurements, we evaluated other factors known to influence the likelihood of housing loss to fire in the region (Syphard *et al.* 2012, 2013). Using the same data as in Syphard *et al.* (2012, 2013), we extracted spatial information from continuous grids of explanatory variables for the locations of all structures in our analysis. Variables included interpolated housing density based on a 1-km search radius; percentage slope derived from a 30-m digital elevation model (DEM); Euclidean distance to nearest major and minor road and fuel type, which was based on a simple classification of US Forest Service data (Syphard *et al.* 2012), including urban, grass, shrubland and forest & woodland.

Analysis

We performed several analyses to determine whether relative differences in home protection are provided by different distances and amounts of defensible space, particularly beyond the legally required 30 m (100 ft), and to identify the effective treatment distance for homes on low and steep slopes.

Categorical analysis

For the first analysis, we divided our data into several groups to identify potential differences among specific categories of defensible space distance around structures located on shallow and steep slopes. We first sorted the full dataset of 2000 structures by slope and then split the data in the middle to create groups of homes with shallow slope and steep slope. We divided the data in half to keep the number of structures even within both groups and to avoid specifying an arbitrary number to define what constitutes shallow or steep slope. The two equal-sized subsets of data ranged from 0 to 9%, with a mean of 8% for shallow slope, and from 9 to 40%, with a mean of 27% for steep slope. Within these data subsets, we next created groups reflecting different mean distances of defensible space around structures. We also performed separate analyses based on whether defensible space measurements were calculated within the property boundary or whether measurements accounted for the total distance of defensible space.

Within all groups, we calculated the proportion of homes that were destroyed by wildfire. We performed Pearson's Chi-square tests of independence to determine whether or not the proportion of destroyed structures within groups was significantly different (Agresti 2007). We based one test on four equal-interval groups within the legally required distance of 30 m (100 ft): 0–7 m (0–25 ft), 8–15 m (26–50 ft), 16–23 m (51–75 ft) and 24–30 m (76–100 ft). A second test was based on three groups (24–30 m (75–100 ft), 31–90 m (101–300 ft) and >90 m (>300 ft) or >60 m (>200 ft)) to evaluate whether groups with mean defensible space distances >30 m (>100 ft) were significantly different from groups with <30 m (<100 ft). When defensible space distances were only measured to the property boundary, few structures had mean defensible space >90 m (>300 ft). Therefore, we used a cut-off of 60 m (200 ft) to increase the sample size in the Chi-square analysis. In addition to the Chi-square analysis, we calculated the relative risk among every successive pair of categories (Sheskin 2004). The relative risk was calculated as the ratio of proportions of burned homes within two groups of homes that had different defensible space distances.

Effective treatment analysis

In addition to comparing the relative effect of defensible space among different groups of mean distances, as described above, we also considered that the protective effect of defensible space for structures exposed to wildfire is conceptually similar to the effect of medication in producing a therapeutic response in people who are sick. In addition to pharmacological applications, treatment–response relationships have been used for radiation, herbicide, drought tolerance and ecotoxicological studies (e.g. Streibig *et al.* 1993; Cedergreen *et al.* 2005; Knezevic *et al.* 2007; Kursar *et al.* 2009). The effect produced by a drug or treatment typically varies according to the concentration or amount, often up to a point at which further increase provides no additional response. The effective treatment (ET50), therefore, is a specific concentration or exposure that produces a therapeutic response or desired effect. Here we considered the treatment to be the distance or amount of defensible space.

Using the software package DRC in R (Knezevic *et al.* 2007; Ritz and Streibig 2013), we evaluated the treatment–response relationship of defensible space in survival of structures during wildfire. To calculate the effective treatment, we fit a log-logistic model with logistic regression because we had a binary dependent variable (burned or unburned). We specified a 2-parameter model where the lower limit was fixed at 0 and the upper limit was fixed at 1. We again performed separate analyses for data subsets reflecting shallow and steep slope, as well as from measurements of defensible space taken within, or regardless of, property boundaries. We also performed analyses to find the effective treatment of percentage clearance of trees and shrubs within the property.

Multiple regression analysis

To evaluate the role of defensible space relative to other variables, we developed multiple generalised linear regression models (GLMs) (Venables and Ripley 1994). We again had a binary dependent variable (burned versus unburned), so we specified a logit link and binomial response. Although the proportion of 0s and 1s in the response may be important to consider for true prediction (King and Zeng 2001; Syphard *et al.* 2008), our objective here was solely to evaluate variable importance. We developed multiple regression models for all possible combinations of the predictor variables and used the corrected Akaike's Information Criterion (AICc) to rank models and select the best ones for each region using package MuMIn in R (R Development Core Team 2012; Burnham and Anderson 2002). We recorded all top-ranked models that had an AICc value within 2 of that of the model with lowest AICc to identify all models with empirical support. To assess variable importance, we calculated the sum of Akaike weights for all models that contained each variable. On a scale of 0–1, this metric represents the weight of evidence that models containing the variable in question are the best model (Burnham and Anderson 2002). The distance of defensible space measured within property boundaries was highly correlated with the distance of defensible space measured beyond property boundaries ($r = 0.82$), so we developed two separate analyses – one using variables measured only within the property boundary and the other using variables that accounted for defensible space outside of the property boundary as well as the potential effect of neighbours having uncleared vegetation within 30 m (100 ft) of the structure. A test to avoid multicollinearity showed all other variables within each multiple regression analysis to be uncorrelated ($r < 0.5$).

Surrounding matrix

To assess whether the proportion of destroyed structures varied according to their surrounding matrix, we summarised the most common cover type at the end of defensible space measurements (descriptions in Table 1) for all structures. These summaries

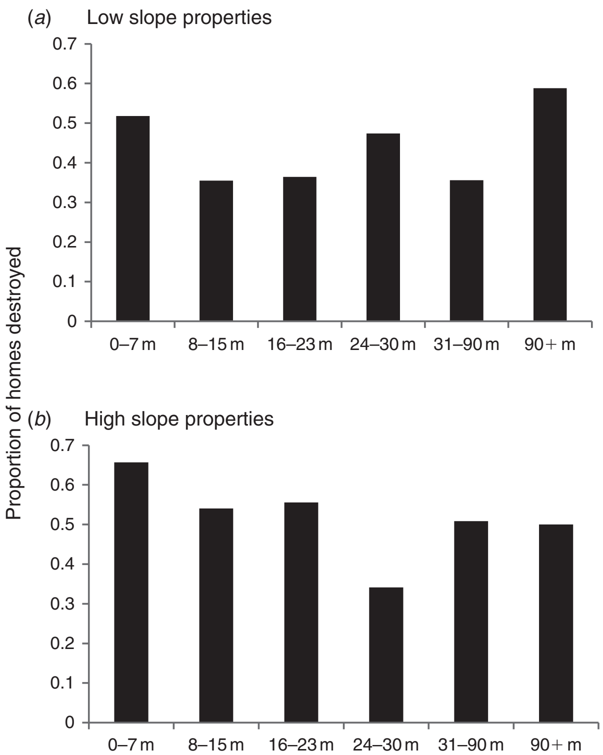

Fig. 3. Proportion of destroyed homes grouped by distances of defensible space based upon total distance of clearance within property boundary, for structures on (*a*) shallow slopes (mean 8%) and (*b*) steep slopes (mean 27%).

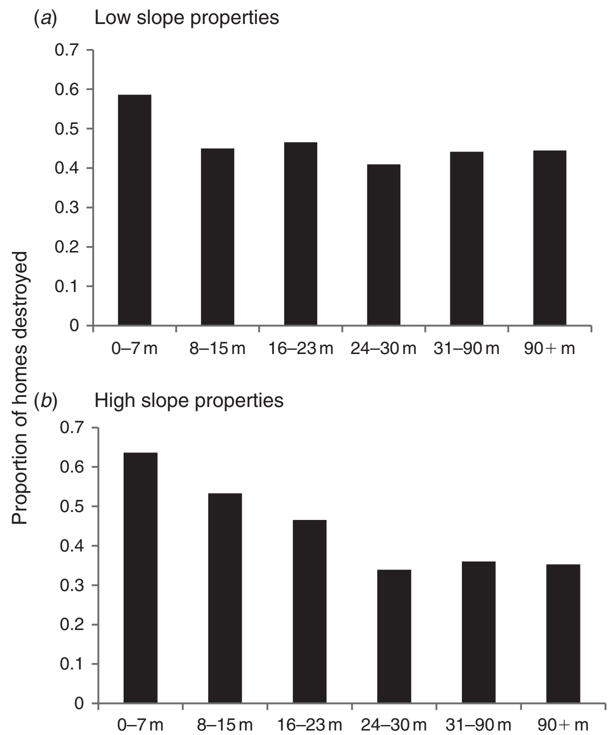

Fig. 4. Proportion of destroyed homes grouped by distances of defensible space based upon total distance of clearance regardless of property boundary, for structures on (*a*) shallow slopes (mean 8%) and (*b*) steep slopes (mean 27%).

were based on the majority surrounding cover type from the four orthogonal sides of the structure. We also noted cases in which there was a tie (e.g. two sides were urban vegetation and two sides were structures).

Results

Categorical analysis

When the distance of defensible space was measured both 'only within property boundaries' (Fig. 3) and 'regardless of property boundaries' (Fig. 4), the Chi-square test showed a significant difference ($P < 0.001$) in the proportion of destroyed structures among the four equal-interval groups of distance ranging from 0 to 30 m (0–100 ft). This relationship was consistent on both shallow-slope and steep-slope properties, although the relative risk analysis showed considerable variation among classes (Table 2) There was a steadily decreasing proportion of destroyed structures at greater distances of defensible space up to 30 m (100 ft) on the steep-slope structures with defensible space measured regardless of property boundaries (Fig. 4*b*). Otherwise, the biggest difference in proportion of destroyed structures occurred between 0 and 7 m (0–25 ft) and 8–15 m (26–50 ft) (Figs 3*a–b*, 4*a*).

When the distance of defensible space was measured in intervals from 24 m (75 ft) and beyond, the Chi-square test showed no significant difference among groups ($P = 0.96$ for shallow-slope properties and $P = 0.74$ for steep-slope properties) (Figs 3, 4), although again, the relative risk analysis showed considerable variation (Table 2).There was a slight increase in the proportion of homes destroyed at longer distance intervals when the defensible space was measured only to the property boundaries (Fig. 3*a–b*). This slight increase is less apparent when distances were measured regardless of boundaries (Fig. 4*a–b*).

The relative risk calculations showed that the ratio of proportions was generally more variable among successive pairs when the distances were measured within property boundaries (Table 2). For these calculations, the risk of a structure being destroyed was significantly lower when the defensible space distance was 8–15 m (25–50 ft) compared to 0–7 m (0–25 ft) on both shallow- and steep-slope properties. On the steep-slope properties, there was an additional reduction of risk when comparing 24–30 m (75–100 ft) to 16–23 m (50–75 ft). However, the risk of a home being destroyed was slightly significantly higher when there was 31–90 m (101–225 ft) compared to 16–23 m (50–75 ft). For distances that were measured regardless of property boundary (total clearance), the only significant differences in risk of burning were a reduction in risk for 8–15 m (25–50 ft) compared to 0–7 m (0–25 ft).

Table 2. Number of burned and unburned structures within defensible space distance categories (m), their relative risk and significance
A relative risk of 1 indicates no difference; <1 means the chance of a structure burning is less than the other group; >1 means the chance is higher than the other group. The relative risk is calculated for pairs that include the existing row and the row above. Confidence intervals are in parentheses

	Distance within property				Total distance			
	Burned	Unburned	Relative risk	*P*	Burned	Unburned	Relative risk	*P*
Shallow slope								
0–7	200	186			162	114		
8–15	109	198	0.69 (0.12)	<0.001	108	132	0.77	0.002
16–23	51	89	1.03 (0.30)	0.850	78	90	1.03	0.770
24–30	36	40	1.30 (0.39)	0.110	50	70	0.90	0.430
31–90	28	47	0.79 (0.24)	0.220	79	99	1.06	0.640
60 or 90+	10	6	1.67 (0.63)	0.040	8	9	1.01	0.830
Steep slope								
0–7	245	128			224	128		
8–15	174	148	0.82 (0.10)	0.001	158	139	0.84	0.008
16–23	85	68	1.03 (0.16)	0.750	73	83	0.87	0.210
24–30	29	56	0.61 (0.17)	0.004	26	50	0.73	0.080
31–	29	28	1.49 (0.48)	0.050	39	68	1.06	0.760
60 or 90+	5	5	0.98 (0.47)	0.950	4	8	0.91	0.830

Table 3. Effective treatment results reflecting the distance (in metres, with feet in parentheses) and percentage clearance within properties that provided significant improvement in structure survival during wildfires
The property mean is the average distance of defensible space or percentage clearance that was calculated on the properties before the wildfires and provides a means to compare the effective treatment result to the actual amount on the properties

	All parcels effective treatment ($n = 2000$)	Parcel mean	Shallow slope (mean 8%) effective treatment ($n = 1000$)	Parcel mean	Steep slope (mean 27%) effective treatment ($n = 1000$)	Parcel mean
Defensible space within parcel	10 (33)	13 (44)	4 (13)	14 (45)	25 (82)	11 (35)
Total distance defensible space	10 (32)	19 (63)	5 (16)	20 (67)	20 (65)	18 (58)
Mean percentage clearance on property	36	48	31	51	37	35

Effective treatment analysis

Analysis of the treatment–response relationships among defensible space and structures that survived wildfire showed that, when all structures are considered together, the mean actual defensible space that existed around structures before the fires was longer than the calculated effective treatment (Table 3). Regardless of whether the defensible space was measured within or beyond property boundaries, the estimated effective treatment of defensible space was nearly the same at 10 m (32–33 ft).

The effective treatment distance was much shorter for structures on shallow slopes (4–5 m (13–16 ft)) than for structures on steep slopes (20–25 m (65–82 ft)), but in all cases was <30 m (<100 ft). Although longer distances of defensible space were calculated as effective on steeper slopes, these structures actually had shorter mean distances of defensible space around their properties than structures on low slopes (Table 3).

The calculated effective treatment of the mean percentage clearance on properties was 36% for all properties, 31% for structures on shallow slopes and 37% for structures on steep slopes (Table 3). In total, the properties all had higher actual percentage clearance on their property than was calculated to be effective. However, this mainly reflects the shallow-slope properties, as those structures on steep slopes had less clearance than the effective treatment.

Multiple regression analysis

When defensible space was measured only to the property boundaries, it was not included in the best model, according to the all-subsets multiple regression analysis (Table 4). However, it was included in the best model when factoring in the distance of defensible space measured beyond property boundaries (Table 5). In both multiple regression analyses, low housing density and shorter distances to major roads were ranked as the most important variables according to their Akaike weights. Slope and surrounding fuel type were also in both of the best models as well as other measures of defensible space, including the percentage clearance on property and whether vegetation was overhanging the structure's roof. The number of sides in which vegetation was touching the structure was included in the best model when defensible space was only measured to the property boundary. The total explained deviance for the multiple regression models was low (12–13%) for both analyses.

Table 4. Results of multiple regression models of destroyed homes using all possible variable combinations and corrected Akaike's Information Criterion (AICc)

Includes variables measured within property boundary only. Top-ranked models include all those ($n = 12$) with AICc within 2 of the model with the lowest AICc. Relative variable importance is the sum of 'Akaike weights' over all models including the explanatory variable

Variable in order of importance	Relative variable importance	Model-averaged coefficient	Number inclusions in top-ranked models
Housing density	1	−0.003	12
Distance to major road	1	−0.0005	12
Percentage clearance	1	−0.02	12
Slope	1	0.03	12
Vegetation overhang roof	1	0.5	12
Fuel type	0.67	Factor	9
Vegetation touch structure	0.49	0.07	6
Distance defensible space within property	0.45	−0.0002	5
South-westness	0.36	−0.0007	3
Distance to minor road	0.28	−0.0002	1
D^2 of top-ranked model			0.123

Table 5. Results of multiple regression models of destroyed homes using all possible variable combinations and corrected Akaike's Information Criterion (AICc)

Includes variables measured beyond property boundary. Top-ranked models include all those ($n = 6$) with AICc within 2 of the model with the lowest AICc. Relative variable importance is the sum of 'Akaike weights' over all models including the explanatory variable

Variable in order of importance	Relative variable importance	Model-averaged coefficient	Number inclusions in top-ranked models
Housing density	1	−0.003	6
Distance to major road	1	−0.0005	6
Total distance defensible space	1	−0.004	6
Percentage clearance	1	−0.01	6
Vegetation overhang roof	0.99	0.4	6
Slope	0.99	0.03	6
Fuel type	0.86	Factor	4
South-westness	0.42	−0.0009	2
Distance to minor road	0.36	−0.0009	2
Neighbours' vegetation	0.27	0.08	1
Vegetation touch structure	0.27	0.18	1
D^2 of top-ranked model			0.125

Surrounding matrix

The cover type that most frequently surrounded the structures at the end of the defensible space measurements was urban vegetation, followed by urban vegetation leading into wildland vegetation, and wildland vegetation (Fig. 5). Many structures were equally surrounded by different cover types. There were no significant differences in the proportion of structures destroyed depending on the surrounding cover type. However, a disproportionately large proportion of structures burned (28 *v.* 9% unburned) when they were surrounded by urban vegetation that extended straight into wildland vegetation.

Discussion

For homes that burned in southern Californian urban areas adjacent to non-forested ecosystems, most burned in high-intensity Santa Ana wind-driven wildfires and defensible space increased the likelihood of structure survival during wildfire. The most effective treatment distance varied between 5 and 20 m (16–58 ft), depending on slope and how the defensible space was measured, but distances longer than 30 m (100 ft) provided no significant additional benefit. Structures on steeper slopes benefited from more defensible space than structures on shallow slopes, but the effective treatment was still less than 30 m (100 ft). The steepest overall decline in destroyed structures occurred when mean defensible space increased from 0–7 m (0–25 ft) to 8–15 m (26–50 ft). That, along with the multiple regression results showing the significance of vegetation touching or overhanging the structure, suggests it is most critical to modify vegetation immediately adjacent to the house, and to move outward from there. Similarly, vegetation overhanging the structure was also strongly correlated with structure loss in Australia (Leonard *et al.* 2009).

In terms of fuel modification, the multiple regression models also showed that the percentage of clearance was just as, or more important than, the linear distance of defensible space.

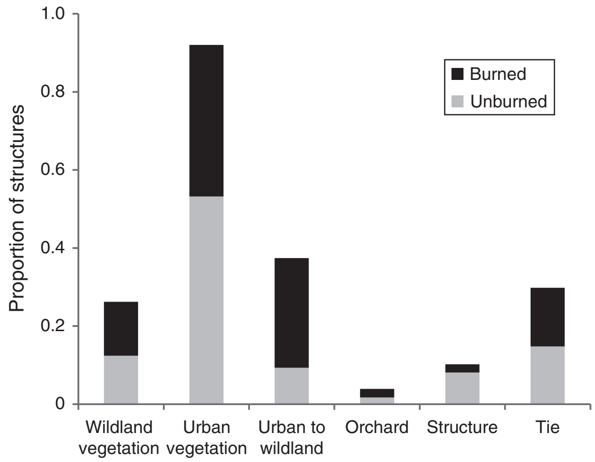

Fig. 5. Proportion of destroyed and unburned structures based on the primary surrounding cover type at the end of defensible space measurements. There were no significant differences in the proportion of burned and unburned structures within cover types ($P = 0.14$). Cover types are defined in Table 1.

However, as with defensible space, percentage clearance did not need to be draconian to be effective. Even on steep slopes, the effective percentage clearance needed on the property was <40%, with no significant advantage beyond that. Although these steep-slope structures benefited more from clearance, they tended to have less clearance than the effective amount, which may be why slope was such an important variable in the multiple regression models. Shallow-slope structures, in contrast, had more clearance on average than was calculated to be effective, suggesting these property owners do not need to modify their behaviours as much relative to people living on steep slopes.

Although the term 'clearance' is often used interchangeably with defensible space, this term is incorrect when misinterpreted to mean clearing all vegetation, and our results underline this difference. The idea behind defensible space is to reduce the continuity of fuels through maintenance of certain distances among trees and shrubs. Although we could not identify the vertical profile of fuels through Google Earth imagery, the fact that at least 60% of the horizontal woody vegetative cover can remain on the property with significant protective effects demonstrates the importance of distinguishing defensible space from complete vegetation removal. Thus, we suggest the term 'clearance' be replaced with 'fuel treatment' as a better way of communicating fire hazard reduction needs to home owners.

The percentage cover of woody shrubs and trees was not evenly distributed across properties, and we did not collect data describing how the cover was distributed. Considering the importance of defensible space and vegetation modification immediately adjacent to the structure, it should follow that actions to reduce cover should also be focussed in close proximity to the structure. The hazard of vegetation near the structure has apparently been recognised for some time (Foote *et al.* 1991; Ramsey and McArthur 1994), but it is not stressed enough, and rarely falls within the scope of defensible space guidelines or ordinances.

In addition to the importance of vegetation overhanging or touching the structure, it is important to understand that ornamental vegetation may be just as, if not more, dangerous than native vegetation in southern California. Although the results showed no significant differences in the cover types in the surrounding matrix, there was a disproportionately large number of structures destroyed (28% burned *v.* 9% unburned) when ornamental vegetation on the property led directly into the wildland. Ornamental vegetation may produce highly flammable litter (Ganteaume *et al.* 2013) or may be particularly dangerous after a drought when it is dry, or has not been maintained, and species of conifer, juniper, cypress, eucalypt, *Acacia* and palm have been present in the properties of many structures that have been destroyed (Franklin 1996). Nevertheless, ornamental vegetation is allowed to be included as defensible space in many codes and ordinances (Haines *et al.* 2008).

One reason that longer defensible space distances did not significantly increase structure protection may be that most homes are not destroyed by the direct ignition of the fire front but rather due to ember-ignited spot fires, sometimes from fire brands carried as far as several km away. Although embers decay with distance, the difference between 30 and 90 m (100 and 300 ft) may be small relative to the distance embers travel under the severe wind conditions that were present at the time of the fires. The ignitability of whatever the embers land on, particularly adjacent to the house, is therefore most critical for propagating the fire within the property or igniting the home (Cohen 1999; Maranghides and Mell 2009).

Aside from roofing or home construction materials and vegetation immediately adjacent to structures (Quarles *et al.* 2010; Keeley *et al.* 2013), the flammability of the vegetation in the property may also play a role. Large, cleared swaths of land are likely occupied at least in part by exotic annual grasses that are highly ignitable for much of the year. Conversion of woody shrubs with higher moisture content into low-fuel-volume grasslands could potentially increase fire risk in some situations by increasing the ignitability of the fuel; and if the vegetation between a structure and a fire is not readily combustible, it could protect the structure by absorbing heat flux and filtering fire brands (Wilson and Ferguson 1986).

The slight increase in proportion of structures destroyed with longer distances of defensible space within parcel boundaries was surprising. However, that increase was not significant in the Chi-square analysis, although there were some significant differences in the pairwise relative risk analysis. Nevertheless, the largest significant effect of defensible space was between the categories of 0–7 m (0–25 ft) to 8–15 m (26–50 ft), and it may be that differences in categories beyond these distances are not highly meaningful or reflect an artefact of the definition of distance categories. These relationships at longer distances are likely also weak compared to the effect of other variables operating at a landscape scale. Although the categorical analysis allowed us to answer questions relative to legal requirements and specific distances, the effective treatment analysis was important for identifying thresholds in the continuous variable.

The multiple regression models showed that landscape factors such as low housing density and longer distances to major roads were more important than distance of defensible space for explaining structure destruction, and the importance of

these variables is consistent with previous studies (Syphard *et al.* 2012, 2013), despite the smaller spatial extent studied here. Whereas this study used an unburned control group exposed to the same fires as the destroyed structures, previous studies accounted for structures across entire landscapes. The likelihood of a fire destroying a home is actually a result of two major components: the first is the likelihood that there will be a fire, and the second is the likelihood that a structure will burn in that fire. In this study, we only focussed on structure loss given the presence of a fire, and the total explained variation for the multiple regression models was quite low at ~12%. However, when the entire landscape was accounted for in the total likelihood of structure destruction, the explained variation of housing density alone was >30% (Syphard *et al.* 2012). One reason for the relationship between low housing density and structure destruction is that structures are embedded within a matrix of wildland fuel that leads to greater overall exposure, which is consistent with Australian research that showed a linear decrease of structure loss with increased distance to forest (Chen and McAneney 2004). That research, however, only focussed on distance to wildland boundaries and did not quantify variability in defensible space or ornamental vegetation immediately surrounding structures. Thus, fire safety is important to consider at multiple scales and for multiple variables, which will ultimately require the cooperation of multiple stakeholders.

Conclusions

Structure loss to wildfire is clearly a complicated function of many biophysical, human and spatial factors (Keeley *et al.* 2009; Syphard *et al.* 2012). For such a large sample size, we were unable to account for home construction materials, but this is also well understood to be a major factor, with older homes and wooden roofs being most vulnerable (Franklin 1996; Cohen 1999, 2000). In terms of actionable measures to reduce fire risk, this study shows a clear role for defensible space up to 30 m (100 ft). Although the effective distances were on average much shorter than 30 m (100 ft), we recognise that additional distance may be necessary to provide sufficient protection to firefighters, which we did not address in this study (Cheney *et al.* 2001). In contrast, the data in this study do not support defensible space beyond 30 m (100 ft), even for structures on steep slopes. In addition to the fact that longer distances did not contribute significant additional benefit, excessive vegetation clearance presents a clear detriment to natural habitat and ecological resources. Results here suggest the best actions a homeowner can take are to reduce percentage cover up to 40% immediately adjacent to the structure and to ensure that vegetation does not overhang or touch the structure.

In addition to defensible space, this study also underlines the potential importance of land use planning to develop communities that are fire safe in the long term, in particular through their reduction to exposure to wildfire in the first place. Localised subdivision decisions emphasising infill-type development patterns may significantly reduce fire risk in the future, in addition to minimising habitat loss and fragmentation (Syphard *et al.* 2013). This study was conducted in southern California, which has some of the worst fire weather in the world and many properties surrounded by large, flammable exotic trees. Therefore, recommendations here should apply to other non-forested ecosystems as well as many forested regions.

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Dead forests burning: the influence of beetle outbreaks on fire severity and legacy structure in sub-boreal forests

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Abstract. Recent regional mountain pine beetle (MPB) outbreaks have generated unprecedented tree mortality across the fire-prone landscapes of western North American forests and could potentially modify fire severity and postfire ecological effects. In 2012, 2013, and 2014, three fires burned through high mortality, gray-phase lodgepole pine-dominated forests in the plateau regions of central interior British Columbia, Canada, providing an opportunity to test for interactions between MPB outbreaks and wildfires. We inventoried 63 plots that spanned gradients of outbreak severity, fire severity, and burning conditions in a wilderness setting. Our objective was to evaluate the influence of outbreak severity on fire severity by assessing typical first-order fire effects as well as legacy structure related to the consumption of woody biomass on snags/trees. We found no evidence of a relationship between outbreak severity and fire severity for six of seven first-order fire effects, with the exception of deep charring. We found evidence that legacy structure in the form of consumed branch structure and deep char development had greater odds of occurrence on MPB-killed snags compared to trees killed during wildfire. Our results indicate two key findings. First, fire severity as it relates to most first-order fire effects measures is not influenced by outbreak severity, instead it is more strongly influenced by the interaction of fuels, weather, and topography during fire events. Second, our results highlight how the interaction between outbreak severity and fire severity alters postfire structural legacies and their functional attributes, which could have important ecosystem implications.

Key words: deep char; *Dendroctonous ponderosae*; legacy structure; mountain pine beetle; *Pinus contorta* var. *latifolia*; short-interval disturbances; snags; sub-boreal; wildfire.

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Introduction

Forest ecosystems across western North America are increasingly experiencing ecological disturbances from wildfires burning through landscapes with abundant tree mortality from insect outbreaks. The recent mountain pine beetle (*Dendroctonous ponderosae*, hereafter MPB) outbreaks circa the 1990s and 2000s are responsible for tree mortality in forests that span over 25 million hectares across the western United States and Canada (Raffa et al. 2008, Bentz et al. 2010, Meddens et al. 2012), and British Columbia (BC) houses nearly 20 million of those hectares (Axelson et al. 2009, Perrakis et al. 2014). The spatial extent and high mortality rates associated with recent outbreaks alter standing woody fuels in affected forests from mostly alive to mostly dead, which changes the composition of the fuel profile and raises concerns for increased fire severity (Hicke et al. 2012, Jenkins et al. 2012). The overlap between MPB outbreak and wildfire disturbances that recur within a short time interval may lead to linked effects in which the first event alters the extent, severity, or probability of occurrence for the second event (Kulakowski and Veblen 2007, Simard et al. 2011). Previous field-based studies have investigated interactions of short-interval MPB-fire disturbances with variable levels of lodgepole pine (*Pinus contorta* var. *latifolia*) mortality in montane regions of the western United States (Harvey et al. 2014*a*, *b*, Agne et al. 2016) and found fire severity to be either weakly linked or unrelated to outbreak severity. However, the magnitude of the MPB outbreaks in BC far exceeds the conditions seen in the western United States (Raffa et al. 2008) and the biophysical environment differs from earlier studies (Harvey et al. 2014*b*, Agne et al. 2016), such that further investigation is required to understand the implications of fire burning through BC's MPB-affected forests.

The changes in fuel profiles from MPB outbreaks and subsequent stand breakdown have raised concerns among land managers for altered fire behavior and potential changes in subsequent fire severity that could generate burning conditions that are more hazardous and more severe than from fire burning through stands of live trees. Severe tree mortality alters the configuration, continuity, and moisture content of fuels over time as stands break down (i.e., needle loss, branch breakage, sloughing bark, snag fall), all of which may influence fire behavior (Page and Jenkins 2007, Hicke et al. 2012). Dry, dead fuels ignite more quickly (Stockstad 1979) and at lower temperatures (Stockstad 1975) compared to live fuels. When these dead fuels are coarse, they are prone to smoldering (Brown et al. 2003), which often extends burning time beyond the initial flaming front (Alexander 1982), thus allowing for dry dead fuels to burn longer and have more biomass consumed (Brown et al. 2003, Hyde et al. 2011) that could alter the structural legacies that persist postfire. The only known empirical study examining the effects of altered fuel profiles on fire behavior found that spread rates increased through red-phase outbreak conditions in lodgepole pine forests in BC (Perrakis et al. 2014). Simulation models posit crown fire to increase during the red phase of the outbreaks, 1–3 years postattack, and then decline as needles are dropped from the canopy and snags transition to the gray-phase of the outbreaks, 3–10 years postattack (Hicke et al. 2012). Alternative models suggest a shift from active crown fire during the red phase to passive crown fire in the gray phase (Klutsch et al. 2011, Simard et al. 2011, Schoennagel et al. 2012) that could result in more biomass consumption and simplification of the legacy structure of snags.

Retrospective data that evaluate fire effects to characterize fire severity provide a complement to measures of fire behavior for understanding interactions between MPB outbreak and wildfire. Fire severity is often characterized by measurements of first-order fire effects (Reinhardt et al. 2001, Ryan and Elliot 2005) and refers to the amount of immediate ecological change associated with vegetation mortality and biomass loss from fire (Keeley 2009). Retrospective studies on MPB-fire interactions are two pronged either using remotely sensed data (e.g., satellite imagery) to quantify the amount of change between prefire and postfire conditions at coarser resolutions, or field studies that measure fire effects on the ground to characterize fire severity at finer scale resolutions. Existing remote sensing studies have shown that outbreak severity does not increase fire likelihood (Meigs et al. 2015), fire severity (Meigs et al. 2016), or area burned (Hart et al. 2015) for forests in the western United States.

Field studies can capture subtleties that may be absent in remote sensing studies and have found that the relationship between outbreak severity and fire severity varies across the western United States. These studies have focused on subalpine lodgepole pine forests (Harvey et al. 2014*a*, Agne et al. 2016) and forests dominated by subalpine fir (*Abies lasiocarpa*) but with substantial basal area of lodgepole pine (Harvey et al. 2014*b*) across topographically complex landscapes. Generally, these studies have suggested that gray-phase outbreak severity results in decreased fire severity (Harvey et al. 2014*a*, Agne et al. 2016), or limited to no change in fire severity (Harvey et al. 2014*b*, Agne et al. 2016)—with the exception of deep char, a metric of fire severity that showed a consistently positive relationship with severity of MPB outbreaks (Harvey et al. 2014*b*). Some measures of fire severity increased under extreme fire weather and were attributed to burning conditions, including deep char (Harvey et al. 2014*b*), suggesting that prefire beetle outbreak and burning conditions contribute to deep charring on wood. Deep char is generated through incomplete combustion of deadwood often from long, smoldering burns (Bird et al. 2015) that result in more biomass consumption and is visually distinct compared to scorch that is generated from flaming combustion and typically occurs on trees that are alive at the time of fire (Campbell et al. 2007). Deep char is distinguished by its iridescent black with patterning like the scales of alligator skin in contrast to the matte black, dusty appearance of scorch. Deep charring on trees changes the structure and function of the postfire landscape (Campbell et al. 2007, Donato et al. 2016) by altering structural legacies and has been clearly recognized as an important severity metric when examined in areas of high-severity reburns (fire + fire; Donato et al. 2016). However, the deep char effect, and the altered structural legacy it contributes to the postfire landscape, has largely been ignored in the context of insect outbreak and wildfire interactions.

Here, we examine the effect of gray-phase outbreak severity on fire severity for lodgepole pine-dominated forests with high prefire mortality rates, in central interior BC. Our objective was to evaluate the influence of outbreak severity on fire severity by assessing first-order fire effects after three recent wildfires that burned in 2012, 2013, and 2014. We wanted to (1) ascertain whether the extensive MPB-induced tree mortality that spans the sub-boreal forests of BC responds similarly in terms of first-order fire effects to forests that have burned and been studied in the western conterminous United States and (2) expand understanding and recognition of how postfire legacies (e.g., snags) can be affected by MPB outbreaks. We anticipated first-order fire effects (e.g., scorch/char height and area on trees, surface char, exposed mineral soil) would be unaffected by the severity of the outbreaks and primarily driven by fire weather, based on previous findings (Harvey et al. 2014*a*, *b*, Agne et al. 2016). In the context of structural legacies, we anticipated that snags killed by the MPB outbreak a decade prior to fire would burn longer, through smoldering combustion that would consume more wood biomass and lead to consistent development of deep char. We also predicted the interaction between outbreak severity and fire would reduce the structural complexity on snags, due to the potential extended duration of smoldering combustion in addition to prefire stand breakdown where MPB-killed trees experience needle loss, branch breakage, and shedding of bark. In contrast, the legacies of trees that were alive at time of fire and then killed by the wildfire (i.e., fire-killed) would have less deep char and retain much more structural complexity.

Methods

Study area

We conducted our field sampling across three fires that burned in 2012, 2013, and 2014 in Tweedsmuir and Entiako Provincial Parks, which are situated in the sub-boreal forests on the southern portion of the Nechako Plateau in BC (Fig. 1). The study area has a mean monthly maximum temperature of 8.5°C (range −3.3 to 19.8°C), a mean monthly nighttime temperature of −2.8°C (range −11.9 to 6.7°C), and total annual precipitation of 507.6 mm with a monthly mean of 42.3 mm (range 22.7–60.8 mm), based on the monthly means from the 1981–2010 climate normals (Abatzoglou et al. 2018). Precipitation accumulates as snow in the winter and rain during the remainder of the year. Although it rains through the summer (Abatzoglou et al. 2018), there are weeks with no rain that are associated with persistent high-pressure ridges (Nash and Johnson 1996). Within the fire perimeters, landscapes are associated with the Sub-Boreal Pine Spruce and Sub-Boreal Spruce biogeoclimatic zones (Meidinger and Pojar 1991), and lodgepole pine is the dominant canopy species (Fig. 1; BCMFLNRO 2012). Moisture gradients dictate composition, structure, and disturbance history, based on historical reconstructions from surrounding areas (Steventon 2001, Francis et al. 2002) and stand age distributions (Delong 1998). Within our fire perimeters, climax lodgepole pine inhabits the driest end of the moisture gradient, seral lodgepole pine persists with mean fire returns of 100–175 years, and climax spruce (*Picea engelmannii* × *glauca*) occupies pockets with high moisture levels such as riparian zones or through succession with long intervals of no fire (Parminter 1992). The landscape is gently rolling with low topographic relief, minimizing the topographic influence on fire behavior. Elevation ranges from 850 to 1300 m, in the region.

Field sampling occurred within three wildfire perimeters (Fig. 1). All fires were lightning ignited and received minimal to no suppression activities due to wilderness management

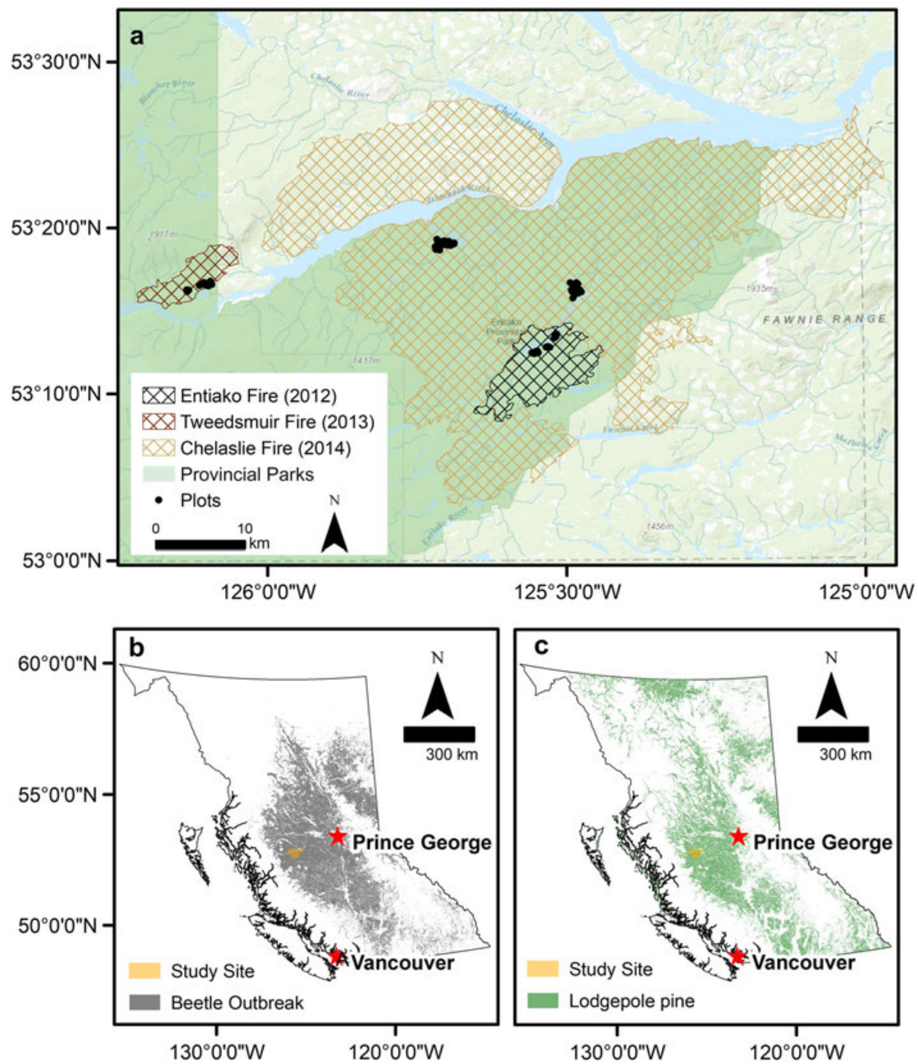

Fig. 1. Maps of the study area, MPB outbreak extent, and lodgepole pine range. Fire perimeters for three study fires that burned in 2012 (Entiako Fire; 7459 ha), 2013 (Tweedsmuir Fire; 3354 ha), and 2014 (Chelaslie Fire; 133,000 ha). (a) Provincial park boundaries are displayed as protected areas and overlaid with fire perimeters. Panel (b) shows the extent of the MPB outbreak across British Columbia based on aerial survey data from 2000 to 2011 (BCMFLNRO 2016). Panel (c) shows the estimated range of lodgepole pine across British Columbia.

objectives for the parks (Rob Krause and Mike Pritchard, *personal communication*). The Entiako Lake fire (R10171) burned 7450 ha from 3 August 2012 to 22 September 2012 (BCWS 2016). The Tweedsmuir fire (R10252) burned 3600 ha from 9 September 2013 to 16 September 2013 (BCWS 2016). The Chelaslie River fire (R10070) burned 133,100 ha from 8 July 2014 to 26 October 2014 (BCWS 2016). All fires burned through gray-phase outbreak conditions of varying severity. The recent MPB outbreaks, circa 1990s and 2000s, peaked in the region around 2003/2004 and have affected forests across much of the province (Fig. 1). MPB activity has declined since 2006 (Wulder et al. 2009). The lag between peak outbreak and the three wildfires was about a decade, with standing dead trees (snags) beginning to transition to coarse woody debris in some outbreak-affected stands. The location of fires within parks provided a rare opportunity to study MPB-fire interactions without the interference of active management (i.e., harvest/salvage activity, fire management/suppression).

Sampling design

Field sampling occurred from late June through August 2016, allowing us to characterize a snapshot of early successional forest communities from two to four years postfire. The study area has no road access, and sampling was limited by hiking and boating distances from three remote cabins within the parks (Fig. 1). Study plots were distributed through forest dominated by lodgepole pine. We selected plots based on a two-pronged approach including an a priori site selection using digital data, followed by verification and final selection in the field. A priori digital data included aerial survey data of MPB outbreak severity (BCMFLNRO 2016), burn severity maps generated from the differenced normalized burn ratio (dNBR; Eidenshink et al. 2007), and vegetation maps from the province's Vegetation Resource Inventory (VRI) to target areas of pure lodgepole pine (BCMFLNRO 2012). Aerial survey data for MPB outbreaks were a coarse resolution (400 m raster data products) and indicated areas within and around our study sites ranged between 50% and 100% canopy mortality (BCMFLNRO 2016). We selected from 12 to 39 plots in each study fire, depending on accessibility, for a total of 63 plots (Appendix S1: Table S1). We distributed plots across a gradient of fire severity within each fire class based on dNBR maps as low (n = 22), moderate (n = 18), and high (n = 23) that equated to light surface, severe surface, and crown fire based on our field measurements.

We visited plots on the ground for data collection. We verified canopy trees were predominantly lodgepole pine and were representative of the fire severity in the surrounding area. From the plot center location, we recorded GPS coordinates using a Garmin hand-held unit (GPSMAP 78s) and established a ten-by-ten-meter (100 m^2; 0.01 ha) survey plot, and divided it into four quadrants along the north–south and east–west axes, identified as NE, SE, SW, and NW. Within each quadrant, we placed a one-by-one-meter (1 m^2) subplot at increasing distances from the center of the plot (NE-1 m, SE-2 m, SW-3 m, and NW-4 m). Within plots, we recorded information for each live or dead tree: species, live/dead, evidence of MPB activity including exit holes and j-shaped galleries, diameter at breast height, and measures of first-order fire effects to characterize fire severity (including legacy structures) described in detail below. Within subplots, we recorded surface fire severity as first-order fire effects including duff depth, exposed mineral soil, terrestrial surface char, and litter. Measured variables at the plot and subplot resolution were used to characterize stand structure, MPB outbreak severity, fire severity and to analyze the relationship between MPB outbreaks and fire severity.

Mortality status of canopy trees and outbreak severity

We identified trees as live or dead at time of sampling and assigned a cause of death to each dead tree. We used these data to quantify MPB outbreak severity, mortality from fire, and cumulative mortality for each plot. We identified a tree's cause of death based on protocols adapted from Harvey et al. (2013, 2014*a*). We attributed each tree as most likely to have been (1) killed prefire by MPB (i.e., MPB-killed), (2) killed prefire by another agent (i.e., other-killed), (3) killed by fire (i.e., fire-killed), or (4) live postfire with no evidence of MPB activity (Table 1). We evaluated snags for MPB activity unless they were alive at time of sampling. We assessed each dead canopy tree for presence or absence of exit holes associated with adult beetles emerging from the tree (Harvey et al. 2013, 2014*a*). Then, we removed bark from each dead tree to identify galleries specific to MPB or other bark beetle species (Harvey et al. 2013, 2014*a*). We classified a tree as MPB-killed if it had the requisite exit holes and j-shaped galleries specific to MPB. While much of the prefire tree mortality present was linked to MPB, we also observed significant Ips beetle (*Ips pini*) activity, which we included as other-killed if there was no evidence of MPB. We classified a tree as other-killed if it was lacking evidence of exit holes and j-shaped galleries, but other evidence suggested death prior to fire such as no needle retention in the canopy, sloughing bark, other insect activity, and decay at the base, which is common in this system due to the moist climate (Table 1); this was a small portion of the total trees sampled (7%). We classified a tree as

Table 1. The criteria and classes used to identify a tree's cause of death for the study region in BC.

Cause of death	Description	Trees sampled (%)
Live tree	Live when sampled; green canopy; no visible beetle activity	4.67
Fire-killed	Dead when sampled; scorched bark, branches, and/or outer sapwood; no evidence of galleries or exit holes from MPB or other bark beetle activity; not highly decayed/weathered particularly at the base and in the canopy	28.95
MPB-killed	Dead when sampled; no needles remaining in the canopy; vacated mountain pine beetle (MPB) galleries in cambium with exit holes in remaining bark	59.38
Other-killed	Dead when sampled; highly decayed/weathered, no bark, missing branches; more advanced decay then MPB-killed trees; full deep char with no identifiable vacated MPB galleries	7.00
Prefire-killed: MPB-killed + Other-killed	All prefire-killed from both MPB-killed and other-killed	66.38

Notes: Methods adapted from Harvey et al. (2014*a*). Trees sampled summarize observed data from field collections.

fire-killed if it had red needles in the canopy or postfire needle drop, and no evidence of prefire MPB or other beetle activity. We estimated a general metric of prefire-killed trees as the combination of MPB-killed and other-killed (Table 1). We calculated plot-level metrics for outbreak severity as the proportion of MPB-killed trees per plot and prefire mortality as the proportion of all prefire-killed trees per plot. Pre-outbreak stand estimates were based on all trees in the plot, regardless of status.

Fire severity recorded as first-order fire effects at the plot level

We characterized fire severity with seven measures of first-order fire effects that were scaled to a plot-level metric. We measured fire effects on standing trees/snags and the terrestrial surface including: height of scorch and/or char on trees, percent cover of scorch and/or char on trees, percent deep charring on trees, litter/duff depth, proportion of remaining litter, proportion of terrestrial surface char, and proportion of exposed mineral soil. Scorch and deep char are visually distinct, scorch with a dusty, matte black appearance and deep char with an iridescent black, scale-like appearance. In some cases, snags had both areas of scorch and deep char. The height of scorch and/or deep char (hereafter scorch/char) was measured to the nearest 0.5 m on each tree with four-meter measuring sticks and converted to mean scorch height per plot. We estimated the percent area covered, as height and circumference, of scorch and/or deep char (hereafter scorch/char) and calculated a mean proportional area per plot. We inverted the mean proportion of area per plot to the proportion of unscorched area per plot for analysis. We recorded deep char for each tree as no deep char, <50% deep char, or 50–100% deep char coverage on the snag and calculated the proportion of snags with deep char for a plot-level variable. The four terrestrial surface fire effects metrics were measured in each subplot in the four quadrants of the plot. Litter/duff depth was measured as the combination of litter plus duff to the nearest millimeter in two opposing corners of each subplot and averaged to a plot-level variable. We recorded the percent of remaining litter, terrestrial surface char, and exposed mineral soil and calculated a mean for each variable from the four subplots to generate plot-level metric. Remaining litter, terrestrial surface char, and exposed mineral soil were converted to proportions for analysis purposes. Because we surveyed plots between two and four years postfire, we captured various early successional stages in postfire litter accumulation and vegetative regrowth.

Fire severity recorded as biomass consumption of legacy structure at the tree level

To characterize the effect of outbreaks and wildfires on postfire legacy structure, we categorized biomass loss on each tree based on the remaining branch structure and deep char. The remaining branch structure refers to the fine, moderate, and coarse branch structure, and it was quantified as presence or absence. A classification of absence meant that there was no remaining branch structure on the tree and no associated branches on the ground in the area of the tree/snag, which indicated that branches were consumed by fire. As described above, deep char was visually distinct from scorch and recorded as absent, <50%, or 50–100% deep char coverage on each snag. We retained these categories to assess the relationship between deep char and remaining branch structure and converted the categories to presence or absence of deep char for each tree/snag to evaluate the relationship between a tree's cause of death and deep char development.

Fire weather and topography

The Entiako, Tweedsmuir, and Chelaslie fires that provided the footprint for our study burned during three different fire seasons (2012, 2013, and 2014) across a landscape with low topographic complexity. Fires burned over a relatively long duration within each season, which allowed us to account for variability in fire weather and day-of-burn conditions (Appendix S1: Table S1). Each plot was assigned a day of burn from day-of-burn progression maps estimated from MODIS hotspot data (Parks 2014), which allowed us to assign the daily fire weather index (FWI) to each plot that was generated from the nearest weather station (Appendix S1: Table S1). The calculated FWI is a metric from the Canadian Forest Fire Weather Index System (Van Wagner 1987) that integrates temperature, relative humidity, and wind speed. We used the FWI to assign each plot a burning condition category of moderate (>13–29) or extreme (>29), based on breakpoints outlined by Alexander and De Groot (1988). All fires experienced moderate burning conditions within a portion of their perimeter; however, extreme burning conditions (FWI $\geq$ 29) only occurred in two of the three fires (2012 Entiako and 2014 Chelaslie fires; Appendix S1: Table S1). For plots, elevation fluctuated between 873 and 1043 m. Plots were relatively flat with a mean slope of 2.6° (range 0–20°). We did not pursue topography as an explanatory variable of fire severity due to the low topographic variability at our study plots.

Statistical analysis

We tested for relationships between each of the seven fire effects metrics and MPB outbreak severity at the plot level, while accounting for burning conditions. Our seven fire effects metrics served as response variables: average scorch/char height, average proportion of unscorched/uncharred area on trees, proportion of trees with deep char, litter/duff depth, proportion of remaining litter, proportion of terrestrial surface char, and proportion of exposed mineral soil. We tested the relationship of each response variable against the proportion of MPB-killed trees (our index of MPB severity) and burning conditions, which was included as a categorical variable of moderate or extreme FWI (Appendix S1: Table S1). An interaction term between burning conditions and proportion of MPB-killed trees was included in all models to assess whether observed relationships changed under different fire weather conditions. Relationships with scorch/char height and litter/duff depth were fit using linear models. The proportion of terrestrial surface char was logit-transformed and fit with a linear model. All other models in which the response variable was a proportion were analyzed with generalized linear models, and each response variable was fit with a distribution appropriate for the type and distribution of the response variable (see Appendix S1: Table S3 for distributions associated with each analysis). We also ran each model and replaced the proportion of MPB-killed trees with the proportion of prefire-killed trees, since dead trees would all be similar in terms of conditions and moisture content regardless of what killed them. The models with the proportion of prefire-killed trees demonstrated similar statistical relations to the proportion of MPB-killed trees. We report all models that were statistically significant, and we kept all fire effects models that were run with the proportion of MPB-killed trees as an explanatory variable, since these models were a more conservative estimate of MPB caused mortality.

We evaluated the effect of outbreak severity and wildfire on postfire legacy structure from tree-level fire effects of deep char and branch structure loss. We analyzed data at the tree level using two different response variables: (1) presence/absence (1/0) of branch structure on individual trees and (2) presence/absence (1/0) of deep char on individual trees. We accounted for burning conditions as a categorical variable of moderate or extreme FWI (Appendix S1: Table S1). An interaction term between burning conditions and cause of death was included in all models. We used generalized linear mixed models with a binomial distribution for presence/absence data using a logit link, and each model included the plot as a random effect and the interaction term between cause of death and burning conditions. Results are reported as probability of occurrence, and the comparison between mortality types (e.g., MPB-killed versus fire-killed) is reported as the odds ratio. Additionally, we determined whether the presence/absence of branch structure was related to the coverage of deep char on the tree. Our explanatory variable of deep char was treated as a three-level categorical variable of no deep char, <50% coverage of deep char, or 50–100% coverage of deep char on the tree, while accounting for burning conditions.

We assessed fit for all models by visually inspecting the residuals, which appeared to be adequately met. We evaluated and corrected for overdispersion in all generalized linear models and generalized linear mixed models when necessary. For our two linear models, assumptions of normality and constant variance of the residuals were checked graphically and appeared to be adequately met. We assessed the interaction term with a drop-in-deviance test. The interaction term was retained in each model regardless of statistical significance, because of the known interaction between fire weather and fuels. All statistical analyses were conducted in R statistical computing software version 3.4.4 with the stats package (R Development Team 2018). For generalized linear models, we used the function glm in the MASS package (Venables and Ripley 2002). For generalized linear mixed models, we used the function glmer in the lme4 package (Bates et al. 2015). We considered $p < 0.05$ as convincing evidence of a relationship and $P < 0.10$ as suggestive of a relationship to minimize the potential of a type II error. Data and code for analyses are available online (Talucci 2019).

Results

We collected data from 943 trees across 63 field plots with 910 lodgepole pine trees/snags and 33 spruce trees/snags. Canopy tree species were predominantly lodgepole pine with a plot mean of 96% (range across plots 63–100%; Appendix S1: Table S2). Estimated mortality from MPB was 59% of all trees sampled, and estimated prefire mortality (i.e., MPB-killed plus other-killed) was 66% of all trees sampled (Table 1). When we evaluated just lodgepole pine mortality across all 63 plots, the estimated mean for lodgepole pine killed by MPB was 63% and the estimated mean for lodgepole pine killed by all agents prior to fire (all prefire) was 70% (Appendix S1: Table S2). Cumulative mortality was estimated at 93% for lodgepole pine as a combination of prefire and fire mortality (Appendix S1: Table S2).

Effect of outbreak severity on first-order fire effects at the plot level

The effect of outbreak severity on fire severity was limited, with six of seven fire effects showing no evidence of an effect (Fig. 2, Appendix S1: Table S3). average scorch/char height, average proportion of unscorched/uncharred area on trees, litter/duff depth, proportion of remaining litter, proportion of terrestrial surface char, and proportion of exposed mineral soil showed no evidence of an effect of outbreak severity (Fig. 2, Appendix S1: Table S3). Outbreak severity did show evidence of an effect on proportion of trees with deep char. Under moderate burning conditions, the proportion of trees with deep char increased with increasing outbreak severity (Fig. 2, Appendix S1: Table S3), which held true when we substituted the proportion of prefire-killed trees for MPB-killed trees (Fig. 2, Appendix S1: Table S3). Under extreme burning conditions, the relationship between the proportion of MPB-killed trees and deep char was not statistically significant, however when we substituted the proportion of prefire-killed trees for MPB-killed trees that relationship was statistically significant (Fig. 2, Appendix S1: Table S3).

Effect of outbreak and wildfire on legacy structure

Outbreak severity and wildfire showed distinct evidence of an effect on the legacy structure of the forest, measured by biomass consumption as deep char and branch structure loss on individual trees. Both deep char development and branch structure loss had greater odds of occurrence when a tree was dead prior to fire (i.e., MPB-killed or prefire-killed) compared to being

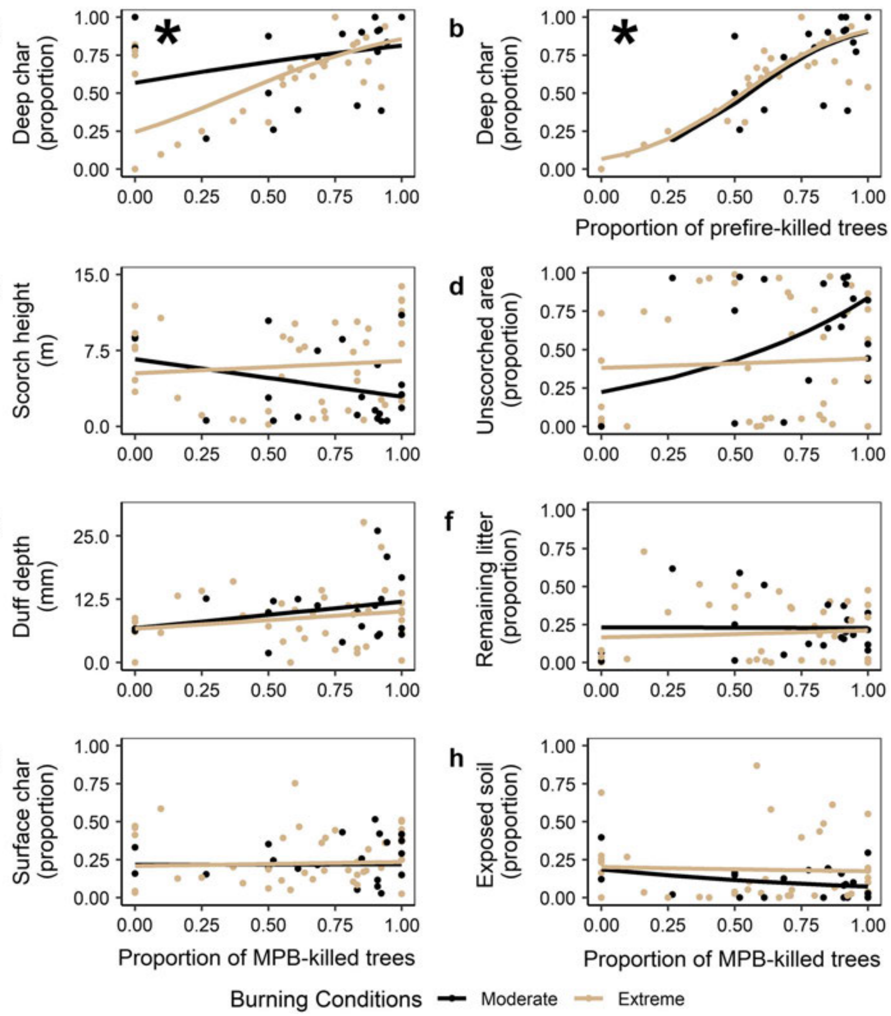

Fig. 2. The relationships between outbreak severity and seven first-order fire effects measured at the plot level: proportion of trees with deep char (a–b), average scorch/char height (c), average proportion of unscorched/uncharred area on trees (d), litter/duff depth (e), proportion of remaining litter (f), proportion of terrestrial surface char (g), and proportion of exposed mineral soil (h). Response variables are along the *y*-axis with the explanatory variable of the proportion of mountain pine beetle (MPB) killed trees or prefire-killed trees (only panel b) along the *x*-axis. Points are the raw data (n = 63 plots), and fitted lines show the estimated statistical relationship. The response variable of deep char is shown in panels a and b, and they were the only two models that indicated a strong statistical relationship (*). Response variables c–h were unrelated to outbreak severity. See Appendix S1: Table S3 for model estimates and confidence intervals.

alive at time of fire, which was consistent across both moderate and extreme burning conditions (Figs. 3, 4, Appendix S1: Table S4). Under both moderate and extreme burning conditions, a MPB-killed and prefire-killed snag had greater odds of developing deep char compared to a fire-killed tree (Figs. 3, 4, Appendix S1: Table S4). There were greater odds of branch structure being consumed on a MPB-killed and prefire-killed snag compared to a fire-killed tree under moderate conditions, and the size of that effect was slightly smaller under extreme conditions but still significant (Figs. 3, 4, Appendix S1: Table S4). We found that branch structure had greater odds of being consumed when deep char exceeded 50% coverage on the tree for both

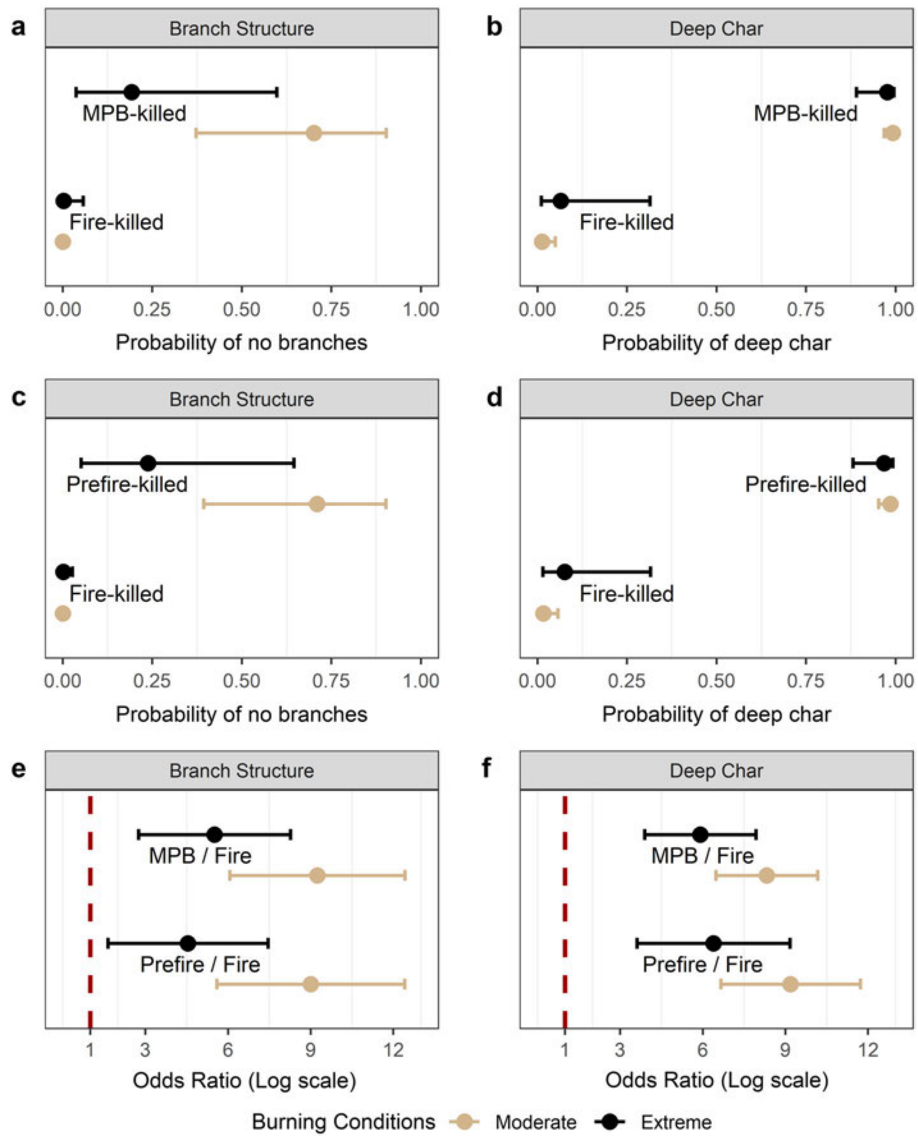

Fig. 3. The effect of outbreak severity and wildfire on legacy structure measured at the tree level. Tree-level fire effects show the role of mountain pine beetle (MPB) outbreak severity (MPB-killed and prefire-killed) on consumption of woody material and simplification of structural legacies in the form of branch loss and deep char development (a–d). Comparison between groups (i.e., MPB-killed versus fire-killed) is shown in (e) and (f) as odds ratios with the red dashed line marking no difference at one. The model estimates are listed in Appendix S1: Table S4.

moderate and extreme burning conditions (Fig. 5, Appendix S1: Table S5).

Discussion

We found that fire severity as measured by scorch/char height and area, and surface fire metrics, is not influenced by MPB outbreak severity but that fire severity measured as biomass loss and legacy structure was consistently influenced by the outbreak history. These findings from BC align with previous field research that evaluated the influence of outbreak severity on fire severity in the western United States

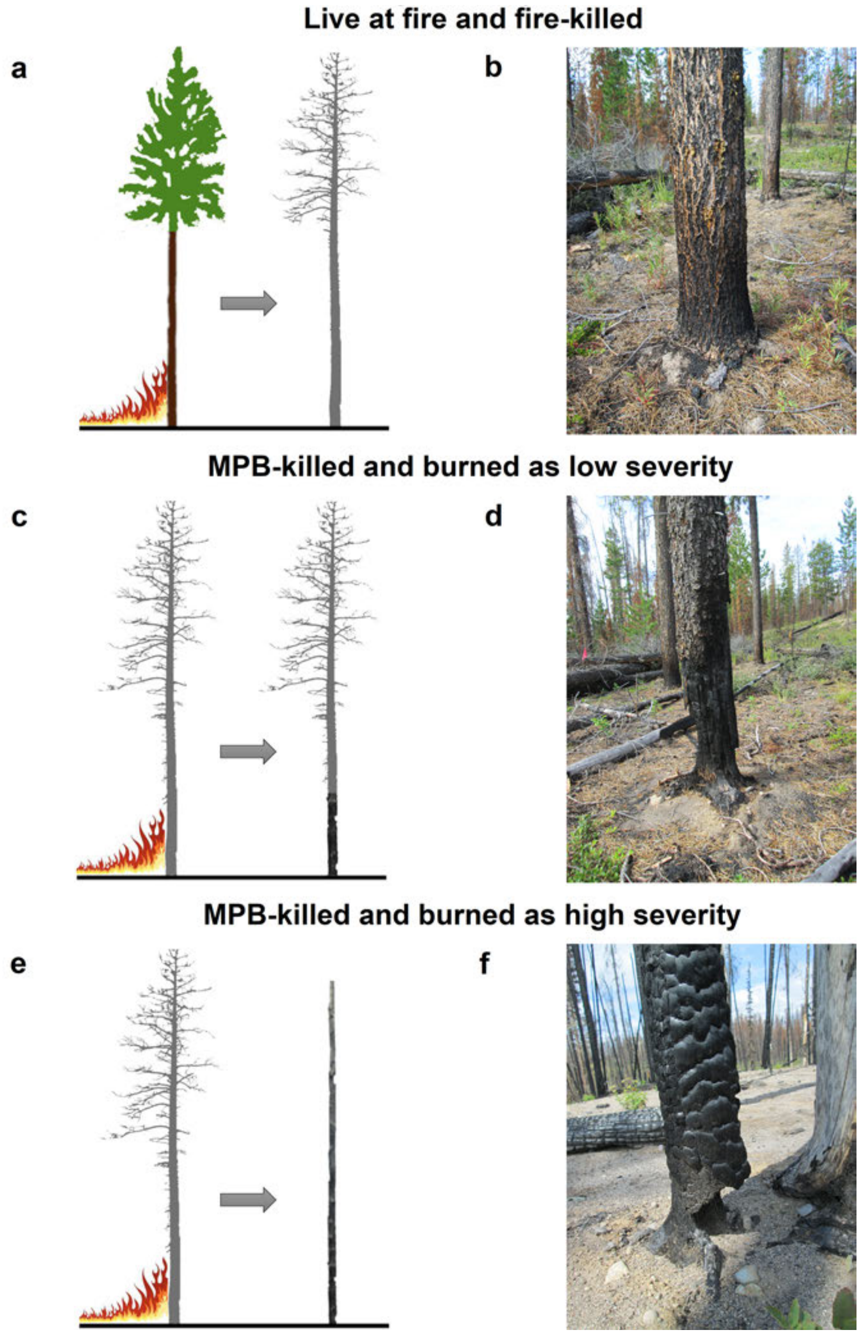

Fig. 4. Tree-level fire effects are dependent on whether a tree is alive or dead at time of fire. Panel (a) illustrates a tree that is live at time of fire and killed by fire, with the adjacent panel (b) showing a photo of a tree live at time of fire and killed by fire with scorched bark but no consumption of the tree. Panel (c) illustrates a MPB-killed tree that burns under low severity conditions, with the adjacent panel (d) showing a photo of deep char development and consumption at the base of the tree, which is attributed to fungal development (Donato et al. 2009). Panel (e) illustrates a MPB-killed tree that burns under high-severity conditions, with the adjacent panel (f) showing a photo of deep char that covers the entire tree in a plot that burned as high severity.

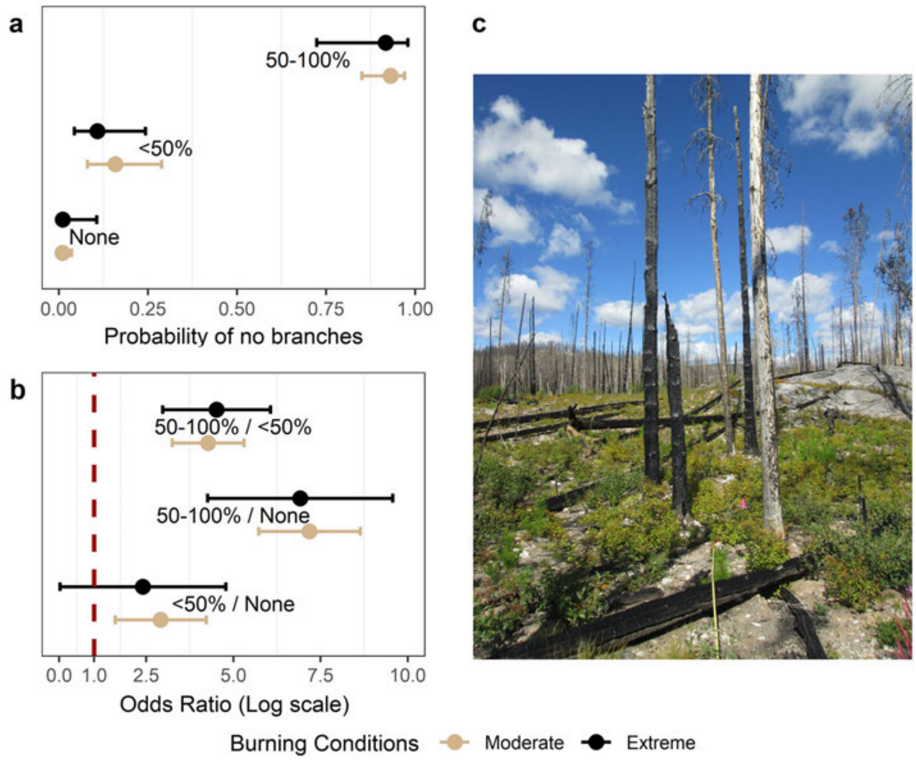

Fig. 5. Deep char coverage influences the consumption of branch structure on a tree. The probability of branch loss from deep char development is shown in (a). Comparisons between groups are shown in (b) with a red dashed line demarcating no difference at one. In (c), a photo of a snag with deep char and no branches adjacent to a snag with branches still intact and no deep char. Model estimates are list in Appendix S1: Table S5.

(Harvey et al. 2014*a*, *b*, Agne et al. 2016), but extends our understanding of these short-interval disturbances by highlighting the synergistic effect on postfire structural legacies. Prefire mortality had a greater likelihood for increased biomass consumption and deep char, which aligns with findings on reburns, where wildfires recur in short intervals (Donato et al. 2016). Prefire mortality, regardless of the mechanism of death, results in an altered legacy structure that is more simplistic and has more deep char. While this effect on structural legacies is generally accepted in the field, it has been broadly overlooked and unquantified in assessments of how outbreak affects fire severity. When asking the question "does MPB outbreak affect fire severity, are they linked disturbances," the answer is yes—specifically through the deadwood structure that remains in these ecosystems.

Effect of outbreak severity on first-order fire effects at the plot level

Outbreak severity did not show evidence of an effect on fire severity for six out of seven measured first-order fire effects; the exception was deep char. This reflects similar findings in gray-phase outbreak conditions found by Harvey et al. (2014*b*) and extends our understanding to the geography of BC's sub-boreal forests. Our finding indicates some inherent noise and uncertainty in our data as well as the influence of fire weather. The six fire effects—scorch/char height and area, duff depth, litter, surface char, and exposed soil, are likely controlled by the combined factors of the fire environment, that is, the interaction of fuels, weather, and topography (Countryman 1972, Krawchuk and Moritz 2011, Whitman et al. 2015) but without a strong signal from outbreak fuel structure, which aligns with previous research evaluating interactions between outbreak severity and fire severity (Harvey et al. 2014*a*, *b*, Agne et al. 2016). Scorch on trees is naturally variable and can be driven by multiple factors including the composition of fuel structures, crown and/or surface fire spread, burning conditions, slope steepness, and ignition patterns (Alexander and Cruz 2012*a*). Our results show no evidence of a relationship between terrestrial surface fire effects and outbreak severity, which was also consistent with findings in previous retrospective studies with gray-phase outbreak conditions (Harvey et al. 2014*b*, Agne et al. 2016). The lag time between needle drop and our study fires would have allowed for the decomposition of fine fuels (Simard et al. 2011, Harvey et al. 2013) thus minimizing the effect of outbreak on surface fuels. Most snags were still standing at time of fire, so that the concern of increased surface fire severity from abundant coarse woody debris was not observed. These findings support the general narrative that low-frequency and high-severity fire regimes associated with lodgepole pine in sub-boreal forests are strongly driven by climate systems of high-pressure, creating dry-hot conditions conducive for burning such that variability in fuel structure/vegetation plays a secondary role (Bessie and Johnson 1995, Nash and Johnson 1996, Whitman et al. 2015).

Effect of outbreak and wildfire on structural legacies

Our findings support the notion that dead wood, which in our landscapes is predominantly snags generated by MPB outbreaks, burns differently than live wood and indicates an important MPB-fire connection. Live trees rarely experience significant combustion, and therefore, little to no consumption occurs on the tree (Campbell et al. 2007), which is attributed to higher moisture content compared to their dead counterparts (Brown et al. 1985). Extended periods of smoldering and glowing combustion (Brown et al. 1985, Page and Jenkins 2007, Hyde et al. 2011) are facilitated by lower moisture content in snags and coarse wood (Stockstad 1979). Lower moisture content in snags could enable passive crown fire or torching of snags (Wenger 1984), which may be the primary mechanism for consumption of branch structure. Some simplification of branch structure may also occur on gray-phase MPB-killed trees prior to fire. The torching of snags and extended periods of smoldering have been demonstrated in areas that experience reburn, wildfires that recur in short intervals (Donato et al. 2016). High-severity reburns have shown there is an eightfold increase in deep char development on snags and the retention of woody biomass is half the amount of once burned areas, in the Klamath Mountains of southwestern Oregon (Donato et al. 2016). Where wind-throw is followed by wildfire in short intervals, snags and coarse wood have been shown to be reduced with a marginal increase in charred material (Buma et al. 2014). In lodgepole pine/Douglas-fir (*Pseudotsuga menziesii*) forests on the Chilcotin Plateau of BC south of our study sites, areas of high prefire mortality from MPB outbreak experienced 13% more consumption of dead wood and the variability in canopy consumption was attributed to mortality status with dead prefire snags having more of their branch structure consumed (Brad Hawkes, *personal communication*). This evidence indicates that it is not necessarily the mechanism of prefire mortality, for example, MPB outbreak, wind-throw, or prior wildfire, but the fact that there is an abundance of deadwood with altered moisture levels and fuel structures compared to live wood, which alters postfire ecological and structural legacies as they relate to standing snags and coarse woody debris.

The consumption of branches and deep char development on snags alters the structural legacies that endure through fire. These altered legacies may introduce long-term implications for ecosystem structure and function including availability of canopy seedbank, accumulation of coarse woody debris, and early seral structure and resources for early seral species (Franklin et al. 2000, Swanson et al. 2011, Johnstone et al. 2016). After MPB outbreak, lodgepole pine snags continue to retain some of their aerial seedbank in the canopy postmortem while some cones fall to the forest floor (Teste et al. 2011). Cones in snags or on the forest floor can be exposed to extended heating from a snag smoldering or slower moving surface fire (Alexander and Cruz 2012*b*), which could reduce seedbanks and influence postfire resilience (Johnstone et al. 2016). The loss in snag biomass and branch structure alters the accumulation of coarse wood that may influence short- and long-term carbon and nutrient cycles (Harmon 2001), structure of habitat for wildlife (Fontaine et al. 2009, House 2014) including nesting and perching habitat, and both structure and function of early seral ecosystems (Swanson et al. 2011). More charring on trees reduces the quality of the snag for saproxylic insects thereby affecting foraging woodpeckers (Saint-Germain et al. 2004, Nappi et al. 2010), which could influence trophic webs. Deep char development can encapsulate the remaining wood, which may limit decomposition, slow decay, and extend long-term carbon storage (Preston 2009, Bird et al. 2015). Together, these changes to dead wood that may alter the long-term structure and function in the postfire forest are considered compound disturbance effects, where the outbreak severity and fire severity work in combination to create unique post-disturbance conditions that are different than the outcomes of the singular disturbance of wildfire (Paine et al. 1998). Further research is needed to determine the long-term implications of compound disturbance effects related to legacy structure, coarse wood recruitment, carbon storage, pyrogenic carbon, habitat structures, trophic webs, and early seral ecosystems in forests where fires are increasingly burning through stands with high volumes of snags from insects, windthrow, drought, and prior fire.

Conclusion

Sub-boreal forest ecosystems of BC have experienced widespread tree mortality from the MPB outbreaks, generating a fuel structure characterized by an abundance of deadwood that is now interacting with wildfires. The contiguous landscape of lodgepole pine-dominated forests situated at the epicenter of the outbreak in western North America allowed us to assess interacting, or linked, effects between outbreak and fire severity. Our results suggest that while many first-order fire effects are not influenced by outbreak severity, legacy structure related to the degree of biomass consumption is strongly influenced by the interaction of outbreak severity and fire severity. These findings are especially important to consider after the 2017 and 2018 fire seasons in BC wherein a record number of hectares burned, with many of the fires burning through snag forests affected by MPB outbreaks.

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Supporting Information

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecs2.2744/full

Perspectives on LANDFIRE Prototype Project Accuracy Assessment

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Chapter 13

Perspectives on LANDFIRE Prototype Project Accuracy Assessment

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Introduction

The purpose of this chapter is to provide a general overview of the many aspects of accuracy assessment pertinent to the Landscape Fire and Resource Management Planning Tools Prototype Project (LANDFIRE Prototype Project). The LANDFIRE Prototype formed a large and complex research and development project with many broad-scale data sets and products developed throughout its various stages. The scope of the project was defined as mapping and modeling vegetation, wildland fuel, and fire regime characteristics (Rollins and others, Ch. 2). Because of the breadth of the investigation, it is important to base our expectations for accuracy on a clear understanding of the intricacies, interdependencies, and scope of mapping and modeling LANDFIRE products. Our goals in this chapter are to: 1) provide relevant background information regarding accuracies and what was realistically achievable in the LANDFIRE Prototype, 2) provide background regarding our strategies for LANDFIRE National, 3) describe our actual LANDFIRE Prototype accuracy results in broad terms, and 4) provide recommendations for the national implementation of LANDFIRE. This chapter is not intended to provide an exhaustive list and description of all of the various accuracy-related issues and conclusions resulting from the LANDFIRE Prototype (for specific details, the reader will be referred to the appropriate chapters). Rather, this chapter is intended to be broad in scope and to place the many accuracy components within the context of the LANDFIRE Prototype and LANDFIRE National projects. Please note that Lunetta and Lyon (2004) provide an in-depth discussion of the current state of accuracy assessment within the science community.

Background

General Accuracy Tenets and Philosophy

First we will provide the reader with several broad tenets used in defining accuracy assessment for the LANDFIRE Prototype Project and thereby lay the foundation for the more in-depth discussion following.

Tenet 1: Assuming that thematic detail and spatial scale are constant, product accuracy is generally inversely correlated with the size of the region being assessed.

Within the remote sensing literature, there are many references to accuracy levels, and many of the reported values are quite high. These high levels may lead to inflated expectations regarding what types of accuracies will be achievable from LANDFIRE. Many previous studies were conducted within relatively small study

In: Rollins, M.G.; Frame, C.K., tech. eds. 2006. The LANDFIRE Prototype Project: nationally consistent and locally relevant geospatial data for wildland fire management. Gen. Tech. Rep. RMRS-GTR-175. Fort Collins: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

areas, often aided by high levels of "hand crafting" during the mapping process and/or in-depth knowledge of the particular study area. We do not have the luxury of spending a great amount of time and effort on any one particular region mapped through the LANDFIRE Project, and the mapping and modeling tasks need to be accomplished through largely automated processes. These limitations do not by any means reduce the value of the products being created through LANDFIRE; however, it should be stated that LANDFIRE products will likely have lower overall accuracies than do data sets derived from more localized studies characterized by large amounts of field data, increased processing effort that may include on-screen digitizing and recoding, and/or iterative refinement of modeled results.

Tenet 2: The higher the thematic detail, the lower the accuracy.

A relatively large number of vegetation classes were mapped for the LANDFIRE Prototype (Long and others, Ch. 6). While the chosen map unit classification system made sense on many levels for the LANDFIRE Prototype, it must be recognized that the proliferation of classes in this or similarly complex systems will imply a relative decrease in accuracy levels. This does not in any way diminish the value of the vegetation products, but is rather simply a result of a more complex map unit classification design. For example, a two-category classification of water and uplands is likely to result in high accuracy, with expected accuracies above 99 percent. This high accuracy does not mean that the *value* of the product is particularly high, but simply reflects that the accuracy for depicting these two classes is high. Additionally, there are difficulties that arise when categorizing continuous phenomena into rigid and discrete classes. For instance, a more detailed map unit classification system might treat juniper and pinyon – juniper ecosystems as several discrete classes even though the boundaries between them are relatively arbitrary and difficult to delineate both in the field as well as within the imagery. With complex vegetation map unit legends, such as that used in the LANDFIRE Prototype, vegetation class accuracy levels can be expected to drop. Nevertheless, LANDFIRE products reliably and consistently describe the distribution of vegetation composition, condition, and structure and associated wildland fuel and fire regimes across broad landscapes. These mapped data are useful for hazardous fuel reduction projects, for a variety of resource management projects, and for both strategic and tactical wildland fire management.

Tenet 3: Field information used for assessing accuracy is not perfect.

As mentioned under Tenet 2, the LANDFIRE Prototype vegetation map unit legends are relatively complex (Long and others, Ch. 6). The map unit classifications are developed using large quantities of field data, and all of the field plots are assigned to one of the many possible classes. Most of these plots are used to generate maps, but some are reserved for use in the accuracy assessment phase of the investigation. We recognize four major potential sources of error associated with field plot data:

- Errors occur frequently in the identification of species and measurement of vegetation structure in the field (for example, in the data for one prototype field plot, a misplaced decimal point indicated a shrub height of 60 feet).
- The vegetation on some field plots has undoubtedly changed between the time the field data were collected and when the imagery was acquired.
- Geo-location errors in plot and imagery data result in inaccurate characterization of some imagery pixels.
- The assignment of plots to specific vegetation classes will have errors associated with the wide array of opinions among professional field ecologists regarding the field classification of any given field plot.

Tenet 4: The modeled results of complex ecological systems will be characterized by ambiguity and controversy.

The products generated from the LANDFIRE Prototype represent our best approximations in depicting the current status of very complex natural phenomena. The information used in our modeling efforts is based on the best available input data and assumptions. However, although our output products represent reasonable and robust depictions of current conditions, we recognize that, due to lack of baseline research, our knowledge of certain ecological systems is imprecise. Use of such information in the modeling process may result in potential flaws in the products, and hence not all of the core LANDFIRE deliverables will be free of error and ambiguity. Nevertheless, the LANDFIRE Project represents an integration of the best available science in remote sensing, ecosystem simulation, landscape fire and succession modeling, predictive landscape mapping, and wildland fire behavior and effects prediction.

We are therefore confident that the products generated represent the best current assessments of the status of these ecosystems with regard to wildland fire and will be of great value to natural resource managers.

Accuracy Assessment Considerations for LANDFIRE

The need for conducting accuracy assessments of the spatial products created from mapping projects has been well documented (Congalton 1991; Foody 2002). Factors that influence map accuracy include (but are not limited to) the remote sensing platform, the quality of ancillary sources of information, the quality of field data, the floristic complexity of the map unit classification system used, and the sampling design. Traditional first-order map accuracy estimates involve generating an error matrix, computing overall accuracy, and estimating "producer's accuracy" and "user's accuracy" (Congalton 1991). In the past, assessment of map accuracy has involved much post-mapping fieldwork in order to develop error matrices. These formal, traditional accuracy assessments involving field campaigns can be labor-intensive, time-consuming, and cost-prohibitive, especially when dealing with projects that cover large regions of diverse and overlapping vegetation composition and conditions (Stehman and others 2000). For this reason, only a few efforts have conducted accuracy assessments across broad expanses such as the entire United States (Stehman and others 2003; Wickham and others 2004).

Techniques that worked well in assessing mapping accuracy across large regions for the 1990s National Land Cover Database (NLCD; Vogelmann and others 2001) employed modifications of traditional accuracy assessment methodologies (Stehman and others 2003; Wickham and others 2004). As background, the 1990s NLCD database was developed using Landsat satellite imagery acquired for the Multi-Resolution Land Characteristics (MRLC) 2001 consortium using methods previously described (Vogelmann and others 1998). During development of the database, it was determined that an accuracy assessment for the large area product was required, and that such an effort would have to be modified from more traditional assessments. The modifications were necessary in part due to the scarcity of field data across the mapped regions, the large size of the area being assessed (and associated high costs of collecting data from a statistically valid number of field locations across the entire conterminous United States), difficulties in assigning unambiguous map unit labels to many field plots, and geolocational errors associated with field plot and satellite-derived mapping information.

Three important lessons learned from the accuracy assessments of the 1990s NLCD effort pertain directly to the accuracy assessment methods used during the LANDFIRE Prototype Project:

- Collecting data for and compiling custom field databases is time consuming and expensive. Similarly, combining data from disparate sources and distilling them into a training database for mapping purposes is time consuming, expensive, and can result in data inconsistencies unless special effort is made to crosswalk and/or standardize input data. On the other hand, using existing field data, rather than collecting custom field data, saves both time and money. In short, for large-area projects, it makes sense to use existing field data for conducting accuracy assessments.
- Determining accuracy values for different subregions is acceptable when mapping large regions. Accuracies are likely to vary across large mapped areas due to region-specific heterogeneity in landscape composition and structure, and it was advantageous to derive an understanding of the geographic variability of accuracies of the products developed for LANDFIRE. To this end, use of a systematic random sampling design can provide optimal results. Such a design ensures that all geographic regions are adequately sampled and thereby ensures that at least some estimates of accuracies exist throughout the entire study region.
- Some errors are more "wrong" than others. For instance, for the LANDFIRE effort, misclassification of a pinyon – juniper stand as a riparian woodland stand will likely have a greater negative impact on the predicted fire behavior than misclassification of a pinyon – juniper stand as a juniper stand. Furthermore, some vegetation types are spectrally and biogeographically very similar to other vegetation types, and even with "perfect" source material, it is difficult to adequately distinguish some of these classes. For example, Douglas-fir and white fir are spectrally very close (fig. 1), and both species inhabit similar ecological niches. In regions where both Douglas-fir and white fir occur, we can expect significant confusion between the two classes. For instance, in central Utah, cross validation accuracies for these two classes were quite low, as anticipated. Nonetheless, we suspect that the errors related to misclassifying similar vegetation types will only minimally impact predicted fire behavior, whereas

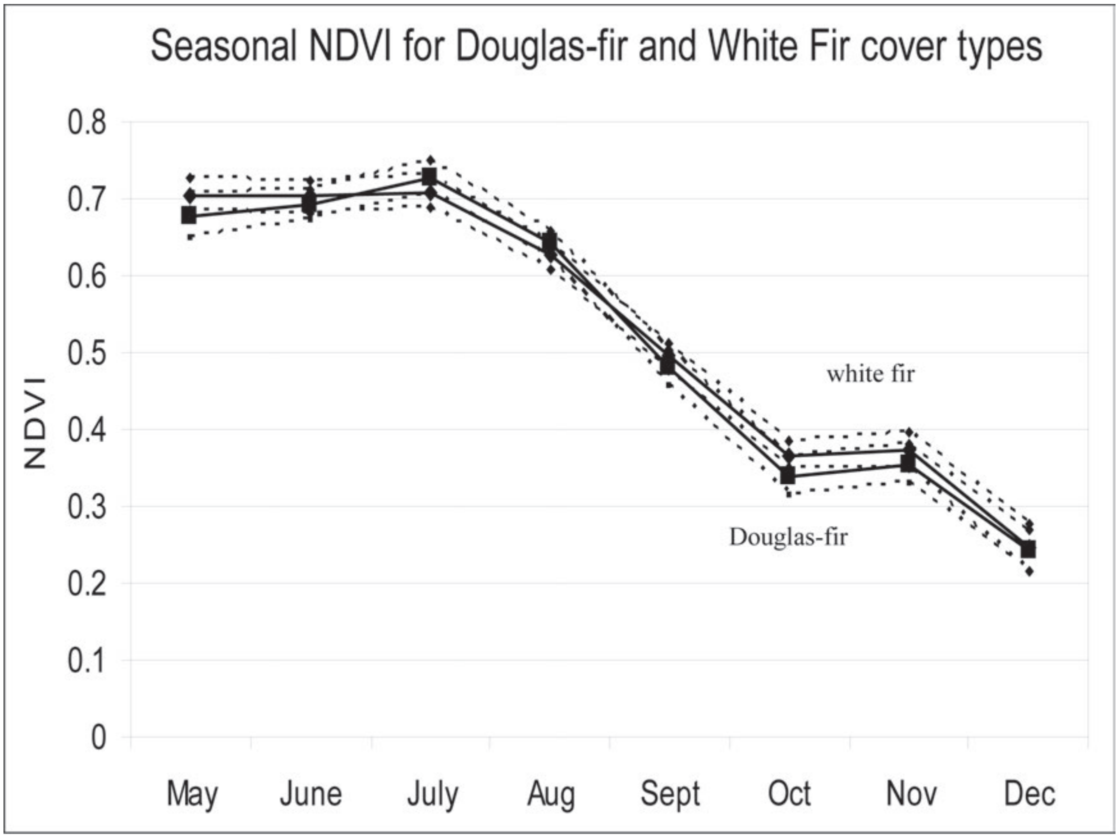

Figure 1—Seasonal normalized difference vegetation index (NDVI) spectral profiles for Douglas-fir and White Fir cover types.

errors related to misclassifications of more dissimilar vegetation types lead to greater negative impact. For this reason, both ecologists and image analysts need to critically analyze error matrices in order to fully understand and characterize the ways in which product errors may affect project objectives.

We took these lessons into consideration in the design of our LANDFIRE accuracy assessment protocol:

- Because LANDFIRE is a large-region project, we tapped into a variety of data sources and made use of existing field data to assess the accuracy of LANDFIRE Prototype products (rather than wasting time and money collecting data for and compiling a custom field database). See Caratti, Ch. 4 for details on the acquisition of data for and compilation of the LANDFIRE reference database.
- Cross-validation error matrices were generated and examined separately for both LANDFIRE Prototype regions.
- For the LANDFIRE Prototype, mappers, ecologists, and wildland fire scientists critically evaluated errors at several stages in prototype product development. These evaluations resulted in aggregation and disaggregation of classes based on the “mappability” and “model-ability” of the vegetation classes. See Keane and Rollins, Ch. 3 and Long and others, Ch. 6 for detailed descriptions of the creation of the final vegetation legends for the LANDFIRE Prototype. This expert-based process for map unit classification refinement is built into the accuracy assessment system for LANDFIRE National.

Overview of Accuracy Assessment Conducted for the LANDFIRE Prototype Project

The LANDFIRE Prototype Project involved many sequential steps, intermediate products, and interdependent processes, each involving evaluations of the accuracy of intermediate and final products. Please see appendix 2-A in Rollins and others, Ch. 2 for a detailed outline of the procedures followed to create the entire suite of LANDFIRE Prototype products.

Role of Input Data

Field data accuracy issues—Field data played a critical role in many stages of the LANDFIRE Prototype. These data were essential inputs for developing the vegetation products, percent canopy cover and height data layers, and potential vegetation data layers. See Caratti, Ch. 4 for detailed information on data acquisition for and compilation of the LANDFIRE reference database.

Described below are a number of data quality issues that needed to be addressed in the LANDFIRE Prototype.

- *Number of field plots*: For the LANDFIRE Prototype accuracy assessment, we used all field plot data that met the stringent quality-control criteria (Caratti, Ch. 4) and represented the large number of classes mapped during the vegetation mapping tasks (for details about the vegetation mapping procedures, see Frescino and Rollins, Ch. 7 and Zhu and others, Ch. 8) We used literally thousands of points for each of the two prototype regions. During this process, we recognized that some vegetation classes had limited numbers of field plots. Short of gathering additional plot information (see Keane and Rollins, Ch. 3 for LANDFIRE Prototype design criteria), there was no obvious solution to this problem. We attempted to map these rarely sampled vegetation types, even when we had limited numbers of field plots for those classes. We believe that most of these rare classes were under-represented in the resultant products.

- *Field plot geolocational accuracy*: Field plots must have accurate geolocational coordinates to geographically rectify with the many spatial databases involved in the LANDFIRE process. This was especially important during the vegetation cover and structure characterization phase of the LANDFIRE Prototype, wherein each field plot was matched with a single Landsat pixel and used in the mapping process. Any significant error in the field location coordinates has the potential to match the wrong spectral information with that particular field plot, thereby resulting in mapping error. For the prototype effort, we overlaid plot locations onto satellite imagery to determine whether there were plots that obviously did not match the imagery. While most plot locations appeared to be reasonable, we observed that many plots representing natural vegetation were actually located on major roads. When plot information was originally acquired for these sites, the actual Global Positioning System (GPS) measurements were apparently made at the road locations adjacent to the field plots, rather than within the field plots. Thus, the GPS locations did not exactly match the locations where the field measurements were made. For these sites in the LANDFIRE Prototype, a new set of geolocations was derived to better represent actual field plot locations.

  In another case, we noted (also based upon imagery assessment) that many putative shrub sites were located in obviously forested areas. We later discovered that those plots corresponded to a particular project in which the main focus was to describe shrub vegetation regardless of whether or not it represented the dominant vegetation type. These plots were consequently discarded from the prototype accuracy assessment. Both cases illustrate the need for assessing field plot information in conjunction with satellite imagery to ensure that the field information is accurately recorded.

  Moreover, it should be recognized that satellite imagery can have georeferencing errors as well. As a general rule, the coordinates of most pixels in the imagery used for the LANDFIRE Prototype are within 30 meters of the actual location – but exceptions occur. Even in the case where a pixel has slightly greater than a 15-meter error associated with it, this may be large enough to create a slight yet definite mismatch between the imagery and field information. While there is little that we can do about this problem, we at least need to recognize that some of the error term associated with the products generated will be attributable to this issue.

- *Assignment of field data into discrete vegetation classes*: One of the challenges in generating land cover maps is the stratification into discrete classes of a very complex natural world composed of multiple continuums. Regardless of which vegetation map unit system is used, many vegetation plots will represent elements of two or even more classes, and thus some plots will defy unambiguous categorization. As an example of one such problem, we mapped Juniper and Pinyon – Juniper (PJ) as two distinct classes. In nature, pinyon pine and juniper often coexist, but sometimes juniper occurs as more-or-less pure stands. We used 25 percent juniper composition as the threshold separating Juniper from Pinyon – Juniper (in other words, if a stand had 75 percent or greater basal area juniper in a stand comprised of both pinyon pine and juniper, it was called "Juniper"; whereas, if it had less than 75 percent juniper, it was called "PJ"). Analysis of seasonal spectral data indicated that many juniper stands were spectrally distinct from many of the PJ stands (fig. 2); however, significant spectral overlap existed between the two classes, as well. After decision tree classification, cross-validation accuracies indicated significant error in the classification of these two cover types (fig. 3). We believe that much of this error is attributable to the artificial boundaries imposed by the classification of a continuum.

- *Temporal correlation between field data and satellite imagery*: Disturbance such as that caused by fire, insects, or logging can alter the sites enough to cause the temporal mismatches between field data and satellite imagery that result in classification problems. For the prototype, we made use of a large volume of existing field data acquired from disparate sources (Caratti, Ch. 4), and much of the field information was acquired over a long period of time. Although information from many plots was relatively old (for example, field data acquired over a 10-year time period prior to imagery acquisition), we determined that many of these plots still contained information that was useful and relevant to the LANDFIRE Prototype. For example, plots located within reasonably intact and undisturbed forests or sagebrush lands, under normal circumstances, do not change much over a 10-year span. After completing the first prototype study in Utah, we recognized the importance of using a change-detection approach and employed such an approach in the northern Rockies prototype region to discard plot information derived from areas that changed between the times when the field information was obtained and when the imagery was acquired.

Geospatial data issues—Landsat imagery data from the MRLC 2001 consortium served as the primary source of spatial data for developing the vegetation and structure products (Homer and others 2004) (refer to

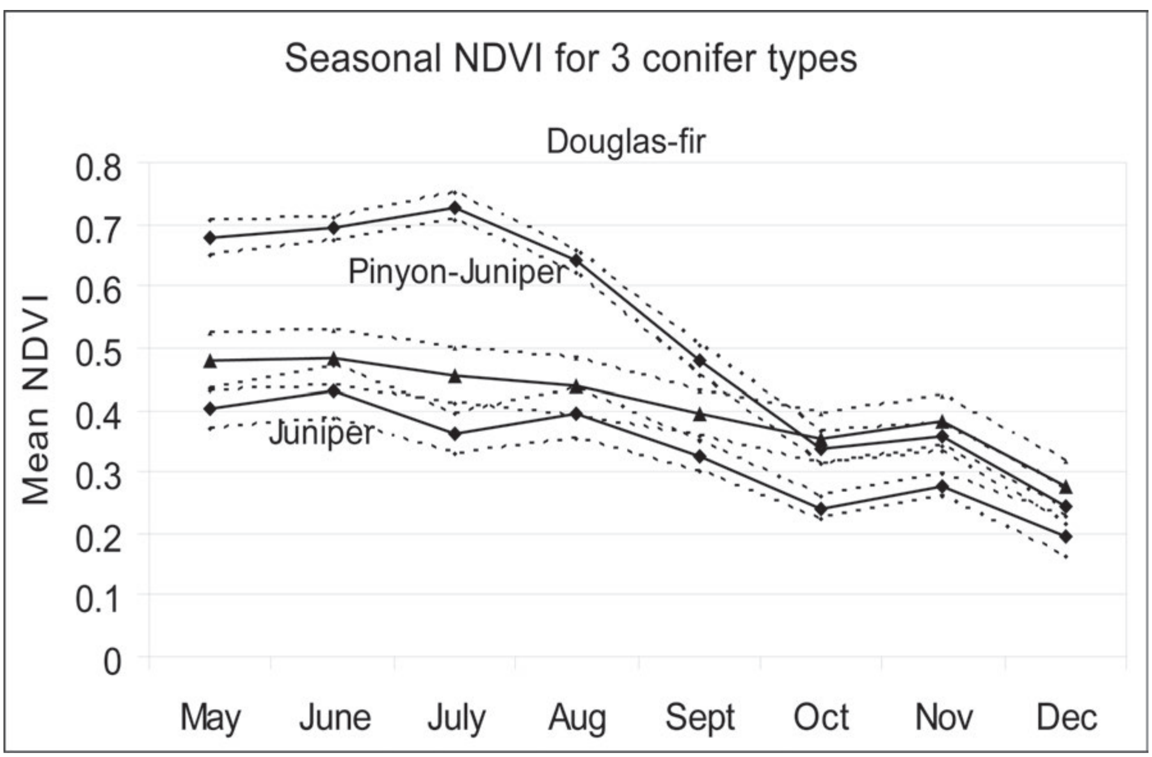

Figure 2—Seasonal normalized difference vegetation index (NDVI) spectral profiles for Douglas-fir, Pinyon – Juniper, and Juniper cover types.

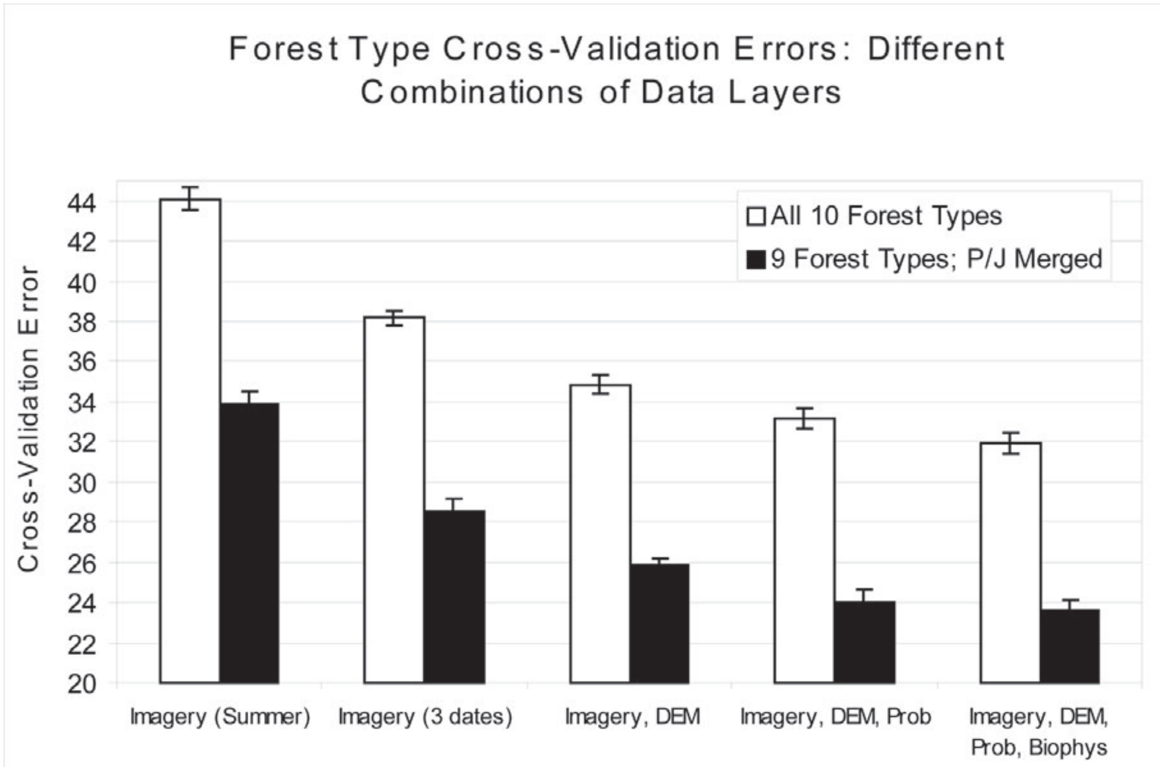

Figure 3—Cross-validation errors for forest types in the Zone 16 prototype study area as a function of different amounts of input source material. Black bars depict the effects of merging the Pinyon – Juniper and Juniper classes.

Zhu and others, Ch. 8 for further discussion regarding the imagery and ancillary data sources used for vegetation mapping in the LANDFIRE Prototype). In general, the images used for the prototype effort were the best data available during the LANDFIRE Prototype and represented three seasonal time periods (leaf-off spring, leaf-on summer, and leaf-on fall). Although the MRLC 2001 data used are of high quality, problems can arise when using any source of remotely sensed information. The foremost imagery-related problems affecting the LANDFIRE Prototype included atmospheric issues, disparate imagery acquisition dates, and geolocational problems.

- *Atmospheric issues*: Most of the acquired image scenes used in the prototype effort were of excellent quality. Even the best scenes, however, have occasional cloud and/or haze problems, which can either totally obstruct the view of portions of landscape or change the digital values enough to impact the mapping process. While not a large problem in the prototype areas, there were a few locations for which imagery quality was sub-par. These issues are inevitable and are likely to be a bigger problem in cloudier locations of the country such as the eastern United States and the upper Midwest.
- *Disparate imagery acquisition dates*: We attempted to use imagery from similar time periods as much as possible; however, due to cloud issues, optimal imagery data were not always available. Using scenes from different dates of the same year, such as using July and September data in the same "leaf-on" mosaic, resulted in problems resulting from phenological differences. Using scenes from different years, such as using one scene from 2002 and an adjacent scene from 2003, resulted in problems related to different weather patterns (for example, vegetation spectral response can be very different during wet versus dry years) and to occasional land cover changes that occurred between years. For the LANDFIRE Prototype, we attempted to minimize these problems through careful selection of scenes and use of spatial "date of acquisition" information in our decision tree and regression tree classifications.

- *Geolocational problems*: Images used in this investigation were processed using the National Landsat Archive Production System methods (USGS Landsat Website 2004). Data were corrected for terrain and projected to a standard projection (Albers Equal Area) using automated software processing. Individual pixel coordinate information was approximately 30 meters from actuality. Thus, even when field information had precise GPS coordinates, the field data were sometimes linked to the wrong pixel due to imagery registration errors. Because of technological, time, and budget constraints, we could not circumvent this problem. Registration methods needed to be consistent and automated to ensure that the process was feasible for application over the entire United States. We simply had to assume that the field data adequately characterized an area broader than the precise location of the plot and that the image pixel used was spectrally representative of its surrounding pixels. Note that in many cases, the quality-control checks performed on the field data mitigated some of these problems.

Ancillary data issues—Other sources of input information for the LANDFIRE Prototype included Digital Elevation Model (DEM) data and derivative products, 1990s NLCD land cover data (Vogelmann and others 2001), 2000s NLCD land cover data (Homer and others 2004), a suite of biophysical gradient data layers (Holsinger and others, Ch. 11; Keane and others 2001; Rollins and others 2004), and potential vegetation information (Frescino and Rollins, Ch. 7). Error terms are associated with each data type. While it is beyond the scope of this chapter to describe in detail all of the sources of errors associated with the many data layers, a few specific points should be made:

- Although not flawless, each data source used in the LANDFIRE Prototype represented the best available science and data quality.
- The source of the DEM data was the National Elevation Dataset (NED) (Gesch and others 2002). Although NED is an excellent source of digital elevation data, it came to our attention during the final stages of the prototype effort that another data source would have been more appropriate: the Elevation Derivatives for National Applications (EDNA) data set (http://edna.usgs.gov). The EDNA data represent a set of data layers derived from an earlier version of the NED. To create the EDNA data layers, the NED data were "smoothed" so that they would be better suited for hydrological modeling purposes. It should also be noted that, regardless of the source of the digital elevation model information, there are horizontal and vertical error terms associated with these data sets tracing back to the original source material. These digital elevation model data sets are regularly improved and updated.
- The 1990s and 2000s NLCD data sets were used for stratification purposes at various stages in the prototype effort, and both data sets have known error terms associated with them. See Yang and others (2001) and Homer and others (2004) for details regarding the accuracies of these products.

Accuracy of Thematic Maps

Cross-validation and points for independent validation—Accuracy assessment is an integral component of land cover mapping work. When a large number of field points are available, a reasonable alternative to generating traditional first-order accuracy estimates (see the above section *Accuracy Assessment Considerations for LANDFIRE*) is cross-validation. To create the LANDFIRE vegetation products, we employed decision tree analysis implemented within the See5 program (Quinlan 1993) using Landsat, DEM, slope, aspect, biophysical gradient, and potential vegetation data layers. The program enables cross-validation, which consists of repeated experiments in which a subset of the sample is used to train a classification model and an unseen subset is used to evaluate the model. In model runs for the prototype effort, we found that a five-fold cross-validation was appropriate. In each model run, the original field point data sets were divided into five subsets of equal size, and each subset was used to evaluate the algorithm trained using the remaining four subsets. Theoretically, this approach is not as thorough as a rigorous, statistically designed post-mapping field accuracy assessment campaign. It has been shown, however, that cross-validation can provide accuracy estimates comparable to these time-consuming and expensive methods (Huang and others 2003). See Frescino and Rollins, Ch. 7 and Zhu and others, Ch. 8 for actual accuracy results and cross-validation error matrices for the vegetation products derived for the LANDFIRE Prototype. For LANDFIRE National, we recommend reserving a set percentage of plots from the decision and regression tree analyses for independent accuracy assessment. See the *Recommendations for National Implementation* section below for details.

Field verification—Although it is not always feasible to conduct a detailed field verification and validation campaign, when possible, field visits at various stages of product development can be highly useful. Field visits, both during and after the product generation phase, provide the technical teams conducting the mapping work with a good basic understanding of the natural vegetation and ecology of the regions in which they are working. Further, field checks of particular sites to determine if they match the modeled results can be very instructive and useful for improving mapping accuracies. For the LANDFIRE Prototype, we made three separate field visits of approximately five days each. We traveled to the central Utah highlands region twice (once before mapping and once after the products were created), and we traveled once to the western Montana region (post-mapping). In all cases, images and/or maps were evaluated in the field, and actual plot measurements were made. Although not statistically rigorous, such efforts provided a better understanding of potential problem areas for future methods improvement. For example, an area of western hemlock was overestimated in the map products, and we were able to trace the overestimation back to problems in the original field sampling methods used to help generate the training data in the mapping process. Although no obvious solution to the problem was apparent, the case illustrates the importance of field visits in methods improvement. In another field activity, spectral measurements of shrub and herbaceous vegetation density were made by one team in the western Montana region to help refine shrub and herbaceous canopy cover methodology. This activity was undertaken in an attempt to improve canopy cover mapping and is being considered for the National Implementation of LANDFIRE.

Consistency checks with data from other sources—Related data sets, generated by other projects and for other applications, are often available and can be used for comparison purposes. The USGS Gap Analysis Program (GAP), for example, generates detailed vegetation maps for conservation management and planning (http://www.gap.uidaho.edu). We compared the GAP products created for the central Utah highlands prototype area with the cover type maps created for the LANDFIRE Prototype. The two sources of data compared reasonably well in some cases and less so in others (see figs. 4 and 5). It should be noted that the GAP products were created using different field databases than those used for the LANDFIRE Prototype. In addition, the vegetation map unit classification systems used were different, which limited the utility of direct, parallel comparison between the GAP products and LANDFIRE products. Although such comparisons may lack statistical rigor, they indicate where major qualitative similarities and differences exist between products and in turn may indicate which classes and regions are the most suspect. In addition, vegetation and structure products should be reviewed by regional experts whenever possible to determine whether noteworthy mapping problems exist and whether additional work is warranted. Such a review is recommended for national implementation of LANDFIRE.

Accuracy of Potential Vegetation Type and Canopy Fuel Maps

We generated potential vegetation type (PVT) data sets using decision tree software and cross-validation routines very similar to those used for generating vegetation maps. We also produced coinciding maps of confidence, which depict the relative prediction errors representing a spatial and visual representation of PVT map accuracy. See Frescino and Rollins, Ch. 7 for detailed descriptions and results of these activities. We estimated the accuracy of canopy fuel layers using regression tree procedures in which correlation coefficients were generated to measure the agreement between the predicted values and actual values. Additionally, we compared with predicted values a set of points randomly selected from the LANDFIRE reference database from each prototype zone. As in the case of PVT, we also produced coinciding maps of confidence. See Keane and others, Ch. 12 for a detailed description of canopy fuel accuracy.

Accuracy of Maps Based on Landscape Simulation Models

Accuracy evaluation of vegetation maps created from satellite imagery and ancillary data is straightforward and is based on a foundation of scientific literature (Foody 2002; Lunetta and Lyon 2004). In contrast, it is often conceptually very difficult to ascertain the quantitative accuracy of many of the products that are generated through complex modeling efforts, such as those employed to create the historical reference conditions for quantifying ecological departure in LANDFIRE. Moreover, it is difficult — if not impossible — to assign an absolute measure of accuracy to an ecological departure product because such a product represents deviation from conditions modeled under a variety of limitations in terms of baseline ecological data. Modeling assumptions, while based on the best available disturbance ecology science, may or may not be completely valid. Without the luxury of time-travel, it is very difficult to validate what the "normal" or historical vegetation condition actually was.

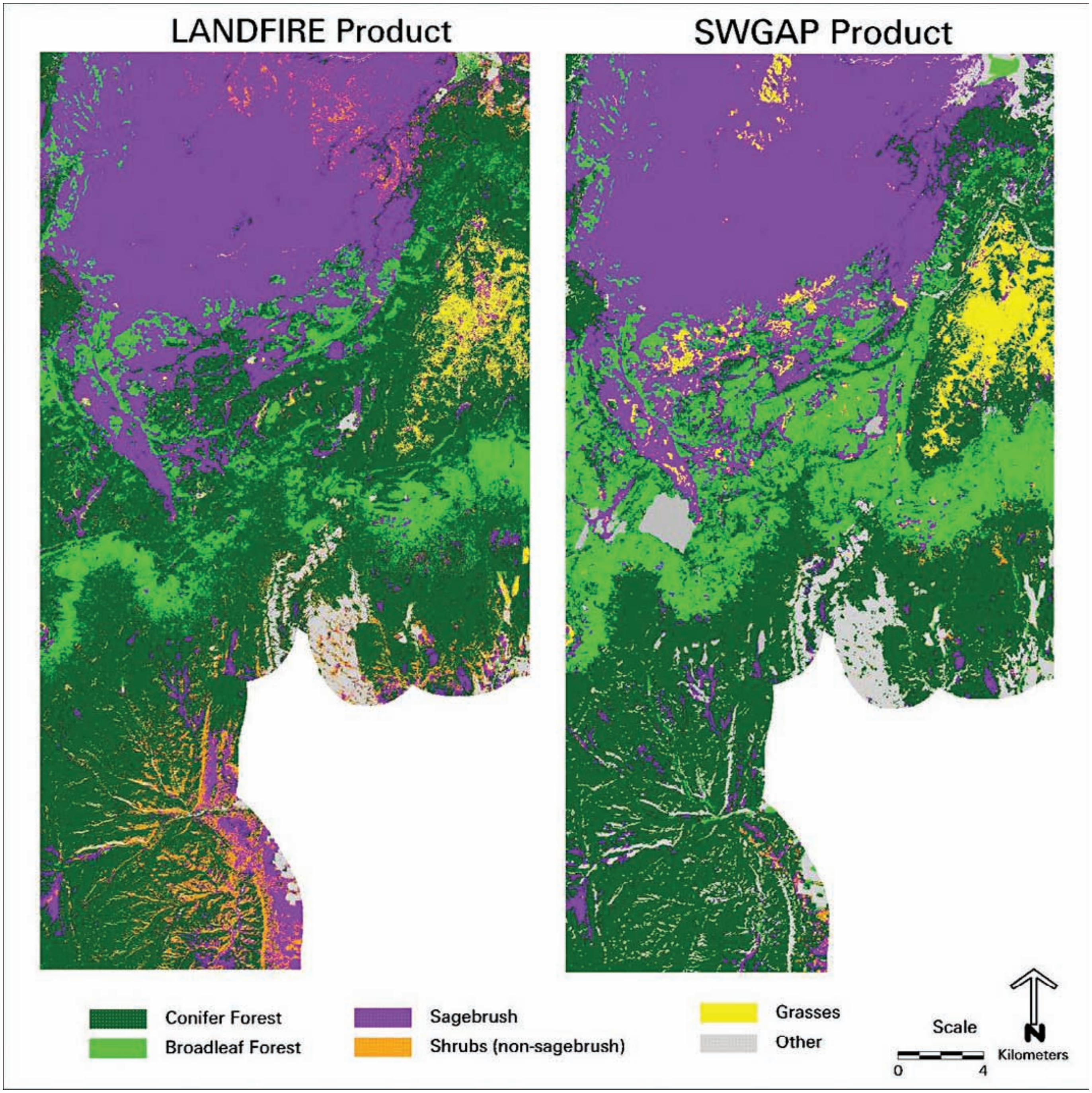

Figure 4—Comparison between a LANDFIRE vegetation type product and a product developed by the Southwest GAP Project in southern Utah. Multiple thematic classes have been combined to facilitate visual comparisons.

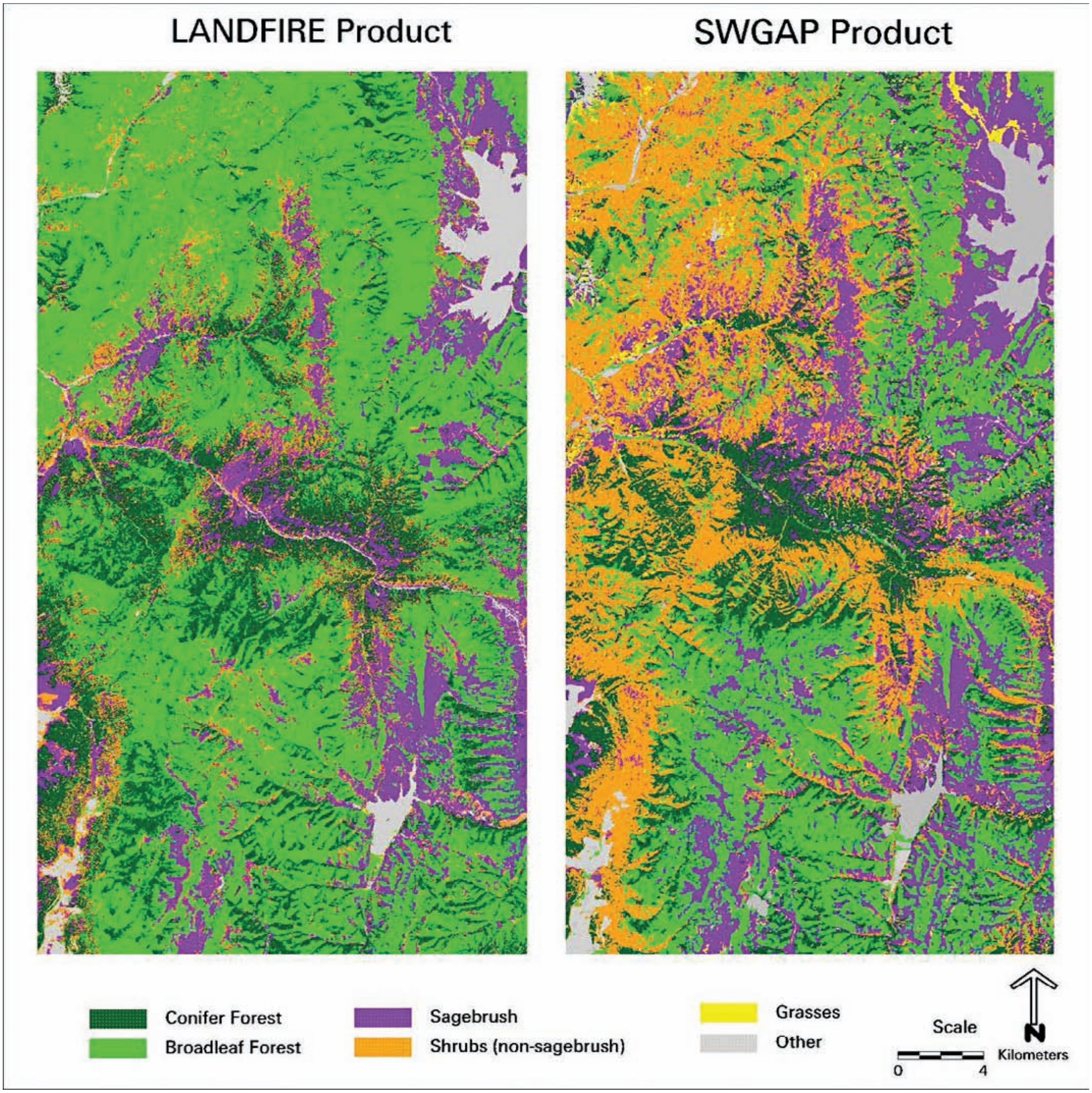

Figure 5—Additional comparison between a LANDFIRE vegetation type product and a product developed by the Southwest GAP Project in southern Utah. Multiple thematic classes have been combined to facilitate visual comparisons. Major differences between shrub and broadleaf forest classes can be traced back to differences in classification systems (Gambel oak and bigtooth maple were categorized as trees in the LANDFIRE map unit classification and as shrubs by GAP).

For accuracy assessment approaches used to evaluate LANDFIRE products based on landscape simulation models, see Pratt and others, Ch. 10 and Holsinger and others, Ch. 11. In addition, see the *Recommendations for National Implementation* section below for suggestions on improving the accuracy assessment of LANDFIRE products based on landscape simulation models.

Recommendations for National Implementation

Source Data

All source data need to be inspected carefully. This is especially true for field data and imagery, which form important foundations for much of the ensuing LANDFIRE tasks. As a matter of course, if field data used for training are inaccurate, then the resulting products will likely have lower levels of accuracy. Imagery quality can also greatly affect accuracy levels of derived products. Although optimal imagery data sets are not always available for a given location, there are usually several excellent options. It is important to ensure that the best possible imagery data sets are used. Below are some specific recommendations regarding the selection of source data.

Number of field plots—As general rule, the more field reference plots, the better. For each LANDFIRE National mapping zone, we anticipate using literally thousands of field plots in order to develop adequate characterizations. These must represent the entire range of conditions that occur throughout the mapping zones. For vegetation map unit classification development, for example, we have a target number of at least 100 plots per class. Fewer plots per class would diminish our confidence in our ability to map that class accurately and would likely result in the inadequate mapping of that particular feature. Rare classes (land cover features limited in occurrence across the landscape) are notoriously difficult to map accurately, largely because there are relatively few field plots representing these classes that can be used for training data. For national implementation of LANDFIRE, we recommend 1) generating vegetation products using all plots, 2) evaluating results, 3) determining which vegetation classes were represented by too few plots, and 4) re-running the map unit classification without these rare classes.

Field plot geolocational accuracy—Field plots with inaccurate coordinates have the potential to cause significant error in mapping results. We recommend that field plot locations be overlaid onto the imagery and that the plot locations be visually inspected to determine if attribute data for each plot are consistent with the imagery. Points located on roads or other locations clearly not characterized by the reference plot should be either omitted or shifted to the appropriate location.

Field data temporal issues—Much of the field information available for the national implementation of LANDFIRE is likely to have been acquired by various organizations over a relatively long period of time. As discussed above, inclusion of plots located in areas where the vegetation has changed between the time the field information was collected and when the imagery was acquired can cause significant mapping problems. The ideal situation is for field data and imagery to be acquired at approximately the same time, but this is impractical due to the large volume of field data necessary for product generation. One option is to discard plots with relatively old information (by imposing an arbitrary cutoff of five or more years); however, including as many plots as possible, even if some include older information, is preferable because even old plots can contain useful information. For this reason, for national implementation, we recommend using the change-detection approach developed for the western Montana prototype area. We recommend using normalized difference vegetation index (NDVI) change between 1990s and 2000s NLCD imagery to locate and isolate plots that have changed markedly over the last 10 years. If a plot is located within a region of high spectral change (based upon imagery analysis) and if the change appears to be related to a land cover change event (such as fire, logging, or insect disease) as opposed to a cloud or cloud shadow, the plot should be flagged and omitted from further analyses.

Imagery data—Imagery acquired by Landsat will likely continue to be the primary source of spatial data for developing vegetation and structure products for LANDFIRE National. The MRLC 2001 consortium, of which the LANDFIRE Project is a partner, is the best source for imagery in part because it is readily obtained and has been consistently pre-processed. Although this imagery represents the best data available, we do anticipate some issues that will need to be addressed. As with the prototype effort, we anticipate the primary imagery-related problems impacting LANDFIRE National to include atmospheric issues, disparate imagery acquisition dates, and geolocational issues (see above section *Geospatial data issues*). It is anticipated that haze and cloud problems will be especially prevalent in the eastern U.S., upper Midwest, and in the Pacific Northwest. Imagery differences related to phenological variables are also likely to impact mapping on a grander scale than was experienced in the prototype effort. When current MRLC data are deemed insufficient for LANDFIRE purposes (based upon visual inspection), additional scenes should be purchased and processed and incorporated into the mosaicking process.

Ancillary data—LANDFIRE will continue to use the best available source data for national implementation. One change that we recommend is using the EDNA data set (USGS EDNA website 2004) as the primary source of digital elevation data. These data are more refined than the data used in the prototype effort. The 1990s and 2000s NLCD data sets will continue to be used for stratification purposes at various stages of LANDFIRE National.

Accuracy of Output Products

Output product inspection—All LANDFIRE products must initially undergo an inspection phase during which the following question is asked: “Do these products make sense?” Although admittedly subjective, many errors will be caught early in the process through such inspections. If performed properly, such an initial evaluation provides a valuable safeguard that can save time and prevent the need to recreate the products.

Cross-validation and error matrices—As in the LANDFIRE Prototype, we recommend the use of cross-validation for approximating accuracies, especially for existing vegetation type and potential vegetation type. Correlation coefficients derived from regression tree analyses should be used when generating continuous variable data sets. Error matrices should be evaluated to facilitate better understanding of the strengths and weaknesses of the vegetation products. Regarding creation of the mapping models, we recommend using 5- or 10-fold cross-validation for each of the individual LANDFIRE mapping zones.

Points for independent validation—For national implementation of LANDFIRE, we recommend reserving a set percentage of plots from the decision tree and regression tree analyses solely for assessing accuracy. Note, however, that the field-referenced data used as input are collected from various projects and agencies, and thus the original source of field data cannot be considered a “random” sample of plots. Any sample of plots selected from a non-random set of points cannot be considered statistically random. Nonetheless, we have determined that withholding a limited number of points for validation purposes provides worthwhile accuracy information.

Nevertheless, we determined that it’s better to produce a more accurate set of products with imperfect accuracy information than a less accurate set of products with better known accuracy estimates. We do not want to withhold plots that would best be used for model and product development. As a compromise, we recommend that two percent of the plots be withheld from the modeling activities. These plots will then be used to estimate accuracies for aggregations of LANDFIRE mapping zones or “superzones”. We plan to merge data sets from three to four adjacent mapping zones and conduct validation activities for these regions. A target of at least 50 plots for each vegetation class per superzone provides useful information for estimating accuracies.

Stratification of accuracy assessment—In addition to providing general accuracy information at the superzone and individual mapping zone levels, we recommend providing more local estimates of accuracy nested within these other levels. This will be accomplished through spatial stratification of broad areas using biophysical gradient modeling information and other sources of spatial data and through thematic aggregation of similar vegetation types for localized regions. The process of stratifying mapping zones into zones based on the biophysical gradient layers developed for LANDFIRE (see Holsinger and others, Ch. 5) will be used as a basis to further our understanding of product errors, which in turn will enable refinement of future mapping procedures. This stratification process may facilitate the discrimination of different vegetation types with similar spectral signatures that occupy sites having very different environmental characteristics.

Field verification—As discussed above, we recommend conducting a modest level of field verification throughout LANDFIRE National. Field visits provide the technical teams with a basic understanding of the natural vegetation and ecology of the regions in which they are working, and field visits to particular sites serve to verify (or invalidate) the modeled results. Ideally, a field visit should take place at the beginning of each zone’s mapping activities for familiarization purposes, and an additional field visit should occur near the end of the mapping process to verify and refine the mapping process.

Consistency checks with data from other sources—Whenever possible, products should be compared with existing independently produced data sets. In some cases, products unrelated to LANDFIRE have been generated for certain local areas, and these can be used to help assess accuracies of LANDFIRE products. Spatial and tabular data potentially provide good general information. In addition, we recommend that LANDFIRE support the generation of local validation data sets, where appropriate.

Accuracy of Maps Based on Landscape Simulation Models

As discussed above, it is generally very difficult to ascertain the quantitative accuracy of products generated through complex landscape modeling efforts. Even so, there are some approaches suitable for assessing the validity of certain LANDFIRE modeled products, such as modeled historical fire regimes.

Although as of yet there are no examples of complete data sets representing historical vegetation conditions for the entire United States at the spatial grain of the LANDFIRE products, there are local historical data sets that can be used to “spot check” the validity of the products generated. For instance, historical aerial photographs and field-based data sets may provide useful information for assessing modeled historical fire regime products. Although not a true quantitative analysis, comparisons with historical data will likely provide information regarding the validity of the products.

As described above, it is important that the outputs from complex modeling activities be scrutinized carefully and checked for obvious flaws or deviations from expected results. As obvious as this seems, we are aware of numerous investigations in which this avenue has been neglected and in which spatial products were produced but not carefully examined. Although this type of evaluation does not yield quantitative error estimates, it can provide valuable insight regarding probable accuracies.

Finally, users of the LANDFIRE data sets should recognize that the inputs to the modeling process, while not always perfect, reflect the most accurate and current information available and are based upon ecologically sound assumptions. For these reasons, LANDFIRE products represent state-of-the-art modeling and technology and thus a significant improvement over other current options.

Conclusion

There is no single recommended procedure for deriving accuracy estimates for LANDFIRE products. Because time- and cost-related constraints, it will not be possible to conduct traditional accuracy assessments for the LANDFIRE mapping region (the entire U.S.). Yet at the same time, we recognize that evaluations of quality and accuracy increase the credibility of the final LANDFIRE products. Additionally, we can learn much by assessing error terms in the products, and this knowledge can be invaluable for future mapping and modeling endeavors. We suggest conducting a suite of accuracy assessment methods for LANDFIRE National, ranging from mostly qualitative assessments (such as the critical inspection of products, consultation with regional experts, and comparisons with existing data sets) to more quantitative analyses (such as cross-validation assessments, traditional accuracy assessments at the superzone level, and select evaluations at local levels). These combined approaches will provide LANDFIRE data users with the accuracy information necessary to facilitate the appropriate use of the data.

For further project information, please visit the LANDFIRE website at www.landfire.gov.

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Chapter 8

Mapping Existing Vegetation Composition and Structure for the LANDFIRE Prototype Project

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Introduction

Overview

The Landscape Fire and Resource Management Planning Tools Prototype Project, or LANDFIRE Prototype Project, required the mapping of existing vegetation composition (cover type) and structural stages at a 30-m spatial resolution to provide baseline vegetation data for the development of wildland fuel maps and for comparison to simulated historical vegetation reference conditions to develop indices of ecological departure. For the LANDFIRE Prototype Project, research was conducted to develop a vegetation mapping methodology that could meet the following general requirements:

- Cover types (species composition) must be characterized at a scale suitable for subsequent mapping of wildland fuel and fire regime condition class (FRCC). The vegetation map unit classification used for mapping cover types must be based on existing national systems, such as the United States National Vegetation Classification System (NVCS; Grossman and others 1998). The alliance (a community with multiple dominant species) or association (a community with a single dominant species) levels of this standard must provide a clearly defined list of map units that can be used as a basis for mapping vegetation classes that are both scaleable and representative of suitable units for modeling historical fire regimes (see Long and others, Ch. 6 for details on the LANDFIRE vegetation map units).
- The mapping of existing vegetation structure must be based on the relative composition of forest, shrub, and herbaceous canopy cover and average forest, shrub, and herbaceous canopy height. Although structural stages are discrete map units describing unique combinations of canopy cover and canopy height by life form, mapping individual canopy cover and height variables as continuous variables is desired to provide additional information for mapping and modeling vegetation and flexibility for setting threshold values.

The task of mapping existing vegetation is interconnected with several major tasks performed in the LANDFIRE Prototype Project. The mapping of existing vegetation requires attribute tables developed from the LANDFIRE reference database (LFRDB) (Caratti, Ch. 4), satellite imagery acquisition and processing, the development of a vegetation map unit classification system (see Long and others, Ch. 6), the development of a biophysical settings stratification (Frescino and Rollins, Ch. 7), and the modeling of environmental gradient layers (Holsinger and others, Ch. 5). The design and testing of the vegetation mapping methodology have substantial influences on the outcome of the overall project because accuracies of subsequent products (such as maps of wildland fuel) are a function of the accuracy of mapped vegetation types and structure. In this chapter, we discuss the design features of the existing vegetation mapping component of LANDFIRE and present

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results of the prototype. We conclude the chapter with recommendations for the national implementation of a consistent vegetation mapping effort.

Technical Problems

Significant technical limitations exist regarding achieving desired accuracies in the mapping of vegetation types and structure variables over broad areas. In the LANDFIRE Prototype, accuracies were affected by the spatial resolution, geographic extent, and information content defined by the project's objectives. The U.S. Geological Survey (USGS) Gap Analysis Program demonstrated the feasibility of mapping many existing vegetation cover types at the regional scale; however, methodologies have been inconsistent between regions (Eve and Merchant 1998). In addition, the mapping of forest canopy cover using imagery and regression techniques has been routinely performed for the operational mapping of vegetation structure variables (Huang and others 2001). Beyond that, however, literature reporting success stories regarding the mapping of vegetation structure using imagery is scant.

We conducted a prototype study to test a methodology for mapping vegetation cover types and structure variables. The three central objectives of the study were to:

- test an adaptable approach for mapping existing vegetation types and canopy structure at a 30-m resolution for the entire prototype area;
- develop digital maps of existing vegetation types and structural stages and conduct an accuracy assessment for the vegetation deliverables; and
- document research findings and limitations to the consistent mapping of existing vegetation composition and structure.

Specifically, this study tested a vegetation mapping protocol that met the design criteria and guidelines of the LANDFIRE Project (Keane and Rollins, Ch. 3). Further, this study investigated the limitations of using data contained within the LANDFIRE reference database (Carrati, Ch. 4) as training data and the applicability of satellite and ancillary data in meeting LANDFIRE's objectives. For vegetation modeling and wildland fuel mapping, the LANDFIRE Prototype Project required a structural stage map classified on the basis of mapped canopy cover (closed and open) and canopy height (high and low) by forest, shrub, and herbaceous life forms. We attempted to generate continuous maps of vegetation height and cover to maximize the utility of these products in a variety of applications.

As described in Rollins and others (Ch. 2), the LANDFIRE Prototype Project was conducted in two mapping zones: Zone 16, located in the central highlands of Utah and covering approximately 4 million ha of forest ecosystems (57 percent of the total land cover) and 2.5 million ha of shrub and herbaceous ecosystems (35 percent of the total land cover); and Zone 19, located in the northern Rocky Mountains of western Montana and northern Idaho and covering approximately 5.4 million ha of forest ecosystems (47 percent of the total land cover) and 5 million ha of shrub and herbaceous ecosystems (44 percent of the total land cover).

Literature Review of Vegetation Mapping

Similar to other natural science problems, the regional-scale mapping of vegetation types and structure variables carries unique technical and organizational challenges (Gemmell 1995). Spatial variations of vegetation types and structure are generally not characterized by unique spectral signatures, as captured by conventional broadband optical sensors (Kalliola and Syrjanen 1991; Keane and others 2001). Although significant improvements can be made by using specialized sensors, such as hyperspectral spectrometer or canopy lidar, data from such sensors having desired spatial resolutions are not available at national or regional scales. The associated enormous data volumes and high costs (in time and labor) make these technologies impractical for large-area applications at the present time.

Various techniques exist for modeling and estimating vegetation type and canopy structure (particularly percent forest cover); these include physics-based canopy reflectance models, empirical models linking ground-referenced data to satellite imagery, spectral mixture analysis, neural networks, and direct measurement using lidar and interferometric synthetic aperture radar. Each of these approaches has limitations in large-area applications, such as those related to cost and consistency. However, recent applications using the classification and regression tree (CART) approach (Breiman and others 1984) have been found to overcome many such limitations, provided sufficient amounts of field and geospatial data are available. Recent studies (Friedl and others 2002; Huang and Townshend 2003; Mahesh and Mather 2003; Yang and others 2003) have demonstrated the utility of CART techniques in mapping land cover, estimating species distribution, modeling percent forest canopy cover, and computing imperviousness at a 30-m grid resolution for large areas and even for the United States. Although CART techniques require relatively little human decision-making during algorithm executions, it is important to note that, ultimately, the knowledge scientists have acquired through studying vegetation patterns and attributes enhances the development mapping models to produce the most accurate results possible. Computer classifiers, regardless of their sophistication, are no substitute for scientists' understanding of the patterns, attributes, and conditions of existing vegetation and associated ecological processes.

Environmental data layers (such as elevation) are important predictor variables for characterizing vegetation patterns and attributes and for stratifying the distribution of vegetation along environmental gradient lines (Balice and others 2000). The use of spectral bands in combination with topographic data (for example, digital elevation models (DEM), slope, and aspect) is common in many land cover and vegetation mapping applications. However, topographic data capture only a part of the overall environmental factors that determine the establishment, growth, distribution, and succession of plant species and associations. The incorporation of a more complete set of environmental gradient layers into the mapping of existing vegetation should lead to increased predictive power and thematic accuracy (Keane and others 2002; Rollins and others 2004). Keane and others (2002) discuss techniques for deriving an entire set of climate, soil, and ecological gradient layers using interpolated weather observations in conjunction with topographic and soil databases and also describe the advantages of using such biophysical gradients in combination with remote sensing and field data to map vegetation, wildland fuel, and general ecosystem conditions.

In addition to the development and use of gradient variables, Keane and others (2001, 2002), Keane and Rollins, Ch. 3, and Rollins and others (2004) also suggest an approach for developing site-specific biophysical settings maps by mapping stable, late-seral communities as a function of certain climate, topographic, soil, and ecological gradients. This mapped "potential" vegetation can be used as a stratification tool in mapping actual vegetation distribution by constraining the distribution of cover types to those geographic strata where growth of the cover types' dominant species is ecologically possible.

Methods

The LANDFIRE Prototype Project involved many sequential steps, intermediate products, and interdependent processes. Please see appendix 2-A in Rollins and others, Ch. 2 for a detailed outline of the procedures followed to create the entire suite of LANDFIRE Prototype products. This chapter focuses specifically on maps of vegetation composition and structure, which served as important precursors to maps of wildland fuel and ecological departure in the LANDFIRE Prototype Project. Figure 1 outlines the technical approach used in LANDFIRE Prototype vegetation mapping and illustrates the data flow between several technically challenging tasks. Details of these tasks are described below.

Satellite Data Acquisition and Processing

The LANDFIRE Project partnered with the Multi-Resolution Land Characterization (MRLC) Consortium (Homer and others 2004) to facilitate the acquisition and processing of Landsat imagery. The consortium has completed the acquisition and processing of a full set of Landsat imagery for the United States with a minimum of three cloud-cover dates (circa 2001) for each pixel corresponding to phenological cycles of leaf-on, leaf-off, and spring green-up. Huang and others (2002) describe the steps involved in processing the MRLC satellite imagery, including terrain-corrected geometric registration and radiometric calibration using at-satellite reflectance models, calculations of normalized difference of vegetation index (NDVI), and tasseled cap transformations. The MRLC Consortium-sponsored development of the National Land Cover Dataset (NLCD) includes general land cover map units such as forest, agriculture, water, and urban areas mapped at a 30-m resolution (Homer and others 2004). The acquisition and processing of satellite imagery and the mapping of NLCD land cover map units were conducted for mapping zones, which were loosely delineated along major ecological regions. The LANDFIRE central Utah highlands and northern Rockies prototype areas were examples of these MRLC map zones.

The LANDFIRE Prototype Project had access to the following data layers from the MRLC catalogue for the Utah and northern Rockies prototype areas: 10 spectral bands for each of the 3 Landsat seasonal acquisitions (6 original spectral bands excluding the thermal band, 3 tasseled cap transformation bands, and 1 NDVI band) and land cover classes mapped to Anderson's Level 1 land cover classification (Anderson and others 1976). Using these data as a starting point, we mapped forest, shrub, and herbaceous cover types and structure attributes. These maps formed the foundation for mapping wildland fuel and fire regime characteristics (Holsinger and others, Ch. 11; Keane and others Ch. 12).

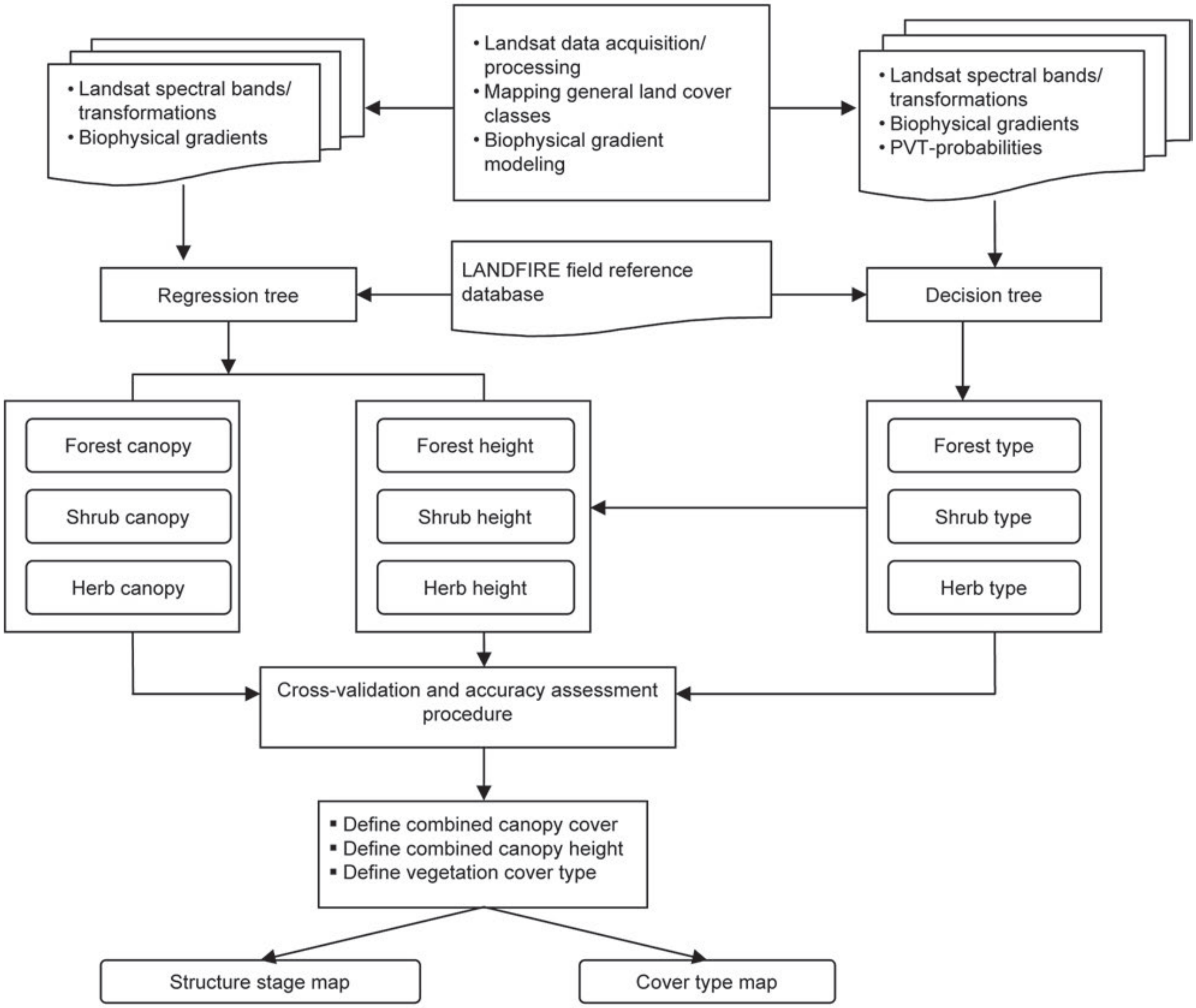

Figure 1—Flow diagram of the methodology used for mapping cover type and vegetation structure in the LANDFIRE Prototype Project.

Use of Biophysical Gradient Variables and Potential Vegetation Maps

In addition to the spectral predictor variables discussed above, the LANDFIRE existing vegetation mapping task incorporated two ancillary data sets that functioned differently in the mapping process. One was a suite of biophysical gradient layers developed as a set of intermediate LANDFIRE products with input from weather, topographic, and soil databases (Holsinger and others, Ch. 5: table 6). Table 1 lists the biophysical gradient variables used in the prototype for mapping existing vegetation; these represent a winnowed set of the entire suite of variables produced for the LANDFIRE Prototype. Biophysical gradients were used in the mapping process to provide a geographic context for the ecological processes that control establishment, growth, and distribution of vegetation communities.

The second data set was a potential vegetation type (PVT) map with attributes describing the probability of specific cover types existing in each PVT. This database was derived by calculating the distribution of cover types within individual PVTs by intersecting the plots contained in the LFRDB with the PVT map (Keane and Rollins, Ch. 3; Frescino and Rollins, Ch. 7). Conceptually, by using the PVT and cover type probability

Table 1—Biophysical and topographic layers used in the LANDFIRE vegetation mapping process.

Symbol	Description	Unit	Source data
SRAD	Daily solar radiation flux	$KW/m^2/Day$	Weather and topographic data
Tmin	Daily minimum temperature	C°	Weather and topographic data
Tmax	Daily maximum temperature	C°	Weather and topographic data
Tnight	Daily average nighttime temperature	C°	Weather and topographic data
Dday	Degree days	C°	Weather and topographic data
PPT	Daily precipitation	cm	Weather and topographic data
RH	Relative humidity	%	Weather and topographic data
PET	Potential evapotranspiration	kgH_2O/yr	Weather and topographic data
AET	Actual evapotranspiration	kgH_2O/yr	Weather, topographic, and soil data
GSWS	Growing season water stress	-Mpa	Weather, topographic, and soil data
PSI	Soil water potential	-Mpa	Weather, topographic, and soil data
KDBI	Keetch-Byram drought index	Index	Weather database
SWF	Soil water fraction	%	Weather, topographic, and soil data
Sdepth	Soil depth to bedrock	cm	Soil and topographic data
LAI	Potential leaf area index	Index	Landsat spectral data
DEM	Digital elevation model	m	National Elevation Database
Slope	Slope	%	National Elevation Database
Aspect	Aspect	Azimuth	National Elevation Database
POSIDX	Topographic position index	Index	National Elevation Database

information in the mapping of vegetation cover types, we implemented a stratification that constrained cover types to the geographic areas where cover types were ecologically possible. Sites (pixels) where certain cover types were not likely to occur would have low probabilities; therefore, these cover types were less likely to be predicted for these pixels. Each cover type was associated with a probability distribution map. The probability layers were implemented in the mapping process much in the same way as the biophysical gradient layers and satellite imagery.

Vegetation Map Unit Classification

Two different approaches were used in the development of the vegetation map unit classification systems for the prototype mapping zones. For the central Utah mapping zone, we formulated the map unit classificaton based on an overall understanding of the presence of vegetation alliances and associations (Long and others, Ch. 6). For the northern Rocky Mountains prototype area, we examined and summarized the LFRDB to form the basis for the vegetation map unit classification. Brohman and Bryant (2005) have described these approaches as the "top-down" and the "bottom-up" approaches, respectively. Long and others (Ch. 6) discuss the criteria and factors used in developing the LANDFIRE vegetation map unit classification systems, the lessons learned in applying them, and recommendations for a national approach to vegetation map unit development.

We were concerned with two technical issues when evaluating the map unit classifications of existing cover types for the prototype: 1) whether each cover type was sufficiently represented by an adequate number of field-referenced data from the LFRDB and, if not, how such "rare map units" should be treated and 2) whether some cover types (such as the Juniper cover type versus the Pinyon – Juniper cover type) would be floristically or ecologically difficult to separate in spectral, biophysical, and geographical domains. The technical issues were considered in the context of four guidelines defined at the beginning of the LANDFIRE Prototype Project: a map unit, whether it is a cover type or a fuel model, must be identifiable, scalable, mappable, and modelable (Keane and Rollins, Ch. 3). Because the prototype study areas were the first mapping zones to be mapped under the LANDFIRE design criteria and guidelines, we were unsure whether the map unit classification systems could perform consistently across different geographic areas.

Reference Data

Caratti (Ch. 4) describes in detail the compilation of the LFRDB for the prototype. The compilation of the LANDFIRE reference database relied on the coordination of three separate and independent efforts: 1) the cooperation and support from the U.S. Forest Service (USFS) Forest Inventory and Analysis (FIA) database collected nationwide on permanent inventory plots (Smith 2002); 2) the collection and processing of existing field data from all land management units such as Bureau of Land Management districts or national parks; and 3) the acquisition of new, supplementary field data from areas where there were no or not enough existing data (for example, various western rangelands in the United States do not currently have adequate field data collection programs).

Because the LFRDB was compiled from various sources collected for different purposes, information gleaned from the LFRDB was highly variable in terms of sampling design. The FIA data represented the most consistent information for forest cover types and canopy height. Rangeland field data usually contained cover type labels, but structure information was rare. In addition, reference data for mapping forest canopy cover were generated by calculating the number of forest cells within a 30-m cell using either high-resolution satellite data (spatial resolution of 1-m or better) or digital orthophotographs (Homer and others 2004).

Quality-control procedures were conducted as a part of the existing vegetation mapping process to detect problems and errors inherent in field-referenced data derived from disparate sources. We assumed that these procedures would identify most existing data problems but would not identify and eliminate all problems. These procedures were as follows:

Detecting outdated field data—Many field plots measured in years past were considered useful if the dominant species had not changed. A substantial number of plots, however, had undergone major disturbances such as fire or logging. We therefore computed the differences between the 1992 and 2001 Landsat NDVI values to flag field plots with conditions that had potentially changed during that 10-year period.

Detecting field data with erroneous geographic coordinates—We identified major geo-coding problems such as coordinates located on roads or located out of mapping areas. We visually examined plot locations overlaid with road networks and general land cover maps (such as NLCD maps).

Detecting field data with major coding errors—We detected such problems by overlaying field data on raw satellite imagery and by sorting variables according to major cover types. For example, if a field plot coded as *sagebrush* was located in the center of an otherwise intact *forest* polygon, or if a shrub plot had a height value taller than that of forest plots, such plots were flagged.

Reducing spatially clumped field plots—The LFRDB contains field data that come from different sources and are collected with different objectives, which occasionally results in spatially clumped plot information. In order to produce a spatially well-distributed and balanced data sample, we sub-sampled clumps of the available data to result in a more even distribution of field data.

The use of these quality-control procedures resulted in the exclusion of a number of available field plots from either the mapping or validation processes. This led to a total of 6,177 field plots (1,809 FIA forest plots and 4,368 non-FIA forest and rangeland plots) for Zone 16 and 7,735 field plots (1,993 FIA forest plots and 5,742 non-FIA forest and rangeland plots) for Zone 19 to be used for subsequent training or accuracy assessment. These numbers differ slightly from other applications of the LFRDB in LANDFRIE mapping because, based on objectives, each mapping effort implemented its own quality control procedure. Although all of the plots contained LANDFIRE cover type labels, only subsets of plots from the LFRDB had attributes of canopy height and canopy cover (table 2). In addition, ten percent of the field data points available for each of the cover type and structure mapping tasks were withheld from the mapping process for the purpose of accuracy assessment (Vogelmann and others, Ch. 13).

Mapping Algorithms

Classification and regression tree algorithms have demonstrated robust and consistent performance and advantages in integrating field data with geospatial data layers (Brown de Colstoun and others 2003; Friedl and Brodley 1997; Hansen and others 2000; Joy and others 2003; Moisen and others 2003, Moore and others 1991; Rollins and others 2004). Nonparametric CART approaches recursively divide feature space into many subsets in a hierarchical fashion to achieve the best overall model performance (lowest error and highest R^2, derived using a cross-validation technique). For this study, we adopted the classification tree algorithm to map vegetation types as discrete map units and the regression tree algorithm to map canopy cover and canopy height as continuous variables using two related

Table 2—Numbers of field reference plots in each mapping zone used in either mapping or accuracy assessment and corresponding to various map products. Forest canopy cover mapping relied on imagery of high spatial resolution instead of field reference plots.

	Mapping zone	Number of cover types	Cover type plots	Canopy cover plots	Canopy height plots
Forest	16	10	1,809	N/A	1,809
	19	14	1,993	N/A	1993
Shrub	16	14	1,595	2,120	1,698
	19	15	1,788	1,788	989
Herbaceous	16	7	300	2,263	1,311
	19	8	597	597	282

commercial applications: See5 (classification trees) and Cubist (regression trees) developed by Quinlan (1993). The mapping models were trained on the compiled data set of spectral bands and biophysical ancillary variables listed in table 1 and cover type and structure variables from the LFRDB.

Vegetation Database Development

Training vegetation mapping models—The creation of the CART-based algorithms for mapping existing vegetation involved several steps: 1) exploration of general data such as correlation analyses and plotting of cover types from the LFRDB against predictor layers, 2) iterations of CART algorithm runs to determine the adequacy of training data and other biophysical layers, 3) visual evaluation of classification and regression trees and final output maps, 4) generation of cross-validation statistics as an initial indicator of map accuracies, and 5) development of vegetation maps by applying the final mapping models. As mentioned above, we withheld data from 10 percent of available field reference plots for accuracy assessment and used the rest of the field plots for training the CART algorithms. We ran classification tree or regression tree classifiers, depending on whether the mapped theme was categorical or continuous, and generated 10-fold cross-validation statistics. Results of the cross-validation were used to determine the quality of training data and the performance of the predictor layers, but not to assess the final accuracy of resulting maps.

Determination of rare and similar map units—Although the LANDFIRE Prototype Project vegetation map unit classifications were developed to meet specific design criteria and guidelines (Keane and Rollins, Ch. 3; Long and others, Ch. 6), two technical questions arose during the mapping of existing vegetation: how to treat 1) rare cover types and 2) spectrally and biophysically similar cover types. We considered a cover type to be rare if it was supported with fewer than 30 reference plots, and those plots were not concentrated in one general location. We retained a rare map unit in the overall mapping process if the resulting spatial pattern made sense (such as when a riparian cover type followed river patterns) and if retaining the map unit did not result in a significant drop in accuracy. Otherwise, the rare map unit would be omitted. Additionally, we decided, based on differences in historical disturbance regimes, to keep cover types that were biophysically and spectrally similar (such as Pinyon – Juniper) separate, even though merging the cover types would significantly improve overall map accuracy.

Stratifications by life form—During the mapping of these vegetation attributes, the question arose as to whether the cover types and structural stages should be constrained by their respective forest, shrub, and herbaceous life forms; that is, we questioned whether a given pixel could be assigned more than one life form for cover type, height, and canopy designations. Multiple life form assignments provided flexibility for the characterization of wildland fuel. Such flexibility would also benefit other potential applications of LANDFIRE data, such as insect and disease or biomass studies. In the process of LANDFIRE vegetation mapping, we therefore modeled each pixel independently for each of the three life forms (forest, shrub, and herbaceous; fig. 1).

Product Validation Plan and Accuracy Assessment

The LANDFIRE accuracy assessment is described in detail in Vogelmann and others (Ch. 13). We tested the approach in which ten percent of the field data points available for cover type mapping were withheld from the mapping process for the purpose of accuracy assessment but found that the approach did not work well because of the uneven availability of field data in support of different cover types in the map unit classification. For several cover types in each of the mapping zones, the amount of data withheld in the 10 percent sample was too low to be statistically meaningful. As the result, we reported overall accuracies for cover types using the results of 10-fold cross-validations. For structure variables, we used a set of independent plots to assess statistical accuracy using regression techniques. This afforded us the opportunity to examine the behaviors of mapping structure variables versus those of categorical variables. Forest canopy cover, mapped with fine-resolution imagery as training data, would be assessed with both a sample of withheld reference points generated from the fine-resolution imagery as well as field estimates obtained from the use of digital cameras equipped with fisheye lenses.

Results

Maps of Cover Type and Structural Stage

We applied the vegetation mapping approach described above to the central Utah and northern Rockies prototype areas. Spectral imagery, biophysical gradients, PVTs, and probabilities were used together with field plot data to produce maps of forest, shrub, and herbaceous cover types, as well as canopy cover and canopy height by life form.

Accuracy of LANDFIRE Prototype Vegetation Mapping

We reported accuracy assessments using a cross-validation approach for cover types by life form (table 3) and by withholding field data for the structure variables by life form (table 4). For cover types, only overall accuracies were reported. For structural stages, R^2 values were variable and ranged from relatively consistent (for forest canopy cover and height) to relatively inconsistent (for shrub and herbaceous canopy cover and height). This variability indicates that forest structure may be mapped reasonably as a continuous variable, whereas consistency and accuracy would be questionable when mapping shrub and herbaceous structure as continuous variables. However, when evaluated as two-class variables (either as closed and open canopy cover or high and low canopy height), results showed that the same shrub and herbaceous structure can perform as consistently and accurately as categorical variables.

Discussion

Analysis of Mapping Consistency for Vegetation Types and Structure

In general, we found that the approach described above for mapping existing vegetation characteristics effectively met LANDFIRE requirements, which was a difficult objective to achieve due to the large number of vegetation map units, reliance on existing field-referenced data, the task of characterizing vegetation structure, and the requirement for a nationally consistent methodology. For the moderately detailed vegetation map unit classification, mapping accuracies of 60 percent or better were achieved at a 30-m spatial resolution.

We explored the mapping of more than two map units for structure variables. For example, we mapped herbaceous height to three map units (0 to 0.5 m, >0.5 to 1 m, and >1 m), shrub height to four map units (0 to 0.5 m, >0.5 to 1 m, >1 to 3 m, and >3 m), and forest height to four map units (0 to 5 m, >5 to 10 m, >10 to 25 m, and >25 m). The tests yielded independent overall accuracies of 73, 61, and 82 percent for herbaceous, shrub, and forest height, respectively. From these results, we concluded that grouping continuous values of the structure variables into several discrete map units would be an acceptable and rational alternative methodology for national implementation of the LANDFIRE methods. Use of this alternative methodology would require the development of a consistent national structural stage map unit classification.

Table 3—Cross validations (10 percent withheld, ten-fold repetitions) conducted separately by mapping zones and by forest, shrub, and herbaceous life forms.

Life form	Mapping zone	Number of classes	Cross validation
Forest	16	10	67
	19	14	64
Shrub	16	14	62
	19	15	68
Herbaceous	16	7	60
	19	8	56

Table 4—Accuracy assessments conducted separately for two structure variables by life forms and map zones. Overall accuracy (OA) was obtained by using holdout withheld field plots (n) that were set aside based on quality and distribution of the total available field plot data (N). Structure variables are treated as both continuous variables measured with the R2 statistic and two-class categorical variables for overall accuracy (OA). The Ttwo canopy cover classes of canopy cover are closed (≥40%) and open (<40%); for canopy height they classes are high (≥10m, 1m, 0.24m) and low (<10m, 1m, 0.24m) for forest, shrub, and herbaceous life forms, respectively.

	Map	Canopy cover			Canopy height		
Life form	zone	n/N	R^2	Overall accuracy	n/N	R^2	Overall accuracy
Forest	16	1,272/20,000	0.78	0.92	220/2204	0.58	0.88
	19	1,200/20,000	0.88	0.89	127/5,541	0.56	0.78
Shrub	16	125/1,253	0.41	0.74	107/1,073	0.36	0.85
	19	119/1,788	0.59	0.79	81/989	0.65	0.86
Herbaceous	16	18/182	0.37	0.71	15/280	0.04	0.86
	19	126/597	0.58	0.69	75/182	0.63	0.70

Consistency in field sampling and data collection affects the consistency of mapping vegetation characteristics. Of the three types of reference data used in mapping existing vegetation, cover type and canopy height values can generally be identified or measured consistently in the field. Canopy cover, on the other hand, can be difficult to measure in the field. This issue does not affect the measurement of forest canopy cover values because training data are derived from high-resolution (1 m or better) imagery by calculating numbers of high-resolution forest pixels within each 30-m Landsat pixel. The use of inconsistently estimated canopy cover values as training data, however, can potentially affect the mapping of shrub and herbaceous canopy percent cover (as happened during the prototype). Shrub and herbaceous canopy results from the two prototype mapping zones were reasonable (table 3), but difficulties in consistently estimating canopy cover in the field indicated that we needed to further research new or alternative methods for mapping shrub and herbaceous canopy cover.

The results of this study may be attributed, in part, to the use of ecologically significant ancillary data layers, which accounts for a moderate but nonetheless significant increase in accuracy (ranging from 1 to 9 percent). The development of biophysical gradient layers and PVT probabilities follows a standardized process for all mapping zones. However, for any given area, satellite reflectance can vary significantly for the same cover type with different canopy cover percentages (either due to land management practices or regeneration stages) or appear similar for different vegetation types or different structural stages during certain seasonal periods. Different cover types or structural stages, however, should respond consistently to the effects of biophysical gradient variables such as soil depth or potential evapotranspiration (PET); this addition of information from the biophysical gradient variables increases the likelihood that these map units will be discriminated by mapping algorithms. For example, one might expect Engelmann spruce (*picea engelmannii*) to grow in relatively deep soil on cool, north-facing sites with low PET, regardless of whether it is found in Zone 16 or Zone 19. Therefore, the incorporation of biophysical and PVT data in the mapping process should contribute to enhanced consistency and thematic accuracy in mapped existing vegetation across the United States.

Even though the existing vegetation maps shown in figures 2 and 3 characterize the vegetation composition of all life forms, it should be noted that each life form was mapped independently, by design, for cover type and structure. Modeling life forms independently preserves the possibility of more than one mapped life form per pixel (in other words, allows for probabilities of multiple canopy layers within a pixel) to improve fuel mapping and enhance the range of the data's ecological applications. However, mapping approaches should be carefully considered when comparing or merging these separate data sets. For example, a final map of cover types may look different depending on the order of precedence between forest, shrub, and herbaceous cover and the threshold values used in defining the life forms (for example, a pixel with 10 percent or greater forest canopy cover may be considered as forested land). It is important that precedence and thresholds be applied uniformly between mapping zones for consistency.

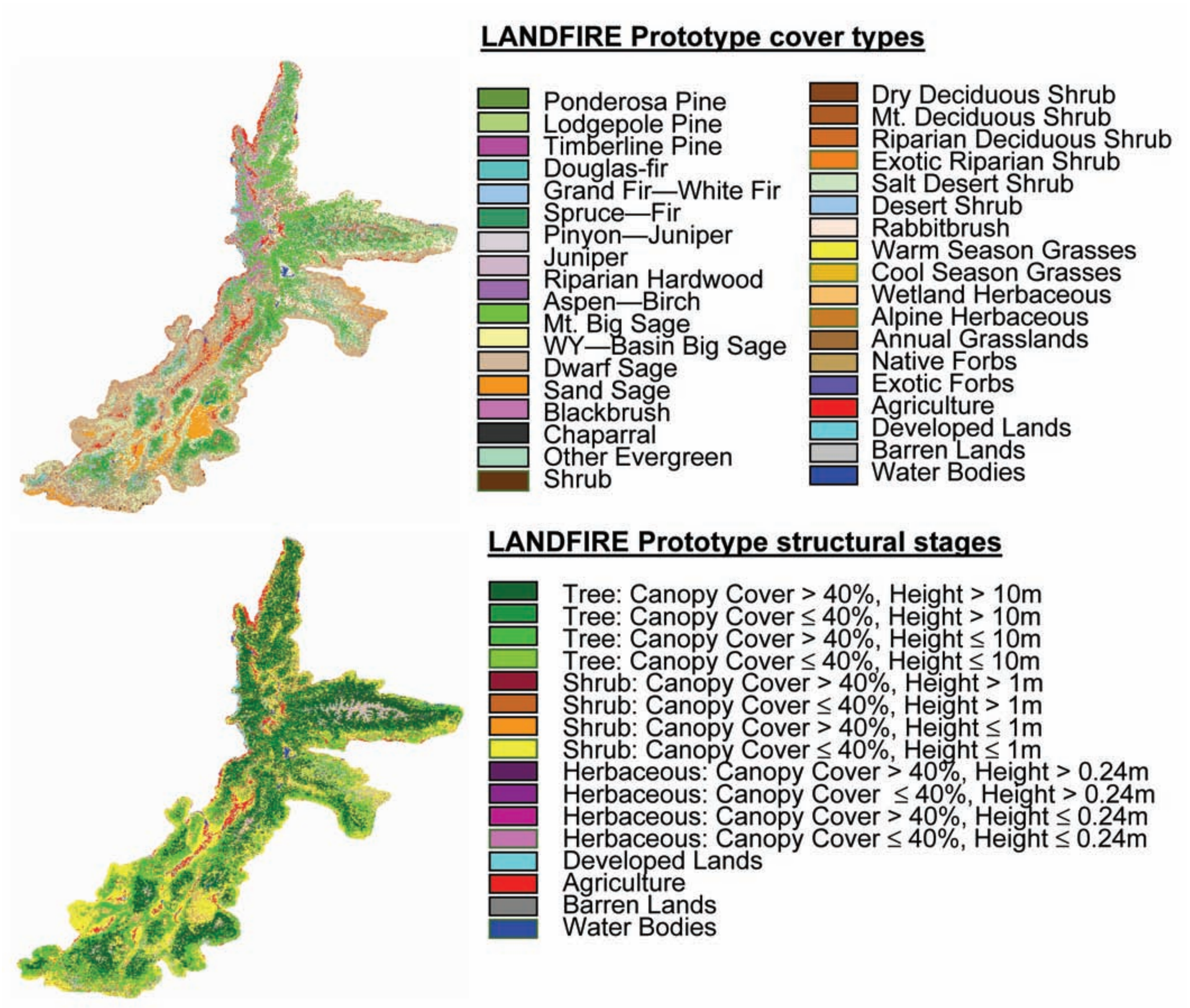

Figure 2—LANDFIRE Prototype cover type (top) and structural stage (bottom) maps for Zone 16. The cover type map is compiled from separate forest, shrub, and herbaceous cover type maps, whereas the structural stage map is grouped from continuous maps of height and cover for display purposes.

Factors that Affect Mapping Accuracies

Several factors should be considered when examining the accuracy estimates for maps of cover types and structure. First, the mapping and accuracy assessment of cover type and structure variables by life form were conducted based on field-referenced databases of different sizes and data collected throughout the study areas using a variety of sampling strategies. As would be expected, vegetation mapping was sensitive to the availability of field data. Test results showed that the number of field-referenced plots used for mapping and accuracy assessment affected not only the level but also the consistency of mapping accuracies, with fewer plots related to greater variability in accuracy estimates and more plots to more robust accuracy estimates (fig. 4). Data for herbaceous vegetation were limited in availability relative to the overall size of the field-referenced data set and hence affected herbaceous mapping accuracy. To improve uncertainties related to shrub and herbaceous cover and height, we determined that these variables should be mapped as categorical map units.

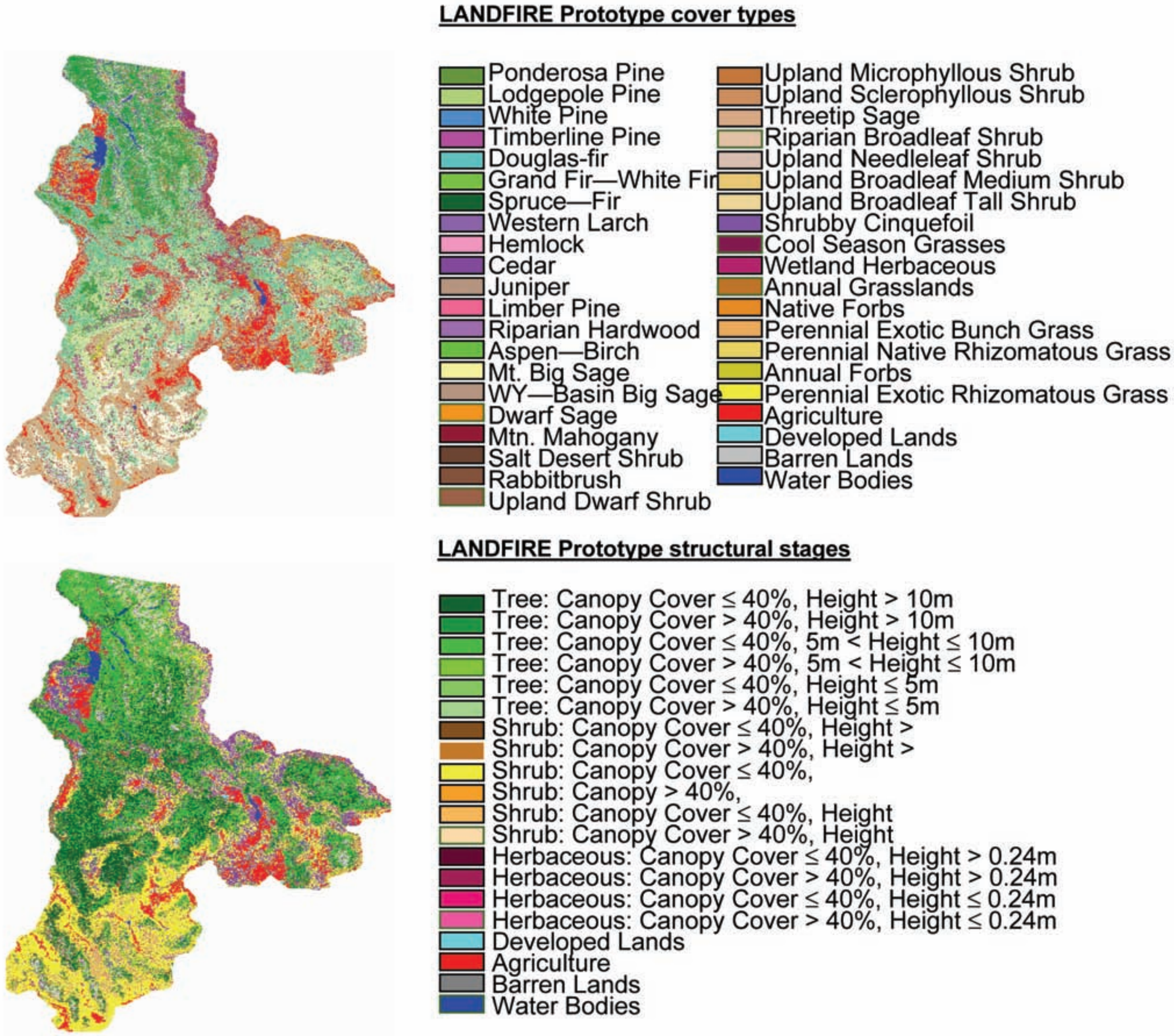

Figure 3—LANDFIRE Prototype cover type (top) and structural stage (bottom) maps for Zone 19. The cover type map is compiled from separate forest, shrub, and herbaceous cover type maps, whereas the structural stage map is grouped from continuous maps of height and cover for display purposes.

Second, field-referenced data, with which mapping models were trained and accuracy assessed, were collected from different sources, for different objectives, and with different techniques. Even though these plot data were quality-screened and standardized through an extensive effort (Caratti and others, Ch. 4), it was inevitable that the differences and errors in field data carried over into map quality and accuracy assessment. For example, certain reference data for forest canopy cover were derived using digital ortho-photographs, viewing forest cover synoptically from above the canopy. On the other hand, field estimates for shrub and herbaceous canopy cover were made using visual estimation from close-range, oblique positions that limited objectivity and consistency. We did not experience these problems when determining forest, shrub, and herbaceous *height*, which was usually directly measured and had a high degree of user-confidence.

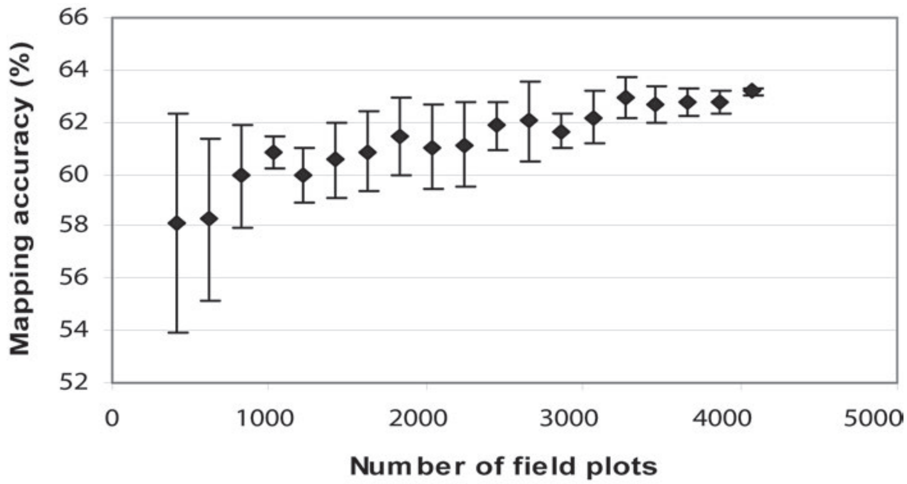

Figure 4—Cross-validation accuracy estimates obtained for the mapping of forest cover types as a function of the number of forest field plots. More plots contributed to better accuracy and consistency (smaller standard deviation) to a certain point, after which the relationship became flat.

Third, as discussed above, rare map units and ecologically and biophysically similar map units affected mapping accuracies. For example, if the Juniper cover type was merged with the combined Pinyon – Juniper cover type, forest cover type accuracy increased by more than 10 percent. The rationale for keeping such similar cover types separate is that, even though they occupy similar ecological niches and have similar site characteristics, separating them increases the utility of the LANDFIRE wildland fuel and fire regime products.

Utility of Biophysical Gradient Data for Vegetation Mapping

Although the use of DEM data for improving mapping results has been widely documented, the effects of a whole host of biophysical gradient layers and PVT-probability data layers is largely untested at the scale and scope of this study. These data layers provide information that supplements satellite imagery. Plant distribution patterns and conditions are strongly linked to a multitude of environmental factors (for example, temperature, soil, weather patterns, day length, soil properties, and rainfall), and the accurate characterization of these variables should, at least in theory, improve mapping results. In addition, spatial information that indicates where particular vegetation types can and cannot exist across a wide region (that is, PVT-probability data layers) should be similarly useful. Figure 5 compares cross-validation results using mapping models with and without the additional biophysical gradients listed in table 1 and using PVT-probabilities as predictor variables. Figure 6 displays mean and standard deviation values of a subset of the biophysical variables intersected with vegetation cover types from field plot data collected in the central Utah prototype area. These figures show that the incorporation of certain biophysical gradients and PVT-probabilities in mapping models contributes to increased mapping accuracy and consistency. These results are consistent with the findings of Keane and others (2002) and Rollins and others (2004).

Vegetation Patterns in Areas of Major Disturbances

Wildfires, insect and disease outbreaks, and forest clear cuts are some of the major disturbances to ecosystems captured by the satellite sensor in terms of their spectral properties. How well did our mapping capture and reflect these changes in vegetation conditions? We evaluated our mapping methods' effectiveness in this regard by looking at known areas of wildland fire, bark beetle infestation, and clear-cuts in the prototype mapping zones.

We evaluated two wildland fires areas that burned in Bryce (summer 2001) and Zion (fall 2001) national parks to determine what differences might exist between pre-fire and post-fire vegetation maps when mapped with the same pre-fire models. Pre- and post-fire map comparisons showed distinct differences between both

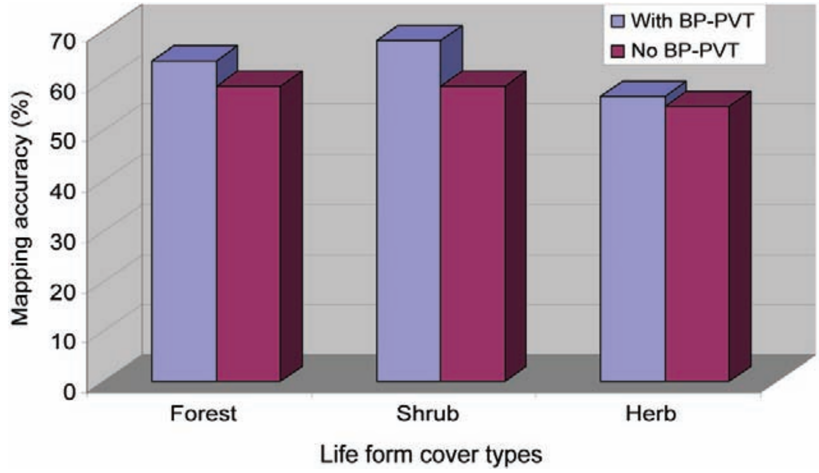

Figure 5—Cross-validation accuracy estimates obtained in the Zone 16 prototype area, by life form, with and without the 15 biophysical gradients and PVT-probabilities in the mapping models. An average of 8 percent increase in cross-validation accuracy was obtained by incorporating the selected biophysical gradients and PVT-probabilities that together describe the habitats of the cover types to be mapped.

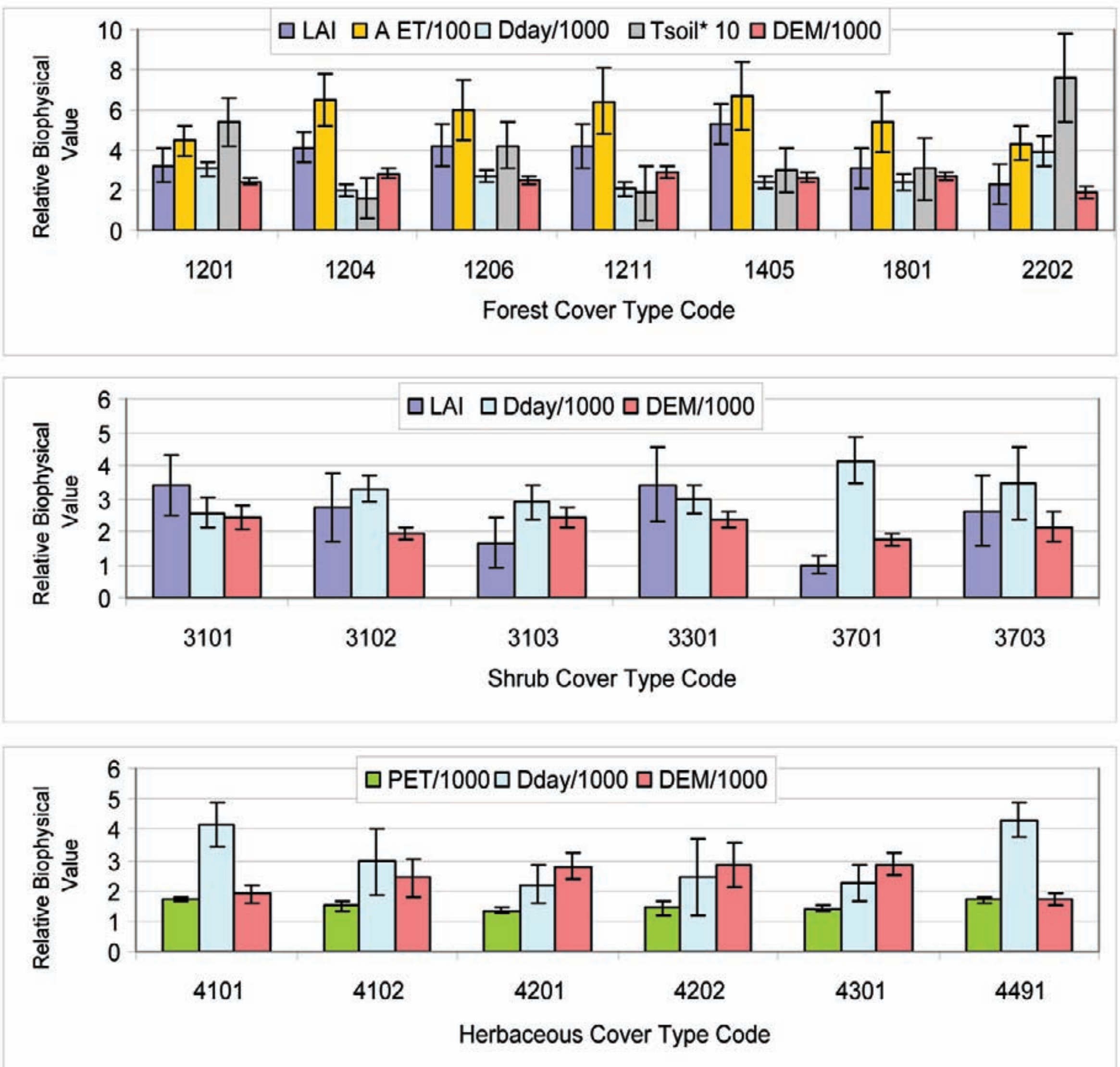

Figure 6—Mean and standard deviation values of selected biophysical variables found effective in mapping cover type against various forest (top), shrub (middle), and herbaceous (bottom) cover types of field-reference data. Most of the biophysical variables were divided or multiplied by a constant for display purposes. Refer to table 1 for definitions and descriptions of the biophysical variables. Refer to Long and others, Ch. 6 appendix 6-A for vegetation cover type coding protocol.

vegetation cover types and structural stages. The Bryce fire, a prescribed fire, showed general shifts from forest to shrub map units and, regarding structure, showed a shift toward increased low-height shrubs. The Zion fire, a wildfire, revealed a shift from predominately deciduous forest types to low shrubs.

Using bark beetle survey data obtained from the Dixie National Forest, we conducted simple zonal statistical analyses. Results indicated that the mapped species composition corresponded fairly well to that of those species identified in the survey data for the years 1998-2000. (Note that the level of actual disturbance varied within the survey data and was not differentiated in this study.) Structure information was not available in the survey data, but mapped structure data indicated that most bark beetle infestations occurred in areas identified as high forest cover (greater than 40 percent canopy cover) and height (greater than 10 m), indicating old-growth forest.

Similarly, we compared clear-cut areas, identified using modeling and masking methods, with mapped vegetation cover type and structure variables. Shrubs and a high percentage of grasses were dominant in clear-cut areas. Structural stages indicated a trend from forests with high canopy cover and canopy height to a high percentage of low cover (less than 40 percent), low height (less than 1 m) shrubs. Herbaceous cover was identified as being high cover (greater than 40 percent) with mixed heights.

Field Data Quality and Quantity Requirements

The acquisition of field-referenced data posed a significant challenge to the LANDFIRE Prototype effort, both logistically and technically. Caratti (Ch. 4) describes the logistical efforts and complications associated with conducting a national field data campaign. Specifically, technical challenges encountered during the mapping process, such as uneven amounts and disparate quality of field data used to meet various vegetation mapping objectives, were tied to the fact that the LFRDB was based on data from varying sources and collected with different objectives. As discussed above, such issues necessitated the careful implementation of a quality-control and quality-assurance (QA/QC) process prior to the training of the mapping algorithms for existing vegetation types and structure. "Lessons learned" from the QA/QC process follow:

- Accuracy and consistency are a function of the amount of available field-referenced data. Greater amounts of field-referenced data contribute to enhanced confidence in mapping accuracy (fig. 4), whereas limited field-referenced data are correlated to reduced confidence in mapping accuracies of affected cover types.
- The use of data from different sources requires that special attention be given to those cover type map units that are not supported with sufficient numbers of field plots. Both prototype mapping zones had map units with only a few field reference data points for training. As discussed above, the question of how to define and treat rare map units arose during the prototype, and we defined rare map units as those having less than 30 field reference plots scattered spatially within a mapping zone. Options for the treatment of these rare map units included keeping the map units in maps, omitting them, or omitting them and then "burning" the few field plots to the map in a post-process and merging them with floristically similar cover types. For the prototype, we chose to retain the rare map units in the models and resulting map products to inform the development of the LANDFIRE vegetation map unit classification system. For national implementation, rare map units that cannot be supported with a sufficient number of field plots will not attain target-level accuracies. We recommend omitting such map units from the mapping of existing vegetation cover types.
- Spatial distribution and a valid probability-based sampling design increase the consistency and accuracy of the map products. Compared with field-referenced data from various agency sources, the use of FIA forest inventory plots for mapping forest cover types and structure produced more consistent and accurate mapping results because the sampling design for FIA data produced training data that were spatially well-distributed across the landscape. Further, FIA data required very little additional processing time and were easy to use; in contrast, non-FIA field data required extensive processing time, related to QA/QC and re-selecting/re-sampling, to derive suitable data sets (in terms of spatial distribution and data quality) from available data points. For example, in Zone 19, a Bureau of Land Management study produced more than 4,800 field plots, mostly describing sagebrush, Douglas-fir, and lodgepole pine vegetation communities, in a relatively small area of approximately 1,152 km^2, near Salmon, ID. Spatially, this data set equated to approximately one plot for every 24 ha, versus a mapping zone average of one plot for every 835 ha. The inclusion of this data set in the

training process overwhelmed the mapping models and overrode areas with sparse plot coverage of different cover types. We therefore determined that the application of locally limited or concentrated data collected using various sampling designs to an entire mapping zone could have adverse effects on the accuracy of final products. For this reason, forest mapping in LANDFIRE National should employ FIA data exclusively. Rangeland mapping in LANDFIRE National, however, will require extensive QA/QC processing steps to transform available field-referenced data to a more suitable data set.

- As noted above, the following critical steps should be taken prior to the development of the mapping models: 1) examine field-referenced data, 2) conduct QA/QC procedures to detect spatial errors as well as information content-related errors, 3) correct these errors if necessary, and 4) derive a final, refined, error-free data set for training and accuracy assessment. This is a time-consuming yet necessary process that will contribute to increased consistency and confidence of map products.

Effects of the Vegetation Map Unit Classification System

Determining accuracy objectives and the appropriate extent of mapping areas are among the factors that need to be considered when defining a workable national vegetation map unit classification system. If floristically or ecologically overlapping cover types (such as Juniper and Pinyon -- Juniper or Upland Microphyllous and Upland Sclerophyllous) are to be mapped for LANDFIRE National, then guidelines must be developed for defining how the mapping accuracy of such overlapping map units is to be assessed.

Next, although our use of the NVCS was a reasonable starting point for vegetation map unit classification and the approach worked fine for each individual mapping zone, vegetation cover types were not always comparable between the two prototype mapping areas, however, as is evidenced by the legends in figures 2 and 3. As a result, accuracy estimates for the two prototype mapping zones could not be compared in a straightforward fashion, particularly for shrub cover types.

As discussed above, another challenge encountered during the application of the two vegetation map unit classification approaches (as discussed above in the *Vegetation Map Unit Classification* section) was answering the question of how to treat rare map units. There were no guidelines for consistently defining and treating rare map units. Moreover, there was no answer as to whether dropping rare map units, instead of using the alternative options discussed above, might affect the utility of LANDFIRE vegetation maps in other future natural resource management projects.

Recommendations for National Implementation

Because of the size and complexity of this research effort, many questions concerning LANDFIRE's national implementation are as of yet unanswered. The field data compilation effort will be an expensive and time consuming task, and a pressing need exists regarding the study of links between mapping performance, resource expenditure, and methods of field data collection. Ecological relationships between mapped potential vegetation and existing vegetation need to be investigated. Further research must be conducted to quantify the relative contributions of the different approaches and data sets used in the prototype. Performance consistency must be tested between adjacent western mapping zones, as well as in one or more prototype areas located in the eastern United States. Repeatability of the methods used in the prototype, both temporally and spatially, must also be evaluated. Furthermore, it is not clear whether the LANDFIRE Prototype methodology will suffice for other vegetation metrics, such as quantifying woody or non-woody biomass; a study in this area could yield information leading to enhanced applications of LANDFIRE vegetation maps. Nevertheless, the LANDFIRE Prototype Project provides sufficient information on which to base several recommendations regarding the national implementation of LANDFIRE.

Ways to Ensure Consistent National Vegetation Mapping

As noted above, several tasks related to existing vegetation mapping for the prototype effort may be standardized and potentially automated to facilitate LANDFIRE's national implementation. These tasks include: 1) the creation of a national vegetation map unit classification system that is mappable using spectral and biophysical/ecological data and is supported with adequate field-referenced data; 2) the consistent acquisition and processing of a multi-seasonal Landsat database; 3) the application of QA/QC procedures to the LFRDB to ensure a robust field-referenced database that can be used for a wide variety of applications; 4) the consistent modeling of biophysical data layers and probabilities of existing vegetation species or types associated with potential vegetation types; and 5) the continued application of CART as the primary mapping algorithms to ensure objectivity and flexibility when using high volumes of field data and predictor variables. We discuss these points in detail below.

Need for a Mappable Vegetation Map Unit Classification System

The vegetation map unit classification system used for the national implementation of LANDFIRE must meet a number of key criteria including the following: 1) the system must be nationally consistent, ecologically logical and hierarchical, acceptable to a wide array of users and groups, and must meet existing Federal Geographic Data Committee (FGDC) standards; 2) vegetation map units must be mappable using operational methodology to achieve reasonable accuracies; and 3) the map unit classification system must include vegetation map units that have high relevance with respect to the core LANDFIRE products. The Ecological Systems classification (Comer and others 2003) developed by NatureServe meets these objectives. This system represents the hierarchical merging of NVCS alliances into a nationally available suite of vegetation map units. Unlike alliances, which have proved exceedingly difficult to map accurately, most Ecological Systems classes are mappable, assuming an adequate number of field plots exist for training purposes. In addition, the Ecological Systems classification was developed by plant ecologists, lending credibility to the approach and resulting in a greater level of acceptance throughout the user community. We anticipate that a few additional "target alliance" map units will be added to the LANDFIRE National map unit classification legend on a case-by-case basis. These will be added only when it is determined that a particular map unit not specifically identified by the Ecological Systems classification has special relevance to LANDFIRE.

Need for National Field-referenced Data Collection and Processing

Many LANDFIRE tasks rely on a comprehensive, consistent, and extensive field-referenced database. The database serves as a reference for the development, testing, and accuracy assessment of all LANDFIRE vegetation, biophysical settings, and wildland fuel data layers and of all vegetation and fire regime simulation models. Field data from existing projects should be incorporated into this database whenever available and should include but not be limited to data sets such as FIREMON fire monitoring databases, USFS Landscape Ecosystem Inventory Systems databases, and the National Park Service fire monitoring databases. In addition, the USFS FIA Program's forest inventory plot database proved a useful source for the majority of forest data. Where data are lacking, supplemental field data collection is required to fill informational needs on rangeland map units. This assortment of field-referenced data should be collectively scrutinized for quality assurance, regularly updated, and maintained as a comprehensive LANDFIRE field-referenced database.

Need for Nationally Consistent Imagery Database

The availability of a quality Landsat imagery catalog is a key prerequisite for national implementation of the approaches developed for the LANDFIRE Prototype Project. Among all predictor variables, it is satellite imagery that usually captures the most current vegetation conditions, and, when used repeatedly over time, identifies changes in vegetation conditions and distributions. Thus, we recommend that LANDFIRE National continue to play an active role in the MRLC Consortium. This membership ensures the continued development of suitable multi-seasonal Landsat image catalogs, optimal levels of image processing (geometric, radiometric, and atmospheric rectification and calibration) for the rest of the country, and mapping zone-based image compilation for national vegetation mapping. In addition, LANDFIRE National should support studies that examine and compare the characteristics of other mid-resolution sensors with those of Landsat. Even though the LANDFIRE Project does not currently require any additional Landsat imagery, the potential benefits of using different satellite data for future updating should be considered.

Need for Nationally Consistent Set of Biophysical Gradient Layers

Biophysical gradients have effects similar to that of Landsat imagery on the spatial and information integrity of existing vegetation maps. Many of the biophysical layers are physiologically and ecologically related to the establishment, distribution, and conditions of plant species, and the incorporation of these gradient layers into the mapping process contributes to increased accuracies. For the national implementation of LANDFIRE, we recommend that a set of biophysical gradient layers similar to those listed in table 1 be used to map vegetation in all mapping zones. In addition, we recommend that further research be conducted to quantify the contribution of the individual biophysical variables to mapping accuracy. Furthermore, research should be conducted to minimize residual coarse-resolution imprints in 30-m biophysical data resulting from the coarser resolution weather and soil databases used to produce these data. The development of standard minimum mapping units in modeling simulations has shown promise in standardizing the process and eliminating coarse imprints.

The Need to Continue with Research and Improvements

Although results of the LANDFIRE Prototype Project indicate that the general approach should effectively meet target accuracy and consistency requirements for national implementation, there are areas where continued research and improvements are needed. One ongoing research effort involves the development of a new and more consistent approach to mapping shrub and herbaceous canopy cover. Current research is testing ways to effectively correlate calibrated Landsat-based NDVI to shrub and herbaceous canopy cover (Liu and others 2004). Other research areas include more efficient use of the individual biophysical gradient layers, more effective mapping of riparian vegetation, and a national accuracy assessment strategy.

Conclusion

The mapping of existing vegetation with complete national coverage at a 30-m spatial resolution is a core requirement of the LANDFIRE Project. National data at this 30-m resolution do not currently exist. As a result, the prototype research was needed to answer questions related to the mapping and characterizing of cover types and structure variables. LANDFIRE's existing vegetation products are expected to provide data not only for use in wildland fire management, but also for use in many other natural resource and environmental applications. Findings from the LANDFIRE Prototype effort are summarized as follows:

If supported with an adequate amount of field-referenced data, target accuracies of 60 percent or better are achievable for a mid-level vegetation map unit classification at the regional scale. The addition or subtraction of floristically or ecologically similar cover types has significant effects on resulting accuracies. Of the three major life forms, herbaceous cover types are the most difficult to map because these species adapt to many general biophysical characteristics and have few unique spectral signatures. Relationships between the floristic complexity of the vegetation map unit classification and mapping accuracies indicate that the national vegetation map unit classification will need to be designed carefully to include adequate flexibility.

For LANDIFRE, vegetation structure is defined by canopy cover and canopy height of forest, shrub, and herbaceous life forms. These structure attributes can be mapped consistently as categorical variables. Mapping these attributes as continuous variables, particularly for shrub and herbaceous height and cover, is inconsistent and, thus, is not recommended for national implementation of the LANDFIRE prototype methods.

Field data collection and processing are the most critical factors in ensuring that LANDFIRE maps of existing vegetation are objective and accurate. The detection and correction of errors existing in field-referenced data are time-consuming but absolutely necessary tasks, particularly for field data from sources other than FIA (as these other data sets tend to be locally limited and have various sampling designs). The objective of repeated field data processing and quality control is to derive a refined, high-quality field data set.

The incorporation of LANDFIRE biophysical gradient layers and cover-type probabilities associated with potential vegetation types into the mapping models contributes to a significant increase in mapping accuracy. In addition, the use of the biophysical and ecological stratifications that describe the environmental effects on species establishment and growth also contributes to enhanced mapping consistency.

For further project information, please visit the LANDFIRE website at www.landfire.gov.

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Imler-Jacquez, Sandra R -FS

From:	Sarah Hyden <sarah.hyden@me.com>
Sent:	Thursday, December 19, 2019 9:09 PM
To:	FS-comments-southwestern-santafe
Subject:	Encino Vista Landscape Restoration Project

December 19, 2019

Rich Nieto, District Ranger
Coyote Ranger District
HC-78, Box 1
Coyote, NM 87012
Submitted electronically via comments-southwestern-santafe@usda.gov

Scoping Comments for the Encino Vista Landscape Restoration Project

Dear Mr. Nieto,

I am respectfully submitting these comments to the U.S. Forest Service concerning the scope of the agency's analysis under the National Environmental Policy Act (NEPA) of the Encino Vista Landscape Restoration Project across approximately 128,400 acres within the Coyote Ranger Districts on the Santa Fe National Forest.

I. General comments:

The Encino Vista Landscape Restoration Project is located on the Coyote Ranger District of the Santa Fe National Forest south of the communities of Cañones, Youngsville, Coyote, and Gallina, New Mexico. The project boundaries encompass over 128,400 acres of which 119,767 acres are national forest lands. The project area overlaps thirteen HUC 12 sub-watersheds. The majority of the project is within these sub-watersheds: Coyote Creek, Cañones Creek, Headwaters Rio Puerco, Poleo Creek, Outlet Rio Puerco, Upper Rio Galina and Rio Capulin.

The project area is situated between the elevation of 6,450 and 10,600 feet. The area includes piñon/juniper woodlands, ponderosa pine, mixed conifer, aspen and spruce-fir forest types.

It includes essential habitat for many wildlife species, including several endangered or at-risk species including Mexican spotted owl, Northern goshawk, New Mexican meadow jumping mouse, Southwestern willow flycatcher, Jemez Mountain salamander, and Rio Grande cutthroat trout.

The project area is an ecologically diverse treasure and valued by many local residents for recreation.

The Proposed Action envisions thinning treatments on a total of up to 88,400 acres of the project area. It envisions prescribed burning on a total of up to 110,213 acres, and maintenance burning would occur in areas treated with prescribed fire at 5-20 year intervals. This is the largest project ever proposed in the Santa Fe National Forest. Only 22,225 acres of forest would be managed for old growth characteristics.

An Environmental Impact Statement *must* be prepared. The Encino Vista Landscape Restoration project, including the proposed Forest Plan amendment, "may' have a significant impact on the environment, and thus the Forest Service must prepare a robust EIS, ensuring that it complies with NEPA's required "hard look." The EIS must analyze the baseline conditions of the project area, and the direct, indirect and cumulative impacts of the proposed timber management activities, road construction and maintenance, and all other activities. The public must be fully included in the project planning process.

Sufficient notice of the project and the comment period was not given. Very few people in the Santa Fe area, where most of the forest protection advocates focused on the Santa Fe National Forest reside, were aware of the project and the ongoing comment period until the comment period was half way complete. There was no notice placed in any newspaper, and the notice was only put out on a very limited mailing list. I strongly suggest the USFS start over on the scoping comment period and put out proper notice so all citizens and conservation groups who are interested will have time to write comprehensive scoping comments.

The best available scientific information (BASI) must be utilized in project planning, and the U.S. Forest Service is require to explain how it met this mandate. There are numerous studies that support a much more conservationist approach to managing the Encino Vista project area that have not even been considered. The USFS must consider a broad range of best available science.

While the SFNF's lower elevations were historically dominated by low-severity fire, there was an under-appreciated and substantial moderate severity and high severity fire component. This forest evolved in a mixed intensity fire regime, and all intensities of fire are important for maintaining ecological processes and integrity. Moderate and high intensity fire should be considered as natural to our forest ecosystem and no attempt should be made to exclude them except for the safety of important values such as human lives (egress from communities and forest recreational areas) and structures. Research shows that medium and long-term the effectiveness of watersheds is not substantially impaired by fire, even high intensity fire.

The model framework the USFS is using is found in GTR-310, a framework developed primarily in and for an entirely different eco-region, focused primarily in reference study sites around Flagstaff. It is appropriate for use for the SFNF. A more appropriate and region-specific framework must be developed. Please provide examples of where the desired condition has been achieved. It is important that areas in which the desired condition have been achieved be evaluated.

Planning treatments based on general landscape categories such as Ecological Response Units is not nearly targeted and strategic enough, and is a broad stroke way of planning that is likely to result in much ecological damage. These days, in cancer treatment, therapies are no longer broad poisoning of much of the system, but targeted and strategic, and often have a much more positive outcome with much less adverse effects. This should be emulated. The forest is not "ill" in a way that it cannot largely recover from through its own regenerative processes.

Forest that were treated with mechanical thinning in fuel treatment projects that have occurred recently and in the more distant past in the Santa Fe National Forest, and subsequently treated with prescribed fire periodically are not healthy forests and do not in any way resemble historical forests. Forest fuel treatments must be re-evaluated and done in a much more limited, light-handed and strategic way, if a comprehensive cost/benefit analysis indicates that it is beneficial that they be done at all.

There must be a strong focus on connectivity, and on preserving the forest as free of adversely impactful interventions as possible. Much more acreage of forest should be managed for old growth characteristics.

No amendment should be made of the existing Forest Plan for this project relating to Mexican Spotted Owl. The Encino Vista Landscape Restoration Project directly contradicts the current injunction on logging activities in Mexican Spotted Owl habitat. It is highly disappointing that the USFS is proposing to contradict an existing court ordered injunction. This should not occur.

The USFS must consider an alternative that truly conserves and protects the SFNF while focusing on fire moderation from the home and other values outward. The Proposed Action does not go nearly far enough in conserving forest ecology and resources. Also, it is essential to incorporate into the project the major elements from the Santa Fe Conservation Alternative, submitted by WildEarth Guardians, Defenders of Wildlife and Sierra Club for the Santa Fe Mountains Landscape Resiliency Project. It is equally applicable to the Encino Vista Landscape Restoration Project, and included below.

Please use the principles of the Santa Fe Conservation Alternative to develop a much more nuanced restoration project, focused primarily on real restoration such as decommissioning unneeded roads, planting in riparian ways and erosion control. A robust monitoring program is of the essence.

II. Additional Concerns:

1) Monitoring — There must be a robust monitoring program developed and put into in place. It should thoroughly consider effects of fuel treatments on overall forest ecology, connectivity, riparian ways, wildlife (especially endangered and at-risk species), tree health, affects on recreation and the health impacts of prescribed burn smoke on humans and wildlife. Mexican spotted owl populations must be monitored. It is necessary for a requirement to be put into place that the project be halted if the monitoring plan is not thoroughly and comprehensively carried out.

2) Assumptions — Many assumptions are based on unproven science or studies that have substantial flaws and invalid conclusions. Assumptions should be thoroughly evaluated using a broad range of research including studies that support a more conservationist view of forest management. The explanation in the Scoping Document of why there is an increased and substantial fire risk is very limited to one scientific perspective (connectivity of fuels), when as delineated in the Common Ground report, there are three important scientific perspectives regarding the condition of our forests. Climate needs to be considered much more in the analysis of this project, both as a causative factor for fire and that there is a need to preserve trees to sequester carbon.

3) Conservation — There should be a general strong bias in project planning towards conserving our forest in as natural a condition as possible, and to allow forest ecology itself to bring our forest into greater balance through natural processes. There are too many substantial adverse impacts related to intensive tree thinning and prescribed burning. Treatments should be very limited, site-specific and strategic, as recommended by the Santa Fe Conservation Alternative.

4) Genuine restoration — Focus should be on genuine restoration activities instead of cutting and burning. Decommissioning of all unneeded roads must be included in project planning. Focus should be on true restoration such as planting in riparian areas as needed and hand-building structures in arroyos to slow flood waters.

5) Reduce prescribed burns for public health — The Proposed Action contains up to 110,213 acres of prescribed burning. That much burning will have very large negative impacts on public health. There is a great deal of upset and controversy among the public about the adverse health effects many are experiencing from the large number of prescribed burns and wildfires expanded with fire accelerants in recent years. The number of days per year that the USFS performs prescribed burns must be capped, so that there is a very limited number of days that create smoke impacts on the public. The effects of volatilized fire accelerants must be analyzed. A system

must be set up to take in and document public health impact reports.

6) Roads — There are 6,900 miles of roads in the SFNF, many of which are leaking sediment into streams and fragmenting wildlife habitat. According to the 2008 Travel Management Record of Decision for the Santa Fe National Forest, 2,878 miles of open system roads were to be closed for public use. A minimum network of roads should be identified for the Encino Vista Landscape Restoration Project area, and all unneeded roads should be closed and/or decommissioned.

7) Invasive species — The Proposed Action does not include sufficient actions for limiting the spread of invasive species via management of livestock grazing, roads, equipment used for thinning and OHVs. A thorough plan must be developed.

8) Thinning — The framework provided by GTR-310 clearly supports over-thinning in the SFNF. Projects done in the SFNF post GTR-310 are not much healthier in appearance than pre-GTR-310. The cost/benefit analysis of thinning, especially the severe thinning recommended by GTR-310, a document not focused on our forest type, has not been done. Any thinning done should be very light-handed, targeted and limited to protect specific discreet values or for extremely dense areas previously damaged by logging.

9) Prescribed burns — There may be some justification for limited prescribed burns where the duff and understory are very thick, but the widespread prescribed burning that is repeated every several years has clearly not supported forest health. The forest understory is never allowed to return, or as soon as it has started to return it is burned off again. The use of prescribed burns needs to be re-evaluated using the full range of the best available science.

10) Connectivity — Connectivity is inadequately addressed in the Proposed Action, having an emphasis on vegetation management in Ecological Response Units. Connectivity should be a major focus and strong and effective wildlife corridors developed.

11) IRAs — No mechanical thinning should occur in Inventoried Roadless Areas, and very little prescribed burning — only when there is a limited, strategic and site-specific reason. IRAs should be left as intact as possible.

12) WUI communities — Thinning should not be done further than 150 feet from structures in WUI areas for the prevention of fire in WUI communities as it has been proven to not be an effective strategy for this purpose by former USFS researcher Jack Cohen and others. Forests adjacent to communities should be left intact and natural as possible to be used for recreation. Support and education should be given to WUI property owners to effectively fire proof their homes and the surrounding 150 feet. The development of alternative egresses for communities that have only one egress should be supported.

III. The Santa Fe Conservation Alternative

The Proposed Action does not adequately meet the conservation requirement of the 2012 forest planning rule. "Conservation. The protection, preservation, management, or restoration of natural environments, ecological communities and species." Alternative 2 includes forest fuel treatments and road development that clearly, in the SFNF, greatly harm the forest ecosystem in numerous ways.

WildEarth Guardians, along with Sierra Club and Defenders of Wildlife have developed an alternative for analysis in both an EA and/or an EIS for the Santa Fe Mountains Landscape Resiliency Project called the Santa Fe Conservation Alternative, to address forest management in a way that conserves forest resources. It is equally applicable to the Encino Vista Landscape Restoration Project and would meet the conservation requirement of the 2012 forest planning rule very well. These principles should be applied to the Encino Vista Landscape Restoration Project

It is based on education, engineering and enforcement. Instead of widespread fuel treatments out in the forest away from the WUI, it recommends the more effective and conservationist steps of educating the public about maintaining a safety zone around WUI structures and campfire safety, engineering to protect communities and values from post-fire flooding in key areas, maintaining power lines, etc. and increased law enforcement to reduce unsafe fire behavior in the forest.

I am submitting it as the preferred general approach for the Encino Vista Landscape Restoration Project.

The basic principles of the Santa Fe Conservation Alternative are:

Thinning
—Limited hand thinning (up to 9") only in dry pine and mixed conifer outside of IRAs.
—Stumps cut down to the ground
—No thinning adjacent to the WUI for the purpose of protection of structures or communities except within 150 feet of structures, and for fire fighter safety zones.
—Maximum trees removed in most thinned areas to 80 BA
—Leave more tree groupings (50% minimum) and maintain a shrub understory. Utilize a wildlife habitat-based determination of tree and vegetation retention
—Identify riparian area concerns and create plan to protect

Slash management
—Pile burning of activity fuels
—Reevaluate slash management timing and methods to avoid potential bark beetle outbreaks, and sterilization of soil under slash piles. No slash over 3" left on the ground during the dry season

Prescribed burning
—Utilize managed wildland fire and pile burning wherever possible. Utilize minimal broadcast prescribed burns only in areas that are not assessable for pile burns.

IRAs
—No thinning in IRAs
—Identify Roadless Area concerns and develop a policy to restore

Monitoring (key means of reaching desired outcomes of healthy forest habitat and protection of public health)
—Test plots for monitoring purposes
—Soil sampling - plot number and spacing to be determined
—Baseline species evaluation (i.e. population capacity and presence/absence)
—Improved air quality standards and monitoring to protect sensitive (human) population

Reclamation and restoration
—Reclamation of any USFS roads deemed unessential in Travel Management Plan
—Hand building of structures (example Zuni bowls) in arroyos to slow flood waters
—Planting native, stream side vegetation where appropriate to slow floodwaters
—Reintroduction of beaver where appropriate

WUI and community forests
—Develop a program to support fire-proofing of structures and surrounding 100 feet, at least

through increased outreach and education. This should be a homeowner responsibility
—If possible, support development of an alternative egress for communities with a single egress
—Leave most areas that the public uses for recreation, including forests adjacent to communities, natural and intact.
--Take into greater account the need to preserve areas that are special to communities, such as Cougar Canyon
—Increased law enforcement to protect against unsafe fire behavior by forest visitors

Scenic quality
—Maintain the scenic quality of all treated areas. Develop a standard for acceptable scenic quality

Sincerely,

Sarah Hyden
PO Box 22654
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(505) 983-3401

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Imler-Jacquez, Sandra R -FS

From: Judi Brawer <jbrawer@wildearthguardians.org>
Sent: Thursday, December 19, 2019 1:58 PM
To: FS-comments-southwestern-santafe
Subject: [CAUTION: Suspicious Link]Encino Vista Landscape Restoration Project
Attachments: Encino scoping letter.pdf

PROCEED WITH CAUTION: This message triggered warnings of **potentially** malicious web content. Evaluate this email by considering whether you are expecting the message, along with inspection for suspicious links.

Questions: Spam.Abuse@usda.gov

To whom it may concern,

Please find attached our scoping comments on this project

Sincerely,
Judi

WildEarth GUARDIANS

JUDI BRAWER
Wild Places Program Director

(208) 871-0596
www.wildearthguardians.org/public-lands/

WildEarth
GUARDIANS
A FORCE FOR NATURE

December 19, 2019

Rich Nieto, District Ranger
Coyote Ranger District
HC-78, Box 1
Coyote, NM 87012
Submitted electronically via comments-southwestern-santafe@usda.gov

Re: Scoping Comments on the Encino Vista Landscape Restoration Project Purpose and Need for Action and Proposed Action

Dear Mr. Nieto,

WildEarth Guardians respectfully submits these comments to the U.S. Forest Service concerning the scope of the agency's analysis under the National Environmental Policy Act (NEPA) of the Encino Vista Landscape Restoration Project across approximately 128,400 acres within the Coyote Ranger Districts on the Santa Fe National Forest (Santa Fe NF). The landscape-scale project includes a number of activities requiring rigorous environmental analysis including forest plan amendments, significant vegetative treatments in TES species habitats, road construction and road management actions. These activities will occur in multiple sub-watersheds, specifically the Coyote Creek, Cañones Creek, Headwaters Rio Puerco, Poleo Creek, Outlet Rio Puerco, Upper Rio Galina and Rio Capuling. *See* Encino Vista Landscape Restoration Project Purpose and Need for Action and Proposed Action (hereinafter "Scoping Notice"), p. 2. Please add my names and organization to the contact list to receive any future public notices regarding this project.

Most, if not all, of the sub-watersheds in the project are impaired or functioning at risk and many of the roads in the project area are in poor condition. *See* Scoping Notice, p. 5. Improving these conditions at a measurable level is an outcome we urge the Forest Service to demonstrate in the subsequent analysis. Given the scope and scale of the proposed action, including two Forest Plan amendments, and thus the potential for significant impacts, we strongly urge the Forest Service to prepare an environmental impact statement (EIS).

I. The Forest Service must ensure that this is truly a "Restoration" project.

> The purpose of the Encino Vista Landscape Restoration project is to restore overall forest health, lower fire risk, improve watershed health, and enhance wildlife habitat across the landscape. There is a need to increase forest ecosystem sustainability and resiliency to insects, disease, and climate change by shifting forest composition and structure toward desired conditions within the historic (or natural) range of variability for each forest type. There is also a need to reduce the risk of uncharacteristic wildfires, to improve species habitat, and overall watershed conditions.

This project focuses on forest restoration and resiliency treatments to:

- Reduced stand densities
- Reintroduce fire on the landscape
- Revitalize meadows and aspen stands
- Promote a diverse forest structure for a variety of wildlife species
- Improve watershed conditions across the landscape, which would safeguard the water supply for villages, towns and ranches within the project area as well as downstream communities, by increasing the quality and quantity of water that flows through the network of streams improving watershed function within the Encino Vista Project area
- Significantly reduce the risk of catastrophic wildfire and its aftermath (flooding, debris flow)

Scoping Notice, pp. 8-9.

Yet, despite being touted as a landscape restoration project, the proposed action provides little in actual restoration activities such as identifying the Minimum Road System and associated actions such as decommissioning system roads, removing non-system roads and fixing poorly placed and/or sized culverts, in-stream and riparian restoration, and reducing the impacts of livestock grazing and motorized use. Without such restoration work, the Encino Vista project is merely a logging and burning project, not restoration. These restoration actions are essential to meeting the stated purpose and need, and achieve the desired results, above.

According to the Scoping Notice,

> The project area overlaps twelve HUC 12 sub-watersheds, but the majority (74%) of the project is within these sub-watersheds: Cañones Creek, Coyote Creek, Headwaters Rio Puerco, and Poleo Creek. These four sub-watershed are a crucial source of recharge of ground water used as drinking water for the communities of Coyote, Youngsville, Cañones and Abiquiu. Portions of four other sub-watersheds make up nearly 23% of the project area, they are: Outlet Poleo Creek, Upper Rio Gallina, Rio Capulin, and Abiquiu Reservoir. The Cañones Creek is an impaired functioning watershed. The other previously mentioned watersheds are functioning at risk.

Scoping Notice, p. 5. Yet, there is no substantive information as to how these watersheds will be improved to ensure the necessary groundwater recharge. Extensive timber management is not the answer, especially without other substantive restoration activities such as road decommissioning, riparian restoration, and addressing the impacts of livestock grazing.

Indeed, according to the Scoping Notice, livestock grazing plays a significant role in why the area is purportedly outside of historic ecological conditions. "Fire suppression is the primary reason which has allowed for change in fuels (suppression combined with grazing and logging are the driving factors for change in fuel abundance, type, and arrangement)." Scoping Notice, p. 5. In the Juniper Grass ERU, "Typically, native understory grasses are perennial species, while forbs consist of both annuals and perennials. Shrubs are characteristically absent or scattered. Due to the effects of long-term fire suppression and grazing in this type, in many locations the current condition is severely departed from historic conditions." *Id.*, pp. 5-6. It is unclear how livestock grazing has impacted the other vegetation communities such as Aspen, mixed conifer, Ponderosa pine, dry mixed conifer, Pinion-Juniper woodlands, and pinion-juniper sagebrush. This must be analyzed as part of the baseline conditions and cumulative effects analysis.

Further, replacing Forest Plan standards with desired conditions and guidelines weakens the Forest Plan. The Forest Service's desired conditions are not based on the best available science and are static conditions used as an excuse for the Forest Service to continuously log in areas where natural fire should be returned. Forests are not static, they are constantly changing, and natural fire is an essential component of this change.

II. Mexican Spotted Owl

We are dismayed at the Forest Service's failure to ensure the protection and restoration of MSO and their habitat. The Encino Vista project directly contradicts the current injunction on logging activities in Mexican Spotted Owl habitat, and the proposed Forest Plan amendments do not comply with the 1996 Standards and Guidelines that still apply on the Santa Fe NF and impose significant restrictions/constraints on management activities in protected habitat (i.e., PACs and steep slopes). One of the assumptions of the current programmatic Biological Opinion for the Santa Fe NF is that the Forest Service will implement the the Forest Plan, including the 1996 Standards and Guidelines. So, since the proposed Forest Plan amendments deviate from the 1996 Standards and Guidelines, that is an action "outside" of the programmatic Biological Opinion that requires a separate "stand alone" Biological Opinion. Further, the Encino Vista project and associated MSO Forest Plan Amendment fails to incorporate significant and essential components of the 2012 Recovery Plan.

The 2012 Recovery Plan states that scientific

> studies suggest that at least some kinds of mechanical forest treatments may negatively affect spotted owls. No clear guidance emerges from these studies relative to types, extents, or spatial arrangement of treatment that might minimize effects to owls. Such information is needed if management is to proceed in owl habitat. Lacking such information, managers should proceed cautiously in terms of treatment intensity and extent. That is, initial treatments should be limited in spatial extent and treatment intensity, and should be aimed at balancing reduced fire risk with maintaining the mature forest structure that seems to be favored by spotted owls. Treatments in owl habitat should be linked to rigorous monitoring of owl response, to allow us to evaluate the effects of different types and extents of treatments in an adaptive management context... The Recovery Team recommends mechanical treatment in PACs only if such monitoring occurs.

2012 Recovery Plan, p. 73.

Pursuant to the 2012 Recovery Plan, "Forest restoration and fuels-reduction treatments must be evaluated over time using appropriate modeling, rigorous monitoring, management experiments, and/or research to assess their effectiveness in maintaining or creating owl habitat and/or their effectiveness in reducing the threat of high severity or stand-replacing wildland fire." *Id.*, p. 250. Accordingly, the NEPA analysis for this project must include the results of past monitoring of the impacts of timber management activities, roads and motorized use, noise and recreational activities, and livestock grazing on MSO. And, this project must incorporate the rigorous monitoring and other recovery recommendations of the 2012 Recovery Plan. We expect to see the results of at least two years of rigorous pre-project monitoring in the NEPA analysis.

The Forest Service must comply with the ESA, its Forest Plan (including the 1996 Standards and Guidelines), and the 2012 Recovery Plan to provide for the recovery of MSO. This includes limiting activities that impact critical habitat, Protected Activity Centers (PACs), and recovery habitat. The FS must consult with the U.S. FWS on the impacts of the project and Forest Plan Amendment on MSO, and these consultation documents must be provided to the public during the NEPA process on the agency's website for this project.

The Forest Service should follow the management recommendations in the 2012 Recovery Plan (see Appendix C of the 2012 Recovery Plan), for PACs, recovery habitat, and other habitats, and must also analyze the impacts of climate change on MSO, as discussed in the 2012 Recovery Plan.

III. As part of the analysis of the Encino Vista Project under NEPA, the Forest Service must not only consider the Santa Fe National Forest's Travel Analysis Report and identify unneeded roads to prioritize for decommissioning or other uses, but it must also identify the Minimum Road System.

The Forest Service faces many challenges with its vastly oversized, under-maintained, and underfunded road system. The Santa Fe National Forest is no exception. According to the 2008 TAR, "we estimate that the Santa Fe National Forest needs over $4 million per year for adequate maintenance for all of our roads, using recommended maintenance frequencies and costs." 2008 Travel Analysis Process Report, p. 18. The TAR identifies as a partial resolution will be to reduce the miles of the designated road system." *Id.* It does not appear that the TAR has been updated since 2008, so this $4 million could have significantly increased since then, and there is no information on the Forest Service's progress in decommissioning unneeded roads.

The impacts from roads to water, fish, wildlife, and ecosystems are well documented in scientific literature. The following is just a small list of examples:

- Increased sedimentation in stream beds has been linked to decreased fry emergence, decreased juvenile densities, loss of winter carrying capacity, and increased predation of fishes, and reductions in macro-invertebrate populations that are a food source to many fish species (Rhodes et al. 1994, Joslin and Youmans 1999, Gucinski et al. 2000, Endicott 2008).
- Roads can act as barriers to [fish] migration (Gucinski et al. 2000). Culverts in particular often interfere with sediment transport and channel processes such that the road/stream crossing becomes a barrier for fish and aquatic species movement up and down stream.
- Where both stream and road densities are high, the incidence of connections between roads and streams can also be expected to be high, resulting in more common and pronounced effects of roads on streams (Gucinski et al. 2000).
- Roads and trails impact wildlife through a number of mechanisms including: direct mortality (poaching, hunting/trapping) changes in movement and habitat use patterns (disturbance/avoidance), as well as indirect impacts including alteration of the adjacent habitat and interference with predatory/prey relationships (Wisdom et al. 2000, Trombulak and Frissell 2000).
- Forman and Hersperger (1996) found that in order to maintain a naturally functioning landscape with sustained populations of large mammals (such as elk), road density must be below 0.6 km/km2 (1.0 mi/mi2).
- The MSO 2012 Recovery Plan identifies the impacts that roads, noise and motorized recreation have on MSO, including breeding.

Indeed, the scoping notice admits that "[m]any of the roads in the project area are in poor condition. These roads do not provide safe and efficient access. Their degraded state may be causing soil erosion." Scoping Notice, p. 5. Yet, despite this project being touted as a "restoration" project, there is no mention of a resilient road system, road decommissioning, or otherwise improving vegetation, habitat, and riparian conditions by removing roads and limiting motorized use. Instead, the Forest Service proposed to construct approximately 5 to 10 miles of temporary roads and conduct road infrastructure improvement and maintenance on existing Forest Service roads within the project area. Scoping Notice, p. 10. In order to meet the restorative purpose and need for the project and eliminate or reduce the impacts of the roads within the project area to water quality, fish and wildlife habitat, the Forest Service needs to take steps related to its road system.

Local communities and visiting recreationists are also impacted by the oversized, under-maintained and under-funded road system. Since roads are not regularly maintained or upgraded, they are highly susceptible to storms. Small culverts become plugged with debris, forcing water over the road and often resulting in the road getting washed out. Gullies can form along roadbeds making it difficult to drive a car on the road. This "storm damage" can eliminate access and often takes years to fix, if it even gets fixed at all. In order to ensure access to beloved trails, campsites, fishing and swimming holes, etc., the Forest Service should target limited road maintenance funding to high priority recreation/community access roads.

The Forest Service recognized the challenges related to the oversized and deteriorating road system nearly two decades ago. In 2001, the Forest Service promulgated the Roads Rule (referred to as "subpart A"). 66 Fed. Reg. 3206 (Jan. 12, 2001); 36 C.F.R. part 212, subpart A. The Roads Rule created two important obligations for the agency. One obligation is to complete a Travel Analysis. 36 C.F.R. § 212.5(b)(2). Another obligation is to identify the minimum road system needed for safe and efficient travel and for the protection, management, and use of National Forest system lands. *Id.* § 212.5(b)(1).

In 2008 the Santa Fe National Forest took the first step and completed its travel analysis report (2008 TAR). The next step under Subpart A is to consider the valid portions of the travel analysis report and begin to identify and implement the minimum road system (MRS) in the analysis of site-specific projects of the appropriate geographic size under NEPA.[1] Here, extensive time that has passed since the 2008 TAR, and it doesn't appear that the Forest Service has made any effort to identify the MRS within the project area. Given that the Encino Vista Project is considering changes to a number of roads, and given its geographic scale, this is precisely the type of project where the Forest Service must consider its Travel Analysis Report and identify unneeded roads to prioritize for decommissioning or to be considered for other for the Santa Fe National Forest, and identify the MRS.[2] This is essential to meeting the restoration purpose and need.

The minimum road system is the road system the Forest Service determines is needed to:

- "meet resource and other management objectives adopted in the relevant land and resource management plan";
- "meet applicable statutory and regulatory requirements";
- "reflect long-term funding expectations"; and
- "ensure that the identified system minimizes adverse environmental impacts associated with road construction, reconstruction, decommissioning, and maintenance."

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[1] See Memorandum from Leslie Weldon to Regional Foresters et al. on Travel Management, Implementation of 36 CFR, Part 212, Subpart A (Mar. 29, 2012) (hereafter 2012 Weldon Memo), page 2 (directing forests to "analyze the proposed action and alternatives in terms of whether, per 36 CFR 212.5(b)(1), the resulting [road] system is needed").
[2] *Id.* (directing forests to use the travel analysis report to identify the minimum road system for proposed actions at the scale of a 6th code subwatershed or larger).

36 C.F.R. §212.5(b)(1). The Forest Service should identify the minimum road system by analyzing whether a proposed project is consistent with the relevant portions of the travel analysis report and consider the minimum road system factors under 36 CFR 212.5(b)(1) for each road the agency decides to keep as part of the specific project.[3]

Subpart A directs the agency to "identify the roads on lands under Forest Service jurisdiction that are no longer needed," and therefore should be closed or decommissioned.[4] The rule refers to all roads, not just National Forest System roads. The rule defines a road as "[a] motor vehicle travelways over 50 inches wide, unless designated and managed as a trail."[5] The forest must assess these proposed actions in relation to the risks and benefits analysis in the Travel Analysis Report, as well as the factors for a minimum road system, with the goal of minimizing adverse environmental impacts.

We are concerned that the scoping notice for this project didn't even reference the Travel Analysis Report and has not identified any roads to be closed, much less decommissioned, or culverts to be replaced. We believe these actions are essential to achieve the restoration purpose and need of the project and move the forest towards a more economical and storm resilient road system. We expect to see the incorporation of the Travel Analysis Report into the draft Environmental Impact Statement, along with reasoning if decisions are made that are different from the recommendations in the report. Sometimes we have seen this as a table and sometimes as an appendix to the analysis. The overall purpose being to minimize environmental impacts, ensure reliable access and be in-line with road maintenance budgets.

IV. The Forest Service must prepare a robust Environmental Impact Statement (EIS) under NEPA.

The Encino Vista project, including the proposed Forest Plan amendment, "may" have a significant impact on the environment, and thus the Forest Service must prepare a robust EIS, ensuring that it complies with NEPA's required "hard look." The agency may not ignore topics if the information is uncertain or unknown. Where information is lacking or uncertain, the Forest Service must make clear that the information is lacking, the relevance of the information to the evaluation of foreseeable significant adverse effects, summarize the existing science, and provide its own evaluation based on theoretical approaches. 40 C.F.R. § 1502.22.

The EIS must analyze the baseline conditions of the project area, and the direct, indirect and cumulative impacts of the proposed timber management activities, road construction and maintenance, and all other activities. There are a number of large landscape projects throughout the Santa Fe NF, and the agency has yet to address the cumulative impacts of so much timber management and associated road building, especially on old growth, MSO and other species' and their habitats. Other issues that must be analyzed include the impacts of livestock grazing, particularly in light of climate change, on forest ecosystems, vegetation, riparian areas, and overall forest health. Livestock grazing and the poor condition of the road system must be assessed as part of the baseline conditions.

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[3] *Id.* ("The resulting decision [in a site-specific project] identifies the [minimum road system] and unneeded roads for each subwatershed or larger scale").
[4] 36 C.F.R. § 212.5(b)(2). See also Center for Sierra Nevada, 832 F. Supp. 2d at 1155 ("The court agrees that during the Subpart A analysis the Forest Service will need to evaluate all roads, including any roads previously designated as open under subpart B, for decommissioning.").
[5] 36 C.F.R. § 212.1.

a. The Forest Service should clearly articulate the statement of purpose to include its duty to identify the minimum road system, and provide support for the claimed need.

The Forest Service must shape the project's purpose and need statement according to applicable statutory and regulatory requirements. When the agency takes an action "pursuant to a specific statute, the statutory objectives of the project serve as a guide by which to determine the reasonableness of objectives outlined in an EIS." *Westlands Water Dist. v. U.S. Dept. of Interior*, 376 F.3d 853, 866 (9th Cir. 2004). The Forest Service has a substantive duty to address its over-sized road system. *See*, 36 C.F.R. 212.5. This underlying substantive duty must inform the scope of the project and be included in the agency's NEPA analysis. It's been nearly 2 decades since the agency finalized the Subpart A rules, and 11 years since the Santa Fe NF conducted its TAR, and the Forest Service can no longer delay in addressing this duty.

b. The Forest Service should accurately define the official road network as the baseline for the NEPA analysis.

The baseline and no-action alternative can differ.[6] Current management direction does not compel the Forest Service to recognize decommissioned roads and unauthorized roads as part of the road system. Disclosure of the number and location of decommissioned routes and unauthorized routes, as well as the impacts of those routes, is a necessary component of the no-action alternative. But it is separate and distinct from the identification of the baseline, which should be the official open route system.

In addition, it is helpful for public understanding to have clearly articulated which roads proposed for closure and decommissioning are already not drivable by the public due to lack of maintenance, road wash-outs and storm damage. It's incumbent upon the Forest Service to accurately describe the road network now, what is planned for the future and why those steps will be taken. There is significant room for improvement on how the agency describes the current challenges and how changes may or may not impact communities.

c. The Forest Service must consider a broad array of impacts related to forest roads in its NEPA analysis.

Impacts from Forest Roads

The best available science shows that roads cause significant adverse impacts to National Forest resources. Erosion, compaction, and other alterations in forest geomorphology and hydrology associated with roads seriously impair water quality and aquatic species viability. Roads disturb and fragment wildlife habitat, altering species distribution, interfering with critical life functions such as feeding, breeding, and nesting, and resulting in loss of biodiversity. Roads facilitate increased human intrusion into sensitive areas, resulting in poaching of rare plants and animals, human-ignited wildfires, introduction of exotic species, and damage to archaeological resources. We will look to see if the Santa Fe National Forest outlines a range of activities focused on reducing road impacts, as part of its' draft Environmental Impact Statement. These activities should include road maintenance, installation of BMPs, culvert replacements, hydrologically-disconnecting roads from streams, fish passage improvements, appropriate road closures (sometimes seasonal) and road decommissioning which can all be beneficial to wildlife, water quality, aquatic species and forest users if

[6] See, e.g., FSH 1909.15, 14.2; Council on Environmental Quality's (CEQ) Forty Most Asked Questions (1981), #3 (explaining "[t]here are two distinct interpretations of 'no action'"; one is "'no change' from current management direction or level of management intensity," and the other is if "the proposed activity would not take place").

properly considered and implemented. As this project moves forward, we ask that the Agency ensure that activities on the ground result in changes to the current net negative impacts from these roads.

Climate Change and Forest Resources

Climate change intensifies the impacts associated with timber management, livestock grazing and roads. Draft guidance from the Council of Environmental Quality (CEQ) indicates the Forest Service must include existing and reasonably foreseeable climate change impacts as part of the affected environment, assessed as part of the agency's hard look at impacts, and integrated into each of the alternatives, including the no action alternative. The Forest Service has a substantive duty under its own Forest Service Manual to establish resilient ecosystems in the face of climate change.[7] The Forest Service should analyze in detail the impact of climate change on the Forest, streams, groundwater, roads, and fish and wildlife habitat. Removing culverts, improving stream/road crossings, upgrading culverts, and decommissioning roads are all very important activities that can increase resiliency to climate change impacts. We encourage the Forest Service to consider climate change impacts – especially related to increasing storm intensity - to ensure that culverts are large enough and/or stream crossings are appropriately designed.

IV. The Forest should not construct temporary roads. If avoidance is impossible, the roads should be immediately reclaimed after use.

We encourage the Forest to take a hard look at the proposed temporary roads in order to be certain that they are needed. Current USFS policy is that road beds be restored to natural condition after such project, yet the scoping notice does not contain any such requirement. And, even when temporary roads are restored to natural condition, there is still an impact when temporary roads are developed. In addition to their hydrologic impact, roads fragment habitat, disturb wildlife, invite more noxious weeds and increase fire danger. Additionally, if they are not properly rehabilitated post-project, they can invite illegal incursions and more damage to natural resources. At minimum, we ask that the Santa Fe National Forest restore these segments as soon as the project activities within that specific area are completed. In addition, we ask that the segments are monitored and enforcement actions taken to ensure proper closure.

Conclusion

As conservationists and visitors to the Santa Fe National Forest, we are certain that with thoughtful planning and clear communication, the Santa Fe National Forest staff can create a true restoration project that includes less logging and road building, and more actual restoration actions such as identifying and implementing a minimum road system, road decommissioning, riparian restoration, returning natural fire to the ecosystem, decreasing livestock grazing, and improving watershed health and groundwater recharge. This endeavor is one of the most important efforts the Forest Service can undertake to restore aquatic systems and wildlife habitat, facilitate adaptation to climate change, ensure reliable recreational and community access, and lower operating expenses.

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7 See, e.g., FSM 2020.2(2) (directing forests to "[r]estore and maintain resilient ecosystems that will have greater capacity to withstand stressors and recover from disturbances, especially those under changing and uncertain environmental conditions and extreme weather events"); FSM 2020.3(4) ("[E]cological restoration should be integrated into resource management programs and projects . . . Primary elements of an integrated approach are identification and elimination or reduction of stressors that degrade or impair ecological integrity.").

If you have questions, please contact me.

Sincerely,

Judi Brawer
Wild Places Program Director
WildEarth Guardians
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(208) 871-0596

From: Chuck Hathcock
To: Imler-Jacquez, Sandra R -FS
Subject: Comments on the Encino Vista project #54965
Date: Monday, December 2, 2019 8:42:51 AM

Hi Sandra,

Please accept my comments below for the Encino Vista Landscape Restoration Project # 54965. Thanks!

Hello,
My name is Chuck Hathcock and I am a wildlife biologist in the Jemez and I also own property and live along the Rio Puerco just downstream of this project's boundary between Youngsville and Coyote. Overall I support this project and agree that this restoration needs to be completed before a stand-replacing wildfire hits this area. I have some comments for you to consider.

--You addressed owl habitat thoroughly, thank you.
--The use of goats is mentioned once as a means to control Gambel oak on page 10. Goats can be very destructive, you should detail the proposed use of goats and how you'll mitigate the impacts to other plant species on the lansdcape from the goats.
--You do not mention the Jemez Mountains salamander. While the project boundary looks to be just outside of designated critical habitat on the southern edge, this area is still within the salamander's historic range. You should have some considerations for salamander habitat along the boundary with the VCNP and just west of there.
--You do not mention any compliance requirements for the Migratory Bird Treaty Act. You should have some mention of this federal law and have best management practices that can help mitigate impacts to migratory birds. The primary impact would be from active nest destruction, so having some guidelines to prioritize tree and shrub removals outside of the peak bird nesting season would be appropriate.
--You do not mention removing non-native species. Areas along the Rio Puerco are choked closed with Russian olive and to a lesser extent salt cedar. You mention this and other watercourses in the background and mention that they are at risk, but I don't see any treatments listed for riparian health improvement. Consider adding treatment options that include non-native tree removals.
--Lastly, why is the northern goshawk the only RFSS species mentioned? Others could be impacted by this work as well such as the boreal owl, pale Townsend's bid-eared bat, spotted bat, water and masked shrews, and several plant species.

Thanks for your consideration.

September 30, 2019
Responsible Official: Jennifer Cramer, Forest Planner
Santa Fe National Forest 11 Forest Lane Santa Fe,
NM 87508 505-438-5442
Submitted via: santafeforestplan@fs.fed.us
Submitted by: Dominick A. DellaSala, Ph. D, Conservation Scientist

Re: **Comments on the Santa Fe National Forest Draft Land Management Plan and Draft Environmental Impact Statement (DEIS)**

Please accept these comments for the public record regarding the Santa Fe National Forest Draft Land Management Plan and DEIS. I am a conservation scientist with over 30 years-experience in forest ecosystems, including documenting the importance of fire-mediated biodiversity in dry pine and mixed conifer forests of the West (DellaSala and Hanson 2015[1]). My relevant expertise also includes developing robust conservation strategies for land managers to accommodate wildfires for ecosystem benefits while reducing fire risks to communities. I am submitting the enclosed publications as pdfs in support of my comments, including how extensive logging has increased fire severity in forests (Bradley et al. 2016), limitations of forest thinning in reducing fire intensity (DellaSala et al. 2018), livestock grazing and climate change cumulative impacts on national forests (Beschta et al. 2012), fire ignitions associated with roads (Ibisch et al. 2016), climate change effects on fire regimes (Abatzoglou and Williams 2017), and ecological importance of high severity burn patches in dry pine/mixed conifer forests including New Mexico (DellaSala and Hanson 2019), among other relevant peer reviewed publications. My comments are meant to improve the Forest Service's approach to forest-fire resilience in the Santa Fe National Forest (SFNF) and surroundings with the intent of showing how the agency can and must do better with respect to using the best available science along with involving scientists with a biodiversity perspective on wildfire and not just a fuel centric perspective dominated by fuel management.

The SFNF encompasses 1.6 million acres (nearly the size of Yellowstone National Park) of diverse conifer forests, woodlands, riparian forests, native grasslands and shrublands that make up the scenic beauty and quality of life for surrounding communities, including unmatched recreation, clean water, hunting and fishing, and iconic wildlife species. The SFNF includes nationally significant roadless areas; designated and proposed Wilderness and Wild and Scenic rivers; tribal-cultural sites; and essential habitat for large carnivores, ungulates, and at-risk wildlife such as Mexican Spotted Owl, Southwestern Willow Flycatcher, Jemez Mountain

[1] Note – a copy of the book – a very large pdf – can be purchased here https://www.sciencedirect.com/book/9780128027493/the-ecological-importance-of-mixed-severity-fires. For the purpose of these comments, I included the relative chapter, however, these included editing notes as the publisher did not provide chapter pdfs.

salamander, Rio Grande cutthroat trout, and New Mexico meadow jumping mouse. These and many other species in the project area require intact areas periodically maintained by wildfires of low and mixed severity effects on vegetation that also include occasional large and small patches of high severity fire effects. The SFNF's low elevation forests are predominately influenced by frequent fires of low severity with fire-flare ups that often kill overstory trees (site and landscape heterogeneity). During drought cycles and under extreme fire weather these flare ups can include small and large high severity patches that are important ecologically (DellaSala and Hanson 2019). Upper elevation spruce-fir forests are on centuries long *fire rotation intervals* (landscape scale) where high severity fire effects are characteristic (Margolis et al. 2002) and climatic factors are the top-down driver of fire behavior, not fuels (see Bessie and Johnson 1995). This variability is not appropriately recognized, planned for, or even properly analyzed in the DEIS, which mostly emphasizes mechanical treatments designed to move substantial amounts of closed canopy forests into low fuel condition conducive of low-severity fire effects lacking diversity/heterogeneity at site or landscape levels.

Much of the Santa Fe National Forest biodiversity is distributed along elevation gradients with changes in life zones prominent from valley bottoms and foothills to montane and alpine. Thus, a primary objective of the DEIS should be to maintain landscape connectivity that accommodates biological diversity across life zones and for focal species, species of conservation concern, and at-risk species and ecosystems. The DEIS is deficient in analyzing how connectivity is being impacted specifically by habitat fragmentation in the project area and surroundings (cumulative effects) caused by roads, extensive thinning and forest canopy reductions, ski area development, mining, livestock grazing and infrastructure, off highway vehicles (OHVs), and other developments. Connectivity cannot simply be maintained at the coarse-filter level via vegetation management and very general site-specific measures incorrectly presented as a fine filter approach. Connectivity maintenance requires proper analysis (species-specific trigger points and population viability analysis, see Noon et al. 2003, Schultz et al. 2013) to meet the best available scientific information (BASI) requirement of the 2012 forest planning rule. None of the alternatives in the DEIS meet the BASI requirement for connectivity (Box 1 and Box 2).

Box 1. Ecological integrity. The quality or condition of an ecosystem when its dominant ecological characteristics (e.g., composition, structure, function, connectivity, and species composition and diversity) occur within the natural range of variation and can withstand and recover from most perturbations imposed by natural environmental dynamics or human influence (36 CFR 219.19).

Box 2. Connectivity. Ecological conditions that exist at several spatial and temporal scales that provide landscape linkages that permit the exchange of flow, sediments, and nutrients; the daily and seasonal movements of animals within home ranges; the dispersal and genetic interchange between populations; and the long distance range shifts of species, such as in response to climate change.

Planning deficiencies regarding integrity and connectivity are summarized as follows:

- Connectivity is inadequately addressed by an emphasis on vegetation management in Ecological Response Units (mostly coarse filter), general site-specific measures (inadequate fine filter), and some road closures/decommissioning. Notably, in a comprehensive analysis of biodiversity strategies in a changing climate, connectivity (site-specific structural features, landscape intactness, corridors) was identified as the single most important strategy for enabling plants and wildlife to adapt to climate change and is critical to achieving climate resilient ecosystems (Haber and Nelson 2015). These authors recommend far more measures for maintaining connectivity than what was provided in the DEIS.
- There are substantial roads (6,900 miles) throughout the SFNF, many of which are leaking sediments into streams and pose a barrier and mortality risk to wildlife (vehicle collisions). Roads, cattle, and logging/thinning all affect the biological and physical environment of focal species, at-risk species, and species of conservation concern and this requires fine-scale analysis ("trigger points," and population viability analysis (PVA); as in Noon et al. 2003, Schulz et al. 2013) along with stepped up conservation (see conservation requirement of the planning rule below) that must be analyzed at the appropriate scale using BASI to take a hard look at connectivity and not just providing unsupported claims that vegetation management actions satisfy this requirement.
- The DEIS must analyze connectivity to maintain viable populations of focal species, at-risk species, and species of conservation concern (i.e., via PVA and trigger points) especially in a changing climate and in the context of both direct and indirect cumulative effects (e.g., analyze habitat fragmentation as the antithesis of connectivity).
- A connectivity analysis needs to incorporate cumulative impacts (e.g., livestock, thinning, roads), importance of intact areas (especially connecting life zones along gradients for species movements), and barriers to terrestrial and aquatic focal species, at-risk species, and species of conservation concern along with specific measures for reconnecting habitat. Examples of connectivity analyses include identification of species-specific road density thresholds (generally >1 mi/square mile is problematic for aquatic species), identification and protection of ungulate migration corridors (e.g., deer and elk winter and summer range movements) and large carnivore travel routes (especially along riparian areas) (i.e., the Forest Service must follow approaches in Haber and Nelson 2015).

Maintaining the mixture of fire severity effects on the SFNF is key to meeting the diversity requirements of the 2012 forest planning rule (see section on diversity of plant and animal communities), including mixed-severity fires that produce high-severity patches having unique ecological functions (DellaSala and Hanson 2019). The DEIS is deficient in this regard as it over emphasizes low-severity fire at the expense of mixed-severity fire effects (including high severity patches) essential to ecological processes, ecological conditions, and ecological integrity (Box 1, 3, 4, 5).

Box 3. The selected set of ecological conditions and key ecosystem characteristics related to the composition, structure, ecological processes, and connectivity of plan area ecosystems (terrestrial, riparian, and aquatic), provide the basis for monitoring ecosystem integrity (36 CFR 219.8(a)(1)) and the diversity of plant and animal communities (36 CFR 219.9).

Box 4. System drivers, including dominant ecological processes, disturbance regimes, and stressors, such as natural succession, wildland fire, invasive species, and climate change; and the ability of the terrestrial and aquatic ecosystems on the plan area to adapt to change (§ 219.8)

Box 5. Ecological conditions. The biological and physical environment that can affect the diversity of plant and animal communities, the persistence of native species, and the productive capacity of ecological systems. Ecological conditions include habitat and other influences on species and the environment. Examples of ecological conditions include the abundance and distribution of aquatic and terrestrial habitats, connectivity, roads and other structural developments, human uses, and invasive species.

The DEIS conflicts with the above planning rule requirements in the following ways:

- Alternative 3 (natural process alternative) is erroneously dismissed for Alternative 2 (preferred alternative) that relies on far more mechanical treatments than natural processes. More natural process features from Alternative 3 need to be incorporated into the final plan. Ostensibly, the main reason for the Forest Service rejecting Alternative 3 stems from inaccuracy problems inherent to LANDFIRE, fire scar analysis sampling biases, and inappropriate reference conditions tied to Forest Service research publication GTR-310 that have led to an over-reliance on mechanical treatments to achieve novel ecosystems devoid of most small trees with remaining trees prone to blowdown.
- The DEIS assumes high-severity fire patches are an anomaly of contemporary fire systems and therefore does not properly analyze positive contributions of high-severity patches in contributing to diverse ecosystems (DellaSala and Hanson 2019), particularly high elevation areas that experience characteristic high-severity fires (the predominant fire regime) on long fire rotation intervals.
- High-severity patches are ecological diverse habitats (DellaSala and Hanson 2019) and are important as foraging habitat for raptors such as Northern Goshawks and Mexican Spotted Owls (see Lee 2018), woodpeckers and songbirds (Hutto et al. 2015), small mammals and ungulates (Bond 2015), and may play a role in snowshoe hare/lynx dynamics. This needs to be acknowledged and the at-risk species tables in the DEIS

adjusted to include positive effects of high-severity fires on wildlife instead of all negative effects as incorrectly noted in the DEIS.

- The DEIS does not include sufficient actions for limiting the spread of invasive species via vector management of livestock (maximum permitted stocking rate of 11,400 AUMs is not sufficiently mitigated), roads, and OHVs. Improved foraging habitat for cattle through thinning and infrastructure changes under the preferred alternative is inadequate for addressing the chronic invasive species problems across the SFNF that will accumulate (cumulative effects) over space and time through active management (thinning entries), continued grazing especially in a changing climate (see Beschta et al. 2012) and road developments (see Ibisch et al. 2016 for a review of road impacts, including spread of invasive species).
- The DEIS is completely inadequate in addressing the critical habitat needs and population dynamics of threatened Mexican Spotted Owls (MSO), which require site specific and region-wide population monitoring and not just knowledge of habitat availability. Notably, a federal judge on September 11 enjoined all "timber management actions" in eleven national forests in New Mexico and Arizona for failing to survey and protect the MSO. The SFNF through Endangered Species Act section 7 consultation must engage in specific and region-wide population monitoring to ensure the MSO population is recovering and its habitat protected from thinning and other project actions (e.g., effects of livestock grazing on prey species).

Overall, the DEIS and supporting documents do not meet the BASI requirement of the 2012 forest planning rule with respect to *accurate, reliable, and relevant issues being considered* (Box 6). There are incorrect reference conditions tied to the Forest Service research publication GTR-310 extrapolated from a completely different region, accuracy problems inherent with the LANDFIRE program at the SFNF scale, uncertainties with fire return interval estimates using fire scar sampling, and arbitrary determinations regarding closed canopy forest conditions that has led to an over emphasis on mechanical treatments to achieve desired open forest canopy conditions at the expense of plant and wildlife diversity.

Box 6. § 219.3 Role of science in planning. The responsible official shall use the best available scientific information to inform the planning process required by this subpart. In doing so, the responsible official shall determine what information is the most accurate, reliable, and relevant to the issues being considered. The responsible official shall document how the best available scientific information was used to inform the assessment, the plan decision, and the monitoring program as required in §§ 219.6(a)(3) and 219.14(a)(4). Such documentation must: identify what information was determined to be the best available scientific information, explain the basis for that determination, and explain how the information was applied to the issues considered. (emphasis added)

The DEIS does not sufficiently meet the *conservation* requirement of the 2012 forest planning rule (Box 7).

Box 7. Conservation. The protection, preservation, management, or restoration of natural environments, ecological communities, and species. Conserve. For purposes of subpart A, § 219.9, to protect, preserve, manage, or restore natural environments and ecological communities to potentially avoid federally listing of proposed and candidate species.

Noted deficiencies in the *conservation* requirement include:

- Lack of preservation and protection of natural environments (especially roadless areas, closed canopy mature forests, riparian areas, critical MSO habitat), ecological communities, focal species, at-risk species, and species of conservation concern. Alternative 2, for instance, emphasizes extensive mechanical treatments that may cause irreparable harm to MSO, focal species, at-risk species, and species of conservation concern through major reductions in canopy closure and understory vegetation. Extensive thinning of forest canopies may constitute an adverse effects determination in section 7 consultation for the MSO (and prey species) that uses closed canopy forests for nesting and may use severely burned areas for foraging (see Lee 2018).

Importantly, the DEIS presents a questionable analysis of fire emissions derived from assumptions in the LANDFIRE program and does not include an appropriate analysis of the emissions from logging (in-boundary and transportation/manufacturing of wood products), livestock grazing and infrastructure, and road building/maintenance. An emissions analysis related to all project activities is necessary to properly assess air quality impacts to surrounding communities and CO_2 contributions to climate change with an alternative chosen that minimizes emissions related specifically to forest plan activities (direct, indirect, cumulative emissions impacts).

In sum, I am requesting that the Forest Service modify or include a new alternative that meets the following requirements:

- Identification and protection of specific connectivity areas (e.g., roadless areas, intact riparian and watersheds) for achieving viable populations of focal species, species of conservation concern, and at-risk species in a changing climate (see Noon et al. 2003, Schulz et al. 2013, Haber and Nelson 2015, especially Table 1). Such areas should be protected from mechanical treatments especially habitat of at-risk species (e.g., MSO).
- Consistent with the guidelines for connectivity in Haber and Nelson (2015:Tables 1 and 2), it is essential for the forest plan to identify key characteristics of connectivity (also

Haber and Nelson 2015:Table 3), including composition, structure, process/function at scale: site, landscape, corridors, riparian areas, and wildlife travel routes.

- An analysis and mitigation of how conditions on the SFNF and surrounding areas (logging, roads, development, grazing especially in riparian areas) affect connectivity (cumulative effects analysis).
- Substantial reduction in livestock AUMs and increase in riparian, native meadows, and aspen grove protections, restoration and invasive species containment. This should include opportunities for local conservation groups to purchase grazing leases from willing sellers with the allotments and AUMs permanently retired by the Forest Service. Livestock should be removed from riparian areas and curtailed in areas with native plant communities.
- Accuracy determination and field verification (and error correction) of LANDFIRE and forest canopy determinations, particularly in relation to site-specific reference conditions and ecologically appropriate definitions of closed canopy forests; the >30% closed canopy definition in the DEIS is arbitrary and has resulted in excessive canopy reduction measures to achieve "open" conditions.
- Use of multiple lines of evidence (e.g., see Odion et al. 2014a, Moritz et al. 2018) in estimating historic fire regimes and recognition/ analysis of the importance of mixed-severity fire effects on plant and wildlife diversity, including small and large patches of high-severity fire effects characteristic of drought cycles, fire flare ups, and upper elevation forests.
- A substantial reduction in mechanical treatments that are otherwise resulting in novel forest conditions that lack integrity and climate resilience because of the over-emphasis on open forest conditions that retain few trees. Forests opened by excessive thinning lack understory shrubs, forbs and small trees that contribute to climate resilience (see Baker and Williams 2015, Baker 2018); small trees may also have mature/old growth characteristics because of slow growth rates and more of them need to be maintained as an important understory cohort for future old-growth development (e.g., by creating small gaps and leaving many more tree cohorts).
- A focus on community wildfire risk reduction through partnerships with private landowners that emphasize defensible space measures for homes instead of extensive thinning in the backcountry.
- A substantial reduction in livestock grazing including large no-grazing zones that more aptly address cumulative effects of livestock, infrastructure, and climate change (see Beschta et al. 2012).
- A reduction in project related carbon dioxide emissions by a project level comparison of emissions alternatives.

My detailed comments and supporting publications follow this signature page.

Sincerely,

Dominick A. DellaSala

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UNCERTAINTIES OF FIRE SCAR METHODOLOGY, REFERENCE CONDITIONS, AND FAILURE TO MEET BASI REQUIREMENTS OF THE PLANNING RULE

The 2012 planning rule requires forest plans to meet the best available scientific information (BASI) standard during planning assessments. Ryan et al. (2018) provide specifics on how best to meet this standard illustrated in their Figure 2:

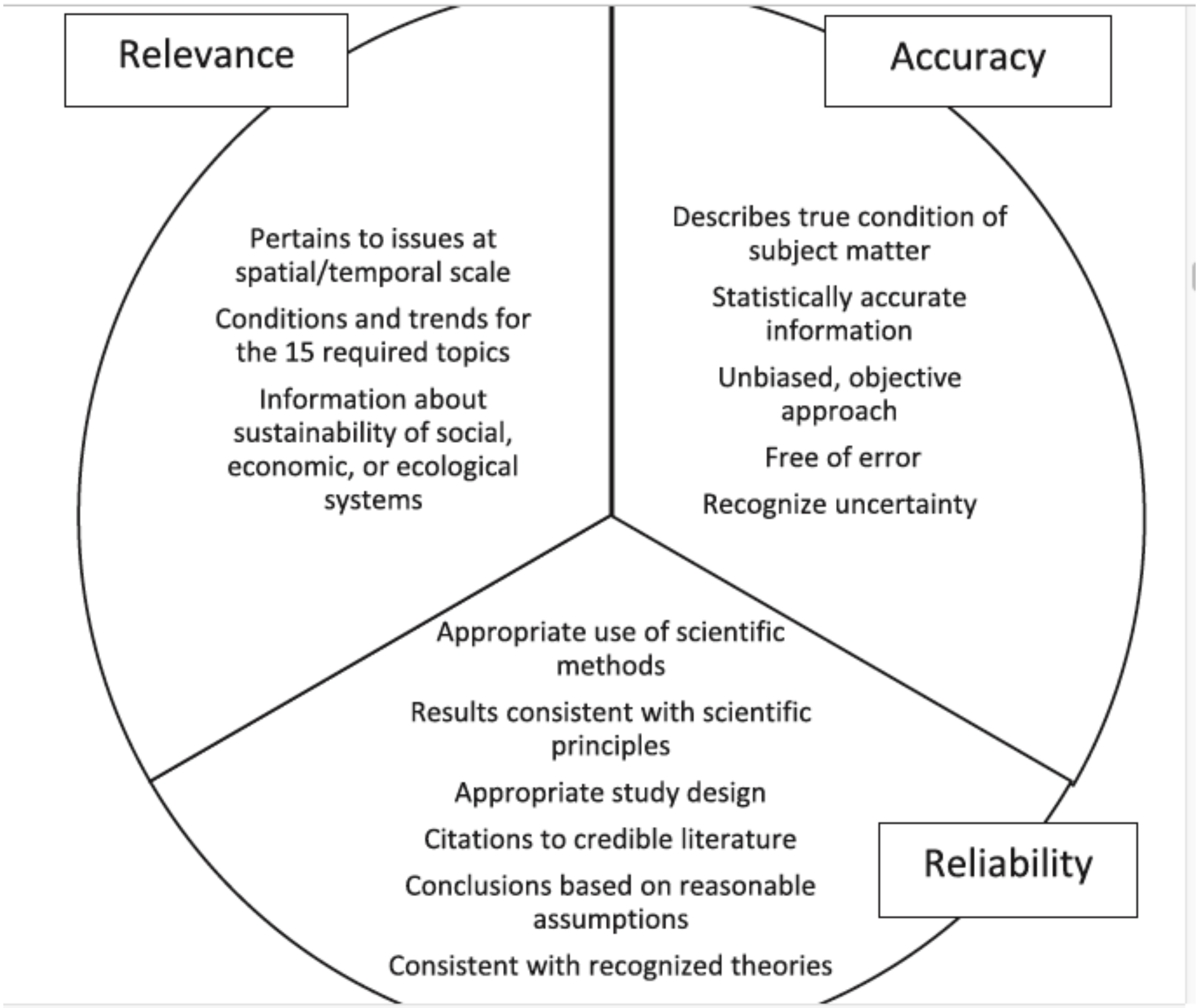
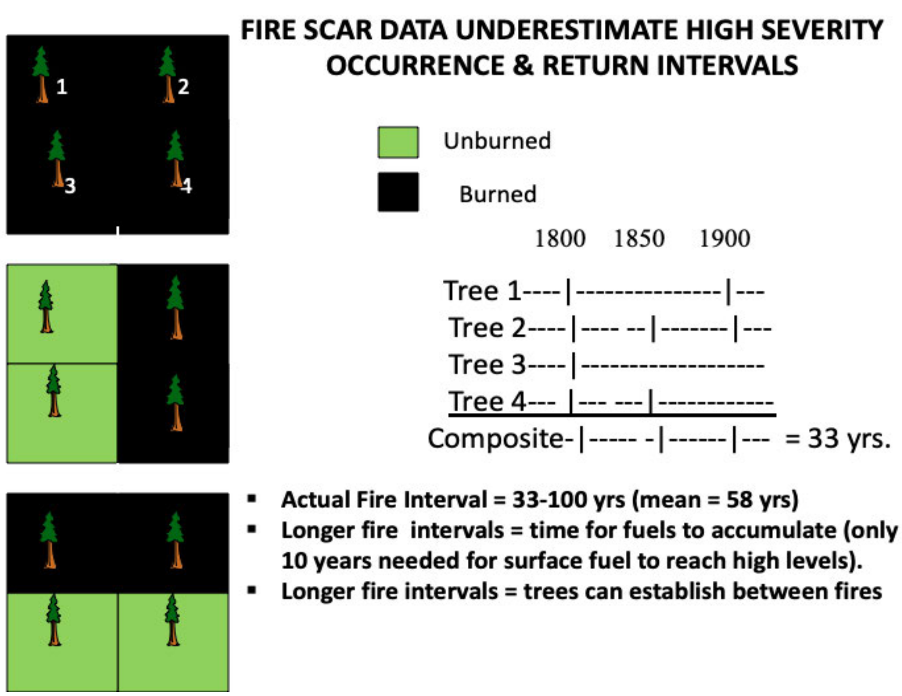


Figure 2 (from Ryan et al. 2018). Criteria for determining best available scientific information (BASI). Source: Forest Service Handbook 1909.12.07.12

According to Ryan et al. (2018) "the definition of BASI is contained in the "zero code" chapter of the handbook and specifies three primary criteria: accuracy, reliability, and relevance (FSH 1909.12.07.12), in addition to referencing the Data Quality Act (PL 106–554) for guidance on evaluating available information (Figure 2). Available is defined as information that currently exists in a form useful for the planning process without further data collection, modification, or validation (FSH 1909.07.01).

Based on the BASI standard above (especially Ryan et al. 2018: Figure 2), there are two main problems with the DEIS fire assumptions: (1) over reliance on fire scar estimates used to determine fire return intervals that are then extrapolated over large areas instead of the more appropriate use of multiple lines of evidence used to calculate *fire rotation intervals* (landscape scale; see Odion et al. 2014a, Moritz et al. 2018); and (2) accuracy and reliability problems with use of LANDFIRE to estimate reference and contemporary conditions in forest plan analyses (discussed below).

Fire return intervals are biased - While local sampling is important for estimating fire return intervals at the stand level, there are significant uncertainties with extrapolating fire scar point sampling data over large landscapes often used by researchers to re-construct historic fire regimes for comparisons to contemporary conditions (Baker 2017). They include sample-site selection bias, lack of tree scars in fire-killed trees (thereby underestimating high severity occurrence), and inappropriate extrapolation of site-specific data to draw landscape-level inferences (Baker 2017). The hypothetical figure below illustrates the inherent sampling bias of grouping individual fire scar data to construct composite fire interval (mean CFI).



In sum, variability in CFI estimates is masked whenever the mean return interval is used (instead of the range or scope-of-inference is inappropriately extrapolated from sites to large areas and whenever measures of central tendency (rather than the range) are used in fire return intervals.

This results in a cascade of errors beginning with a bias toward very short fire return intervals (i.e., because the mean and not the range was used), conclusions that historic conditions were predominately open forests (especially when open is arbitrarily defined using LANDFIRE, see below), conclusions that contemporary forest conditions are way out of bounds, and, the inappropriate need for aggressive mechanical treatments. To fix this problem, the best estimator of fire intervals at landscape scales is to use the fire rotation interval (Baker 2017).

Baker (2017) notes that fire rotations at the landscape scale can be derived from:

1. Areas burned in recent fires from agency fire records or records from remotely sensed data.
2. Historical areas burned reconstructed from scarred trees or plot locations.
3. Historical areas burned reconstructed using a ratio method and scarred-tree or plot records, or comparable data in a table or graph.

The Forest Service must provide information on the fire rotations using methodologies in Baker (2017) supplemented wherever possible with the paleo-ecology literature that can be used to reduce sampling bias associated with shorter sampling timelines and lack of high severity detectability from fire scar extrapolations. For instance, Baker (2017) goes through each source of bias in tree-ring reconstructions and shows that using corrected estimators actually yields longer fire rotation periods for dry pine/mixed conifer areas. Note that Figure 3 and Figure 4 in Baker (2017) show the diversity of fire rotations (longer intervals) in the Santa Fe area and the S2 Table has individual estimates for New Mexico. The sampling bias in fire-scar data must be disclosed as the DEIS is based mainly on fire-scar interpolation from plots to landscapes thereby compounding errors.

To correct for sampling bias, the Forest Service must account for variability in fire-free intervals using more robust methodologies, disclose whether there are historic accounts of fires in the DEIS area beyond just fire-scars, and include paleo-ecology studies from comparable sites to illustrate variability in fire regimes over longer time intervals. Significant discrepancies and debate among researchers about fire scar sampling must be disclosed (e.g., see Odion et al. 2016 response to Stephens et al. 2016 and Moritz et al. 2018).

As an example, a key fire-history study for the Santa Fe watershed is Margolis and Balmat (2009). These researchers indicate that the historical low-severity fire rotation in this watershed for dry pine forests was estimated at 39.80 years. They define frequent fire as < 25 years. Using their definition means that the Santa Fe watershed would not qualify as a frequent-fire regime, as this is a sufficient mean number of years between surface fires to allow understory fuels including shrubs and small trees to accumulate levels that would certainly enable the occurrence of some mixed and high-severity fires and some dense forests overtime. Moreover, this longer period corresponds with the paleo-record from charcoal sediments showing that when wet periods are followed by successive droughts, large fires, including patches of high severity, do indeed occur (Meyer 2010).

It is important to accommodate this variability in fire return interval estimates as heterogeneity in the ensuing burn severity patches at the landscape scale results in high levels of biodiversity (i.e., pyrodiversity of fire severity patches begets biodiversity, DellaSala and Hanson 2015). Notably, even slight differences in fire-return intervals are consequential. Baker (2017) reports that understory fuels in dry forests recover after fires in 7-25 years. If mean fire-return intervals were <25 years, understory fuels would be limited. However, if the interval was >25 years, as reported by Margolis and Balmart (2009), then shrubs and small trees would recover across the landscape and excessive thinning to shift forest to more open-canopy forests with minimal small tree and shrub cover would be inappropriate at large spatial scales.

The role of shrubs and understory vegetation is also a key ecosystem component in dry forests allowing for nutrient cycling and below-ground processes, water absorption and retention, provision of wildlife habitat, pollination and other ecosystem services. Spatial heterogeneity in fire-return intervals at landscape scales is a key indicator of resilience as it allows for both fire refugia (longer return intervals) and fire-maintenance (short return intervals). It is essential to manage for this variability at the site and landscape scale to accommodate wildlife that require a range of severity effects on vegetation: low, moderate and high severity. In other words, when it comes to fire, nature is complex (e.g., mixed severity) while management tends for uniformity (mainly low severity) typically at the expense of fire-mediated biodiversity.

The following Baker (2017) observations about fire interval estimators need to be addressed in the DEIS:

> "Dry-forest landscapes until recently were thought to have historically been primarily old growth forests, with a history of frequent low-severity fire, across their extent (e.g. [72]), but this has been refuted by GLO reconstructions and early aerial photographs (Table 6), paleoecological evidence [24], and early forest-reserve reports and other evidence [63 , 73]. Even in Arizona, which had abundant old forests with frequent fire (Fig 3), denser forests and high severity fire were extensive at certain times and in certain places, as on Black Mesa and parts of the Mogollon Plateau [60 , 73]. It is sensible to restore low-severity fire to its former dominance in the parts of dry-forest landscapes with a history of primarily low-severity fire, historically averaging about 34% of western dry-forest landscapes (Table 6). Estimated mean PMFI/FRs [population mean fire interval/fire rotation] here provide a guide for restoration and management of low-severity fire in extant old-forest parts of landscapes. For most dry-forests today, which are not old, using frequent fire (PMFI/FR <25 years) in restoration is not supported, and fuels do not need to be substantially reduced, because historical PMFI/FRs naturally allowed historical shrubs and small trees to fully recover after fires. Restoration of low-severity fire is still

> needed. The most appropriate approach, given likely long but uncertain mean rates of historical low-severity fire, is for most dry forests today to receive at most one prescribed fire, followed by managed fire for resource benefit, with the goal of mimicking mean historical PMFI/FRs and variability in fire (fire-size distributions, unburned area) as forests reach old age."

Thus, based on Baker (2017) and the problems noted in estimating fire return intervals, the DEIS needs to greatly scale back thinning except where thinning of small trees is needed to re-introduce fire nearest homes.

Problems with GTR-310 reference conditions - The DEIS tiers to GTR-310 (Restoring Composition and Structure in Southwest Frequent Fire Forests). However, GTR-310 does not even align with the geographic scope of the SFNF, as the SFNF is within the Colorado Rockies Forest Ecoregion yet GTR-310 is predominately within the Arizona Mountain Forest Ecoregion, which has a different climate, soil types, historical conditions, and fire regime. Extrapolating from one region to another is inappropriate (Moritz et al. 2018) and thus GTR-310 cannot be relied on for project-specific descriptions or actions.

The DEIS relies upon General Technical Report 310 as a primary source for desired conditions in the SFNF. This is inappropriate because none of the reference studies were from the Sangre de Cristo Mountains, and the two locations in the Jemez Mountains represent just 12 acres of sampled forest. The DEIS should instead rely on site-specific reference conditions and exercise caution when extrapolating fire regimes and forest structures from one geographic location to another given differences in vegetation, fire rotation intervals, elevation gradients, regional climate, and the influence of a rapidly changing climate on contemporary and future fire conditions (see Moritz et al. 2018). Thus, applying the "Flagstaff fire model" derived from GTR-310 is completely inappropriate for the SFNF.

ACCURACY PROBLEMS WITH LANDFIRE NEED FULL DISCLOSURE AND CORRECTION

Fire regime condition class (FRCC) and LANDFIRE vegetation models and maps are used by the Forest Service in planning assessments. These approaches are useful at large spatial scales (national) but they have well known accuracy problems at the project level that need full disclosure, cross validation with field data, and error correction.

For instance, Swetnam and Brown (2010) examined accuracy of LANDFIRE and FRCC assessments in Utah for similar vegetation types as in the DEIS planning area (Box 7).

Box 7. "LANDFIRE map data were found to be ~58% accurate for BpS and ~60% accurate for existing vegetation types. Results suggest that limited sampling of age-to-size relationships by different species may be needed to help refine reference condition definitions used in FRCC assessments, and that more empirical data are needed to better parameterize FRCC vegetation models in especially low-frequency fire types."

Zhu et al. (2006) tested the vegetation mapping protocol of LANDFIRE and likewise *concluded mapping accuracies of 60% or better at 30-m spatial resolution*. Notably, such low accuracy determinations do not comport with Ryan et al. (2018) summary of BASI criteria (their Figure 2 above) and the intent of the Data Quality Act.

Helmbrecht and Blankenship (2016) tested the ability of LANDFIRE to accurately reflect the true or accepted geographic location of features finding problems with errors in feature locations, source data, precision of field measurements, and data entry. Problems in map unit assignments may arise through "*errors or limitations in remote sensing data, field plots, statistical modeling, processing logic, or a combination of these and other factors*" (emphasis added). This is especially the case for forest vegetation and fuels data depending on the age of the source data. For instance, LANDFIRE data are updated every two years but by the time the data are delivered to the user, the data are 3 or more years out of date.

To correct for these problems, Helmbrecht and Blankenship (2016) recommend (and the DEIS should as well) include the following:

1. update for landscape changes that have occurred since the LANDFIRE version,
2. calibrate to local data and knowledge,
3. improve the thematic agreement (accuracy),
4. change the spatial or thematic resolution (e.g. lump or split map units),
5. modify the map unit classification,
6. create additional data versions that reflect temporal variability (e.g. peat soils being available for burning in drought situations, or exotic annual grasses being present in wet years but not dry years),
7. facilitate comparative analysis by creating data versions (e.g. analyzing pre- and post-treatment effects or comparing treatment alternatives),
8. analyze future conditions (e.g. modifying data to represent future conditions under a climate change scenario).

In Northern Idaho, Hyde et al. (2015) evaluated two LANDFIRE fuel loading raster options: (1) Fuels Characteristic Classification Systems (LANDFIRE-FCCS); and (2) Fuel Loading Model (LANDFIRE-FLM) vs. measured fuel loadings for a 20,000-ha mixed conifer study area. They found that the LANDFIRE-FCCS layer showed 200% higher duff loadings relative to measured loadings that led to 23% higher total mean consumption and emissions when modeled in FOFEM. The LANDFIRE-FLM layer showed lower loadings for total surface fuels relative to measured data, especially in the case of coarse woody debris that led to 51% lower mean total consumption and emissions when modeled in FOFEM. Additionally, LANDFIRE-FLM consumption was *59% lower relative to that on the measured plots, with 58% lower modeled emissions*. The authors concluded that these differences in fuel loadings led to significant differences in consumption and emissions depending upon the data and model chosen. The DEIS therefore needs to disclose how errors in fuel loading consumption were addressed in emissions determinations regarding wildfires and how these errors were corrected.

In the Sierra Nevada region, Odion and Hanson (2006) found FRCC *was not able to accurately predict occurrence of high-severity fire* (Box 8).

Box 8. "We found that the proxy for fire suppression effects, Condition Class, was not effective in identifying locations of high-severity fire. Condition Classes 2, 3, and 3+ in the McNally fire all had similar fire severity proportions. When the same Condition Class criteria were applied to the other two fires, we found that fire severity generally decreased rather than increasing from Condition Class 2 to 3+. In short, Condition Class identified nearly all forests as being at high risk of burning with a dramatic increase in fire severity compared to past fires. Instead, we found that the forests under investigation were at low risk for burning at high-severity, especially when both spatial and temporal patterns of fire are considered. The lack of an observed relationship between Condition Class and fire severity suggests that exogeneous forces such as weather, climate, topography, and neighboring vegetation (for example, pyrogenic shrubs) largely determine fire-severity patterns in forests."

Vogelmann et al. (2014) recognized four major potential sources of error associated with field plot data used in LANDFIRE:

1. Errors occur frequently in the identification of species and measurement of vegetation structure in the field (for example, in the data for one prototype field plot, a misplaced decimal point indicated a shrub height of 60 feet).
2. The vegetation on some field plots has undoubtedly changed between the time the field data were collected and when the imagery was acquired.
3. Geo-location errors in plot and imagery data result in inaccurate characterization of some imagery pixels.
4. The assignment of plots to specific vegetation classes will have errors associated with the wide array of opinions among professional field ecologists regarding the field classification of any given field plot.

To correct for these problems, Vogelmann et al. (2014) suggest (and the DEIS should follow) that the Forest Service conduct a suite of accuracy assessment methods for LANDFIRE, ranging from mostly qualitative assessments (such as the critical inspection of products, consultation with regional experts, and comparisons with existing data sets) to more quantitative analyses (such as cross-validation assessments, traditional accuracy assessments at the superzone level, and select evaluations at local levels). These combined approaches will provide the Forest Service with the accuracy information necessary to facilitate the appropriate use of the data in the DEIS.

Cruz and Alexander (2010) note additional problems with related fire modeling summarized in their abstract. The Forest Service needs to disclose errors associated with use of these models in the DEIS:

Abstract. To control and use wildland fires safely and effectively depends on creditable assessments of fire potential, including the propensity for crowning in conifer forests. Simulation studies that use certain fire modelling systems (i.e. NEXUS, FlamMap, FARSITE, FFE-FVS (Fire and Fuels Extension to the Forest Vegetation Simulator), Fuel Management Analyst (FMAPlus[!]), BehavePlus) based on separate implementations or direct integration of Rothermel's surface and crown rate of fire spread models with Van Wagner's crown fire transition and propagation models are shown to have a significant underprediction bias when used in assessing potential crown fire behavior in conifer forests of western North America. The principal sources of this underprediction bias are shown to include: (i) incompatible model linkages; (ii) use of surface and crown fire rate of spread models that have an inherent underprediction bias; and (iii) reduction in crown fire rate of spread based on the use of unsubstantiated crown fraction burned functions. The use of uncalibrated custom fuel models to represent surface fuelbeds is a fourth potential source of bias. These sources are described and documented in detail based on comparisons with experimental fire and wildfire observations and on separate analyses of model components. The manner in which the two primary canopy fuel inputs influencing crown fire initiation (i.e. foliar moisture content and canopy base height) is handled in these simulation studies and the meaning of Scott and Reinhardt's two crown fire hazard indices are also critically examined.

DellaSala et al. (2015) further summarize the problems with fuel models and simulations not comporting with field data and resulting in over-emphasis of efficacy of fuel reduction treatments and these uncertainties need to be addressed in the DEIS as follows:

"Fuel reduction also has been overpromised to be effective, using questionable logic and unvalidated models. First, fire intensity in most forest types is much more strongly affected by wind than by fuel. High fire-line intensity, the primary fire characteristic that promotes crown fires, is the product of the energy released by burning fuel and the rate of spread of fire (Alexander, 1982). Energy release by fuel varies over perhaps a 10-fold range, however, whereas rate of spread can vary over more than a 100-fold range; thus a high rate of spread caused by strong winds can easily overcome the limited reductions in fuel that are feasible (Baker, 2009). This was confirmed by a recent analysis of the 2013 Rim Fire in California, which concludes: "Our results suggest that even in forests with a restored fire regime, wildfires can produce large-scale, high-severity fire effects under the type of weather conditions that often prevail when wildfire escapes initial suppression efforts. . . . During the period when the Rim fire had heightened plume activity... no low severity was observed [in thinned areas], regardless of fuel load, forest type, or topographic position" (Lydersen et al., 2014, p. 333). Second, common fire models used to show that forests would be fire-safe after fuel reductions have an underprediction bias and are not validated. These flawed models include NEXUS, FlamMap, FARSITE, FFE-

FVS, FMAPlus, and BehavePlus (Cruz and Alexander, 2010; Alexander and Cruz, 2013; Cruz et al., 2014). The underprediction bias means that these models often predict that fuel reductions would reduce or eliminate the potential for crown fires in forests, when in fact fuel reductions do not achieve this effect. Fixing these models would be difficult and has not yet occurred (Alexander and Cruz, 2013). Also, these models have not been sufficiently tested and validated using a suite of actual fires, in which case they would likely be shown to fail (Cruz and Alexander, 2010). Alternative validated models are available and could be further developed, but they are not being used (Cruz and Alexander, 2010). Further, studies of tree mortality in thinned areas following fire do not typically take into account the mortality caused by the logging itself before the fire, leading to further biased results."

As further noted by DellaSala et al. (2015) "these concerns should raise red flags about the effectiveness of fuel treatments, as well as issues regarding liability and responsibility. Imagine if a company sold airplanes with identified flawed designs and without adequate test flights, which then crashed. There are thus sound scientific reasons to closely scrutinize government wildland fuel-reduction programs. Meanwhile, we need to be honest and warn the public that living within or adjacent to natural forests prone to burn is inherently hazardous. Only treating fuels in the immediate vicinity of the homes themselves can reduce risk to homes, not backcountry fuel reduction projects that divert scarce resources away from true home protection (Cohen, 2000; Gibbons et al., 2012; Calkin et al., 2013; Syphard et al., 2014)."

Closed Canopy Conditions Arbitrarily Defined - the DEIS arbitrarily defines closed canopy conditions in the mixed conifer-frequent fire and ponderosa pine ERUs as when *woody cover exceeds 30%* (DEIS: Figure 14, Figure 16), using LANDFIRE to determine the reference/baseline condition and contemporary departure indices for alternative analyses. The preferred alternative is based on moving closed canopy forests into desired open canopy conditions over 50 years. Closed canopy forests in some cases currently exceed 70% overstory cover and thus extensive thinning in the preferred alternative constitutes a major change in overstory cover impactful to species requiring closed canopy conditions. Large interspaces will be created across the landscape with substantial reductions in canopy cover and percent of forests in closed conditions to meet this arbitrarily defined "open" reference condition, creating novel ecosystems that do not comport with the ecological integrity or diversity requirements of the planning rule.

Importantly, Scott (2008) documented seven potential shortcomings with the canopy and fuel related provisions of LANDFIR, including:

- canopy cover values are too high,

- data discontinuities exist within and between map zones,
- canopy bulk density values are too low for use in FARSITE,
- canopy base height is too high to generate crown fire,
- treelist data sources may not be best for canopy fuel calculations
- alternative canopy fuel calculation programs may produce different results
- Refreshing and calibrating LANDFIRE data is needed

Scott (2008) reported that the dead fuel moisture model is especially sensitive to errors in canopy cover and concludes:

"Moreover, canopy cover mapping errors may lead to significant indirect fire modeling effects. Because canopy cover is a keystone variable, these indirect effects are difficult to quantify. If canopy cover is overestimated, LANDFIRE may subsequently map the incorrect fuel model, incorrect CBD, incorrect CBH, etc., all of which can strongly affect fire modeling outputs in a geospatial fire analysis."

"Because it is used as an independent variable, the importance of an accurate canopy cover layer in the LANDFIRE process should not be underestimated."

THE DEIS NEEDS TO RECOGNIZE THE ECOLOGICAL IMPORTANCE OF MIXED-SEVERITY FIRES, INCLUDING LARGE AND SMALL HIGH SEVERITY PATCHES FOR POSITIVE CONTRIBUTIONS TO PLANT AND WILDLIFE DIVERSITY

While low elevation pine and mixed conifer forests are predominately maintained by frequent fires of low severity effects on vegetation, there are occasional canopy flare ups and high-severity patches related to local fire-weather conditions, slope, aspect, elevation, and vegetation condition. This variability in fire effects needs to be maintained under the diversity requirement of the planning rule. Instead, the DEIS includes no analysis of the positive effects of mixed-severity fire influences on plant and wildlife communities, especially in upper elevation forests where fires are on centuries long rotation intervals (landscape scale) and produce diverse ecosystem effects including the creation of complex early seral forests (Swanson et al. 2011).

Notably, high-severity fire patches generate "biological legacies" (large live and dead trees, logs, shrubs) essential to forest succession and the maintenance of complex early seral forest conditions (Swanson et al. 2011, DellaSala et al. 2014, DellaSala and Hanson 2015, DellaSala and Hanson 2019). Large and small high severity patches provide important foraging habitat for Mexican Spotted Owls (federally listed, Lee 2018), Northern Goshawks (at-risk species), ungulate foraging habitat (Bond 2015), snowshoe hare/lynx dynamics, woodpeckers (including at-risk species: Lewis's woodpecker) and songbirds (Hutto et al. 2015), bats (Chambers and Saunders 2013), and boreal owls (at-risk species) in upper elevation spruce-fir forests. The DEIS inappropriately and arbitrarily assumes high-severity patches constitute wildlife habitat degradation (e.g., DEIS Volume 1: Tables 51, 57; "catastrophic fire analysis" p. 244).

Using LANDFIRE, the DEIS inappropriately assumes that current fire return intervals are highly departed from reference conditions (86%) as is fire severity leading to what the DEIS claims is a departure from NRV (DEIS Volume 1:89). However, based on a study of high-severity patches in dry pine and mixed conifer forests across the West, including New Mexico, large (>400 ha) high-severity fire patches have not been increasing since the 1990s (DellaSala and Hanson 2019.

From DellaSala and Hanson (2019):

Over the entire time series, 1984-2015, there was a significant increasing trend in the combined total area of CESF [complex early seral forests] patches >400 ha in each year ($\tau = 0.407$, $p = 0.001$), but no trend in patch size ($\tau = 0.009$, $p = 0.802$). However, when the data were analyzed by time periods, there was only a significant difference in the annual area of CESF habitat created by high-severity fire relative to the earliest time period (1984-1991), but no significant differences were detected among time periods since the early 1990s (Table 1, Figure 2). With regard to size of individual large CESF patches, there were no significant differences detected among time periods (Table 2).

Table 1. Critical values ($q_{0.05, 4}$), absolute difference between mean of ranks ($|R_A\text{-}R_B|$), standard errors (SE), and test statistics (q) to assess statistical significance, at $\alpha = 0.05$, of any differences between the four time groups ("1" = 1984-1991, "2" = 1992-1999, "3" = 2000-2007, and "4" = 2008-2015) for total annual area of CESF patches >400 ha using the Nemenyi non-parametric test for multiple comparisons between groups with an equal sample size (n = 8 years for each time group). Statistical significance of levels of q are shown as "Y" (significant) or "N" (not significant).

Time group comparison	$q_{0.05,4}$	$\|R_A\text{-}R_B\|$	SE	q	Significant? ($q > _{0.05,4}$?)
1-2	3.63	45.0	26.53	1.70	N
1-3	3.63	108.0	26.53	4.07	Y
1-4	3.63	107.0	26.53	4.03	Y
2-3	3.63	63.0	26.53	2.37	N
2-4	3.63	62.0	26.53	2.34	N

3-4	3.63	1.00	26.53	0.04	N

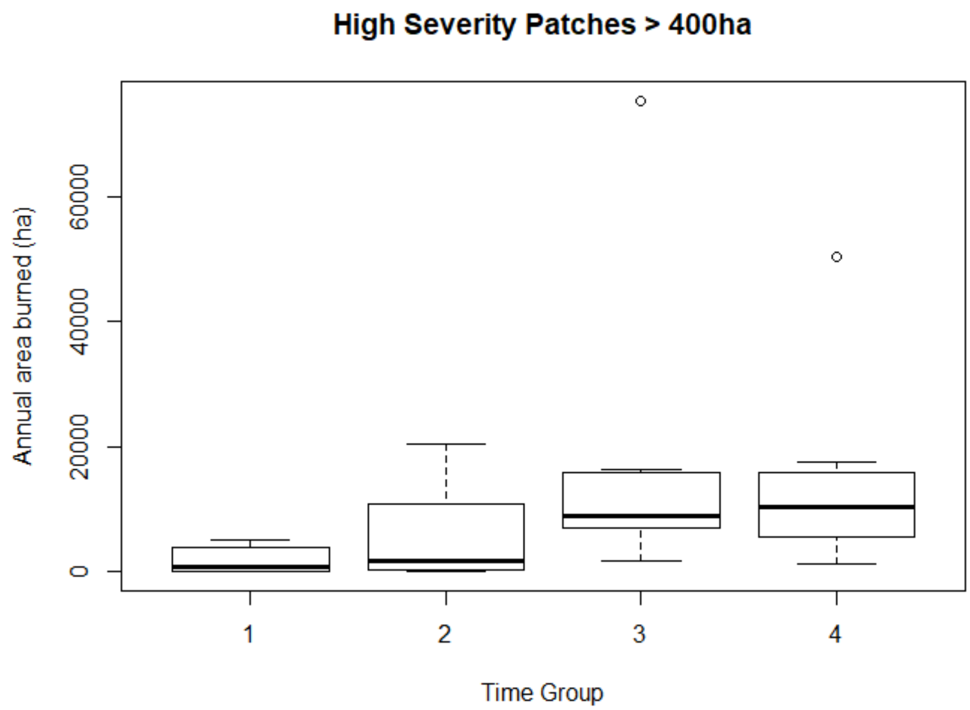


Figure 2 from DellaSala et al. Annual area of large patches (>400 ha) of high-severity fire in the four time periods ((“1” = 1984-1991, “2” = 1992-1999, “3” = 2000-2007, and “4” = 2008-2015).

Thus, the DEIS claims about uncharacteristically severe fires, for which mechanical treatments are based upon, cannot be substantiated by empirical data (including from New Mexico) and thus the DEIS does not meet the BASI requirements.

Importantly, Hutto et al. (2016) recommended that managers maintain ecological integrity of western dry pine and mixed-conifer forests through a more informed approach to the importance of mixed and high-severity fires. Here is their abstract:

Abstract. *We use the historical presence of high-severity fire patches in mixed-conifer forests of the western United States to make several points that we hope will encourage development of a more ecologically informed view of severe wildland fire effects. First, many plant and animal species use, and have sometimes evolved to depend on, severely burned forest conditions for their persistence. Second, evidence from fire history studies also suggests that a complex mosaic of severely burned conifer patches was common historically in the West. Third, to maintain ecological integrity in forests born of mixed-severity fire, land managers will have to accept some severe fire and maintain the integrity of its aftermath. Lastly, public education messages surrounding fire could be modified so that people better understand and support management designed to maintain ecologically appropriate sizes and distributions of severe fire and the complex early-seral forest conditions it creates.*

DellaSala et al. (2017) recommend that managers include mixed-severity effects in dry pine and mixed conifer forests to achieve ecological integrity and plant diversity. And while much of the

DEIS project area can be assumed to be in a xeric pine condition, mixed-severity fire effects, including large and small high-severity patches are indeed characteristic, need to be maintained, and are being grossly underestimated in ecological importance throughout the DEIS. Thus, the DEIS does not meet the BASI requirements of the planning rule as well as the diversity, ecological processes, ecological conditions, and integrity provisions as noted.

BIASED APPROACHES, AREAS OF AGREEMENT & DISAGREEMENT NEED TO BE ACKNOWLEDGED AND CORRECTED

Bias: The DEIS needs to clearly state scientific disputes (disagreements) and avoid biased perspectives on fire as generally noted by Iftekhar and Pannell (2015) and Moritz et al. 2018 (below). The following biased perspectives are inherent in the DEIS:

- Action bias – tendency to take actions even when it is better to delay action (in this case the impacts of aggressive thinning and roads may be more significant than effects of fire on ecosystems given uncertainties of treatment effectiveness as noted).
- Framing effect – tendency to respond differently to alternatively worded but objectively equivalent descriptions of the same item (use of catastrophic fire terminology in the DEIS that fails to account for ecosystem benefits of mixed-severity fires, including periodic flare-ups of high severity patches).
- Reference-point bias – tendency to overemphasize a pre-determined benchmark for a variable when estimating the level of that variably (i.e., over-reliance on fire scar sampling in the DEIS rather than presenting more robust and multiple lines of evidence).
- Satisficing rule – tendency to stop searching for a better decision (i.e., a NEPA based range of alternatives) once a decision that seems sufficiently good is identified.
- Loss aversion – tendency to value losses more highly than similar gains (i.e., managing wildfire of moderate-high intensity for ecosystem benefits instead of avoiding it by mechanical thinning and fire suppression as in the DEIS).
- Limited reliance on systematic learning – tendency to use information from past successful efforts rather than using information from both successful and failed efforts via extensive and well-funded ecosystem monitoring (adaptive management and learning is not possible without well-funded monitoring).

The best way to avoid these biases is to use multiple lines of evidence in re-constructing fire regimes, not rely mainly on fire scars, and conduct well-funded monitoring studies that fully assess project effects on species of conservation concern and ecological and cultural values. Multiple lines of evidence and monitoring are discussed in Odion et al. 2016 and Moritz et al. (2018) in the Common Ground Report (see below).

Areas of Agreement/Disagreement (Common Ground): I participated as one of the respondents in the so-called "Common Ground" report and am thoroughly familiar with the report's findings. The DEIS should pay particular attention to the following key findings in relation to areas of agreement, uncertainty, and disagreement and adjust project actions accordingly.

Areas of Agreement (high certainty):

- The role of changing climatic conditions is increasingly important in influencing fires.
- Multiple fire ecology and fire history research can be useful.
- Heterogeneity of fire effects, including patterns of patches created by fires of all severities, is important to forest resilience to future fires.
- Generalized models of historical fire regimes vary by ecoregion and forest type.
- Even within the same ecoregion and forest type, there is variation in historical fire regimes among differing environmental gradients.
- Historically, some degree of low-, moderate-, and high-severity fire has occurred in all forest types, but in substantially different proportions and patch size distributions at different locations.
- Classification of historical fire regimes according to forest types can be coarse; thus, failure to recognize variation of historical fire regimes *within* forest types can lead to overgeneralization and oversimplification of landscape conditions.
- Presence of roads, road density and railways, livestock grazing, invasive species, and mining can alter fire regimes. Even a single one of these influences can have profound effects on vegetation and fire behavior conditions. When present in combinations, cumulative effects will arise that may push ecosystems past tipping points (Paine et al. 1998, Lindenmayer et al. 2011).
- Knowledge of historical range of variability (HRV) is useful but does not dictate land management goals. HRV findings from one area may or may not have relevance elsewhere.
- Recent trends in many western forest regions of more large fires and more area burned are linked to recent climatic trends of hotter droughts and longer, more severe fire seasons.
- Respondents who emphasized the longer time scales of charcoal records noted that most areas of predominantly low-severity fires showed some incidence of moderate- or high-severity fire over longer time frames.
- It is desirable to use multiple methods to reconstruct historical fire regimes. More can be learned using multiple approaches and considering data from diverse temporal and spatial scales.

- Importance of local context in management of fire-prone landscapes underscores the need to move away from oversimplified narratives that encourage application of fire research beyond its original scope of inference. Note: the scope of inference is of particular concern here as over reliance on fire scar sampling for landscape scale interpolation has inherent biases and uncertainties.

Areas of Disagreement (high uncertainty):

- Fire regime inferences from historical and modern tree inventory data, simulation models, and other approaches have levels of uncertainty.
- Whether large, high-severity fires have increased to a significant and measurable degree in all forest types *in comparison to historical fire regimes* (i.e., prior to modern fire suppression) remains debatable.
- Fuel treatments are urgently needed across nearly all forests remains debatable.
- Fuel treatments should be focused around communities and plantations; but hazard fuel reduction elsewhere remains debatable.
- There is high uncertainty about where and when fuel treatments are beneficial.
- Commonly used vegetation classification schemes as a suitable basis for generalizing about fire regimes remains debatable. Known geographic variation in fire regimes within forest types argues for improved forest and fire regime classifications.
- Tree-ring evidence sometimes supports conclusions that contrast with those derived from landscape-scale inventory and monitoring data using different sampling frames creates uncertainty.
- General applicability of "thinning and prescribed burning remedies" to offset human influences is debatable.
- Human impacts on forest successional conditions in moist and cold forests remains debatable.
- Extent to which landscape tipping points have been reached as a result of high-severity fires is debatable.
- Effectiveness of fuel treatments under projected climate futures and associated more extreme fire weather is uncertain.
- Interpretation of any research evidence and the scope of related inferences is limited by scaling (uncertainty) and sampling concerns associated with the methods, and these limitations apply to all research methods.
- All methods for reconstructing historical fire regimes are necessarily indirect and have degrees of uncertainty. They may include, but are not limited to, interpreting evidence of past fires or the extent of fire-dependent ecosystems from historical documents, land surveys, aerial photographic reconstructions, fire-scar and growth-release data from tree rings, tree age and death dates from tree-ring data, climatic data linked with past fires,

charcoal and pollen deposits, current characteristics of stands (i.e., structure, species, and stand age distribution), fire perimeter mapping, historical timber survey data, and use of statistical distributions for modeling stand-replacing fire.

ROAD IMPACTS AND ROADLESS AREA IMPORTANCE NEED TO BE ANALYZED TO COMPLY WITH CONNECTIVITY REQUIREMENTS OF THE PLANNING RULE

Roads – Given the extensive and cumulative impacts of roads on ecosystem processes, wildlife, water quality, and fire ignitions (see below), *a minimum road density analysis* needs to be conducted to assure the public that there are no excessive roads and that more roads can and should be decommissioned and obliterated rather than improving and building more roads. The DEIS needs to provide a transportation plan analysis to fully assess road-related fire ignitions associated with improved access and to come up with an alternative that reduces them.

Simply improving culverts and surfacing primitive dirt roads with poor drainage also may not be enough to improve water quality. Notably, the DEIS provides no information on Clean Water Act 303d water-quality limited streams and how project-related impacts will be minimized to comply with state and federal water quality standards[2]. Water quality must be assessed in relation to road improvements, greater road access, thinning impacts, and road-stream intersections.

In sum, the DEIS needs to fully disclose road-related impacts as follows:

- Roads and thinning contributions to soil erosion and sediment inputs affecting water-quality even when roads are improved.
- Probability of human-caused wildfire ignitions associated with improved road access (see Balch et al. 2017 for human-caused ignitions, pdf enclosed).
- Fragmentation and degradation of wildlife habitat at road densities > 1 mi/sq mi, particularly impacts to large carnivores and aquatics.
- Spread of invasive species and their effects on fire regimes.
- Likelihood of mass-wasting events on steep erosive slopes along the road prism.

2Particularly in relation to EPA standards see https://nepis.epa.gov/Exe/ZyNET.exe/00001O9W.TXT?ZyActionD=ZyDocument&Client=EPA&Index=1986+Thru+1990&Docs=&Query=&Time=&EndTime=&SearchMethod=1&TocRestrict=n&Toc=&TocEntry=&QField=&QFieldYear=&QFieldMonth=&QFieldDay=&IntQFieldOp=0&ExtQFieldOp=0&XmlQuery=&File=D%3A%5Czyfiles%5CIndex%20Data%5C86thru90%5CTxt%5C00000001%5C00001O9W.txt&User=ANONYMOUS&Password=anonymous&SortMethod=h%7C-&MaximumDocuments=1&FuzzyDegree=0&ImageQuality=r75g8/r75g8/x150y150g16/i425&Display=hpfr&DefSeekPage=x&SearchBack=ZyActionL&Back=ZyActionS&BackDesc=Results%20page&MaximumPages=1&ZyEntry=1&SeekPage=x&ZyPURL

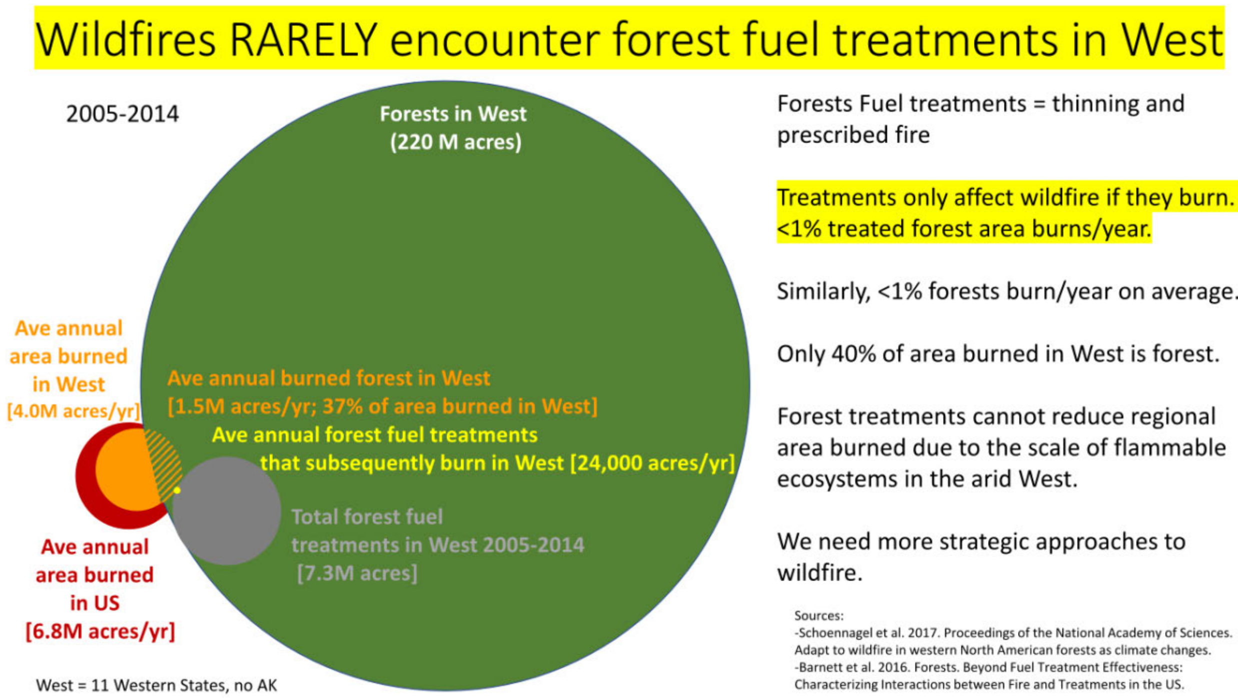
Ibisch et al. (2016) provide a global synthesis of road-related impacts including: wildlife mortality (vehicle collisions); poaching pressure; sediment increases (runoff); chemical contamination; carbon emissions; spread of invasive species; fire ignitions; and habitat fragmentation among others. These impacts can extend out to 1 km on either side of the road prism. Thus, road impacts need to be fully addressed and properly mitigated to assess planned extensive road upgrades and access.

Roadless Areas - The ecological importance of roadless areas is well-documented in the literature (Strittholt and DellaSala 2001, Loucks et al. 2003, Crist et al. 2005, Ibisch et al. 2016) and emphasized in landmark Forest Service policies such as the Roadless Conservation Rule[3] and Interior Columbia River Basin strategy[4]. At a minimum, the DEIS needs to disclose any treatments proposed in inventoried roadless areas and low density roaded areas (<1 mi/sq mi) and must avoid thinning in these areas because of their high conservation value, particularly as relatively unfragmented blocks of wildlife habitat. Roadless areas and low-density roaded areas are of considerable importance to ecosystem integrity (as defined by the 2012 planning rule) as they are often at the headwaters of watersheds essential in maintaining water quality and terrestrial and aquatic ecosystem integrity (DellaSala et al 2011). Roadless areas also tend to be of much lower priority for fuels reduction given their fire regimes are less altered by suppression and they lack the ignition problems associated with roaded areas (e.g., see Roadless Conservation Rule, Columbia River Basin strategy, DellaSala and Frost 2001).

LIMITATIONS OF THINNING ON FIRE BEHAVIOR IN A CHANGING CLIMATE NEED TO BE RECOGNIZED AND CORRECTED

The figure below illustrates uncertainties of relying on thinning to reduce fire intensity given that the period of when fuels are lowest is generally short lived and fires rarely encounter thinned sites when fuels are lowest (Schoennagel et al. 2017). The extremely low probability of fire and thinned site co-occurrence invalidates the DEIS assumptions about lowering fire intensity. Simply increasing the area thinned does not change these odds appreciably given one cannot accurately predict when and where a fire will occur and many areas are inaccessible (Schoennagel et al. 2017).

[3] https://www.fs.usda.gov/roadmain/roadless/2001roadlessrule
[4] https://www.fs.fed.us/r6/icbemp/html/ICBEMP_Frameworkmemorandum-and-strategy_2014.pdf



Moreover, the DEIS needs to disclose the difference between prescribed fire that is applied at the stand level (where impacts to soils can be dispersed and limited) vs. pile burning to consume slash that can cause localized soil damage (excessive soil heating) facilitating the spread of invasive plants and delaying forest succession (especially if livestock grazing also occurs, Besctha et. al 2012).

Excessive opening of the tree canopy can also lead to higher wind penetrance and rapid fire spread, particularly if thinning is conducted on steep slopes and in remote areas with limited access making fine fuel consumption via pile burning impractical. In a warming climate where more extreme fire weather is likely, thinning is even less likely to alter fire behavior (Abatzoglou and Williams 2017, Schoennagel et al. 2017).

CUMULATIVE IMPACTS OF LIVESTOCK GRAZING IN A CHANGING CLIMATE NEED TO BE FULLY ANALYZED AND GREATLY REDUCED

Livestock grazing and associated infrastructure in combination with climate change are causing extensive cumulative effects in the SFNF that are not properly analyzed or mitigated by the DEIS. The DEIS acknowledges that livestock have contributed to degradation of ecosystem resilience (DEIS Volume 1:5) but the alternatives contain numerous contradictions stating, for example, that the DEIS (Volume 1:13) "aims to provide *healthy* forested and non-forested lands that would supply forage for both livestock and wildlife" (Volume 1:13) and that it will "provide sustained multiple uses, products, and services in an *environmentally acceptable manner* (including timber, livestock forage, recreation opportunities, and leasable and locatable minerals) (emphasis added, DEIS Volume 1:16), all the while maintaining grazing at ecologically unacceptable levels (maximum of 11,400 AUMs).

The DEIS (Volume 1:37) recognizes that livestock grazing is "*not a natural process*" (emphasis added), yet, continues grazing under all planning alternatives even though it is inconsistent with ecological processes, ecological integrity, and ecological condition requirements of the planning rule (as noted in the boxes above). None of the alternatives meet these requirements given the high level of grazing maintained.

Importantly, the DEIS does not meet the BASI requirement of the planning rule by failing to analyze cumulative impacts of livestock from roads, infrastructure, and especially climate change. Besctha et al. (2012) noted livestock use affects a far greater proportion of BLM and Forest Service lands than do roads, timber harvest, and wildfires combined by altering vegetation, soils, hydrology and wildlife species composition and abundance "*in ways that exacerbate the effects of climate change on these resources*" (emphasis added). Livestock also contribute to greenhouse gas emissions globally (18% of the total anthropogenic emissions) and in the SFNF, thus, the DEIS needs to analyze livestock-related emissions.

Beschta et al. (2012) recommended large areas free of livestock use to "help initiate and speed the recovery of affected ecosystems as well as provide benchmarks or controls for assessing the effects of grazing versus no grazing at significant spatial scales in a changing climate."

The DEIS analyzed and dismissed Alternative 3 (lower livestock use) and dismissed a no grazing alternative as out of scope. However, Alternative 2 is deficient in meeting the ecological integrity, ecological condition, and ecological processes requirement of the planning rule. Therefore, the Forest Service needs to develop a new alternative or modify Alternative 2 to meet the specific recommendations of Beschta et al. (2012: Table 2) as follows.

Beschta et al. (2012:Table 2). Priority areas for permanently removing livestock and feral ungulates from Bureau of Land Management and US Forest Service lands to reduce or eliminate their detrimental ecological effects

- Watersheds and other large areas that contain a variety of ecotypes to ensure that major ecological and societal benefits of more resilient and healthy ecosystems on public lands will occur in the face of climate change
- Areas where ungulate effects extend beyond the immediate site (e.g., wetlands and riparian areas impact many wildlife species and ecosystem services with cascading implications beyond the area grazed)
- Localized areas that are easily damaged by ungulates, either inherently (e.g., biological crusts or erodible soils) or as the result of a temporary condition (e.g., recent fire or flood

disturbances, or degraded from previous management and thus fragile during a recovery period).

- Rare ecosystem types (e.g., perched wetlands) or locations with imperiled species or communities (e.g., aspen stands and understory plant communities, endemic species), including fish and wildlife species adversely affected by grazing and at-risk and/or listed under the ESA
- Non-use areas (i.e., ungrazed by livestock) or exclosures embedded within larger areas where livestock grazing continues.
- Such non-use areas should be located in representative ecotypes so that actual rates of recovery (in the absence of grazing impacts) can be assessed relative to resource trend and condition data in adjacent areas that continue to be grazed.
- Areas where the combined effects of livestock, wild ungulates, and feral ungulates are causing significant ecological impacts.

Notably, Ratner et al. (2018) document extensive impacts of livestock grazing on aspen groves in in Utah and their findings are generally applicable west-wide and therefore to the DEIS. These researchers found livestock significantly suppressed aspen sprout growth and trampled soils in study plots. They noted that livestock tended to concentrate in aspen groves due to forage availability and shading, even on allotments where livestock grazing is "controlled" and under "moderate" grazing. Ratner et al. (2018) recommended reducing livestock pressure via exclosures at least until aspen height exceeds browsing height and this will require periodic repetition (exclosures) to ensure proper aspen regeneration. At a minimum, exclosures should include entire aspen clonal areas and this needs to be incorporated into the DEIS.

Finally, the DEIS needs to allow for permanent allotment retirement and significantly reduced livestock grazing. This needs to include an analysis of the cumulative effects of *livestock grazing and climate change* and emissions related to livestock use, roads, and infrastructure. The DEIS (Volume 1:31) only allows for continuation of even vacant or understocked allotment and therefore should be modified or a new alternative developed to permanently retire vacant or understocked allotments and allow for voluntary buyout of grazing leases by conservation groups.

RIPARIAN AREAS NEED MORE EFFECTIVE PROTECTION, CONSERVATION, AND RESTORATION ESPECIALLY FROM LIVESTOCK AND THINNING TREATMENTS

The DEIS (Volume 1:153) correctly notes that although riparian areas occupy < 3% of the landscape, they support ~ 80% of the forest's plant and animal diversity, including several at-risk species (e.g., Mexican Spotted Owl, Lewis's Woodpecker, Arizona willow, Jemez Mountain salamander, masked and water shrew, New Mexican meadow jumping mouse, Northern leopard frog, Rio Grande chub, cutthroat trout, and sucker). Hubbard (1977; cited in Kauffman et al.

1984) report that 16-17% of the entire breeding avifauna of temperate North America reside in just 2 New Mexico river valleys and 77% of 166 nesting birds in the southwest depend on riparian habitat (Johnson et al. 1977 cited in Kauffman et al. 1984). Thus, riparian areas need stepped up conservation measures, especially protection from livestock grazing, given their exceptional importance in southwestern dry ecosystems.

Riparian areas also congregate livestock that have a strong preference for stream-side areas and wet montane meadows with high forage production. Livestock degrade this important wildlife habitat type in many ways, including soil compaction, spread of invasive species, stream-bank erosion, hydrological alterations, water quality and stream temperature degradation, and trampling effects (Kauffman et al. 1984, Besctha et al. 2012).

Kauffman et al. (1984) list several ways livestock grazing impacts can be reduced in riparian areas that should be readily adopted by the DEIS:

- Exclusion of livestock grazing;
- Alternative grazing schemes (e.g., late season – after bird nesting);
- Salting, alternative water sources, fencing, range riders to keep livestock out;
- Instream structures (e.g., trash catchers, gabions, small rock dams, individual boulder placement, rock jetties, and silt log drops) for increasing water table in areas of former wet meadows as well as improving fish habitat;
- Combining rest rotation with check dams (although the rest-rotation system may increase trailing and trampling damage, causing streambank erosion and instability);
- Selection of cattle with a preference for upland areas over riparian (cattle are known to have group-specific preferences)

Because of the disproportionate use of wildlife in riparian areas (especially at-risk species) and the extensive livestock damage in the area, the DEIS should incorporate the best elements from Alternative 3 with some notable additions as follows:

- Double the objectives in Alternative 2 (DEIS Volume 1:Table 3, p. 58) for restoring composition and structure in riparian vegetation.
- Within the riparian management zone, move toward desired conditions for vegetation types that are outside of or trending away from natural range of variability by restoring the composition and structure of 30 miles of stream every 10 years. Actions that could improve riparian areas would include removing invasive plant species, stabilising stream channels, planting native species, promoting natural revegetation of bare ground, and redirecting other uses (e.g., providing other watering sources or closing areas to camping – note this redirection needs to include redirecting cattle and not just "other uses").

- Complete aquatic restoration on priority projects on 60 miles of aquatic habitat (e.g., increasing pool quantity, providing stream cover, removing or installing fish barriers, restoring beaver populations, treating invasive aquatic species, etc.) every 10 years to benefit aquatic species.
- Every 10 years, restore native fish species to 40 miles of streams where nonnative fish are absent and where natural or human-made fish barriers exist.
- Further reductions in road densities throughout the forest and avoidance of permanent or temporary roads, particularly those that parallel or cross streams.
- Additionally, an emphasis on beaver reintroduction is complimentary with the above improvements.
- The DEIS should include large no-grazing riparian zones where cattle are fenced out and permanently removed to allow riparian recovery and reintroduction of beaver populations.

FOCAL SPECIES, AT-RISK SPECIES, SPECIES OF CONSERVATION CONCERN NEED TO BE MONITORED AND HABITAT PROTECTED FROM THINNING AND GRAZING

The DEIS (Volume 2:312) states that "the 2012 Rule does not require or prohibit monitoring of population trends of focal species. Instead, it allows the use of any *existing or emerging approaches for monitoring the status of focal species* that are supported by current science" (emphasis added). However, the DEIS is deficient in meeting the BASI requirement of the planning rule as it inadequately monitors population viability of species and does not provide enough habitat protection measures for focal species, species of conservation concern, and at-risk species. Specifically, the DEIS needs to meet the BASI requirement for these species with respect to connectivity (Haber and Nelson 2015), PVA (Noon et al. 2003), and species-specific "trigger points" (Schulz et al. 2012).

The DEIS largely bases management of these species on coarse-filter approaches. The DEIS site specific measures are very general and insufficient as a fine-filter approach.

Importantly, The Committee of Scientists (COS 1999[5]) stated, ''Habitat alone cannot be used to predict wildlife populations'' and ''diversity is sustained only when individual species persist; the goals of ensuring viability and providing for diversity are inseparable. For this reason, the fine-filter species assessment is critical." To meet the BASI requirements, therefore, the Forest Service must appropriately provide fine-filter approaches following recommendations of the COS (1999), Noon et al. (2003) and Schultz et al. (2012) as follows.

[5]COS (1999) https://www.fs.fed.us/emc/nfma/includes/cosreport/Committee%20of%20Scientists%20Report.htm

Noon et al. (2003) note: “to assess the viability of species, at least three assumptions must hold true: (1) attributes that define the coarse filter (i.e., dominant vegetation types) are sufficient and reliable surrogates for habitat and can effectively predict the occurrence of a given species; (2) managing coarse-filter attributes will address the factor(s) currently limiting abundance, density, and persistence of each species; and (3) the spatial resolution of the coarse filter matches the scale at which given species respond to environmental heterogeneity. Although these assumptions may be valid for some species in many circumstances, especially species that are small-bodied, abundant, and tightly linked to a particular vegetation community, the likelihood that the assumptions are met for all, or even most, species in an assemblage is low. For that reason, landscape planning employs “fine-filter” assessments, which are based on direct measures of the status and trends of individual species or on models of population viability to evaluate the needs of species at risk of decline. Noon et al. (2003) report numerous prediction errors associated with coarse-filter approaches that need supplementation with species-specific analyzes. For instance, forest planning needs to include PVA methods in its monitoring and adaptive management approach to better ensure coarse-filter requirements are representative of the community of interest.”

Similarly, Schultz et al. (2012) indicate monitoring plans must include species-specific trigger points that initiate a review of management actions and provisions to ensure species-specific (fine filter) monitoring will be well funded and implemented. The *trigger points must be enforceable and ensure* that specific project actions cease should they further impair the viability of select species (especially the case for at-risk and listed species).

Schultz et al. (2012) note the 2012 planning rule requires “at least, some amount of direct species measurement may be needed to assess the effectiveness of the ecological conditions provided under the coarse-filter approach in achieving the goal of conserving the biological diversity of the area (USFS 2012:124).”

Schultz et al. (2012) provide recommendations for incorporating more specific fine-filter monitoring lacking in the DEIS, as summarised:

- Focusing on distribution, rather than traditional measures of population size and growth rate, which greatly increases the efficiency of broad-scale monitoring programs.
- Advancements in wildlife monitoring, based on detection/non-detection data, including the use of sign surveys, genetic evaluation, and historical presence–absence survey data decrease the cost of monitoring changes in distribution, which can be inferred from the proportion of sample units at which the species is detected.
- Area occupied by a species can be used as an effective measure of a species’ spatial distribution.

- Temporal and spatial patterns in detection/non-detection monitoring data allow inference to changes in animal abundance, the single most influential parameter that provides insights into likelihood of species persistence.

The methods above recommended by Noon et al. (2003) and Schultz et al. (2012) along with connectivity measures recommended by Haber and Nelson (2015) should be applied to all 36 at-risk species, all 32 species of conservation concern, and all 7 focal species in the project area.

Mexican Spotted Owl (MSO) - The Santa Fe National Forest contains 198,888 acres of designated critical habitat for this owl. MSO requires dense conifer forests for nesting; however, will forage in recently severely burned areas (Lee 2018). The main factor involved in the decline of this species has been habitat destruction from logging; severe fire is not necessarily a habitat loss (Lee 2018), yet the DEIS assumes this to be the case. Large and small patches of severe burns juxtaposed with fire refugia for nesting may provide optimal habitat for MSO (Lee 2018). And while much is not known about how thinning affect MSO and its prey, declines in habitat and prey species have been noted for Northern Spotted Owl (see Odion et al. 2014b for review and analysis) and California Spotted Owl (Stephens et al. 2014). For all three subspecies of owls, removal of large trees (before/after fire) and reducing canopy cover (e.g., below 60% for NSO) constitutes habitat degradation that has been linked to nest occupancy failures (Lee 2018).

Thus, at a minimum, thinning units need to be dropped from MSO critical habitat and demographic monitoring implemented for this at-risk species.

FIRE EMISSIONS ARE OVER-ESTIMATED USING LANDFIRE AND EMISSIONS FROM PROJECT ACTIVITIES NEED TO BE ANALYSED FOR DIRECT, INDIRECT, AND CUMULATIVE EFFECTS

The DEIS pays an inordinate amount of attention to emissions from wildfires yet includes no analysis of emissions from livestock grazing, livestock infrastructure and transport, thinning and road development and maintenance. Therefore, the DEIS is deficient in assessing cumulative impacts of emissions and air quality to the surrounding community from project activities.

With respect to fire emissions, the DEIS needs to pay attention to the literature on wildfire emissions from related studies in dry pine and mixed conifer forests as follows.

For instance, Mitchell (2015: chapter 10 in DellaSala and Hanson 2015) has an excellent summary of ineffectiveness of thinning and reduction of carbon stores from thinning.

"While such treatments [referring to thinning and prescribed burning] can sometimes be effective in reducing fire severity, if and when fires occur in thinned areas (Rhodes and Baker, 2008), they can come at the expense of carbon storage. The majority of carbon stored in leaves, leaf litter, and duff is typically consumed by high-severity wildfire and often constitutes the majority of the carbon emissions during the a given fire, yet most of the carbon stored in forest biomass (stem wood, branches, and coarse, woody debris) remains unconsumed even by high-severity wildfires. Consequently, fuel removal via forest thinning almost always reduces carbon storage more than the additional carbon that a stand is able to store when made more resistant to wildfire. For this reason, removing large amounts of biomass to reduce the fraction by which other biomass components are consumed via combustion is inefficient (Mitchell et al., 2009). Fuel reduction treatments that involve the removal of overstory biomass (i.e., intermediate-sized and large trees) are, perhaps unsurprisingly, the most inefficient methods of reducing wildfire-related carbon losses because they remove large amounts of carbon for only a marginal reduction in expected fire severity (Figure 10.2)."

"The majority of carbon stored in montane forest ecosystems of western North America remains unconsumed, even in high-severity wildfires. Large carbon stores in the bole biomass of large forest trees are not consumed, and the substantial proportion of carbon stored in forest soils is only slightly consumed. Most of the carbon emissions in a wildfire are from combustion of litter, duff, and woody debris. In the 2002 Biscuit Fire, CFs for total forest biomass (i.e., trees, snags, shrubs, woody fuels, litter, duff, and soil), weighted according to their respective prefire biomass, were 0.13, 0.15, and 0.21 for low-, medium-, and high-severity fires, respectively. Such factors can be even lower among stands with a higher proportion of carbon storage in bole biomass that likewise remains unconsumed in high-severity wildfires, such as Sitka spruce (P. sitchensis)/Western Hemlock (T. heterophylla) forests in the coast range of the Pacific Northwest (Smithwick et al., 2002;Mitchell et al., 2009). The application of fuel treatments can be effective in reducing fire severity and carbon emissions, but such treatments come at the cost of a net reduction in carbon storage relative to fire alone (Mitchell et al., 2009)."

In a recent global study of pyrogenic carbon emissions, Jones et al. (2019) concluded that "large wildfires convert a significant fraction of the burned vegetation biomass into pyrogenic carbon that can be stored on site for centuries to millennia and this stored carbon is underestimated in emissions calculations. The amount of carbon emitted globally from wildfires is in fact buffered by pyrogenic carbon production resulting in burned landscapes becoming a significant carbon sink." The value of this sink is not even reported in the DEIS nor is it estimated in LANDFIRE and it needs to be in the forest plan. Here is the Jones et al. (2019) abstract, the pdf is attached.

Abstract

Landscape fires burn 3–5 million km2 of the Earth's surface annually. They emit 2.2 Pg of carbon per year to the atmosphere, but also convert a significant fraction of the burned vegetation biomass into pyrogenic carbon. Pyrogenic carbon can be stored in terrestrial and marine pools for centuries to millennia and therefore its production can be considered a mechanism for long-term carbon sequestration. Pyrogenic carbon stocks and dynamics are not considered in global carbon cycle models, which leads to systematic errors in carbon accounting. Here we present a comprehensive dataset of pyrogenic carbon production factors from field and experimental fires and merge this with the Global Fire Emissions Database to quantify the global pyrogenic carbon production flux. We found that 256 (uncertainty range: 196–340) Tg of biomass carbon was converted annually into pyrogenic carbon between 1997 and 2016. Our central estimate equates to 12% of the annual carbon emitted globally by landscape fires, which indicates that their emissions are buffered by pyrogenic carbon production. We further estimate that cumulative pyrogenic carbon production is 60 Pg since 1750, or 33–40% of the global biomass carbon lost through land use change in this period. Our results demonstrate that pyrogenic carbon production by landscape fires could be a significant, but overlooked, sink for atmospheric CO_2.

We repeat from above our concerns about problems with LANDFIRE fire emissions as follows.

In Northern Idaho, Hyde et al. (2015) evaluated two LANDFIRE fuel loading raster options: (1) Fuels Characteristic Classification Systems (LANDFIRE-FCCS); and (2) Fuel Loading Model (LANDFIRE-FLM) vs. measured fuel loadings for a 20,000 ha mixed conifer study area. They found that the LANDFIRE-FCCS layer showed 200% higher duff loadings relative to measured loadings that led to 23% higher total mean consumption and emissions when modeled in FOFEM. The LANDFIRE-FLM layer showed lower loadings for total surface fuels relative to measured data, especially in the case of coarse woody debris that led to 51% lower mean total consumption and emissions when modeled in FOFEM. Additionally, LANDFIRE-FLM consumption was *59% lower relative to that on the measured plots, with 58% lower modeled emissions*. The authors concluded that these differences in fuel loadings led to significant differences in consumption and emissions depending upon the data and model chosen. The DEIS therefore needs to disclose how errors in fuel loading consumption were addressed in emissions determinations regarding wildfires and how these errors were corrected.

CONCLUSIONS AND NEED FOR GREATLY IMPROVED PREFERRED ALTERNATIVE

Based on the above analysis, deficiencies in the DEIS, and need for an improved or new alternative to better meet the BASI and planning rule requirements, I am requesting that the SFNF revise the DEIS to include the following actions.

- **Prioritize community wildfire safety and fire-risk reduction, including home-hardening, defensible space, additional road closures/decommissioning to reduce ignitions, and identification/maintenance of community evacuation routes.** The most prudent means of community fire protection is to *work from the home-out* rather than the *wildlands-in* (emphasis added) according to retired Forest Service researcher Jack Cohen

(2000; also see Youtube interviews[6]) and related home fire-risk reduction work (Syphard et al. 2013, 2014). Community and fire-fighter safety actions should be directed at home protection and anthropogenic fire-ignitions along high-use roads (especially ingress/egress; see Balch et al. 2017). As noted above, research demonstrates that there is a very low (<1%) probability of thinned areas encountering a fire when fuels are lowest (Schoennagel et al. 2017). Therefore, it is imperative that the Forest Service strategically direct limited resources at protecting homes rather than extensive thinning in the backcountry that does nothing for home protection.

- **Reduce human-caused wildfire ignitions (see Balch et al. 2017) associated with road access**. The Forest Service needs to conduct project-specific transportation plans to determine the probability of human-caused fire ignitions in relation to road densities, road improvements, and increased human access along improved roads. These plans should address a broad scope of road-related impacts and choose an alternative based on minimal road access.
- **Protect high value conservation areas from logging/thinning/road improvements.** The DEIS needs to fully disclose impacts of road improvements and thinning on low-density (<1 mi/sq mile) and inventoried roadless areas (see below) and make clear how late-successional (closed canopy) forests within the project area will be maintained and restored to levels comparable to historic or documented reference conditions.
- **Disclose limitations and uncertainties of fire-scar sampling, importance of fire-free periods to shrub and tree recruitment and include more robust fire occurrence/ severity estimators that account for variability in fire-free and frequent-fire intervals.** The DEIS primarily relies on fire-scar sampling to determine the dominant fire regime present yet does not disclose uncertainties and limitations in sampling approaches (i.e., confidence levels). Notably, paleo-ecology studies conducted over longer timelines (millennia) than fire scar sampling show high variability in fire regimes related primarily to regional and local microclimatic factors (slope, aspect, elevation) over time (Meyer 2010). Large fires historically included high severity patches during alternating cycles of wet followed by droughts (Margolis et al. 2011). This is particularly important as extreme fire-weather (top-down driver) is known to over-ride bottom up influences (fuels) on fire behavior in the Rockies (Bessie and Johnson 1995, Schoennagel et al. 2004) and elsewhere (Abatzoglou and Williams 2017). The effect of global heating and increased likelihood of regional droughts may (Margolis et al. 2011) or may not (Parks et al. 2016, Margolis et al. 2017) increase fire severity. This uncertainty is most significant and must be analyzed to determine the need for and limitations of extensive fuels treatments based predominately on assumptions regarding frequent-fire regimes that may become increasingly unlikely in a rapidly changing climate. Additionally, variability in fire return

[6] National Fire Protection Association presentations by Jack Cohen - https://www.youtube.com/watch?v=vL_syp1ZScM; https://www.youtube.com/watch?v=RqKFDDBGd5o

(point/plot scale) and fire rotation (landscape scale) intervals accounts for longer fire-free periods that allow for shrub and small tree recruitment, including both dense and open forest conditions (see below). Thus, the DEIS needs to fully disclose its characterization of a low-severity fire regime, and "open" forest conditions (reference sites) with respect to heterogeneity and in relation to tree canopy mortality, shrub and small tree densities. Notably, even low severity systems have occasional fire-flare ups that kill dominant overstory trees and allow for sufficient shrub and small tree recruitment (see Baker 2017).

- **Substantially reduce livestock grazing in riparian areas and high value conservation areas.** Stepped up conservation and restoration need to be in the forest plan, including large no-grazing zones (exclosures), additional riparian and wet meadow/spring protections, road obliteration, invasive species removals, and beaver reintroductions.
- **More fully disclose and avoid impacts to at-risk species like the Mexican Spotted Owl (MSO).** There is no discussion of importance of mixed-severity wildfires in maintaining foraging habitat for spotted owls (Lee 2018, pdf enclosed). Instead, the DEIS incorrectly assumes, without site-specific data on owl occupancy or region-wide population trends, that wildfire (mostly high severity) degrades MSO habitat. However, Lee (2018) conducted a meta-analysis of fire effects on all three owl subspecies concluding that mixed-severity fire, including patches of large severity, was not the main cause of owl nest abandonment; pre- and post-fire logging was the predominant factor. Also, full disclosure of incidental take under the Endangered Species Act is required and the Forest Service needs to conduct population monitoring to assess MSO demographics and region-wide population trends.
- **Analyze and maintain connectivity especially for at risk, focal, and species of conservation concern.** The forest plan needs to properly analyze connectivity as noted herein including PVA, trigger points, and species/landscape specific measures that properly integrate coarse and fine-filter approaches under the BASI and connectivity requirements of the 2012 forest planning rule and the noted literature cited herein.
- **Reduce emissions from logging and roads.** A stated intent of the DEIS is to provide for resilience to climate change yet there is no requirement of an analysis of project-related emissions from tree clearing and road improvements. Notably, emissions from wildfires are typically much lower than landscape-level logging projects aimed at reducing wildfires (e.g., see Mitchell et al. 2009, Campbell et al. 2016, Law et al. 2018 as examples of appropriate methodologies). Project-specific alternatives must be developed to minimize emissions with alternatives selected that produce the lowest emissions. Alternatives should be compared in CO_2 equivalents, including the social cost of carbon[7].

[7]See https://19january2017snapshot.epa.gov/climatechange/social-cost-carbon_.html

- **Provide a cost-benefits analysis of managing wildfires for ecosystem benefits by working with fire under safe conditions.** The DEIS must disclose project-related costs of thinning, prescribed fire, and road improvements in comparison to managing fire for ecosystem benefits as a viable alternative (e.g., refer to the Cohesive Wildland Fire Management Strategy for wildfire ecosystem benefits[8] and 2012 forest planning rule regarding ecosystem integrity, vegetation diversity, and wildfire maintenance). Thus, it must be disclosed under what conditions will wildfires be managed for ecosystem benefits vs. suppressed so that when fires do eventually occur appropriate actions are taken based on pre-fire response planning and the Forest Service is accountable for implementing those actions accordingly.
- **Thinning to create open canopy forests at the expense of closed canopy forests needs to be greatly reduced and more strategically (surgically) applied.** The over-reliance on thinning stems from accuracy problems noted in the LANDFIRE program, biased fire scar fire estimates, inappropriate extrapolations from the Forest Service research publication GTR-310, and a failure to recognize site-specific and landscape heterogeneity. Thus, thinning treatments need to be greatly scaled back and strategic in application (mostly nearest homes).
- **In limited cases where thinning occurs, forest canopies need to be more fully maintained for closed canopy species associates by**: (1) stops and gaps (explain for the general reader) in thinning to for increased site heterogeneity; (2) retention of much more basal area (as compared to site-specific reference sites) especially around tree cohorts to make them wind firm; (3) retention of old/mature trees on site (based on increment core analysis and not just diameter at breast height); (4) in cases where tree thinning is necessary within the drip line of large mature trees, girdle those trees and leave standing on site as biological legacies; (5) retain more shrubs, forbs, and native grasses by reducing the interval between successive prescribed fires to allow for understory recruitment; and (6) fell and tip large trees in stream-side areas to create in-stream structures rather than thin and remove those trees from the site.
- **"Surgically" applied thinning treatments should be limited to the most drastically altered forests,** most notably, pine plantations in the Jemez and spruce/fir clearcuts on the eastern side of the SFNF.
- **Restoration and conservation measures should be greatly increased to address the following needs not sufficiently met in the DEIS**: (1) beaver reintroduction in riparian areas; (2) large livestock exclosures especially in riparian areas, wet meadows, and aspen groves; (3) road closures and road obliterations to provide connectivity; (4) defensible space within a narrow buffer (~60 feet) around homes; (5) ingress/egress routes for community protection; (6) increases in invasive species removal and containment; and

[8] See https://www.forestsandrangelands.gov/strategy/

(7) identification and protection of site and landscape specific habitat for focal species, species of conservation concern, and at risk species.

- **Compartmentalize the SFNF into fire management units** to determine when to suppress fire for community safety vs. working with fire for ecosystem benefits.[9]
- **Conduct a minimum road access analysis** and decommission/obliterate more roads to reduce impacts to water quality, wildlife habitat and human-caused fire ignitions.

In closing, while I respect the ability of the Forest Service to apply BASI to forest planning decisions on the Santa Fe National Forest, I remain greatly concerned that the noted inadequacies in the DEIS have not met the BASI standard. Instead the preferred alternative will (1) fragment and degrade important wildlife habitat; (2) jeopardize at-risk species (MSO), focal species, and species of conservation concern; (3) degrade water quality (mainly from roads, livestock, tree thinning), impact mature forests and riparian areas (along with wildlife and cultural values); and (4) uses methodologies (e.g., LANDFIRE, fire scar sampling, GTR-310) inappropriate to the SFNF. There is a heavy reliance on fire-scar sampling without disclosure of biases and uncertainties and thinning in stands that may possess old growth characteristics by moving them increasingly into open canopy conditions that lack overstory and understory structures. The efficacy of Alternative 2 mechanical treatments is highly uncertain because of the likelihood that the region's fire regimes will increasingly shift to larger burns due primarily to climate change (Abatzoglou and Williams 2017) and the extremely low odds that thinned sites will encounter a fire when fuels are lowest (Schoennagel et al. 2017).

Additionally, and contrary to what is often claimed by the Forest Service, insect and disease outbreaks are not associated with increased fire intensity. Insect-fire studies, including analysis of outbreaks and fire intensity in the Rockies and elsewhere (Romme et al. 2006, Kauffman et al. 2008, Bond et al. 2009, Black et al. 2011, Six et al. 2014, Hart et al. 2015, Meigs et al. 2016, Talucci and Krawchuck 2019) have repeatedly shown that there is no coupling of increased fire intensity with insect outbreaks. Instead, outbreaks may actually lower fire intensity once the needles of dead trees fall to the ground (within 1-3 years) as canopy fuels and therefore crown fires become highly unlikely. Dead trees also do not contribute to fire spread as they do not fall all at once nor result in accumulation of fine fuels (fine fuel accumulation is associated with logging). Dead trees are keystone legacies that provide essential habitat for cavity nesting birds, denning mammals, and numerous other wildlife. Their role in forest ecosystems needs to be better disclosed and maintained.

While wildfire clearly can be devastating to human communities, it is not an ecological catastrophe as often claimed. The Forest Service needs to develop better supported consensus

[9]see https://www.fs.fed.us/rmrs/publications/framework-developing-safe-and-effective-large-fire-response-new-fire-management; https://www.fs.fed.us/rmrs/publications/spatial-optimization-operationally-relevant-large-fire-confine-and-point-protection

alternatives that focus first and foremost on community protection where there is strong scientific agreement (see Moritz et al. 2014, Schoennagel et al. 2017, Moritz et al. 2018).

Impact of anthropogenic climate change on wildfire across western US forests

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Increased forest fire activity across the western continental United States (US) in recent decades has likely been enabled by a number of factors, including the legacy of fire suppression and human settlement, natural climate variability, and human-caused climate change. We use modeled climate projections to estimate the contribution of anthropogenic climate change to observed increases in eight fuel aridity metrics and forest fire area across the western United States. Anthropogenic increases in temperature and vapor pressure deficit significantly enhanced fuel aridity across western US forests over the past several decades and, during 2000–2015, contributed to 75% more forested area experiencing high (>1 σ) fire-season fuel aridity and an average of nine additional days per year of high fire potential. Anthropogenic climate change accounted for ∼55% of observed increases in fuel aridity from 1979 to 2015 across western US forests, highlighting both anthropogenic climate change and natural climate variability as important contributors to increased wildfire potential in recent decades. We estimate that human-caused climate change contributed to an additional 4.2 million ha of forest fire area during 1984–2015, nearly doubling the forest fire area expected in its absence. Natural climate variability will continue to alternate between modulating and compounding anthropogenic increases in fuel aridity, but anthropogenic climate change has emerged as a driver of increased forest fire activity and should continue to do so while fuels are not limiting.

wildfire | climate change | attribution | forests

Widespread increases in fire activity, including area burned (1, 2), number of large fires (3), and fire-season length (4, 5), have been documented across the western United States (US) and in other temperate and high-latitude ecosystems over the past half century (6, 7). Increased fire activity across western US forests has coincided with climatic conditions more conducive to wildfire (2–4, 8). The strong interannual correlation between forest fire activity and fire-season fuel aridity, as well as observed increases in vapor pressure deficit (VPD) (9), fire danger indices (10), and climatic water deficit (CWD) (11) over the past several decades, present a compelling argument that climate change has contributed to the recent increases in fire activity. Previous studies have implicated anthropogenic climate change (ACC) as a contributor to observed and projected increases in fire activity globally and in the western United States (12–19), yet no studies have quantified the degree to which ACC has contributed to observed increases in fire activity in western US forests.

Changes in fire activity due to climate, and ACC therein, are modulated by the co-occurrence of changes in land management and human activity that influence fuels, ignition, and suppression. The legacy of twentieth century fire suppression across western continental US forests contributed to increased fuel loads and fire potential in many locations (20, 21), potentially increasing the sensitivity of area burned to climate variability and change in recent decades (22). Climate influences wildfire potential primarily by modulating fuel abundance in fuel-limited environments, and by modulating fuel aridity in flammability-limited environments (1, 23, 24). We constrain our attention to climate processes that promote fuel aridity that encompass fire behavior characteristics of landscape ignitability, flammability, and fire spread via fuel desiccation in primarily flammability-limited western US forests by considering eight fuel aridity metrics that have well-established direct interannual relationships with burned area in this region (1, 8, 24, 25). Four metrics were calculated from monthly data for 1948–2015: (*i*) reference potential evapotranspiration (ETo), (*ii*) VPD, (*iii*) CWD, and (*iv*) Palmer drought severity index (PDSI). The other four metrics are daily fire danger indices calculated for 1979–2015: (*v*) fire weather index (FWI) from the Canadian forest fire danger rating system, (*vi*) energy release component (ERC) from the US national fire danger rating system, (*vii*) McArthur forest fire danger index (FFDI), and (*viii*) Keetch–Byram drought index (KBDI). These metrics are further described in the *Materials and Methods* and *Supporting Information*. Fuel aridity has been a dominant driver of regional and subregional interannual variability in forest fire area across the western US in recent decades (2, 8, 22, 25). This study capitalizes on these relationships and specifically seeks to determine the portions of the observed increase in fuel aridity and area burned across western US forests attributable to anthropogenic climate change.

The interannual variability of all eight fuel aridity metrics averaged over the forested lands of the western US correlated significantly ($R^2 = 0.57$–0.76, $P < 0.0001$; Table S1) with the logarithm of annual western US forest area burned for 1984–2015, derived from the Monitoring Trends in Burn Severity product for 1984–2014 and the Moderate Resolution Imaging Spectroradiometer (MODIS) for 2015 (*Supporting Information*). The record of standardized fuel aridity averaged across the eight metrics (hereafter, all-metric mean) accounts for 76% of the variance in the burned-area record, with significant increases in both records for 1984–2015 (Fig. 1). Correlation between fuel aridity and forest fire area remains highly significant ($R^2 = 0.72$, all-metric mean) after removing the linear-least squares trends for each time series for 1984–2015, supporting the mechanistic relationship between fuel aridity and

Significance

Increased forest fire activity across the western United States in recent decades has contributed to widespread forest mortality, carbon emissions, periods of degraded air quality, and substantial fire suppression expenditures. Although numerous factors aided the recent rise in fire activity, observed warming and drying have significantly increased fire-season fuel aridity, fostering a more favorable fire environment across forested systems. We demonstrate that human-caused climate change caused over half of the documented increases in fuel aridity since the 1970s and doubled the cumulative forest fire area since 1984. This analysis suggests that anthropogenic climate change will continue to chronically enhance the potential for western US forest fire activity while fuels are not limiting.

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The authors declare no conflict of interest.

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See Commentary on page 11649.

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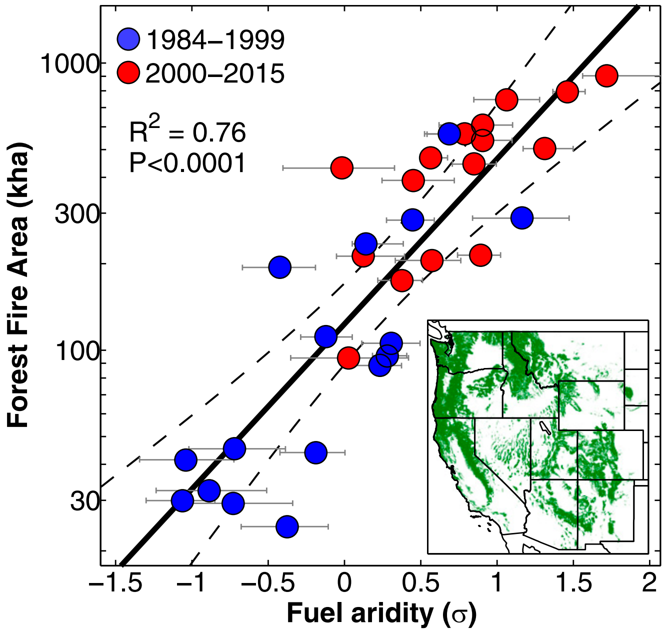

Fig. 1. Annual western continental US forest fire area versus fuel aridity: 1984–2015. Regression of burned area on the mean of eight fuel aridity metrics. Gray bars bound interquartile values among the metrics. Dashed lines bounding the regression line represent 95% confidence bounds, expanded to account for lag-1 temporal autocorrelation and to bound the confidence range for the lowest correlating aridity metric. The two 16-y periods are distinguished to highlight their 3.3-fold difference in total forest fire area. *Inset* shows the distribution of forested land across the western US in green.

forest fire area. It follows that co-occurring increases in fuel aridity and forest fire area over multiple decades would also be mechanistically related.

We quantify the influence of ACC using the Coupled Model Intercomparison Project, Phase 5 (CMIP5) multimodel mean changes in temperature and vapor pressure following Williams et al. (26) (Fig. S1; *Methods*). This approach defines the ACC signal for any given location as the multimodel mean (27 CMIP5 models) 50-y low-pass-filtered record of monthly temperature and vapor pressure anomalies relative to a 1901 baseline. Other anthropogenic effects on variables such as precipitation, wind, or solar radiation may have also contributed to changes in fuel aridity but anthropogenic contributions to these variables during our study period are less certain (22). We evaluate differences between fuel aridity metrics computed with the observational record and those computed with observations that exclude the ACC signal to determine the contribution of ACC to fuel aridity. To exclude the ACC signal, we subtract the ACC signal from daily and monthly temperature and vapor pressure, leaving all other variables unchanged and preserving the temporal variability of observations. The contribution of ACC to changes in fuel aridity is shown for the entire western United States; however, we constrain the focus of our attribution and analysis to forested environments of the western US (Fig. 1, *Inset*; *Methods*).

Anthropogenic increases in temperature and VPD contributed to a standardized (σ) increase in all-metric mean fuel aridity averaged for forested regions of +0.6 σ (range of +0.3 σ to +1.1 σ across all eight metrics) for 2000–2015 (Fig. 2). We found similar results with reanalysis products (all-metric mean fuel aridity increase of +0.6 σ for two reanalysis datasets considered; *Methods*), suggesting robustness of the results to structural uncertainty in observational products (Figs. S2–S4 and Table S2). The largest anthropogenic increases in standardized fuel aridity were present across the intermountain western United States, due in part to larger modeled warming rates relative to more maritime areas (27). Among aridity metrics, the largest increases tied to the ACC signal were for VPD and ETo because the interannual variability of these variables is primarily driven by temperature for much of the study area (28). By contrast, PDSI and ERC showed more subdued ACC driven increases in fuel aridity because these metrics are more heavily influenced by precipitation variability.

Fuel aridity averaged across western US forested areas showed a significant increase over the past three decades, with a linear trend of +1.2 σ (95% confidence: 0.42–2.0 σ) in the all-metric mean for 1979–2015 (Fig. 3*A*, *Top* and Table S1). The all-metric mean ACC contribution since 1901 was +0.10 σ by 1979 and +0.71 σ by 2015. The annual area of forested lands with high fuel aridity (>1 σ) increased significantly during 1948–2015, most notably since 1979 (Fig. 3*A*, *Bottom*). The observed mean annual areal extent of forested land with high aridity during 2000–2015 was 75% larger for the all-metric mean (+27% to +143% range across metrics) than was the case where the ACC signal was excluded.

Significant positive trends in fuel aridity for 1979–2015 across forested lands were observed for all metrics (Fig. 3*B* and Table S1). Positive trends in fuel aridity remain after excluding the ACC signal, but the remaining trend was only significant for ERC. Anthropogenic forcing accounted for 55% of the observed positive trend in the all-metric mean fuel aridity during 1979–2015, including at least two-thirds of the observed increase in ETo, VPD, and FWI, and less than a third of the observed increase in ERC and PDSI. No significant trends were observed for monthly fuel aridity metrics from 1948–1978.

The duration of the fire-weather season increased significantly across western US forests (+41%, 26 d for the all-metric mean) during 1979–2015, similar to prior results (10) (Fig. 4*A* and Table S2). Our analysis shows that ACC accounts for ~54% of the increase in fire-weather season length in the all-metric mean (15–79% for individual metrics). An increase of 17.0 d per year of high fire potential was observed for 1979–2015 in the all-metric mean (11.7–28.4 d increase for individual metrics), over twice the rate of increase calculated from metrics that excluded the ACC signal (Fig. 4*B* and Table S2). This translates to an average of an additional 9 d (7.8–12.0 d) per year of high fire potential during 2000–2015 due to ACC.

Given the strong relationship between fuel aridity and annual western US forest fire area, and the detectable impact of ACC on fuel aridity, we use the regression relationship in Fig. 1 to model

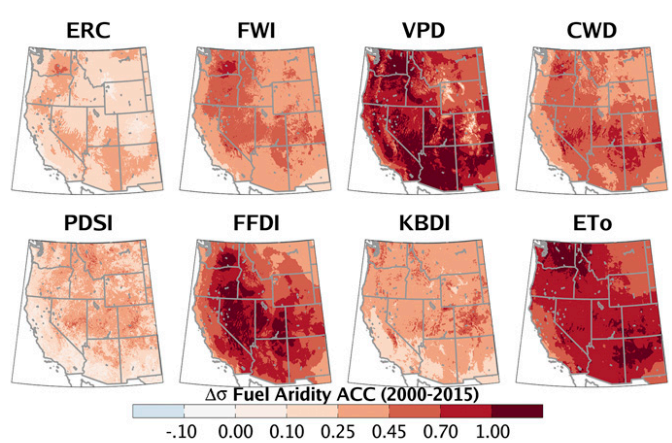

Fig. 2. Standardized change in each of the eight fuel aridity metrics due to ACC. The influence of ACC on fuel aridity during 2000–2015 is shown by the difference between standardized fuel aridity metrics calculated from observations and those calculated from observations excluding the ACC signal. The sign of PDSI is reversed for consistency with other aridity measures.

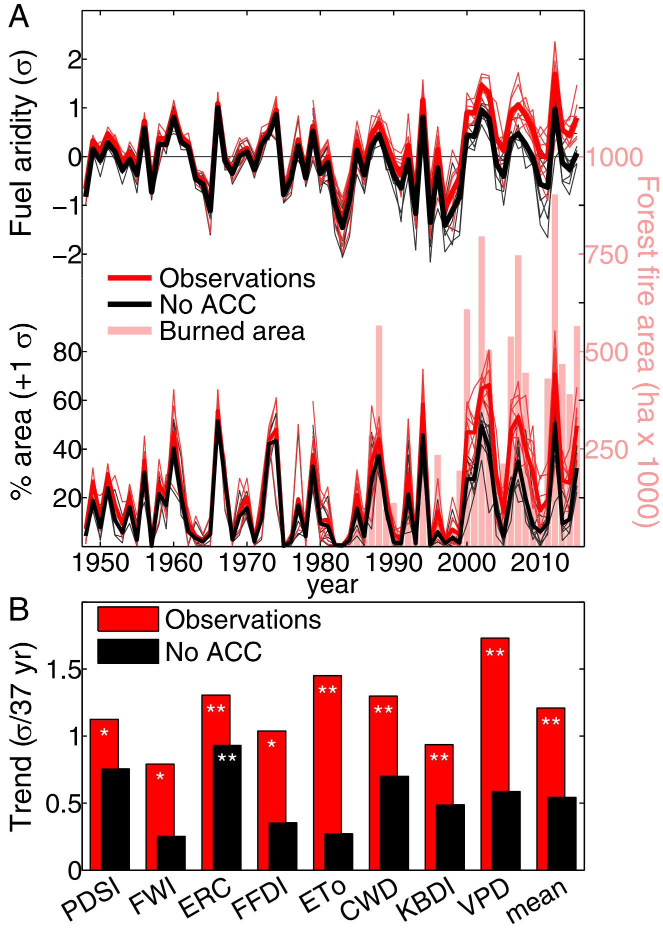

Fig. 3. Evolution and trends in western US forest fuel aridity metrics over the past several decades. (*A*) Time series of (*Upper*) standardized annual fuel aridity metrics and (*Lower*) percent of forest area with standardized fuel aridity exceeding one SD. Red lines show observations and black lines show records after exclusion of the ACC signal. Only the four monthly metrics extend back to 1948. Daily fire danger indices begin in 1979. Bold lines indicate averages across fuel aridity metrics. Bars in the background of *A* show annual forested area burned during 1984–2015 for visual comparison with fuel aridity. (*B*) Linear trends in the standardized fuel aridity metrics during 1979–2015 for (red) observations and (black) records excluding the ACC signal (differences attributed to ACC). Asterisks indicate positive trends at the (*) 95% and (**) 99% significance levels.

the contribution of ACC on western US forest fire area for the past three decades (Fig. 5 and Fig. S5). ACC-driven increases in fuel aridity are estimated to have added ~4.2 million ha (95% confidence: 2.7–6.5 million ha) of western US forest fire area during 1984–2015, similar to the combined areas of Massachusetts and Connecticut, accounting for nearly half of the total modeled burned area derived from the all-metric mean fuel aridity. Repeating this calculation for individual fuel aridity metrics yields ACC contributions of 1.9–4.9 million ha, but most individual fuel aridity metrics had weaker correlations with burned area and thus may be less appropriate proxies for attributing burned area. The effect of the ACC forcing on fuel aridity increased during this period, contributing ~5.0 (95% confidence: 4.2–5.9) times more burned area in 2000–2015 than in 1984–1999 (Fig. 5*B*). During 2000–2015, the ACC-forced burned area likely exceeded the burned area expected in the absence of ACC (Fig. 5*B*). A more conservative method that uses the relationship between detrended records of burned area and fuel aridity (2) still indicates a substantial impact of ACC on total burned area, with a 19% (95% confidence: 12–24%) reduction in the proportion of total burned area attributable to ACC (Fig. S5).

Our attribution explicitly assumes that anthropogenic increases in fuel aridity are additive to the wildfire extent that would have arisen from natural climate variability during 1984–2015. Because the influence of fuel aridity on burned area is exponential, the influence of a given ACC forcing is larger in an already arid fire season such as 2012 (Fig. 5*A* and Fig. S5*C*). Anthropogenic increases in fuel aridity are expected to continue to have their most prominent impacts when superimposed on naturally occurring extreme climate anomalies. Although numerous studies have projected changes in burned area over the twenty-first century due to ACC, we are unaware of other studies that have attempted to quantify the contribution of ACC to recent forested burned area over the western United States. The near doubling of forested burned area we attribute to ACC exceeds changes in burned area projected by some modeling efforts to occur by the mid-twenty-first century (29, 30), but is proportionally consistent with mid-twenty-first century increases in burned area projected by other modeling efforts (17, 31–33).

Beyond anthropogenic climatic changes, several additional factors have caused increases in fuel aridity and forest fire area since the 1970s. The lack of fuel aridity trends during 1948–1978 and persistence of positive trends during 1979–2015 even after removing the ACC signal implicates natural multidecadal climate variability as an important factor that buffered anthropogenic

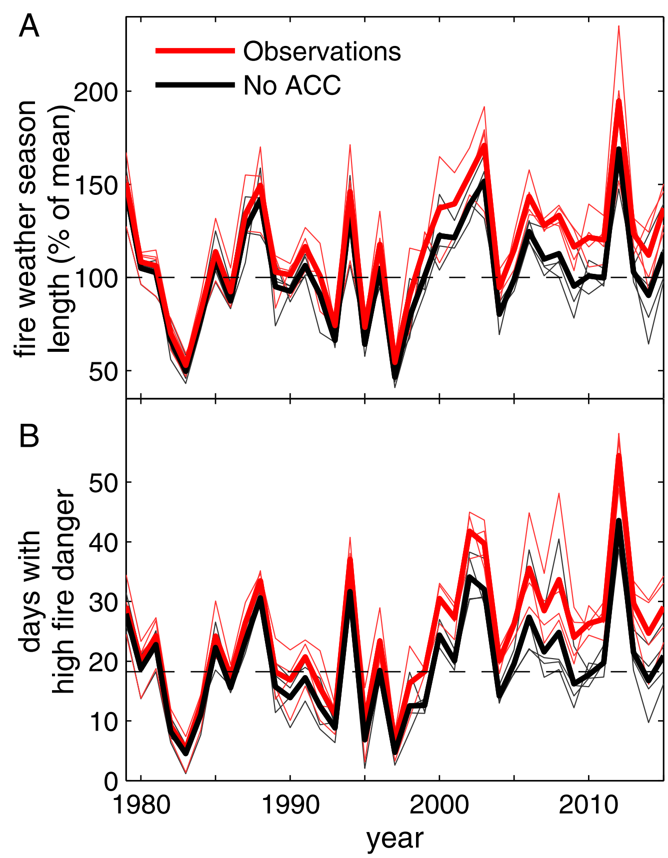

Fig. 4. Changes in fire-weather season length and number of high fire danger days. Time series of mean western US forest (*A*) fire-weather season length and (*B*) number of days per year when daily fire danger indices exceeded the 95th percentile. Baseline period: 1981–2010 using observational records that exclude the ACC signal. Red lines show the observed record, and black lines show the record that excludes the ACC signal. Bold lines show the average signal expressed across fuel aridity metrics.

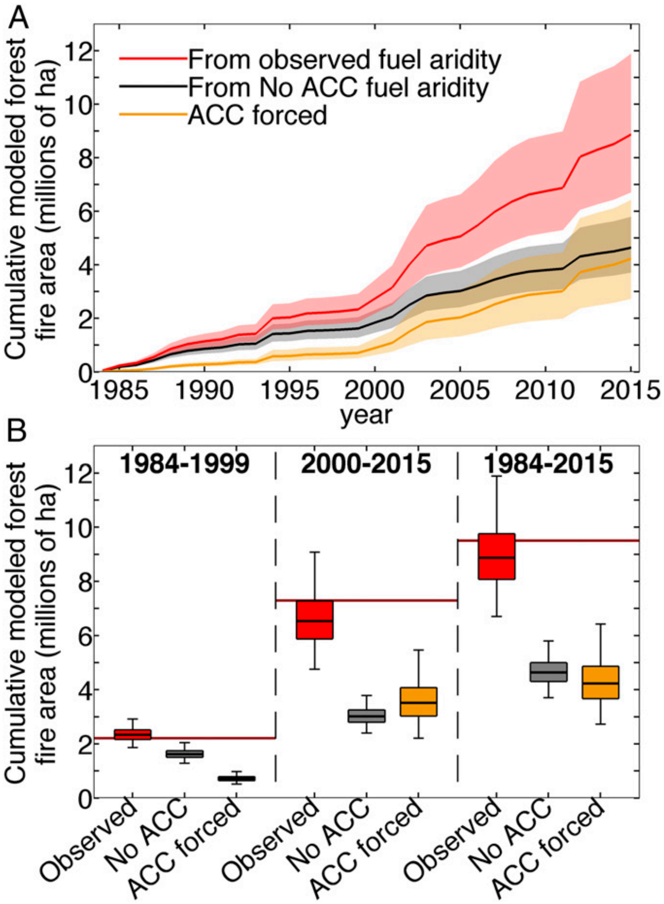

Fig. 5. Attribution of western US forest fire area to ACC. Cumulative forest fire area estimated from the (red) observed all-metric mean record of fuel aridity and (black) the fuel aridity record after exclusion of ACC (No ACC). The (orange) difference is the forest fire area forced by anthropogenic increases in fuel aridity. Bold lines in *A* and horizontal lines within box plots in *B* indicate mean estimated values (regression values in Fig. 1). Boxes in *B* bound 50% confidence intervals. Shaded areas in *A* and whiskers in *B* bound 95% confidence intervals. Dark red horizontal lines in *B* indicate observed forest fire area during each period.

effects during 1948–1978 and compounded anthropogenic effects during 1979–2015. During 1979–2015, for example, observed Mar–Sep vapor pressure decreased significantly across many US forest areas, in marked contrast to modeled anthropogenic increases (Fig. S6) (34). Significant declines in spring (Mar–May) precipitation in the southwestern United States and summer (Jun–Sep) precipitation throughout parts of the northwestern United States during 1979–2015 (Fig. S7 *A* and *B*) hastened increases in fire-season fuel aridity, consistent with observed increases in the number of consecutive dry days across the region (10). Natural climate variability, including a shift toward the cold phase of the interdecadal Pacific Oscillation (35), was likely the dominant driver of observed regional precipitation trends (36) (Fig. S7 *B* and *D*).

Our quantification of the ACC contribution to observed increases in forest fire activity in the western United States adds to the limited number of climate change attribution studies on wildfire to date (37). Previous attribution efforts have been restricted to a single GCM and biophysical variable (14, 16). We complement these studies by demonstrating the influence of ACC derived from an ensemble of GCMs on several biophysical metrics that exhibit strong links to forest fire area. However, our attribution effort only considers ACC to manifest as trends in mean climate conditions, which may be conservative because climate models also project anthropogenic increases in the temporal variability of climate and drought in the western United States (34, 38, 39). In focusing exclusively on the direct impacts of ACC on fuel aridity, we do not address several other pathways by which ACC may have affected wildfire activity. For example, the fuel aridity metrics that we used may not adequately capture the role of mountain snow hydrology on soil moisture. Nor do we account for the influence of climate change on lightning activity, which may increase with warming (40). We also do not account for how fire risk may be affected by changes in biomass/fuel due to increases in atmospheric CO_2 (41), drought-induced vegetation mortality (42), or insect outbreaks (43).

Additionally, we treat the impact of ACC on fire as independent from the effects of fire management (e.g., suppression and wildland fire use policies), ignitions, land cover (e.g., exurban development), and vegetation changes beyond the degree to which they modulate the relationship between fuel aridity and forest fire area. These factors have likely added to the area burned across the western US forests and potentially amplified the sensitivity of wildfire activity to climate variability and change in recent decades (2, 22, 24, 44). Such confounding influences, along with nonlinear relationships between burned area and its drivers (e.g., Fig. 1), contribute uncertainty to our empirical attribution of regional burned area to ACC. Our approach depends on the strong observed regional relationship between burned area and fuel aridity at the large regional scale of the western United States, so the quantitative results of this attribution effort are not necessarily applicable at finer spatial scales, for individual fires, or to changes in nonforested areas. Dynamical vegetation models with embedded fire models show emerging promise as tools to diagnose the impacts of a richer set of processes than those considered here (41, 45) and could be used in tandem with empirical approaches (46, 47) to better understand contributions of observed and projected ACC to changes in regional fire activity. However, dynamic models of vegetation, human activities, and fire are not without their own lengthy list of caveats (2). Given the strong empirical relationship between fuel aridity and wildfire activity identified here and in other studies (1, 2, 4, 8), and substantial increases in western US fuel aridity and fire-weather season length in recent decades, it appears clear from empirical data alone that increased fuel aridity, which is a robustly modeled result of ACC, is the proximal driver of the observed increases in western US forest fire area over the past few decades.

Conclusions

Since the 1970s, human-caused increases in temperature and vapor pressure deficit have enhanced fuel aridity across western continental US forests, accounting for approximately over half of the observed increases in fuel aridity during this period. These anthropogenic increases in fuel aridity approximately doubled the western US forest fire area beyond that expected from natural climate variability alone during 1984–2015. The growing ACC influence on fuel aridity is projected to increasingly promote wildfire potential across western US forests in the coming decades and pose threats to ecosystems, the carbon budget, human health, and fire suppression budgets (13, 48) that will collectively encourage the development of fire-resilient landscapes (49). Although fuel limitations are likely to eventually arise due to increased fire activity (17), this process has not yet substantially disrupted the relationship between western US forest fire area and aridity. We expect anthropogenic climate change and associated increases in fuel aridity to impose an increasingly dominant and detectable effect on western US forest fire area in the coming decades while fuels remain abundant.

Methods

We focus on climate variables that directly affect fuel moisture over forested areas of the western continental United States, where fire activity tends to be flammability-limited rather than fuel- or ignition-limited (1) (study region shown in Fig. 1, *Inset*). There are a variety of climate-based metrics that have been used as proxies for fuel aridity, yet there is no universally preferred metric across different vegetation types (24). We consider eight frequently used fuel aridity metrics that correlate well with fire activity variables, including annual burned area (Fig. 1 and Table S1), in western US forests.

Fuel aridity metrics are calculated from daily surface meteorological data (50) on a 1/24° grid for 1979–2015 for the western United States (west of 103°W). Although we calculated metrics across the entire western United States, we focus on forested lands defined by the climax succession vegetation stages of "forest" or "woodland" in the Environmental Site Potential product of LANDFIRE (landfire.gov). Forested 1/24° grid cells are defined by at least 50% forest coverage aggregated from LANDFIRE. We extended the aridity metrics calculated at the monthly timescale (ETo, VPD, CWD, and PDSI) back to 1948 using monthly anomalies relative to a common 1981–2010 period from the dataset developed by the Parameterized Regression on Independent Slopes Model group (51) for temperature, precipitation, and vapor pressure, and by bilinearly interpolating NCEP–NCAR reanalysis for wind speed and surface solar radiation. We aggregated data to annualized time series of mean May–Sep daily FWI, KBDI, ERC, and FFDI; Mar–Sep VPD and ETo; Jun–Aug PDSI; and Jan–Dec CWD. We also calculated the aridity metrics strictly from ERA-INTERIM and NCEP–NCAR reanalysis products for 1979–2015 covering the satellite era (*Supporting Information*).

Days per year of high fire potential are quantified by daily fire danger indices (ERC, FWI, FFDI, and KBDI) that exceed the 95th percentile threshold defined during 1981–2010 from observations after removing the ACC signal. Observational studies have shown that fire growth preferentially occurs during high fire danger periods (52, 53). We also calculate the fire weather season length for the four daily fire danger indices following previous studies (10).

The ACC signal is obtained from ensemble members taken from 27 CMIP5 global climate models (GCMs) regridded to a common 1° resolution for 1850–2005 using historical forcing experiments and for 2006–2099 using the Representative Concentration Pathway (RCP) 8.5 emissions scenario (Table S3 and *Supporting Information*). These GCMs were selected based on availability of monthly outputs for maximum and minimum daily temperature (T_{max} and T_{min}, respectively), specific humidity (*huss*), and surface pressure. Saturation vapor pressure (e_s), vapor pressure (e), and VPD were calculated using standard methods (*Supporting Information*). A variety of approaches exist to estimate the ACC signal (26). We define the anthropogenic signals in T_{max}, T_{min}, e, e_s, VPD, and relative humidity by a 50-y low-pass-filter time series (using a 10-point Butterworth filter) averaged across the 27 GCMs using the following methodology: For each GCM, variable, month, and grid cell, we converted each annual time series to anomalies relative to a 1901–2000 baseline. We averaged annual anomalies across all realizations (model runs) for each GCM and calculated a single 50-y low-pass-filter annual time series for each of the 12 mo for 1850–2099. We averaged each month's low-pass-filtered time series across the 27 GCMs and additively adjusted so that all smoothed records pass through zero in 1901. The resultant ACC signal represents the CMIP5 modeled anthropogenic impact since 1901 for each variable, grid cell, and month (*Supporting Information*).

We bilinearly interpolated the 1° CMIP5 multimodel mean 50-y low-pass time series to the 1/24° spatial resolution of the observations and subtracted the ACC signal from the observed daily and monthly time series. We consider the remaining records after subtraction of the ACC signal to indicate climate records that are free of anthropogenic trends (26).

Annual variations in fuel aridity metrics are presented as standardized anomalies (σ) to accommodate differences across geography and metrics. All fuel aridity metrics are standardized using the mean and SD from 1981 to 2010 for observations that excluded the ACC signal. Although the selection of a reference period can bias results (54), our findings were similar when using the full 1979–2015 time period or the observed data (without removal of ACC) for the reference period. The influence of anthropogenic forcing on fuel aridity metrics is quantified as the difference between metrics calculated with observations and those calculated with observations that excluded the ACC signal. Area-weighted standardized anomalies and the spatial extent of western US forested land that experienced high (>1 σ) aridity are computed for each aridity metric. Annualized burned area as well as aggregated fuel aridity metrics calculated with data from ref. 50 and the two reanalysis products are provided in Datasets S1–S3.

We use the regression relationship between the annual western US forest fire area and the all-metric mean fuel aridity index in Fig. 1 to estimate the forcing of anthropogenic increases in fuel aridity on forest fire area during 1984–2015. Uncertainties in the regression relationship due to imperfect correlation and temporal autocorrelation are propagated as estimated confidence bounds on the anthropogenic forcing of forest fire area. This approach was repeated using a more conservative definition of the regression relationship, where we removed the linear least squares trend for 1984–2015 from both the area burned and fuel aridity time series before regression to reduce the possibility of spurious correlation due to common but unrelated trends (Fig. S5). Statistical significance of all linear trends and correlations reported in this study are assessed using both Spearman's rank and Kendall's tau statistics. Trends are considered significant if both tests yield $P < 0.05$.

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SYNTHESIS & INTEGRATION

Transitioning western U.S. dry forests to limited committed warming with bet-hedging and natural disturbances

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Abstract. Historical evidence suggests natural disturbances could allow more forest persistence, than expected from models, over 40 yr of transition to the net-zero emissions needed to limit warming to <2.0°C (e.g., Paris Agreement). Forests must ultimately equilibrate with committed warming from accumulated emissions. Historical dry-forest landscapes were heterogeneous from large, infrequent disturbances (LIDs) that reduced tree density and basal area, followed by slow, variable tree regeneration and recovery for 1–3 centuries. These together effectively provided bet-hedging through stand- and landscape-level heterogeneity that enhanced resistance and resilience to a diversity of unpredictable subsequent disturbances. Recent disturbances have not yet exceeded historical variability in rates and patterns, but could cause mortality of ~26–51% of dry-forest area in the transition. This also means 1/2 to 3/4 of dry-forest area could escape most mortality and the mortality area could also have substantial forest persistence. Projections are unavailable for droughts or beetle outbreaks, but they recently caused about 3–4 times as much tree mortality as did moderate- to high-severity fires. Mortality could reduce forest area if new trees do not regenerate, but 24 studies showed recent regeneration after high-severity fires was slow, but indistinct from historical variability. Survival of smaller trees provided regeneration after beetle outbreaks and droughts. Regeneration in general was projected by 2060 to decline by ~10% in one study and increase by 50% in another. If openings from disturbances increased, some grasslands and shrublands could be restored, increasing landscape heterogeneity and resistance to disturbance spread. Given these trends and our limited ability to prevent LIDs, I suggest (1) refocusing restoration to increase bet-hedging resilience to droughts and beetle outbreaks by retaining small trees and diverse tree species, (2) expanding development of fire-safe landscapes to protect people and infrastructure from unavoidable increased fire, (3) enabling more managed fire to restore and enhance stand- and landscape-scale bet-hedging, and (4) accepting that LIDs will revise resistance, resilience, and adaptation, which enhance forest persistence, particularly if post-disturbance survivors are not logged and trees are not planted. Natural disturbance and slow recovery, if bet-hedged to increase resistance and resilience, could enable substantial forest persistence.

Key words: adaptation; beetle outbreaks; bet-hedging; climate change; disturbances; droughts; dry forests; fire; natural recovery; resilience; succession.

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Introduction

Since the 2015 Paris Agreement, the world plans to reduce emissions to limit warming to much less than 2.0°C, possibly 1.5°C, and it is worthwhile to focus on how major ecosystems may transition to this more limited level of warming that is now a global commitment. Extensive disappearing climates and ecosystem changes and the need for widespread assisted migration by the mid- to late 21st century under continuing moderate- to high emissions (e.g., Rehfeldt et al. 2014) are less likely. Understanding is now needed of impacts of more limited warming for specific ecosystems. Here, I review how bet-hedging and natural-disturbance processes (Baker and Williams 2015) could help transition current dry-forest landscapes in the western United States to limited committed warming. Bet-hedging uses small trees, large trees, and diverse trees to hedge against diverse disturbances. Committed warming occurs because once emissions are reduced so they are at net zero (emissions balanced by fixation), the long persistence of emitted CO_2 in the atmosphere and high oceanic heat capacity cause global temperatures to remain elevated for centuries near where they are at net zero (Collins et al. 2013, Mauritsen and Pincus 2017).

Dry forests are major montane ecosystems (Fig. 1), covering ~25.5 million ha of the western United States (Baker 2015). Dry forests include (1) dry pine forests most often dominated by ponderosa pine (*Pinus ponderosa*) or similar pines with relatively few associated trees, and (2) dry mixed-conifer forests with pines plus several other trees (e.g., *Abies concolor*, *Abies grandis*, *Populus tremuloides*, *Pseudotsuga menziesii*). Dry-forest landscapes historically also included grasslands and shrublands, as well as younger forests (Fig. 2), some of which were seral stages after high-severity fires in dry forests, although others were more persistent (Baker 2017*a*).

To keep committed warming below 2.0°C across dry forests of the western United States, emissions may need to be net zero by A.D. 2050 when 80% of projections show 2.0°C of warming would be reached with current emissions (Karmalkar and Bradley 2017). However, globally committed warming of well below 2.0°C that might allow 2.0°C of committed warming across dry forests of the western United States could also be achieved if net-zero emissions are reached by A.D. 2060 after rapid near-term reductions (Sanderson et al. 2016). The 2.6 representative concentration pathway (RCP), the lowest scenario of the Fifth Assessment Report (AR5) from the Intergovernmental Panel on Climate Change, was thought to feasibly constrain warming to <2.0°C (IPCC 2015), but this now appears unlikely (Sanderson et al. 2016). The next IPCC report (AR6), with newer scenarios congruent with 1.5–2.0°C of committed warming, is not due until 2022. However, Shared Socio-Economic Pathways (SSPs) that are being developed suggest a 1.9 RCP could feasibly constrain warming to 1.5°C (Rogelj et al. 2018). Updated global carbon-emissions accounting and pathways make 1.5°C feasible (Tokarska and Gillett 2018, Van Vuuren et al. 2018). Thus, net-zero emissions by 2060 are needed and feasible to avoid rising above 1.5–2.0°C (Tanaka and O'Neill 2018). Therefore, I consider A.D. 2060, ~40 yr, as the main period for transitioning dry forests, after which further, slower adjustment to committed warming continues.

No projections yet exist for extent of climate loss (current climate moves elsewhere or is changed) or its effects on tree populations in dry forests for pathways leading to net-zero emissions by 2060, but perspective is still possible now. Projections of climate loss in dry forests, primarily from bioclimate models, were mostly for A.D. 2060–2100 and/or RCPs of medium to high emissions (Table 1). Loss of climate would likely be lower than in RCP 2.6 (Table 1), but specific projections are lacking. Nonetheless, by 2015, total human-induced global warming was 0.93°C (Millar et al. 2017), about 1/2 to 2/3 of the way to 1.5–2.0°C, suggesting that effects that will occur are well underway. Here, I synthesize what might ensue in dry forests based on recent trends in natural disturbances, tree mortality, and tree regeneration, aided by projections and scenarios to 2060 for low or modest emissions, where available. Further refinement will be needed, but substantial evidence is available now that can provide useful perspective.

Also, bioclimate models do not reveal ecological effects, since they usually lack demography, dispersal, or natural disturbance, and mostly

Fig. 1. Dry forests covered about 25.5 million ha of the western United States, including about 12.6 million ha of dry pine forests and 12.9 million ha of dry mixed-conifer forests. Data are Landfire biophysical settings, which predict historical vegetation (http://www.landfire.gov).

only show how the climate of an ecosystem may change, not effects (Campbell and Shinneman 2017). Climate loss is expected to move upward from lower-elevation and northward from southerly trailing edges of dry-forest ranges, and tree mortality may follow, but unpredictably. Adult ponderosa may be most vulnerable in interior populations (var. *scopulorum*) and less in Pacific populations (var. *ponderosa*) of ponderosa pine, but vulnerability in dry forests may be heterogeneous in general (McCullough et al. 2017). These models are generally only for

Fig. 2. Historical dry-forest landscapes included forests as well as openings with grasslands and shrublands, as shown here in this Whitman Cross photograph from 1897 looking south at Mesa Verde (on the skyline), southwestern Colorado, across a ponderosa pine landscape with Gambel oak (*Quercus gambelii*) shrublands and montane grasslands. Reproduced from a scanned print of the original photograph (Cross 297) at the U.S. Geological Survey Denver Library, Photographic Collection, Denver, Colorado.

adult trees, but tree regeneration may ultimately control tree persistence and expansion (Bell et al. 2014, Dobrowski et al. 2015, Petrie et al. 2017). Natural disturbances (droughts, beetle outbreaks, wildfires, and diseases) will likely cause the tree mortality as climate is lost. Forest resilience could be exceeded and a tipping point (Reyer et al. 2015) crossed. However, inertia from long tree life spans, changing disturbances, and tree survival and regeneration might allow more forest persistence (Campbell and Shinneman 2017).

Here, I first review the historical roles of large, infrequent disturbances (LIDs), post-disturbance legacies, and slow natural recovery in dry forests. Then, I review recent natural disturbances, tree regeneration, and how persistence of tree populations in dry forests to warming could be aided by bet-hedging. Emergence of climates at higher elevations may offset losses in current ranges, if dispersal succeeds (Campbell and Shinneman 2017), but is not addressed here.

Historical Variability in Natural Disturbance and Recovery in Dry-Forest Landscapes

Large, infrequent disturbances historically accomplished most renewal in dry-forest landscapes

Historical dry-forest landscapes included open, low-density stands with large, old trees and a history of low-severity fires, but probabilistic landscape-scale studies found these open forests over only about 34%, on average, of dry-forest area (Baker 2017*a*). The other 66% historically had more diverse stand structures (examples in Table 2, reviews in Odion et al. 2014, Hanson et al. 2015). Historical forests were often younger, denser, and had been burned in fires varying in intensity and severity, as described explicitly in Hessburg et al. (2007:19): "Instead, area was dominated by forest structures that were intermediate between new and old forests, i.e., by pole to medium sized, rather than large trees.... This observation suggested that before any extensive management had occurred, the influence of fire in the dry forest was

Table 1. Projected losses of current dry-forest climates for individual species that occur in current dry forests of the western United States, based on bioclimate and process-based (only Mathys et al. 2017) models.

Emissions level/location	Species	Change (%)†	Date	Emissions scenario or RCP‡	Author(s)
Low					
Arizona–New Mexico Plateau	*Pinus ponderosa*	−58.0	2075–2100	2.6	Mathys et al. (2017)
North America	*Pseudotsuga menziesii*	−22.0	2075–2100	2.6	Mathys et al. (2017)
Idaho Batholith	*Pseudotsuga menziesii*	−19.0	2075–2100	2.6	Mathys et al. (2017)
Wyoming Basin	*Pseudotsuga menziesii*	−1.0	2075–2100	2.6	Mathys et al. (2017)
Medium–high					
North America	*Abies concolor*	−13.4§	2071–2100	A2/B2 mean	McKenney et al. (2007)
North America	*Abies grandis*	−49.6§	2071–2100	A2/B2 mean	McKenney et al. (2007)
North America	*Picea pungens*	−51.2§	2071–2100	A2/B2 mean	McKenney et al. (2007)
North America	*Pinus jeffreyi*	−68.6§	2071–2100	A2/B2 mean	McKenney et al. (2007)
North America	*Pinus ponderosa*	−40.4§	2071–2100	A2/B2 mean	McKenney et al. (2007)
North America	*Pinus ponderosa* var. *ponderosa*	−45.0	2060	6.0	Rehfeldt et al. (2014)
North America	*Pinus ponderosa* var. *scopulorum*	−77.0	2060	6.0	Rehfeldt et al. (2014)
North America	*Populus tremuloides*	−24.7§	2071–2100	A2/B2 mean	McKenney et al. (2007)
North America	*Pseudotsuga menziesii*	−31.5§	2071–2100	A2/B2 mean	McKenney et al. (2007)
North America	*Pseudotsuga menziesii* var. *glauca*	−35.0	2060	6.0	Rehfeldt et al. (2014)
North America	*Pseudotsuga menziesii* var. *menziesii*	−18.0	2060	6.0	Rehfeldt et al. (2014)
High					
Southwestern USA	*Picea pungens*	−81.0	2070–2099	A2	Notaro et al. (2012)
Southwestern USA	*Pinus ponderosa*	−47.0	2070–2099	A2	Notaro et al. (2012)
Southwestern Colorado	*Populus tremuloides*	−52.0	2060	6.0/8.5 mean	Rehfeldt et al. (2015)
Southwestern USA	*Pseudotsuga menziesii*	−50.0	2070–2099	A2	Notaro et al. (2012)
North America	*Pseudotsuga menziesii*	−59.0	2075–2100	8.5	Mathys et al. (2017)
Southwestern USA	All needleleaf evergreen trees	−100.0	2099	A2	Jiang et al. (2013)

Note: Area outside current climates may also emerge with some new area of suitable dry-forest climates, not shown here.
† The change (%) is relative to the present.
‡ Emissions scenarios are A2 (High emissions), B1 (Low), and B2 (Low–Medium); RCP = representative concentration pathway, which is the change in radiative forcing (W/m^2) in 2100 relative to pre-industrial conditions, as defined for emissions scenarios by the Intergovernmental Panel on Climate Change (IPCC). RCP 2.6 is Low, 4.5 is Medium, 6.0 is Medium–High, and 8.5 is High emissions.
§ This is the “no dispersal” projection result.

of a frequency and severity that intermittently regenerated rather than maintained large areas of old, fire tolerant forest.” The intermittent regeneration likely followed LIDs which varied in intensity, but were at least partly intense enough to kill substantial woody plants. Large, infrequent disturbances included fires, insect outbreaks, diseases, droughts, and blowdowns (Foster et al. 1998).

Many historical LIDs in dry-forest landscapes occurred in periodic climatic episodes. Large fires were often during droughts, as in 1848 when 41 of 63 fire-history sites across southwestern dry forests recorded this fire year (Swetnam and Baisan 1996), and in 1910 when 1.2 million ha burned in the northern Rocky Mountains (Odion et al. 2014). About 10 bark beetles had large outbreaks in dry forests (Bentz et al. 2010) when tree defenses were weakened by drought or other events, weather favored beetle reproduction, and mass attack could overcome tree resistance (Bentz et al. 2010, Negrón and Fettig 2014). An example is the 200,000- to 300,000-ha 1895–1909 mountain pine beetle (MPB; *Dendroctonus ponderosae*) outbreak in the Black Hills, South Dakota (Graham et al. 2016). Historical droughts, such as the A.D. 1574–1594 drought in the Southwest, also likely led to extensive tree mortality in dry forests (Swetnam and Betancourt 1998, Williams et al. 2013).

Large disturbances were likely infrequent in historical fire regimes and in other disturbance regimes. Modern fire regimes globally nearly all have log-normal fire-size distributions in which large fires are exponentially less frequent than small fires (Hantson et al. 2016). Historical fire-

Table 2. Examples of probabilistic studies and ancillary supporting sources that showed evidence of historical mixed-severity fire regimes, with substantial area of high-severity fire, that fostered heterogeneous historical dry-forest landscapes in the western United States.

Data source	Author(s)	Location(s)
Probabilistic		
Early aerial photographs	Hessburg et al. (2007)	WA, OR
Forest Inventory and Analysis data	Odion et al. (2014)	W USA
Early forest-reserve reports	Baker et al. (2007), Baker (2012, 2014), Williams and Baker (2014)	AZ, CA, OR, Rocky Mountains
Reconstructions–General Land Office surveys	Williams and Baker (2012*a*, *b*)	AZ, CO, OR
Reconstructions–Tree-rings at landscape scale	Sherriff et al. (2014)	CO
Ancillary supporting sources		
Early historical accounts	Baker (2012, 2014)	CA, OR
Early photographs	Baker (2009)	Rocky Mountains
Reconstructions–Paleo-charcoal	Compilation in Baker (2015)	W USA

Note: AZ, Arizona; CA, California; CO, Colorado; OR, Oregon; WA, Washington; W USA, western USA.

size and patch-size distributions in dry forests also had inverse-J shapes suggesting log-normal distributions (Williams and Baker 2012*a*, Baker 2017*a*). While rare, LIDs could be concentrated in episodes across large land areas, as were severe fires in the late 1800s in the southern Rocky Mountains (Veblen et al. 2000, Schoennagel et al. 2011, Baker 2017*b*), and large MPB outbreaks across the western United States and Canada (Jarvis and Kulakowski 2015).

The severely disturbed extent of LIDs had historical rotations (the expected time to affect the area of a landscape once) of one or more centuries. High-severity fires that killed >70% of basal area in dry forests historically had rotations of about 2–8 centuries (Baker 2015); moderate- to high-severity fires that killed 20% or more of basal area had rotations of 235–319 yr (Odion et al. 2014). Tree age distributions and early observations suggest large insect outbreaks and droughts were also infrequent events in dry forests (Blackman 1931, Swetnam and Betancourt 1998). The historical rotation for outbreaks of MPB, the main outbreak beetle in the western United States (Meddens et al. 2012), might be somewhat longer in ponderosa pine than lodgepole pine (*Pinus contorta*) forests, since ponderosa pine forests are more heterogeneous (Chapman et al. 2012). Jarvis and Kulakowski (2015) reconstructed MPB outbreaks in lodgepole pine at 10 sites in 200,000 ha of western Colorado and found four episodes from 1742 to 1910 that affected 0.5, 0.7, 0.5, and 0.4 of the 10 sites, a rotation of about 80 yr (168 yr/2.1). The rotation for drought-caused mortality in dry forests is unknown as there are no historical reconstructions. Pan-continental droughts that affected several U.S. regions occurred historically in ~12% of the last 1000 yr, but megadroughts of a decade or more, mainly in the Southwest and Central Plains, were rare in the last 500 yr (Cook et al. 2014). The 1574–1594 event, mentioned earlier, is the only historical one known to have caused extensive mortality in dry forests.

In the case of fires, the few percent that are large typically account for most of the total burned area (Strauss et al. 1989) and are often more intense (Swetnam and Baisan 1996). This importance of only a few percent of fires, the largest fires, to total burned area is evident in modern dry forests (Farris et al. 2010) and other forests (Baker 2009). Larger fires often have a mix of intensities and higher intensity, since fires become large because of rapid spread, driven by wind and drier fuels that allow more fuel consumption, increasing fire intensity (Alexander 1982). Large beetle outbreaks and lengthy droughts also appear to cause most tree mortality (Allen et al. 2010, Baker and Williams 2015, Graham et al. 2016), likely because resistance thresholds in trees are difficult to cross with smaller, less severe events (Romme et al. 1998).

Large, infrequent disturbances updated resistance, resilience, and legacies that facilitated recovery and bet-hedging

Large, infrequent disturbances with varying severities historically provided episodic adjustment across dry-forest landscapes, reducing area, density, and basal area of less disturbance-resistant trees, while increasing more disturbance-resistant trees, although current trees simply regenerate at times. Competition was lessened and the canopy was opened (Fig. 3a, b), often reducing vulnerability to subsequent disturbances for decades or longer (Parks et al. 2016).

Large, infrequent disturbances episodically tested and updated resistance, resilience, and bet-hedging across changing landscapes. Large, infrequent disturbances fostered diverse surviving tree species, sizes, and regeneration strategies that provided resistance and resilience to subsequent diverse disturbances (Table 3). This diverse stand and landscape structure and composition after LIDs could effectively provide stand- and landscape-level bet-hedging against an uncertain array of subsequent disturbances (Baker and Williams 2015). At the stand scale, bet-hedging was provided by combinations of large, old trees with thick bark that resisted mortality in fires and some beetle outbreaks (Graham et al. 2016, Welch et al. 2016), abundant small trees that resisted mortality in beetle outbreaks and droughts (Baker and Williams 2015), and diverse tree species so that some trees were not vulnerable to particular insects or diseases. At the landscape scale, areas of large trees, other fire-resistant trees, and low tree density provided landscape resistance to severe fires. Low-to-moderate fuel continuity allowed fires to spread, but with patchiness. Openings reduced ignitions, slowed disturbance spread, and reduced severity, while natural breaks could slow or terminate fires. Low–moderate contiguity of large trees and diverse patches may have reduced beetle spread and limited the size of patches of tree mortality (Graham et al. 2016). Young, recovering forests had high tree survival in beetle outbreaks (Graham et al. 2016) and droughts (Allen et al. 2010).

Natural recovery exemplifies resilience: "...the capacity of a system to absorb disturbance and reorganize while undergoing change so as to retain essentially the same function, structure, identity and feedbacks" (Walker et al. 2004:2). Large, infrequent disturbances in dry forests left behind complex effects from variable disturbance types and severities (Fig. 3), and these legacies (Foster et al.

(a)

(b)

(c)

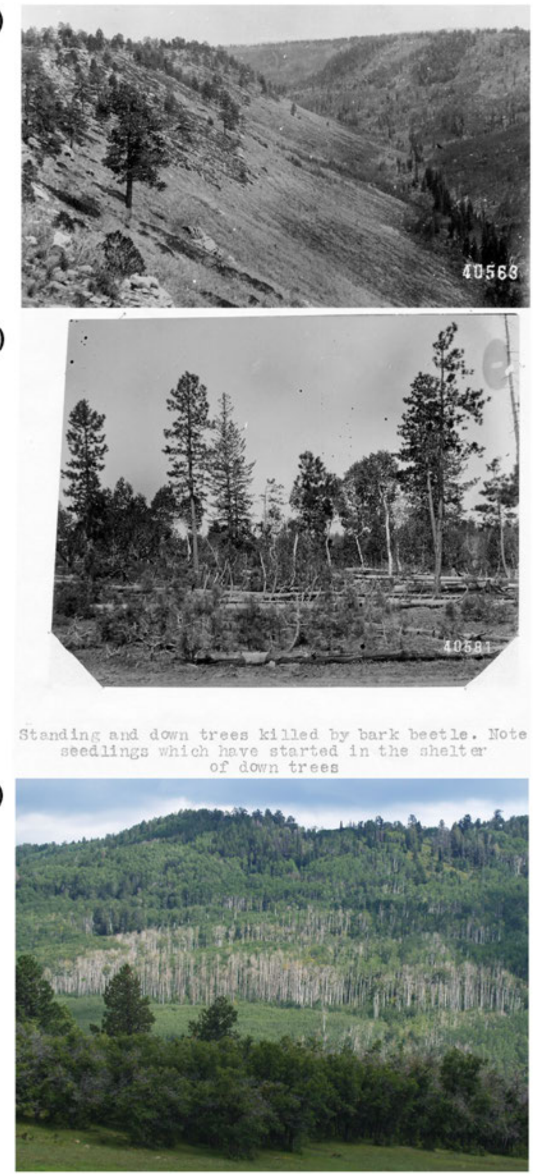

Fig. 3. Legacies after large, infrequent disturbances in dry forests: (a) a historical moderate- to high-severity fire in dry forests on the Uncompahgre Plateau, western Colorado, photograph in 1903 from Riley (1904); (b) a historical beetle outbreak in dry forests on the Uncompahgre Plateau, western Colorado, photograph in 1903 from Riley (1904); and (c) sudden aspen decline (SAD), a recent drought-linked disturbance, in southwestern Colorado, photograph by W. L. Baker, in 2006.

Table 3. Some historical structures (table entries), created by LIDs and environmental heterogeneity, that provided resistance and resilience at the stand and landscape scales to the three main types of LIDs in dry forests.

Property	Moderate- to high-severity fires	Beetle outbreaks	Droughts
Resistance–stand scale	Abundant large trees, some small trees	Abundant small trees, some large trees	Abundant small trees
	Fire-resistant trees	Diverse tree species	Diverse tree species
	Moderate fuel continuity (e.g., patches of rocks, low fuels)	Contiguous patches of small trees	Diverse topo-edaphic settings, some with more moisture
	Lower tree density/fuels, where this occurred, reducing fire severity	Lower tree density, where this occurred	Lower tree density, where this occurred
	Higher tree density/cover leading to shaded, moister fuels, where this occurred		
Resistance–landscape scale	Areas of large trees	Low–moderate contiguity of areas of large trees	Low–moderate contiguity of areas of large trees
	Areas of fire-resistant trees	Diverse patches dominated by different tree species	Diverse patches dominated by different tree species
	Areas of low tree density/fuels, where they occurred	Areas of low tree density, where they occurred	Areas of low tree density, where they occurred
	Limited areas of young, recovering forests	Large areas of young, recovering forests	Large areas of young, recovering forests
	Moderate fuel continuity	Discontinuous suitable host trees	
	Areas of higher tree density/ cover leading to shaded fuels		
	Openings that slowed fire spread (e.g., grasslands, wetlands)	Openings that broke up contiguous suitable host trees	
	Natural fire breaks (e.g., rock outcrops, streams, moist stands)	Natural openings with few or no host trees	
Resilience–stand scale	Resprouting trees and shrubs	Resprouting trees and shrubs	Resprouting trees and shrubs
	Surviving large seed trees, some patches of surviving small trees	Abundant small trees, some large surviving trees for seed	Abundant small trees, some large surviving trees for seed
		As much diversity in tree species as possible	As much diversity in tree species as possible
Resilience–landscape scale	Large seed trees, likely to survive, every 50–100 m, limited patches of small trees	Large areas with abundant small trees likely to survive, some patches of large trees	Large areas with abundant small trees likely to survive, some patches of large trees
	Diverse tree densities, basal areas, and tree species composition	Diverse tree densities, basal areas, and tree species composition	Diverse tree densities, basal areas, and tree species composition
	Most severely burned area within 100–200 m of an unburned edge	Patches with a diversity of dominant tree species	Patches with a diversity of dominant tree species

Note: LIDs, large, infrequent disturbances.

1998) or ecological memory (Johnstone et al. 2016) facilitated natural recovery. Resilience was enhanced by resprouting trees and shrubs, large old trees that provided post-disturbance seed, and variable tree densities and basal areas that provided diverse post-disturbance recovery (Table 3).

Highly variable historical tree regeneration, particularly in the Southwest

Successful ponderosa pine regeneration was limited by a required coincidence of favorable processes from seed formation to seedling survival (Pearson 1923, Feddema et al. 2013, Savage et al. 2013). However, land-survey records from 22,206 km of transects across 1.7 million ha of dry forests in the late 1800s showed that seedlings and/or saplings were present over 35–57% and dense over 20–30% of dry-forest area in Oregon, California, and part of northern Arizona (Baker and Williams 2015). Pulses of regeneration seen in some age structures were favored by canopy-reducing disturbances, particularly fire that created mineral seedbeds and reduced competition by grass, followed by fire-free periods or pluvials, that sustained regeneration (Dugan and Baker 2015). Moderate- to high-severity fires led to more regeneration than did low-severity fires (Wu 1999, Ehle and Baker 2003, Schoennagel et al. 2011, Baker and Williams 2015).

About 14% of dry-forest area, mostly in the Southwest, had sufficiently frequent low-severity fire (Baker 2017*a*) and drier climate to potentially limit regeneration to exceptional pluvials and fire-free periods (Covington and Moore 1994, Savage et al. 1996, 2013). Land-survey records document that seedlings and/or saplings were present over only 4–13% of two large landscapes in Arizona and one in Colorado (Baker and Williams 2015). However, forest age structure in two cases showed more continuous regeneration not limited to wet or fire-free periods, with broad peaks evident in one case (Mast et al. 1999). Broad episodes bring into question whether regeneration was rare and confined to unusual climatic episodes (Savage and Mast 2005).

Contrasting regeneration findings in the Southwest are also documented in early forest-reserve reports. Leiberg et al. (1904:28) said of the 329,000-ha San Francisco Peaks forest-reserve area on the western part of the Mogollon Plateau in northern Arizona:

> Reproduction of the yellow pine is, generally, extremely deficient as regards seedling and young sapling growth, except in an area lying east of Stoneman Lake and south of Morman Lake. Apparently there has been an almost complete cessation of reproduction over very large areas during the past twenty or twenty-five years, and there is no evidence that previous to that time it was at any period very exuberant.

What happened to favor regeneration near the lakes is unexplained, but a nearby landscape also had abundant regeneration. Stabler (1906:7) said of the eastern extension of the Mogollon Plateau onto Black Mesa and into the White Mountains:

> The reproduction of the yellow pine portion of the commercial forest type is wonderfully good. This in spite of the fact that the pine bunchgrass is as a rule very thick and vigorous and but little of it kept down by grazing. The fact that the grass is not grazed makes the numerous ground fires more serious than they otherwise would have been, but in spite of these fires. . .the reproduction is good and occurs in all ages.

A compelling explanation is lacking for contrasts in historical regeneration over large land areas.

Historically slow and incomplete natural recovery after LIDs in dry forests

Severely disturbed dry forests historically regenerated variably, but often slowly, and could remain unforested or sparsely forested for ≥100 yr (Table 4). Post-fire regeneration was at times very dense over extensive area in the Southwest (Fig. 4a, b). High-severity fires could be followed by extended tree regeneration lasting 20–60 yr, which could also be lagged by 15–20 yr and even have >50-yr lags with little or no tree regeneration (Table 4). Openings (grasslands, shrublands) created or maintained by high-severity fires could persist for 130–150 yr or more (Tables 4, 5) and be quite large. For example, in the Sierra, Show (1924:83) reported:

> Perhaps the most striking characteristic of the timber region of northern California. . .is the very large area occupied by brushfields. The brushfields, for the most part, are the results of fires which have destroyed the timber and allowed the brush to occupy the ground; in round numbers 1,500,000 acres [607,000 ha] are now in this condition. Of this million and a half acres probably 75% is restocking naturally, scattered individuals and groups of trees having survived the fires of the past, and can be depended on to take care of themselves. . ..

Forests often, but not always, recovered after intense fires, particularly if surviving seed trees were nearby; if so, trees regenerated and tree density and basal area increased, and forests often became denser (Fig. 4c). Probabilistic studies found dense middle-aged forests and created or maintained grasslands and shrublands in all dry-forest landscapes (Table 2). However, many pathways of forest recovery likely occurred (Kashian et al. 2007). In dry forests, open forest patches and some dense forest patches may have simply persisted and grown older, and some dense forest patches may have been thinned by competition or disturbances (Oliver 1995, Zhang et al. 2013) until a mature forest re-established (Moir and Dieterich 1988).

Including the lag before tree regeneration, recovery of a mature forest after high-severity fire historically required >100 yr (Table 6). Old growth could be reached within 150–200 yr (Mehl 1992, Hamilton 1993), but 150–300 yr for

Table 4. Historical lags in tree regeneration and the length of successful episodes of natural tree regeneration after high-severity fires in dry forests, based on tree-ring reconstructions and early observations.

Topic/Author(s)	Location	Observation
Huckaby et al. (2001)	Front Range, Colorado	Tree regeneration delayed on average by 18 yr after high-severity fires, ranging from 0 to 33 yr for 16 fires from A.D. 1531–1880
Boerker (1915:15)	Western Sierra, California	"Unlike the chaparral regions of southern California, this brush is only a temporary type and is, in most cases, the result of fire having destroyed the forest cover. . .In most cases, in from 5 to 10 years after the fire has consumed the timber, the brush takes possession of the land. . .after the brush has established itself, if seed trees are nearby, seedlings will get started and fight their way through the brush. It takes from 15 to 30 years for a seedling to get large enough to overtop the brush. . ."
Wu (1999)	San Juan Mts., Colorado	Tree regeneration concentrated within 20 yr after higher-severity fires
Baker (2017*b*)	Uncompahgre, Colorado	Tree regeneration sparse or lacking in a stand 24 yr after high-severity fire
Ehle and Baker (2003)	Front Range, Colorado	Tree regeneration concentrated within 20–25 yr after high-severity fires
Nagel and Taylor (2005:448)	Lake Tahoe Basin, California	"Tree regeneration into the chaparral stands was highest during the first two or three decades after the fire [6 fires in 1861–1882], but tree establishment continued for at least five decades after the last fire in all of the stands"
Lauvaux et al. (2016:82)	Southern Cascades, California	"Tree populations were multi-aged. Initial establishment [after 6 fires in 1864–1918] was slow and typically peaked five or more decades after the fire"
Duthie (1914:14)	Front Range, Colorado	Tree regeneration sparse or lacking for first 50 yr: "A careful reconnaissance of the region made in 1911 showed that there are over 10,000 acres of land from which all forest cover was consumed by these fires half a century ago, and upon which there has been practically no natural restocking"
Sherriff (2004)	Front Range, Colorado	Tree regeneration concentrated within 19–60 yr after high-severity fires
Huckaby et al. (2001:25)	Front Range, Colorado	". . .openings were created by a fire in 1851, and remained unforested 149 years later. . .the northern part of the area may have burned again in 1880, slowing tree regeneration"
Kaufmann et al. (2003:239)	Front Range, Colorado	". . .historical mixed severity fires and delays of regeneration into openings created by fire contributed to a very open, spatially complex and temporally dynamic landscape structure"
Pearson (1914:249)	Arizona and New Mexico	"A characteristic feature of the timbered mountains in Arizona and New Mexico at altitudes above 8000 feet is the occurrence of extensive burns. The original forests below 9500 feet were composed mainly of western yellow pine (*Pinus ponderosa*), Douglas fir. . ., limber pine. . ., Mexican white pine. . ., and white fir. . .The greater portions of the burns have grown up to quaking aspen. . ., but extensive areas are practically bare. Scattering trees of the original forest usually remain, and where this condition exists or where the burn is comparatively small conifers are generally restocking the land. . ."

conifers to regain dominance over aspen in mixed-conifer forests (Table 6). Historical high-severity fire rotations of 2–8 centuries (Baker 2015) would have often allowed full recovery to old growth before the next high-severity fire.

Fluctuating historical dry-forest landscapes of recovery had heterogeneous structure

Overall, historical dry-forest landscapes in the western United States fluctuated from infrequent large natural disturbances that included substantial severe fires, beetle outbreaks, and droughts that killed many trees, leaving a diversity of legacies, followed by 100–300 yr of natural recovery. Where tree density and basal area were reduced, vulnerability to droughts and beetle outbreaks often declined; where old trees persisted, vulnerability to severe fires was reduced. Slow, variable post-disturbance tree regeneration and growth made natural recovery after LIDs a dominant

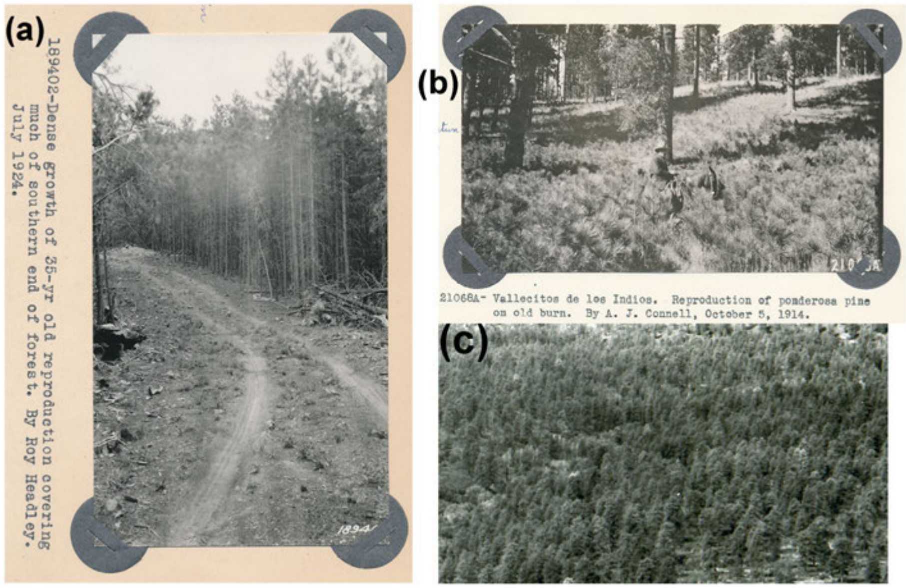

Fig. 4. Dense historical ponderosa pine regeneration after fire in the Southwest: (a) after likely large high-severity fire in the late 1800s in ponderosa pine forests, southern Coconino National Forest, Mogollon Plateau, Arizona, photograph taken in 1924 by Roy Headley, Historical Photo Collection, Region 3, U.S. Forest Service, Albuquerque, New Mexico; (b) after fire in the Jemez Mountains, New Mexico, photograph taken in 1914 by A. J. Connell, Historical Photo Collection, Region 3, U.S. Forest Service, Albuquerque, New Mexico; and (c) an example of a dense middle-aged historical forest, reproduced from a zoom of the right center of Fig. 2, an 1897 photograph by Whitman Cross.

ongoing process in most historical dry-forest landscapes. Episodes of LIDs across large areas meant that large land areas may have been synchronously recovering from natural disturbances. Infrequent disturbances and slow, variable natural recovery explain why historical dry-forest landscapes were spatially heterogeneous with a mix of old forests, middle-aged forests, recently disturbed forests, large and small openings with incipient or nearly completed regeneration, and more persistent openings, as documented by probabilistic landscape-scale studies (Table 2). This stand and landscape diversity conferred resistance, resilience, bet-hedging, and adaptation to diverse, unpredictable future disturbances, but substantial fluctuation still occurred.

Emerging Patterns of Transition to Committed Warming

Tree-mortality agents in dry forests over the last few decades

Increased tree mortality and regeneration decline or failure are expected during the transition. Increasing tree mortality is evident around the world (Allen et al. 2010). In dry forests, mortality is occurring from fires, beetle outbreaks, and directly from drought and temperature stress (Anderegg et al. 2013). Background rates of tree mortality (non-catastrophic, including all agents) increased significantly (3.3% per year, a doubling time of 22 yr), likely from warming, in the 15 old-forest plots most likely in dry forests, since they had short mean fire intervals (Van Mantgem et al. 2009: Table 1). In all plots censused from 1955 to 2007 across the western United States, 19% of trees died over the roughly 50-yr period (Van Mantgem et al. 2009), which is a 263-yr rotation (50/0.19). That would not lead to lasting loss of old forests, as 263 yr is ample time to regrow old trees, but if mortality doubled further, then it could become very limiting, and drought and heat stress could become the main cause of tree mortality (Allen et al. 2010).

Even with more severe (non-background) mortality from beetle outbreaks, droughts, and fires, there are survivors that play key stand-level roles

Table 5. Longer-term studies and observations of post-fire creation or maintenance of grasslands and shrublands after historical high-severity fires in dry-forest landscapes of the western United States.

Author(s)	Location	Years after fire	Observation
Guiterman (2016)	Jemez Mts., New Mexico	>115	Most of the area of 5 large patches (totaling 1142 ha) of mixed montane shrubland, dominated by Gambel oak (*Quercus gambelii*) that originated primarily in 1894–1900 remained largely unforested. Originating fires were likely mixed- to high-severity
Baker (2014)	Sierra Nevada, California	109–118	About 22% of montane chaparral, likely burned in high-severity fires in the late 1800s, did not become forested, and instead remained as montane chaparral, over periods of 109–118 yr
Nagel and Taylor (2005)	Northern Sierra, California	~120–140	About 38% of montane chaparral patches that originated after high-severity fires in 1861–1882 had not become forested by the 2000s
Lauvaux et al. (2016)	Southern Cascade Mts., California	~100–150	About 35% of montane chaparral patches that originated after high-severity fires in 1864–1918 had not become forested by the 2010s
Huckaby et al. (2001), Baker (2009:249)	Front Range, Colorado	~120–150	By about A.D. 2000 [120–50 yr after fires], some forests burned in high-severity fires in 1851 or 1880 had recovered to dense, middle-aged forests, but some openings were still unforested grasslands that were slowly reforesting
Baker (2017*b*)	Uncompahgre Plateau, Colorado	~130–150	About 40% of a large ponderosa pine and mixed-conifer landscape with evidence of high-severity fire in the late 1800s was nonforested (e.g., shrubs, small trees, grasslands); about half the nonforested area that was a mixture of grasslands, shrublands, recent burns, and areas with small trees was not forested by 2010, likely indicating at least century-scale stability after high-severity fires

in forest resilience (Table 3). In beetle outbreaks, most smaller trees survive as do some percentage of larger trees. A 1965–1978 MPB outbreak in the Colorado Front Range killed 25% of ponderosa pines of all sizes, especially 20–36 cm dbh, and reduced basal area by 38% (McCambridge et al. 1982). In British Columbia, a more severe 2005–2008 MPB outbreak, also with western pine beetle (*Dendroctonus brevicomis*), killed ~80% of trees, including 23–42% <15 cm dbh, 81% 15–30 cm, and 94% >30 cm over >175,000 ha, with little variation across a wide range of tree densities (Klenner and Arsenault 2009). In the Black Hills of South Dakota and Wyoming, an MPB outbreak over ~157,000 ha in 2004–2014 mostly killed ponderosas 23–43 cm dbh (Graham et al. 2016). Stands with <21 m^2/ha of basal area were little affected, but mortality increased up to 28–34 m^2/ha, where 74% of trees were killed, but 60% for >34 m^2/ha (Graham et al. 2016). Many trees survived, with means of 141 trees/ha of tree density and 11.7 m^2/ha of basal area. After beetle outbreaks, there were surviving trees of all sizes, especially small trees, as well as patches of surviving trees (Six et al. 2014). Dry forests were substantially renewed, and yet able to persist.

Mortality from droughts in dry forests has not been isolated, as beetles often ultimately kill many drought-affected trees. However, similar mortality patterns were evident with trees of all sizes killed and the highest percent mortality in larger trees (Ganey and Vojta 2011). Droughts put tall, old conifers especially at risk of replacement by shorter trees and shrubs (Bennett et al. 2015, McDowell and Allen 2015, McDowell et al. 2015), because taller trees are more physically vulnerable to failure to conduct water. A surprising 70% of a global sample of trees, in both dry and wet environments, operates with low physiological safety margins for escaping mortality from drought (Choat et al. 2012). Mortality consistent with these drought vulnerabilities is already occurring (Bennett et al. 2015). In contrast, larger trees generally better survive fires, because of thicker bark, elevated branches, and other adaptations (Baker 2009).

Table 6. Longer-term studies and observations of post-fire recovery to forest after historical high-severity fires in dry-forest landscapes of the western United States.

Author(s)	Location	Years after fire	Observations of post-fire succession in dry forests
MacKenzie et al. (2004)	Western Montana	60–100	Tree density/basal area approached pre-fire level within about 60–100 yr, basal- area increase slowed ~100 yr after high-severity fire
Baker (2014)	Sierra Nevada, California	109–118	About 78% of chaparral, likely burned in high-severity fires in the late 1800s, became forested over periods of 109–118 yr
Smith and Smith (2005), Baker (2017*b*)	Uncompahgre Plateau, Colorado	100–137	Conifers can begin to overtop aspen within about 100 yr, with mixed conifer–aspen stands at about 137 yr after high-severity fires
Nagel and Taylor (2005)	Northern Sierra, California	~120–140	About 62% of montane chaparral patches that originated after high-severity fires in 1861–1882 had become forested by the 2000s
Lauvaux et al. (2016)	Southern Cascade Mts., California	~100–150	About 65% of montane chaparral patches that originated after high-severity fires in 1864–1918 had become forested by the 2010s
Huckaby et al. (2001), Baker (2009:249)	Front Range, Colorado	~120–150	By about A.D. 2000 [120–150 yr after fires], some forests burned in high-severity fires in 1851 or 1880 had recovered to dense, middle-aged forests, but some openings were still reforesting
Baker (2017*b*)	Uncompahgre Plateau, Colorado	~130–150	About 40% of a large ponderosa pine and mixed-conifer landscape was nonforested (e.g., shrubs, small trees, grasslands) in the late 1800s; about half that area had become forested by 2010, likely indicating natural recovery after high-severity fires
Leiberg (1902:74)	Western Sierra, California	150	"The yellow pine on these tracts is mostly old growth; that is, the greater percentage of suitable size for mill timber is over 150 years of age"
Wu (1999:134)	San Juan Mts., Colorado	~150–200	"Even-aged stands still maintain their structure, such as a prominent post-fire cohort of aspen or ponderosa pine, 150 years after their last lethal fires, which occurred in the period from 1850 to 1880. . .therefore, this study estimates that all-age structure requires at least two hundred years to develop"
Kercher and Axelrod (1984)	Western Sierra, California	~250	Simulation suggested that about 250 yr would be required for Sierran mixed-conifer forests to recover and stabilize after severe disturbance
Baker (1925:89)	Central Rocky Mountains	≥250	"On the assumption that conifers found in the aspen zone will bear seed at 80 years, most areas ought to be well seeded in with reproduction in three tree generations or about 250 years in the Douglas fir-white fir zone. . .certain areas in the lower zones may require more than 250 years. . ."
Duthie (1914:14)	Front Range, Colorado	200–300	"It is estimated that two or three centuries would elapse before these burns would again be fully reforested if natural regeneration were depended upon to produce a satisfactory forest cover" (describing recovery after high-severity fires that occurred a half century earlier)
Zier and Baker (2006:261)	San Juan Mts., Colorado	Long periods	Over about a century, 40% of mixed-conifer forests visible in 25 scenes in early photographs showed increased conifers, while in 60%, there was no change in proportions of aspen and conifers, suggesting that ". . .long periods of time may be needed for conversion from aspen to conifers, if it occurs at all"

Given these vulnerabilities and documented mortality effects, what were recent sources of mortality; were severe fires, beetle outbreaks, or droughts the largest cause of non-background tree mortality over the last few decades? Details of analysis are in Appendix S1. Most important from this analysis is that insects-disease, on average, overall led to 2.1 times as much mortality area as did moderate- to high-severity fires, 1.7 times in ponderosa, and 2.6 times in dry mixed conifer (Table 7). Estimated rotations were 565 yr for moderate- to high-severity fires and 221 yr for insects-disease in dry mixed-conifer forests (Table 7). Rotations were 408 yr for moderate- to high-severity fires and 247 yr for insects-disease in ponderosa pine. These are similar to 2003–2012 mortality rotations of 500 yr for fire and 286 yr for beetles across all forests of the western United States from the inverse of annual mortality of 0.20% for fire and 0.35% for beetles, and the ratio of insects-disease to moderate- to high-severity fire of 1.75 is also similar (Berner et al. 2017). Under a hypothetical California-type drought scenario moving across dry forests (Appendix S2), an affected area of 5.5 million ha every six years would have a mortality area of 1.3 million ha (24%); if half from direct drought mortality, this would be a drought mortality rotation of 191 yr (6 yr/(0.65 million ha/20.7 million ha)). If so, droughts and insects-disease would likely account for about 3–4 times as much mortality area as severe fires.

Recent sizes and rates of beetle outbreaks, droughts, and moderate- to high-severity fire in dry forests are probably not yet outside the historical range of variability (Table 8), although evidence about historical variability is limited and some individual events have been exceptional locally (e.g., 2012–2016 California drought). Large beetle outbreaks have individually affected up to about 175,000 ha in dry forests, approaching the same scale as the 200,000- to 300,000-ha outbreak in the Black Hills in 1895–1909. The estimated recent beetle mortality rotation of 241 yr would not preclude full recovery of old-growth forests during or after the transition. The historical beetle-outbreak mortality rotation is too poorly known to be certain that this recent rate is or is not similar. Available evidence is insufficient to be able to assess historical vs. recent drought impacts on dry forests, but drought rates themselves are in general not outside historical variability in the western United States (Wuebbles et al. 2017; Appendix S3). However, if the frequency distribution of droughts does not change, $\sim 1°C$ elevated temperature alone will cause an increase in drought events sufficient to kill ponderosa pine seedlings by about 1.8 events by A.D. 2100 under RCP 2.6 (Adams et al. 2017). Larger recent moderate- to high-severity fires have individually affected about 30,000–60,000 ha, except for the 128,000-ha Rodeo–Chediski fire in Arizona (Table 8). Historical fire-size evidence is limited, in general, but the area burned at moderate to high severity on the Uncompahgre Plateau, Colorado, likely in 1879, was at the scale of about

Table 7. Affected area and estimated mortality area in dry forests across the western United States from 1999 to 2012 (n = 14 yr) from moderate- to high-severity fire and insects-disease.

Area and measure	Ponderosa pine	Dry mixed conifer	Total
Affected area			
Fire area (ha)	518,580	415,971	934,551
Insects-disease area (ha)	2,319,651	2,874,101	5,193,752
Fire rotation (yr)	265	367	311
Insect rotation (yr)	59	53	56
Ratio: insects-disease/fire area	4.5	6.9	5.6
Mortality area from multiplying affected fire area by 0.65 and affected insects-disease area by 0.24			
Fire area (ha)	337,077	270,381	607,458
Insects-disease area (ha)	556,716	689,784	1,246,500
Fire rotation (yr)	408	565	478
Insect rotation (yr)	247	221	233
Ratio: insects-disease/fire area	1.7	2.6	2.1
Fire area in 40 yr (% of total), if no change	9.8	7.1	8.4
Insects-disease area in 40 yr (% of total), if no change	16.2	18.1	17.2
Fire area in 40 yr (% of total), if projected	15.4	10.5	12.8
Total analysis area (ha)	9,825,679	10,910,705	20,736,384

Notes: Data on affected areas and total analysis areas are from Baker and Williams (2015). See Appendix S1 for an explanation of estimation of mortality area.

Table 8. Comparative sizes, durations, and rotations of recent large infrequent disturbances in dry forests and the expected mortality area during the 40-yr transition.

Attribute	Insects-diseases	Droughts	Moderate- to high-severity fires
Example events among the largest (ha) events since 1984 in dry forests†	~ 157,000 ha SD/WY‡ >175,000 ha BC§	~5, 500,000 ha CA¶ ~700,000 ha U.S.#	30,146-ha 2012 Whitewater Baldy, NM 34,432-ha 2002 Hayman, CO 36,611-ha 2012 Ash Creek, MT 50,287-ha 2013 Rim, CA 56,174-ha 2011 Wallow, AZ 127,667-ha 2002 Rodeo–Chediski, AZ‖
Duration of these example events (yr)	4–14	5	1
Estimated recent mortality rotation (yr) across total dry-forest area††	233	191	478
Estimated historical mortality rotation (yr) across total dry-forest area for reference	>333‡‡	Unknown	362–491§§
Expected mortality area (% of total dry-forest area) in transition if no change in rotation¶¶	17.2	20.9	8.4
Projected mortality area (% of total area) in transition if climate change shortens rotation¶¶	Not available	Not available	12.8

Note: Province and state abbreviations: AZ, Arizona; BC, British Columbia; CA, California; CO, Colorado; NM, New Mexico; SD, South Dakota; WY, Wyoming.

† These are affected areas, in ha, not mortality areas.

‡ Graham et al. (2016).

§ Klenner and Arsenault (2009).

¶ From Tree Mortality Task Force (2017) and Potter (2017).

\# From Worrall et al. (2013) for roughly the area of aspen decline affecting dry mixed-conifer forests.

‖ From MTBS data (www.mtbs.gov); the area is the sum of the moderate- and high-severity classes in the MTBS pdf map of each fire.

†† From the text, in the case of drought, and from Table 7, in the case of fire and insects-diseases; rotation is the time, in years, it is expected to take for these disturbances to affect land area equal to whole landscapes.

‡‡ The original rotation estimate of >80 yr for affected area is given in the text. The rotation for mortality area can be estimated by dividing by 0.24, which is the estimate from Hicke et al. (2016) used in Table 7.

§§ The original rotation estimate from Odion et al. (2014) was 235–319 yr for affected area, and the rotation for mortality area can be estimated by dividing by 0.65, as explained in the text and used in Table 7. After division, the original 235–319 yr range becomes 362–491 yr.

¶¶ From Table 7.

75,000–90,000 ha (Baker 2017*b*), thus similar to larger recent fires. The geometric mean higher-severity patch size was 47% lower in the recent than the historical period across 624,156 ha of dry forests in the Colorado Front Range (Williams and Baker 2012*a*). Moderate- to high-severity fire in dry forests was not operating from 1984 to 2012 at rates that exceeded historical rates, and the fraction of fires that burned at high severity had not increased (Baker 2015). The recent fire-mortality rotation for moderate- to high-severity fires of 478 yr is within, but toward the long end of the estimated historical rotation of 362–491 yr (Table 8).

Assuming no increase, in ponderosa pine, a resulting fire-mortality rotation of 408 yr would lead to expected mortality area of 9.8% over the 40-yr transition (Table 7). In dry mixed conifer, a fire-mortality rotation of 565 yr would lead to mortality area of 7.1%. In ponderosa pine, an insects-disease mortality rotation of 247 yr would lead to mortality area of 16.2% over the 40-yr transition (Table 7). Similarly, in dry mixed conifer, an insects-disease mortality rotation of 221 yr would lead to mortality area of 18.1% (Table 7). Under a California-type drought scenario, a 191-yr mortality rotation would lead to a mortality area of 20.9%. Overall, if no change in rates over the 40-yr transition, actual mortality area from fire and insects-disease would total ~26% of dry-forest area, 1/3 from moderate- to high-severity fires and 2/3 from insects-disease, a mortality rotation of 154 yr, which could still leave substantial area of old forests by the end of the transition. However, if a California-drought scenario ensued, an added 21% in 40 yr could lead to a total mortality area of $\sim$47%, a rotation of $\sim$85 yr, which could leave much less old forest, since it is particularly vulnerable to droughts and beetles.

Projecting possible increases in these disturbances during the 40-yr transition period is only roughly possible, and only for fire (Table 7; Appendix S3). There are no specific projections for only 1.5–2.0°C of warming on drought, fire, and insects. The U.S. Global Change Research Program reported low-to-medium confidence in a current anthropogenic climate-change effect on fire in the western United States (Wuebbles et al. 2017). Nonetheless, to estimate an upper bound on possible increases in moderate- to high-severity fire in dry forests, I used the midpoint of the low range of projected increases in area burned by A.D. 2046–2065 across 23 analysis areas under moderate emissions (RCP 4.5), which is 1.57 in ponderosa pine and 1.48 in dry mixed conifer (Baker 2015). These were the most recent area-burned projections, which are needed to estimate future mortality area. Using these, the percentage of mortality area from fire would increase from 9.8% to 15.4% in ponderosa pine and from 7.1% to 10.5% in dry mixed-conifer forests (Table 7). If combined with the hypothetical California-drought scenario, the total could reach about 51%.

Recent and projected tree regeneration in dry forests

Is there evidence of tree-regeneration decline in dry forests that could make the forest loss from tree mortality more permanent? Current rates and patterns of tree regeneration in all dry forests are relevant, but the only systematic monitoring is by the Forest Inventory and Analysis Program (FIA). Since 1995, FIA data are remeasured at 5- to 10-yr intervals on plots each representing about 2429 ha (Bechtold and Patterson 2005). Forest Inventory and Analysis data were used to analyze recent recruitment of juvenile vs. adult trees relative to climate in the western United States (Bell et al. 2014, Dobrowski et al. 2015). Ponderosa pine and Douglas-fir seedlings were much less likely to be present than were adults (28,177 plots), particularly along the warmer western and southern range margins of ponderosa pine (Bell et al. 2014). Similarly, for most conifers (13 species in 33,665 plots) in dry forests, juveniles occupied moister sites than did adults (Dobrowski et al. 2015). A caveat is that historical variability in tree regeneration was naturally high, as reviewed earlier, leaving in question whether these short periods of observation represent lasting trends.

These studies provide context, but tree regeneration after severe disturbances in dry forests is most relevant to the transition, since disturbances leave forests most dependent on regeneration. A focus has been on regeneration after high-severity fires; 24 studies, all I found, showed tree regeneration after these fires was almost universally heterogeneous (Table 9). Within the first 30 yr, substantial area lacked any conifer regeneration, while other area had adequate or dense regeneration (Table 9). Where studied, regeneration density was nearly always lowest in high-severity areas, relative to low- or moderate-severity areas. Ponderosa regeneration was commonly highest adjacent to the unburned margin of the fire and declined into the fire to low levels within 100–200 m, often attributed to seed-dispersal limitations, the hotter environment of open areas, or competition with shrubs or deciduous trees. Studies that analyzed topographic effects found regeneration especially deficient at low elevations and on south-facing slopes. Regeneration after high-severity fires in Colorado was concentrated in only three years with unusually high growing season precipitation over a 24-yr period, based on precisely dated seedlings (Rother and Veblen 2017). Less concentrated years of regeneration were evident in young adult trees, less precisely datable, after older New Mexico fires (Savage et al. 2013). Although regeneration was still sparse and favored near unburned margins at 28 and 45 yr post-fire, it was extrapolated to extend within $\sim$50 yr across the 28-yr-old high-severity burn (Haire and McGarigal 2010). Substantial declines in post-fire tree regeneration occurred from warmer and drier conditions since 2000, suggesting possible declines with warming (Stevens-Rumann et al. 2018).

Tree regeneration after recent high-severity fires was often considered unnatural or deficient, but historical evidence now does not support this. Dense regeneration was earlier considered hyperdense and outside the natural range of variability (Savage and Mast 2005). Since then, we have found (1) dense regeneration occurred

Table 9. Studies of tree regeneration up to 64 yr after high-severity fires in dry forests of the western United States arranged by the number of years since fire.

Author(s)	Location	Years after fire	Post-fire seedling/sapling density (trees/ha)
Bonnet et al. (2005)	South Dakota Black Hills	2	>700 ha^{-1} in burn (19 transects in 1 fire) within 12 m of unburned edge, declining inward, still some at 120 m, 180 m; positive effect, scorched needles on burned mineral soil; negative, high understory cover
Keyser et al. (2008)	South Dakota Black Hills	2–5	By year 5 (36 sites in 1 fire), >1000 ha^{-1} in unburned, low and moderate severity; little in high severity
Meigs et al. (2009)	Oregon Eastern Cascades	4–5	Range 0–62,134 conifers/ha (64 plots in 4 fires); no difference among unburned, low, and moderate severity. In ponderosa forests, no ponderosa regeneration in high-severity fires and in mixed conifer limited conifer regeneration in high-severity fires
Ouzts et al. (2015)	Northern Arizona–New Mexico	7–10	Range 0–1433 conifers/ha (46 plots in 8 fires); 2 fires had 0 conifers/ha 7–10 yr after fire, 5 fires had <50 conifers/ha, 1 fire had 1500 conifers/ha; litter cover positively associated with seedlings
Crotteau et al. (2013)	California Southern Cascades	9–10	Mean 2235 conifers/ha in unburned (60 units in 1 fire), 2252 conifers/ha in low-severity, 7868 conifers/ha in moderate-severity, and 733 conifers/ha in high-severity fire; *Abies concolor* dominated regeneration over pines in more severe fire areas
Dodson and Root (2013)	Oregon Eastern Cascades	10	Mean 362 conifers/ha (18 plots in 1 fire); Range 0–1807 with 0 in 5 of the 7 plots <1000 m elevation
Collins and Roller (2013:1807)	Northern California Sierra	2–11	Omitting plots with post-fire management (leaving 21 patches in 5 fires), "there was no pine regeneration in over 90% of sampled patches". No significant effect from distance to unburned forest. Negative effect from shrubs, low seed production or soil moisture
Welch et al. (2016: Fig. 5)	California Sierra, Klamath, Southern Cascades	5–11	Mean about 500 trees/ha in yellow pine (246 plots in 12 fires), about 2000 trees/ha in dry mixed conifer (489 plots in 10 fires), interpolated from a bar graph. Overall across all vegetation types, not just yellow pine and dry mixed conifer, 54% of plots had 0–1 conifers, in interiors of severe burns, in dry areas, where more shrubs
Hanson (2018)	California Sierra Nevada/San Bernardino Mts.	1–12	Mean 3803 conifers/ha at ≤50 m into fire (20 plots in 7 fires), 1850 ha^{-1} at 51–150 m into fire (15 plots in 7 fires), 798 ha^{-1} at 151–300 m into fire (22 plots in 7 fires), and 336 ha^{-1} at >300 m into fire (25 plots in 7 fires). More within 50 m, but no significant difference among other distances. Percent shrub cover not correlated with density of conifer regeneration
Kemp et al. (2016)	Idaho–Montana Northern Rocky Mountains	5–13	Mean 7047–8153 conifers/ha (182 sites in 21 fires); Range 0–127,500 conifers/ha, but 5% of 182 sites had 0 conifers within 500 m; seedling presence probable if within 95 m of live seed source, especially if high basal area; fire severity little effect as most burn area was within 95 m of live trees
Owen et al. (2017)	Northern Arizona	12–13	Mean 84.1 conifers/ha in edge plots (6 plots in 2 fires), 41.4 conifers/ha in interior plots having no surviving trees within 200 m (6 plots in 2 fires); Range 13.0–153.8 conifers/ha in edge plots, 12.0–124.0 conifers/ha in interior plots. Regeneration significantly lower in interiors. Some long-distance dispersal (>300 m) found
Rother and Veblen (2016)	Colorado Front Range	8–15	Mean 37–1424 conifers/ha (302 plots in 6 fires), nearly all lower than pre-fire density, and 59% of plots had 0 conifers in 100 m^2 plot, with 83% of plots having <370 conifers/ha. Few seedlings in hot, dry lower elevations or on south-facing slopes, more seedlings within 50 m of live seed source, also in more southerly locations with summer rainfall
Ziegler et al. (2017)	South Dakota, Northern Colorado	11–15	Mean 43.0 trees/ha (18 plots in 3 fires)
Foxx (1996)	Northern New Mexico	0–16	Two sites in 1 fire had no seedlings in year 1, 0 and 210 trees/ha in year 8, and 218 and 318 trees/ha in year 16
Haffey (2014)	Arizona–New Mexico	6–16?	Only 24% of plots (179 plots in 9 fires) had ponderosa pine regeneration; within 150 m of a seed source, 38% of plots had tree regeneration; no regeneration beyond 250 m from a seed source. Nearly half of ponderosa pine seedlings were near a nurse structure, most often a log or large branch
Roccaforte et al. (2012)	Arizona	1–18	Range 0–11,234 conifers/ha (399 plots at 14 sites in 11 fires); 8 sites had 0 conifers/ha 1–12 yr after fire, 3 sites had 37–74 conifers/ha, 2 sites had 297–336 conifers/ha, and 1 site had 11,234 conifers/ha. Deciduous regeneration was dominant at all but 2 sites

(Table 9. *Continued*)

Author(s)	Location	Years after fire	Post-fire seedling/sapling density (trees/ha)
Chambers et al. (2016)	Colorado Front Range	11–18	Mean 225 trees/ha (305 plots in 5 fires) across unburned, low, and moderate severity. Mean tree density lowest in high severity (118 trees/ha) and in only 25% of plots, whereas 60% of low- to moderate-severity plots had regeneration. Regeneration greatest at high elevations and adjacent to unburned, declining to 10 conifers/ha at 200 m
Shatford et al. (2007)	Southern Oregon–Northern California	9–19	Mean 1694 trees/ha (24 plots in 8 fires); Range 83–8188 trees/ha. Plots showed a wide range from immediate and rapid regeneration to slow and constant to chronically limited. No significant effect of distance from seed source on seedling density; up to 84–1100 trees/ha >300 m from a seed source. Positive effect of shrub and hardwood cover
Guiterman et al. (2015)	Northern New Mexico	20	Mean 11 conifers/ha (10 plots in 1 fire); conifers present in 4 of 10 plots; maximum distance from a ponderosa seedling to unburned edge was 77 m
Rother and Veblen (2017)	Colorado Front Range	8–23	Ponderosa pine and Douglas-fir regeneration was concentrated in years with especially high growing season precipitation (413 dated seedlings at 10 sites in 5 fires); for all sites combined, three years (1995, 1998, and 2009) in twenty-four (1988–2011) accounted for most of the post-fire regeneration. Regeneration lags after the 5 fires were 0–4 yr
Passovoy and Fulé (2006)	Northern Arizona	3–27	Range 0–1052 conifers/ha (210 plots in 7 fires). Four of seven fires in years 4–8 had <50 conifers/ha and one had 26 conifers/ha at year 27, the other two fires had 170 conifers/ha and 1052 conifers/ha in years 4 and 9, respectively
Haire and McGarigal (2010)	Northern Arizona–New Mexico	28, 45	Little within years 1–8 (68 plots in 1 fire) or 1–15 (79 plots in 1 fire); ~8000, 2000 trees/ha near low-severity edge; most within 200 m of low-severity edge, but some to 304 m, 410 m; could reach all of fire area within ~50 yr
Savage and Mast (2005)	Northern Arizona–New Mexico	25–54	Regeneration began within 1–2 yr at 7 sites, within 6–10 yr at 3 sites (300 plots in 10 fires); 5 sites <200 trees/ha, 5 sites >400 trees/ha
Savage et al. (2013)	New Mexico	47–64	Regeneration did not begin for 3–20 yr (5 fires); Range (from 150 plots in 5 fires) per fire: 96–443 adult conifers/ha (≥1.4 m height and >6 cm dbh), 94–1629 seedlings and sapling conifers/ha for a total of 201–2112 trees/ha

Note: ? indicates that the Years after fire entry is uncertain.

historically over 20–30% of dry-forest areas in Oregon, California, and part of northern Arizona (Baker and Williams 2015); (2) dense younger established forests were historically common in nearly all dry-forest landscapes, suggesting past regeneration had been successful and dense (Williams and Baker 2012*b*); and (3) very dense post-fire trees are shown here to have covered large area on the southern Mogollon Plateau in northern Arizona (Fig. 4a) and occurred in the understory of burned forest in the Jemez Mountains, New Mexico (Fig. 4b). Dense and even very dense regeneration, in general and after high-severity fires, was within the historical range of variability in dry forests.

Some also considered poor regeneration after high-severity fires to indicate potentially unnatural type conversion of forests to shrublands or grasslands (Savage and Mast 2005, Haffey 2014), possible indicators of emerging tipping points (Reyer et al. 2015). However, of 24 studies, 21 (88%) covered only up to 27 yr after high-severity fires (Table 9). In general, 27 yr is insufficient, as historical tree regeneration after high-severity fires in dry forests could extend over periods of up to 60 yr (Table 4). Some large areas could even lack regeneration for ≥50 yr (Table 4) in part because of few climatically favorable periods for tree regeneration (Savage et al. 1996, Rother and Veblen 2017). A way to offset insufficient post-fire records is to extrapolate spatially (Haire and McGarigal 2010), but this has not generally been done (Table 9). Evidence is insufficient to conclude that post-fire tree regeneration is outside historical variation.

Historical tree regeneration after high-severity fires in dry forests failed or was slow at times, creating forest openings (Tables 4, 5), but recent studies often did not show modern failure was outside historical variability (Lauvaux et al. 2016). Opening creation by high-severity fire is likely operating at or below historical levels, since high-severity fires are at or below historical rates in dry forests (Baker 2015). Some openings have declined (Coop and Givnish 2007); thus, creation of new openings by high-severity fires is likely restorative (Baker 2017*b*, Boisramé et al. 2017). Openings also enhance resistance to fire spread (Boisramé et al. 2017, Owen et al. 2017) and increase the heterogeneity of landscape structure (Kaufmann et al. 2003), enhancing resistance and resilience (Table 3); thus, added openings in the transition are generally beneficial.

In contrast, tree regeneration after beetle outbreaks and droughts is not currently thought to be declining, because advance regeneration continues. In the multi-decadal period that background tree mortality increased in dry forests as temperatures rose, tree recruitment was unchanged (Van Mantgem et al. 2009). In beetle outbreaks, (1) 75% of trees <20 cm dbh survived (McCambridge et al. 1982), (2) 77% of trees <7.5 cm dbh and 58% of trees 7.5–15 cm dbh survived a severe outbreak (Klenner and Arsenault 2009), and (3) >95% of trees survived in stands with <18 m^2/ha of basal area, about 170 trees/ha in trees up to 37 cm dbh (Graham et al. 2016).

Future regeneration of dry-forest trees in general, not just after disturbances, was projected. Dobrowski et al. (2015) modeled the recruitment niche of 10 dry-forest trees relative to minimum temperature, evapotranspiration, and climatic water deficit. They then projected recruitment prevalence across the West through A.D. 2100 under RCP 8.5 (high emissions) and found recruitment declines of only about 10% or less (estimated from graphs) by A.D. 2060, at the end of the transition. Petrie et al. (2017) modeled climatic effects on stages in ponderosa regeneration (Feddema et al. 2013, Savage et al. 2013) and then projected future conditions with a water-balance model under RCP 4.5 and 8.5. Regeneration potential would be increased by +50% $\pm$ 106%, at 47 sites across the West by A.D. 2020–2059, from more flowering, seed production, and germination, especially in Arizona, Colorado, and New Mexico. After A.D. 2060, at the end of the transition, tree regeneration would decline, due to lower seedling production and survival, especially in the Pacific Northwest (−67%), but less so in the Intermountain region (−29%).

In summary, background rates of tree mortality are increasing in dry forests, and major recent droughts and beetle outbreaks have killed many trees. Recent droughts and beetle outbreaks together account for perhaps 3–4 times as much tree mortality as do moderate- to high-severity fires. Together, natural disturbances could cause tree mortality over 26–51% of dry forests in the transition. Tree regeneration is not apparently outside historical variability and is projected to only slightly decline or even increase. Some opening creation from tree mortality followed by tree-regeneration failure could actually restore grasslands and other openings. Current dry-forest area is not all at risk, as 1/2 to 3/4 could escape substantial mortality under committed warming, and the remainder could have more resistant and resilient forests that persist more than expected.

Transitioning Dry-Forest Landscapes

Large, infrequent disturbances that will enact tree mortality during the transition are capable of rapidly affecting millions of hectares and are generally beyond control. The spatial extent (25.5 million ha) of dry-forest landscapes and associated human communities and infrastructure provides large inertia for preparations. Our ability to control LIDs by manipulating forest structure is limited, and structurally ideal or restored landscapes may help, but a broader tie-in strategy, with a refocus on bet-hedging to enhance resilience to natural-process management may be more feasible and effective.

Limited ability to directly prevent LIDs or reduce their impacts on dry forests in the transition

Our ability to directly prevent LIDs or reduce their impacts is limited. Graham et al. (2016) reviewed the long history of failed attempts at controlling bark beetles through direct suppression or indirect manipulation of forest structure. At best, evidence suggests thinning, the most common manipulation, might modify the extent and pattern of tree mortality over limited area. Fettig et al. (2014) found thinning treatments to reduce tree mortality from MPB were costly and did not work during outbreaks without added direct control; thinning worked in some cases in ponderosa pine forests but had no significant effect in others. Six et al. (2014) also found thinning could possibly work at times, but failures occurred during outbreaks, and unthinned stands may actually have more survivors. Droughts are not directly controllable. Some drought treatments aim to protect particular trees by reducing competition (McDowell and Allen 2015), but this will likely ultimately fail under hotter droughts (Bennett et al. 2015). Fuel treatments to reduce fire spread and severity have also not been very effective: "Mechanical fuels treatments on U.S. federal lands over the last 15 yr (2001–2015) totaled almost 7 million ha. . .but the annual area burned has continued to set records" (Schoennagel et al. 2017:4586). Schoennagel et al. (2017) explained that treatments can reduce fire severity and increase low-severity fire in some dry forests, but the probability of having an effect is low, as only about 1% of treatments actually experience wildfire each year. Thinning treatments have been ineffective for LIDs in dry forests, in general, and are best as short-term, small-area holding actions (Six et al. 2014).

Can ideal or restored landscapes discourage LIDs from crossing tipping points?

Evidence that ideal or restored landscapes can discourage tipping points is also limited. To maintain MPBs in an endemic condition, discouraging an outbreak, Graham et al. (2016:157) suggested, based on high tree survival in an outbreak: ". . .heterogeneous landscapes composed of stands with heterogeneous structures and containing densities in the neighborhood of 80 feet^2 [18.3 m^2/ha] of basal area are resistant to MPB infestations. . ." However, they said forests in the late 1800s were dominantly in that condition when the largest known MPB outbreak in ponderosa pine forests occurred, the 200,000- to 300,000-ha 1895–1909 outbreak in the Black Hills. Thus, ideal landscapes might only be resistant to some beetle outbreaks. Lundquist and Reich (2014:472) said: "Existing models show that diverse composition and configuration is the best and possibly only long-term, large-scale approach to bark beetle management. . ." For droughts, ideal stands and landscapes have not emerged, and there is little historical evidence. For wildfires, low-density stands with large, old ponderosa pines and few understory trees and shrubs are most resistant and resilient to subsequent wildfires (Allen et al. 2002). However, probabilistic studies (Table 2) have shown this structure was a significant, but not dominant component of most historical dry-forest landscapes, which had more heterogeneous stands across heterogeneous landscapes (Table 2). Thus, historical and ideal landscapes appear congruent, and achievable through restoration, for droughts and beetle outbreaks, and at least partly for fires.

Idealized and historical stand and landscape structures are unlikely to prevent LIDs from causing substantial tree mortality, some tree-regeneration failures, and some opening creation, as these were natural components of historical processes of disturbance and recovery in dry forests. Large, infrequent disturbances occurred in historical dry-forest landscapes and led to substantial landscape change and large fluctuations. Dry-forest landscapes appear to have been capable of general recovery after LIDs (Table 6), but some nonforest, created by disturbance, persisted for 100–150 yr or more (Table 5). Whether tipping points were crossed or this simply represents slow natural recovery is uncertain, but in either case dry-forest landscapes were dynamic and subject to large fluctuations that created and renewed resistance and resilience features that fostered bet-hedging (Table 3).

Natural fluctuation means that restoration and management in dry forests are less a matter of restoring and managing forest structures (Table 3) and more a matter of restoring and managing natural disturbance and recovery processes. Most structures are inherently ephemeral, persisting for only years or decades, and are quickly recreated by disturbances, and thus do not warrant intentional restoration. Widespread micro-management of fuel loads and forest structures after LIDs, based on fears of hypothetical mass fires (Stephens et al. 2018), is likely a waste of resources, because extensive structure management to reduce severe fires has been ineffective (Schoennagel et al. 2017). However, old trees and their associated stand- and landscape structures could persist for centuries, are not recreated by disturbances, and have been lost to excessive logging. Structure restoration and management make sense for these long-persisting structures not created quickly by disturbances, but process management, and associated facilitative structures (e.g., bet-hedging) now make sense for most landscape restoration and management.

A tie-in strategy using bet-hedging and process management of disturbances in the transition

Given substantial uncertainty and limited ability to control LIDs, a broad tie-in strategy, using actions beneficial for people and nature no matter what occurs, could likely facilitate more forest persistence in the transition. Suggested actions include (1) refocusing intentional ecological restoration on bet-hedging using historically congruent structures that provide resistance and resilience to diverse future disturbances (Baker and Williams 2015), (2) expanding development of fire-safe landscapes for people and infrastructure (Schoennagel et al. 2017), (3) expanding managed fire, and (4) accepting that LIDs will beneficially revise resistance, resilience, and genetic adaptation (Six et al. 2014). Restoring forest structure is costly, and resistance structures may fail, favoring structures that facilitate more process-based restoration (Millar et al. 2007).

Droughts and beetle outbreaks are likely to be 3–4 times as important as fires during the transition, which means that abundant small trees and high tree species diversity are now the more important resistance and resilience structures for transitioning dry forests (Table 3). Most large restoration programs (Reynolds et al. 2013, Addington et al. 2018) are likely to be ineffective, as they are focused on structures resistant to fire, when it is more likely that drought and beetles will determine the structures that persist in the transition. These programs to thin forests to resist damage by moderate- to high-severity fires have unfortunately reduced the small trees and diverse tree species that most provide resilience to droughts and beetle outbreaks. These programs could be quickly modified to instead retain small and diverse trees. In forests already deficient in small and diverse trees, if only one prescribed fire occurs before managed wildfire for resource benefit ensues, that last fire will likely stimulate some tree regeneration to repopulate small trees. If low-severity fires are generally managed to mimic historical spatial and temporal variability, opportunities will likely occur for diverse trees to repopulate (Baker 2017*a*).

The unpredictability of future disturbances suggests hedging bets (Millar et al. 2007, Baker and Williams 2015) in stand-level restoration by maintaining large and small trees and available tree species diversity. After restoration, most stands, even open low-density stands, can have numerical dominance by small trees of all available species, but also sufficient replacement larger trees of all available species. Early land surveys across 1.7 million ha of dry forests showed small trees (typically <40 cm dbh) were, on average, 62% of total trees (Baker and Williams 2015). Given loss of large trees to logging, retaining all large trees, and mid-sized trees that are their future replacements, is sensible. After disturbances, successful tree regeneration is favored by large surviving trees that provide seed within about 100–200 m (Table 9). Larger trees may later be lost to hotter droughts and beetle outbreaks. However, if there were 20–50 larger (>40 cm dbh) trees per ha, and >5% survived, that could provide needed surviving large trees. Bet-hedging in restoration leaves abundant trees of all species and sizes with small trees dominant.

At the landscape scale, diverse historical forest structures could reduce the spread and effects of natural disturbances (Table 3) and bet-hedging at this key scale of LIDs is very important now. For fires, areas of large fire-resistant trees, openings, and naturally moist areas or shaded fuels provide resistance and favor survivors that aid post-fire resilience. For beetle outbreaks and droughts, diverse tree species and smaller trees provide the most important resistance and resilience. Recovering younger to middle-aged forests were common historically, based on studies in Table 2, and naturally conferred resistance and resilience to beetles and droughts. Kautz et al. (2017:534) found that "...more than 60% of global forests are in various stages of recovery from a past disturbance at any given time." Protecting young, naturally recovering forests is thus feasible, congruent with historical forests, and a key landscape part of a process-restoration approach (Baker 2017*b*). Young forests can survive beetle outbreaks and possibly droughts at much higher rates than older forests (Graham et al. 2016). To maximize bet-hedging, mixtures of diverse resistance and resilience structures across landscapes, with much more focus on beetle outbreaks and droughts, in addition to fire, are now more congruent with expected LIDs.

It would benefit both people and nature to rapidly increase protection of infrastructure, homes, and communities from increased wildfires, and this would also enable more managed use of natural disturbances. With ~7 million ha of fuel-reduction treatments, but fires still burning homes (Schoennagel et al. 2017), we need to prevent the expansion of developments into fire-prone settings and finish full fire protection around all homes, infrastructure, and communities. Effective ways to reduce vulnerability and live with wildfire have been articulated (Cohen 2000, Baker 2009, Calkin et al. 2014, Moritz et al. 2014, Smith et al. 2016, Schoennagel et al. 2017). Tools include fire-safe construction, zoning, building codes, incentives, easements, growth boundaries, insurance policies, and other means (Kennedy 2006, Baker 2009, Schoennagel et al. 2017). Homeowners can use fire-safe construction focused on the home-ignition zone (Cohen 2000). Possibly most effective is for communities and developments to designate growth boundaries that enclose a wide margin of open, fire-resistant land uses that can serve as an effective fire break (e.g., ball fields, wetlands, irrigated agricultural fields), whether they are already in place or require construction. This alone would definitively stop expansion into fire-prone vegetation, protect key concentrations of people and infrastructure from fire, and make it more feasible to manage wildfires for resource benefit on adjoining public lands (Baker 2009).

Among LIDs, using more managed fire for resource benefit would be effective wherever it is safe and feasible, especially in the early part of the transition. Moderate- to high-severity wildfire has the longest recent rotation (Table 8) and is the only LID that can be directed. Prescribed fires are typically not sufficiently intense for effective restoration (Van Wagtendonk and Lutz 2007, Baker 2014), but prescribed burning once across landscapes and near homes and infrastructure is best before initiating managed fire (Baker 2017*a*). Managed wildfires can accomplish more renewal and enhancement of resistance and resilience, and also help prepare communities for future LIDs (Schoennagel et al. 2017). Expanding managed fire is scientifically supported (North et al. 2015, Schoennagel et al. 2017), and solutions to institutional barriers are identified (Stephens et al. 2016). Managed fires early in the transition are especially important to reduce tree density and basal area, which can lower vulnerability to droughts and beetle outbreaks more likely with higher temperatures later in the transition. Early managed fires could also stimulate tree regeneration, when it is favored (Petrie et al. 2017). Recovering small trees and entire stands recovering after fires provide resilience to droughts and beetles and foster asynchrony in tree populations that can slow disturbance spread (Millar et al. 2007, Seidl et al. 2016). If openings or low-density patches are created by early disturbances, those could also reduce later vulnerability. Openings are less likely to ignite (Baker 2009), may slow fire, and could hinder beetle spread.

Acceptance of the benefits of LIDs and protection of the post-LID environment are sensible, since we cannot prevent LIDs in the transition. For example, bark-beetle outbreaks may naturally thin and diversify forest structures (Oliver 1995, Graham et al. 2016), updating resistance and resilience, while increasing biodiversity and furthering genetic adaptation to emerging climates and LIDs (Six et al. 2014, Beudert et al. 2015). Large, infrequent disturbances also provide selection against individual trees not resistant to the LID or post-LID environment (Six et al. 2014). Survivors and post-disturbance regeneration can revise tree adaptations to both emerging climate and patterns of LIDs. Rapid evolutionary response to extreme climatic events is possible, even in long-lived trees (Grant et al. 2017). For example, MPB outbreaks favor survival of slower-growing ponderosa pines, even though faster-growing trees may outcompete them at other times (De la Mata et al. 2017). Also, since post-LID tree regeneration is favored within 100–200 m of surviving trees (Table 9), and LIDs can leave isolated patches of surviving trees that, by chance, have different gene frequencies, the opportunity for locally adapted genetic change is high. As Howe (1976:263) said: "Prevention of major conflagrations... would eliminate the ingredients for drift, i.e., the replacement of large, continuous populations by tiny islands of isolated interbreeders from which most ensuing regeneration would emanate..." To preserve genetic adaptation of trees to emerging climate and LIDs, it is important to not prevent LIDS, not plant trees, and not log post-disturbance survivors (Lindenmayer et al. 2008). Genetic adaptation to committed warming could enhance possibilities for more dry-forest persistence in the transition and during the extended period of adjustment after the initial transition to committed warming.

Conclusions

Limiting warming, as with the Paris Agreement, should enable more persistence of current dry forests in the transition to committed warming than projected by models. Here, I reviewed evidence that (1) LIDs historically produced diverse forest stands and landscapes that naturally provided resistance and resilience to subsequent disturbances; (2) LIDs cannot be generally prevented through direct control or indirect manipulation of forest structure; (3) fires, droughts, and beetle outbreaks are not yet having effects in dry-forest landscapes that appear outside historical variability; (4) in the last few decades, droughts and beetle outbreaks have caused roughly 3–4 times as much tree mortality as fires; (5) primary opportunities to enhance forest persistence are from expanded bet-hedging at stand and landscape scales focused on resistance and resilience to droughts and beetle outbreaks, and facilitating adaptation as disturbances occur; and (6) 1/2 to 3/4 of dry-forest area could possibly escape most mortality during the transition.

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RESEARCH ARTICLE

Restoring and managing low-severity fire in dry-forest landscapes of the western USA

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Abstract

Low-severity fires that killed few canopy trees played a significant historical role in dry forests of the western USA and warrant restoration and management, but historical rates of burning remain uncertain. Past reconstructions focused on on dating fire years, not measuring historical rates of burning. Past statistics, including mean composite fire interval (mean CFI) and individual-tree fire interval (mean ITFI) have biases and inaccuracies if used as estimators of rates. In this study, I used regression, with a calibration dataset of 96 cases, to test whether these statistics could accurately predict two equivalent historical rates, population mean fire interval (PMFI) and fire rotation (FR). The best model, using Weibull mean ITFI, had low prediction error and $R^2_{adj} = 0.972$. I used this model to predict historical PMFI/FR at 252 sites spanning dry forests. Historical PMFI/FR for a pool of 342 calibration and predicted sites had a mean of 39 years and median of 30 years. Short (< 25 years) mean PMFI/FRs were in Arizona and New Mexico and scattered in other states. Long (> 55 years) mean PMFI/FRs were mainly from northern New Mexico to South Dakota. Mountain sites often had a large range in PMFI/FR. Nearly all 342 estimates are for old forests with a history of primarily low-severity fire, found across only about 34% of historical dry-forest area. Frequent fire (PMFI/FR < 25 years) was found across only about 14% of historical dry-forest area, with 86% having multidecadal rates of low-severity fire. Historical fuels (e.g., understory shrubs and small trees) could fully recover between multidecadal fires, allowing some denser forests and some ecosystem processes and wildlife habitat to be less limited by fire. Lower historical rates mean less restoration treatment is needed before beginning managed fire for resource benefits, where feasible. Mimicking patterns of variability in historical low-severity fire regimes would likely benefit biological diversity and ecosystem functioning.

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Introduction

Low-severity wildfires significantly shaped dry forests in the western USA, but historical rates (e.g., mean interval, area burned) of these fires remain uncertain in a time of altered and further changing fire regimes. Low-severity fires periodically burned the understory of historical dry forests, changing fuel loads, composition, diversity, and ecosystem processes without killing most canopy trees [1–2]. Dry forests in the western USA cover 25.5 million ha and include dry pine forests, dominated by ponderosa pine (*Pinus ponderosa*) or other dry pines, and dry mixed-conifer forests that also have firs (*Abies concolor*, *A*. *grandis*, *Pseudotsuga menziesii)* and other trees [3]. Past reconstructions of low-severity fire in dry forests, using tree-rings, focused on long records of dated fire years in small plots, and most were not intended to accurately estimate key rate parameters of low-severity fire [1–2] needed to restore and manage low-severity fire across large landscapes. These small-plot reconstructions have known inaccuracies and biases if inappropriately used for this purpose [1, 4–11]. Fortunately, new landscape-scale and small-plot reconstruction methods [1, 11] overcome many known inaccuracies and biases in estimating historical low-severity fire rates, but limited new estimates are available.

This situation leaves a weak current basis for restoring and managing low-severity fire, using historical rates as a guide, across dry-forest landscapes. Here I: (1) develop regressions for estimating mean historical rates of low-severity fire from past reconstructions using a calibration dataset, and then (2) apply these regressions to estimate mean historical rates of low-severity fire for a large dataset of past reconstructions across the western USA, and (3) assess the applicability of these new estimates across dry-forest landscapes. These new estimates are directly usable in restoring and managing low-severity fire in the parts of dry forests of the western USA where low-severity fire was historically predominant, and provide a West-wide perspective on variability in historical mean rates of low-severity fire in these parts of dry forests. As discussed later, variability around mean rates is also an essential attribute of a low-severity fire regime.

Estimated mean historical rates of low-severity fire need to be fairly accurate, for restoring and managing low-severity fire, because key effects of fires on biological diversity, ecosystem functioning, and post-fire recovery operate significantly differently across a narrow range of mean rates. For example, understory fuels in dry forests, reduced by a single fire, often recover to pre-fire levels in about 7–25 years [12–14]. If mean fire intervals for low-severity fires were 10–15 years, understory fuels would often have been kept at relatively low levels, but if mean intervals were 25 years or more, then understory fuels would more often have been fully recovered and generally higher. Fires that are too frequent can reduce the ecological roles of the forest floor in replenishing soil nutrients and organic matter, enhancing absorption of water and nutrients, and providing habitat for microbial communities, potentially reducing long-term forest productivity [15]. Habitat for wildlife that use snags or down wood could be adversely affected by fire that is too frequent [15], which can also reduce understory plant species richness, possibly due to depletion of soil nitrogen [16]. Native shrubs, historically abundant in some dry forests, may also be reduced by fire at intervals less than about 20–30 years [17]. However, fire-stimulated shrubs in the understory of dry forests may also decline if low-severity fire rates are too low [18]. Insufficient low-severity fire can allow tree density or other understory shrubs to increase, reducing nutrient cycling and understory diversity, and increasing fire severity [16, 19].

Maintenance of tree populations in dry forests also depends on the balance between tree natality and mortality, a balance strongly shaped by rates and patterns of fires. Fire intervals for successful tree regeneration were likely long relative to historical mean intervals, as fires at short intervals can kill most small trees [6]. Patchy surface fires could alone allow survival of small trees in unburned areas [20]. Also, seedlings regenerating in openings may produce limited fuels, enhancing fire patchiness that favors seedling survival [21]. Where fire kills overstory trees, a resulting mineral seedbed and reduced competition with grass can enhance tree regeneration, if other factors (e.g., seed production) co-occur [22]. A fire-quiescent period is also needed [23]. Long intervals may occur over large areas in wet periods, or stochastically from variability in fire. In contrast, mortality of larger trees from single low-severity fires can reach 7–8%; if repeated every 10 years, larger trees could be reduced by half in a century, but, assuming the same 7–8% rate repeated every 50 years, larger trees would be halved in 500 years [17]. Thus, tree populations, both young and old trees, are sensitive to rates and patterns of low-severity fire.

Rates and patterns of low-severity fire also affect how resistant and resilient dry forests are to future fire, drought, and beetle outbreaks [24]. Open, low-density forests relatively free of shrubs and small trees can be produced by repeated low-severity fires, and may be more resistant to subsequent higher-intensity fires than are denser forests, with more shrubs and small trees [25]. Forests subject to repeated low-severity fires could even be self-limiting, if the rate of fires is high, possibly promoting continuing low-severity fire rather than higher-severity fires [26]. However, if a deficiency in tree regeneration occurs because of too-frequent fires, dry forests would be vulnerable to subsequent regeneration lags or failures after droughts and beetle outbreaks that are a higher current risk than are severe fires [24]. Too little low-severity fire could increase fire severity, but too much could reduce higher-severity fires that enhance spatial heterogeneity, a key source of forest resilience to future disturbances [3].

Research has enhanced understanding of the importance of rates and patterns of low-severity fire to biological diversity, ecosystem functioning, and sustainability of dry forests, but estimated historical rates and patterns of low-severity fire remain uncertain. Newer methods for accurately reconstructing rates of historical low-severity fire promise to eventually resolve uncertainty, but improved estimates, the focus here, might be possible from past research.

Measures and estimators of mean rates of low-severity fires

Terms and measures

A *low-severity fire* in this study is a fire that burns in the understory of a forest, and is often defined as causing mortality or topkill of no more than about 20% of stand basal area [27–28]. These fires are not usually burning in the canopy independently, instead torching upwards from surface fuels into single or small groups of trees. These fires could also be called low-moderate severity to reflect some canopy mortality, but the extent of canopy mortality from these fires is poorly known [17].

Several measures of mean rates of fire also need explanation. At a point in a landscape, the average interval between fires is the point *mean fire interval* (point MFI). The average MFI across multiple points in a landscape provides a sample estimate of the *population mean fire interval* (PMFI) for a particular landscape, which is the grand mean fire interval across the landscape [6]. Fire-interval data at points have interval distributions that often are skewed, not normally distributed. Alternative measures of central tendency, such as the median, can characterize these distributions. These distributions often can also be fit by the flexible two- or three-parameter Weibull distribution, which has a shape parameter that describes the form of the distribution (e.g., lognormal), a scale parameter that represents the 63rd percentile of the distribution, and a shift parameter to set the location of the distribution [29]. The mean and median of the fitted Weibull distribution, which can offset unusual values in actual data [29], are useful alternative measures of central tendency. Descriptors of variation (e.g., standard deviation) are relevant for all measures. The *fire rotation* (FR) is the expected time for fire to burn an area equal to the area of a landscape of interest [17]. The FR for a landscape is equivalent to the PMFI, which was shown analytically [6] and through simulation [7–8]. Fire-interval data at points can be used to estimate the PMFI, or area-burned data across a landscape can be used to estimate the FR. PMFI estimates at points and FR estimates across areas are the fundamental, equivalent estimators of mean rates of fire, as they show how often points experience fire and the equivalent time it takes for fire to burn across a landscape.

Estimators of the Population Mean Fire Interval (PMFI)

For reconstructions of mean low-severity fire rates in the pre-EuroAmerican period, which are predominantly derived using tree-ring and fire-scar methods, the actual intervals needed for estimating PMFI can be sampled and processed in several ways. Fires do not physically leave a scar on every tree that burns [30], and the scarring fraction (SF), the fraction of live trees that receive a scar from a fire, may be moderate or even low. The intervals derived from scarred trees are thus simply estimators of the actual fire intervals that occurred at a point.

The most widely used fire-interval estimator is the mean composite fire interval (mean CFI), often also called the mean fire-return interval (MFRI) or even, to confuse matters, the MFI itself, which is not the estimator but instead what is being estimated. This estimator seeks to offset the fact that SF is < 1.0 by compositing scar records across a set of nearby trees, which together are expected to contain a more complete record of fires that burned the point. To calculate mean CFI, the user creates a pooled "composite" list of fire years that burned any tree in a set of sample trees, then the estimated intervals are those between fire years in the composite list. However, this composite list of all fires may contain small spot fires that have little ecological effect, and users often also report estimates for larger fires that scarred more than 10%, 25%, or another percentage of scarred trees. Various measures of central tendency can be calculated, including the mean, median, and Weibull measures. I distinguish variants here using combined terms, such as mean CFI-all fires, mean CFI-10% scarred, or median CFI-25% scarred. Mean CFI-10% scarred, for example, is the mean composite fire interval for fires recorded on $\geq$ 10% of scarred trees.

Another commonly used estimator is the mean individual-tree fire interval (mean ITFI). This estimator is calculated in two steps. First, the intervals between fires on an individual scarred tree are used to estimate the MFI for that tree. Second, the grand mean of each tree's estimated MFI is calculated across a set of sample trees. In this case, restrictions (e.g., 25% scarred) are not used, but alternative measures of central tendency are, so there are fewer variants.

Finally, we developed an estimator, the mean all-tree fire interval (mean ATFI), which seeks to offset SF < 1.0 by using an estimated SF to predict the total number of scars that would have occurred if SF was 1.0 [7–8, 11]. This estimator has been shown to be the best available estimator of PMFI [11], but it is not used in this paper because few ATFI estimates are currently available.

Estimators of the Fire Rotation (FR)

Area-burned estimates for calculating FR can be derived from three main sources: (1) area burned in recent fires from agency polygon fire records or fire-atlas records or from remotely sensed data, (2) historical area burned from fire-year maps reconstructed from scarred-tree or plot locations, or (3) historical area burned reconstructed using a ratio method and scarred-trees or plot records, or comparable data in a table or graph.

Polygon fire records or fire-atlas records are available from public land-management agencies, and are most complete and accurate after about A.D. 1980. Early data are often from fire perimeters sketched on a map, but later data may have been from remotely-sensed data [31]. Small fires were not always mapped. Accuracy of boundaries of fires in fire-atlas data, relative to tree-ring reconstructions and remote-sensing data, was moderately high in one study, sufficiently accurate to use in some research [31]. In another study, tree-ring methods underestimated fire extent relative to fire-atlas maps, which also had some errors [32]. A larger study showed closer agreement between fire-atlas data and tree-ring reconstructions of fires [1].

Fire-year maps are typically reconstructed from tree-ring and fire-scar data collected at a grid of points or a set of random points. Fire scars near the points are dated, dates are displayed on a map or in GIS, and a fire perimeter is placed around the points common to a fire year [33–34]. The boundary is positioned using a set of fire-spread principles [35], Voronoi polygons centered on the points [1], convex hulls [32], fuzzy-set methods [36], inverse-distance weighting [23, 33], or indicator-kriging [33–34]. If grid points are close, unburned area may be most accurately mapped, but a larger grid spacing is often needed to allow sufficient area to be sampled, leading to less precision in boundaries and unburned areas [34]. Smaller fires also will be missed more often with larger grid spacing. Larger fires that contribute most to fire rotation are mapped the best. Fire rotation has been shown to be estimated within about 10% of the value obtained from fire-atlas data [1, 11].

A non-spatial ratio method estimates area burned within a study area as proportional to the percentage of sample trees scarred in a particular fire year or the percentage of plots in which a particular fire year is recorded on sample trees. The equation [37] is:

$$A_i = (AT^* NS_i)/(NST - NRE) \quad (1)$$

where A_i is area burned in year i, AT is the study area size, NS_i is the number of scarred trees or plots recording a fire in year i, NST is the total number of scarred trees or plots, and NRE is the number of scarred trees or plots eliminated by subsequent fires. This method is most accurate when the number of scarred trees or plots is large and these are well distributed across a sample area [1, 37]. However, scarred trees are often clustered [30], which could lead to ratio estimates that are biased and too short. Because the location of scarred trees or plots is not used, unburned area may also be underestimated. In a large modern corroboration study, the ratio method accurately estimated area burned of larger fires (> 100 ha), that accounted for 97% of total area burned, and fire rotation from total plots was 89% of fire rotation from fire-atlas data [1].

FR can be calculated, using any of the three sources of data, by the equation [17]:

$$FR = (ObservationPeriod/FractionBurned) \quad (2)$$

where FR is fire rotation, in years, $ObservationPeriod$ is the period, in years, for which there are mapped or reconstructed records of fire, and $FractionBurned$ is the fraction of the study area estimated to have burned during the observation period, obtained by summing the areas of fires or the estimated fraction burned from ratio estimates.

Perspectives on estimating PMFI/FR and interpreting mean CFI

A central area of analysis and discussion by our research group has been about whether past mean CFI and ITFI estimates from small plots accurately estimate PMFI/FR. Other studies (e.g. [38]) were more focused on reconstructing a long history of dated fire years across a network of locations, not so much accurate rates of fire across landscapes. I continue the rate focus here. An earlier review suggested mean CFI is too short and mean ITFI is too long as an estimator of PMFI/FR [6]. This study suggested mean CFI was often too short from compositing across too much area or samples and mean ITFI was too long, as it does not offset unrecorded fires that occur because SF is < 1.0 [6].

Reflecting a need for rate estimates, some studies mostly used mean CFI as comparable to, or effectively an estimator of FR [39–40]. Others also used historical median CFI as an estimator of historical FR [41]. Another compared estimated median CFI, ITFI, and FR, found median ITFI was closest to FR, and suggested median ITFI might be used to estimate FR in low-severity fire regimes [42]. In contrast, other studies suggested fire scars provide estimates of the PMFI/FR that are generally too long: ". . . our findings clearly demonstrate that analysis of fire scars will likely underestimate past fire occurrence" ([10]:1500). However, when compositing fire-scar records over larger areas and more trees, mean CFI declines toward 1.0, a fire every year [1, 43], an estimate of PMFI/FR that is nearly always too short. Given uncertainty about estimators of low-severity fire rates, some studies suggested that summary statistics, such as mean CFI or FR, should not even be used in restoring and managing low-severity fire (e.g. [44]).

Other studies suggested that multiple descriptors of fire regimes (i.e., including mean CFI) are desirable (e.g. [1]). Studies, that favored mean CFI and ITFI as one of multiple statistics, suggested they must be interpreted correctly. For example, regarding mean CFI-all fires, one study said it was not designed to estimate area burned, and if it does not, that is not a problem in mean CFI, but an error in interpreting it [1]. Other studies also suggested it is a problem if mean CFIs are interpreted as indicating how often the entire stand burned ". . . since fires are quite variable in burn patterns" ([2]:1091). Similarly, other studies suggested managers need to recognize that fires indicated by mean CFI burned in variable spatial and temporal patterns, including unburned areas [45]. A study in California said: ". . . the composite MFIs are not equivalent to average point fire intervals, population means [sic] fire intervals or natural fire rotation. They are an estimation of average intervals between fires of any size, or of an estimated size class, occurring anywhere within a study area" ([46]:52). That mean CFI declines with increasing sampling area is also interpreted by some not as a fundamental flaw [6], but instead as an added descriptor of a fire regime [47–49]. Complex power-function patterns across spatial scales, observed as mean CFI declines toward 1.0 with more samples, are thought in this study to elucidate cross-scale spatial properties of fire regimes. Thus, ". . . measures of fire frequency are area dependent, and . . . fire return intervals cannot be described by a single number independent of spatial scale" ([48]:820). However, scale-dependent values are only known for CFI measures, not other rate measures. In summary, there is now general agreement that mean CFI and its variants (e.g., median CFI) and ITFI are not intended to estimate the PMFI/FR. Mean CFI is accepted to not indicate area burned, the pattern of the fire, or PMFI/FR.

Accurate estimators of the PMFI/FR are still needed. Fortunately, recent modern calibrations have validated new methods for estimating PMFI and FR that do not need to use mean CFI or ITFI and have promising accuracy [1, 11]. However, it may be decades before better estimates from these new methods become sufficiently common to be able to guide restoration and management of low-severity fire. In the meantime, past mean CFI and ITFI plot estimates are abundant, and required large efforts to gather and process. Moreover, plot data on fire history likely will remain a fundamental sampling component of spatial fire histories, and could provide detail about spatial variability in FR and MFI across landscapes. Mean ITFI is less studied; it remains unclear how it might perform as an estimator of PMFI/FR, but it may suffer from the unrecorded fire problem, so that mean ITFI may be too long [6]. Now that there are more spatial estimates of FR, further analysis of the relationships of CFI, ITFI, and PMFI/FR is warranted, to see whether a variant of CFI or ITFI may estimate PMFI/FR.

Materials and methods

I assembled two datasets for analyzing the relationships of CFI, ITFI, and PMFI/FR in dry forests of the western USA (Fig 1) using an analysis of bias and inaccuracy followed by regression analysis. I also recorded and analyzed fire-history sampling measures (e.g., number of samples) and their effects on these relationships.

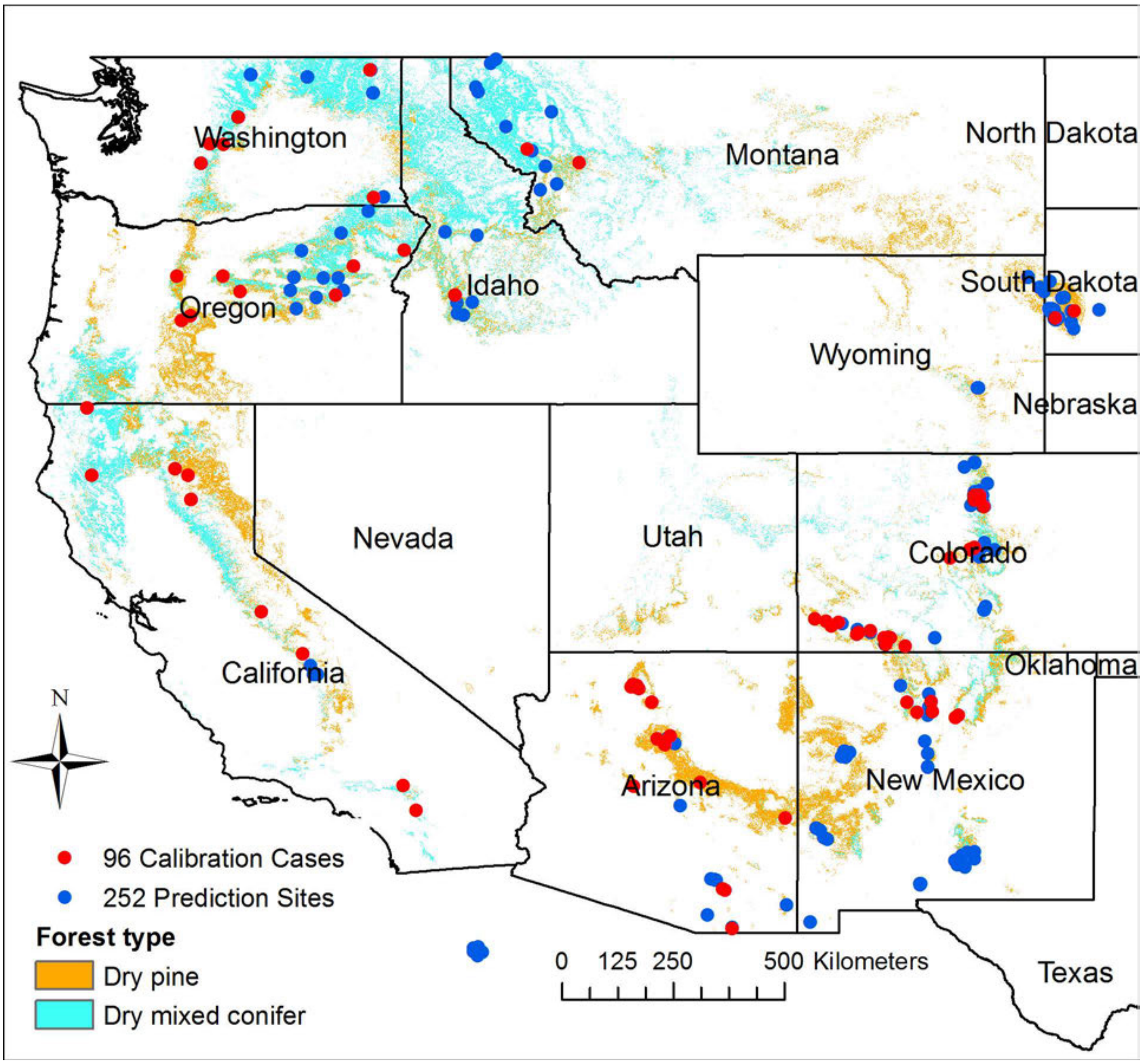

Fig 1. The 96 calibration cases and 252 prediction sites from the International Multiproxy Paleofire Database. Note that multiple plots were often done near one site, thus the number of dots is fewer.

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The 252-site fire-history dataset

To obtain a large sample of fire-history sites in dry forests to use to analyze methods and estimators in common use, I searched the International Multiproxy Paleofire Database (IMPD) (https://www.ncdc.noaa.gov/data-access/paleoclimatology-data/datasets/fire-history) for all fire-scar sites between 102˚ and 125˚ west longitude and 30˚ and 60˚ north latitude, finding 436 sites. I excluded 77 sites not clearly in dry pine or dry mixed-conifer forests. Some were also excluded because their FHX file (containing the fire-history data) in the IMPD was not usable (n = 26), the dataset was too small (n = 6) or calculations could not be completed (n = 12). I also removed 63 sites usable in a calibration dataset, described next, which left 252 sites. I left in 9 sites from Mexico and one from Canada that are nearby and relevant to the western USA.

I downloaded from the IMPD an FHX file containing fire-history records for each site and used the Fire History Analysis and Exploration System (FHAES; frames.nbii.gov) Version 2.0.2 [50] to calculate CFI and ITFI estimators. To reduce differences in the period of record, I restricted calculations for all sites to the period extending from the earliest fire to the latest fire within the A.D. 1600–1900 period. The purpose of restricting analysis to fire-to-fire periods is that scar-to-scar fire intervals are traditionally used. I did not want to introduce a possible confounding variable by using an arbitrary period. After restriction, I omitted sites with < 50 years of record, an arbitrary criterion aimed at minimizing short records.

For each case, I also recorded ancillary information, from the original publication reporting the study or from the FHX file, including the sample area, the number of sampled scarred trees, the total number of fire scars across all sampled trees, the analysis years used. and the types of targeted sampling used, including: (1) seeking the best information/longest record, (2) seeking multi-scarred trees, (3) seeking clusters of scarred trees, (4) seeking scars on dead wood, or (5) placing plots or selecting study areas in areas with many scarred trees or in old forests with long records of fire. I also recorded whether fire severity was studied, and I recorded the location of the samples found in the FHX file or publication.

The 96-case calibration and analysis dataset

To analyze the relationship of CFI and ITFI estimators and FR, I searched for and found 44 fire-history studies with 96 fire-history reconstructions and alternative calculations of fire rates in dry forests in which the study: (1) estimated CFIs and/or ITFIs and (2) in areas of at least 80 ha, also estimated FR or provided data sufficient to allow FR to be calculated from data in the paper or in an FHX file (S1 Table). The purpose of this dataset was to analyze whether CFI and ITFI estimators can predict FR. I included all sites from the IMPD, meeting the criteria defined earlier, for which sample area was given and was ≥ 80 ha, and for which there was a usable FHX file. Other sites > 80 ha were included that did not have an FHX file, but were documented in a publication. If area was reported as a range, I used the midpoint. The 80-ha minimum is an arbitrary limit to increase the area used for estimating FR. Analysis periods did not need to be pre-EuroAmerican or identical among sites, but had to have ≥ 50 years of record. If measures were not calculated in the study, I restricted analysis to scar-to-scar intervals, beginning with the first scar after ≥ 10 samples had accumulated, and ending with the last fire.

FR was calculated in the study, or by me if the study did not do this, using the previously-described area-burned estimates: (1) area burned from agency polygon fire records (n = 1) or fire-atlas records (n = 2), (2) estimates of area burned from fire-year maps reconstructed from scarred-tree or plot locations (n = 24), or (3) estimates of area burned from the ratio method and scarred-tree or plot locations (n = 63), or data in a table or graph (n = 6). For published studies, I recorded whether FR was estimated from total number of scarred trees/plots or recorders. In a few cases, this was uncertain and I recorded the most likely. A recorder is a tree scarred at least once, which increases the probability of recording fires [30]. If the study did not estimate FR, I used FHAES and Minitab to estimate FR from fire-history data in the IMPD for sites for which an FHX file was available and usable. I copied the summary table, provided in FHAES for each FHX file, into Minitab 17 [51] to do calculations. I made ratio estimates, and calculated them separately based on both total number of scarred trees and number of recorders. Sites were included more than once if different methods to calculate FR were provided in the study or could be calculated. As in the case of the 252-site dataset, I obtained and recorded ancillary information for each site.

The 342-site merged dataset

To allow calculation of histograms for particular attributes across all the sites, I merged sites in the two datasets. I removed post-EuroAmerican sites from the 96-case calibration dataset, then merged it with the 252-site prediction dataset, yielding a dataset of 342 sites (S2 Table). These include some alternative estimates from the same site or area by different studies or from using different methods, data sources, time periods or with different boundaries or other differences.

I did a rough analysis of whether sampled stands were old forests in the pre-EuroAmerican era. Old-growth dry forests are generally at least 150–200 years old, but also have attributes other than age [52], so here I call forests older than 150–200 years just "old forests." To roughly estimate the age of sampled forest stands, I used the beginning year of analysis for each stand, as defined in the study (first fire year if not). Stands with beginning years before A.D. 1700 were likely generally $\geq$ 200 years old in A.D. 1900, thus would have been old forests in the pre-EuroAmerican era. Although some could have been younger, if the oldest sample trees were not abundant, often the beginning year of analysis was defined by a minimum number of sample trees (e.g. [10]). Although imprecise, this should roughly estimate sampling in old forests. I also reviewed GLO-survey and aerial-photo reconstructions of fire severity to assess the percentage of historical landscapes with a history of predominantly low-severity fire. The GLO reconstructions use a calibrated and validated low-severity fire model [53]. The calibrated model predicts low-severity fire where historical tree density was $<$ 178 trees/ha, percentage of large trees was $>$ 29.2%, and percentage of small trees was $<$ 46.9% [53].

Can CFI and ITFI measures predict PMFI/FR?

The calibration dataset included 21 estimators of the rate of low-severity fires based on CFI, ITFI, and PMFI/FR and three sample-size variables. Sample-size variables included sample area (ha), total number of scarred trees, and scar density, expressed as total scarred trees per 100 ha (e.g. [54]). These variables are included because previous analyses found that CFI estimators were related to sample size [6]. The 21 estimators of the rate of low-severity fires included five measures of central tendency (mean, median, Weibull scale, Weibull mean, and Weibull median) for CFI-all fires, CFI-10% scarred, CFI-25% scarred, ITFI, plus the PMFI/FR based on recorders.

These 21 variables are used to individually predict PMFI/FR based on total scarred trees/plots, not based on recorders, for several reasons. Most of the best available estimates, from fire-year maps and ratio estimates using plots in a grid, are based on fires from total scarred trees in the plot. For ratio estimates from just scarred trees, recorders or all scarred trees each have strengths and limitations (S1 Text), summarized here. The use of all scarred trees is consistent with most plot-scale fire-year estimates. Recorders are two to three times less abundant than single-scarred trees, so area burned is inherently less detailed if only recorders are used, likely generally inflating area burned and shortening the estimated PMFI/FR. However, recorders do have a higher probability, than do unscarred trees, of recording a fire or of documenting it did not burn at a particular point [30]. Recorders are also multi-scarred trees, that inherently omit unscarred and single-scarred trees, that can indicate where fires did not burn, also inflating area burned and shortening PMFI/FR. PMFI/FR estimates from targeted trees (typically multi-scarred) were reduced to about 86–95% of estimates from equal-size probabilistic samples [55], supporting this expected effect. Also, about 1/3 of fires may be missed if only recorders are used [S1 Text]. More research is needed on using unscarred trees, single-scarred trees, recorders ($\geq$ 2 scars), or all scarred trees to estimate area burned, but all scarred trees likely provide the best estimates.

To understand the direction and magnitude of differences between the 21 estimators and the PMFI/FR, I calculated bias and inaccuracy for the 21 estimators relative to PMFI/FR-total scarred trees/plots for the calibration dataset. Bias is quantified by relative mean error (RME):

$$RME = \sum_{i=1}^{n} [(M_i - FR_i)/FR_i]/n \tag{3}$$

where M_i is value i of n total available estimates for CFI or ITFI estimator M of the 21 estimators and FR_i is the corresponding estimate of PMFI/FR-total scarred trees/plots [56]. RME measures relative bias as sample sizes differ. I also calculated the standard error of each mean and tested the null hypothesis that mean bias is zero using a one-sample *t*-test in Minitab 17 [51]. Inaccuracy or error was also calculated using a relative measure, relative mean absolute error (RMAE):

$$RMAE = \sum_{i=1}^{n} [|(M_i - FR_i)|/FR_i]/n \tag{4}$$

where symbols are as above. This quantifies the difference or error between the 21 estimators versus PMFI/FR-total scarred trees/plots as a percentage of this PMFI/FR estimate [56]. I also calculated the standard error of each mean and then tested the null hypothesis that mean inaccuracy is zero using a one-sample *t*-test in Minitab 17 [51].

Can bias and inaccuracy be overcome by adjusting estimators using regression models? Scatter plots showed that PMFI/FR-total scarred trees/plots versus CFI and ITFI estimators were generally linear (e.g., Fig 2A), thus I fit linear regression models, using the lm function in R version 3.2.3 [57], to predict PMFI/FR-total scarred trees/plots from each of the 21 estimators. Sample size differed among the regressions, because individual estimators were not available for all 96 cases. After initial fitting, for each measure I removed 1–2 outliers with the largest studentized residuals (i.e., > 3.0). After refitting, I examined a plot of residuals versus fitted and a normal probability plot to identify trends in residuals, which were lacking for all models.

To estimate prediction error, which is useful itself but also provides a model-selection criterion, I completed a 10-fold cross-validation using the cv.lm function in the DAAG package in R. The output is the mean square error (MSE) of predicted estimates, and its square root is the root mean square error (RMSE), a prediction analog of the standard error of the estimate in fitted regression equations. Prediction error from cross-validation is asymptotically equivalent to Akaike's information criterion (AIC), a commonly applied model-selection criterion [58], but low prediction error is most germane for this application.

Do sample-size variables improve these models? To test this, I redid the regressions with three sample-size predictors (sample area, total scarred trees, scarred trees/100 ha) in addition to each of the 21 estimators in the previous models. This time I used best-subset regression in Minitab 17 [51], and the best predictor models were chosen by the lowest Mallow's C_p statistic, where each included variable also had to be significantly ($\alpha = 0.05$) related to FR-total scarred trees/plots. I again removed 1–2 outliers based on studentized residuals and examined histograms of residuals and normal probability plots, but found no trends in residuals.

Results

Bias, inaccuracy, and regression models to estimate PMFI/FR

Bias was significantly different from 0.0 for all estimators except mean ITFI and inaccuracy was significantly different from 0.0 for all estimators (Table 1). Mean RMEs of -69% to -75% for CFI-all fires, -60% to -69% for CFI-10% scarred, and -38% to -49% for CFI-25% scarred estimators, combined with low standard errors, show that CFI measures all lead to estimates of PMFI/FR that are consistently too short (Table 1). Bias diminished from CFI-all to CFI-25%, but all estimators, except mean ITFI, were still biased. Inaccuracy for CFIs had similar

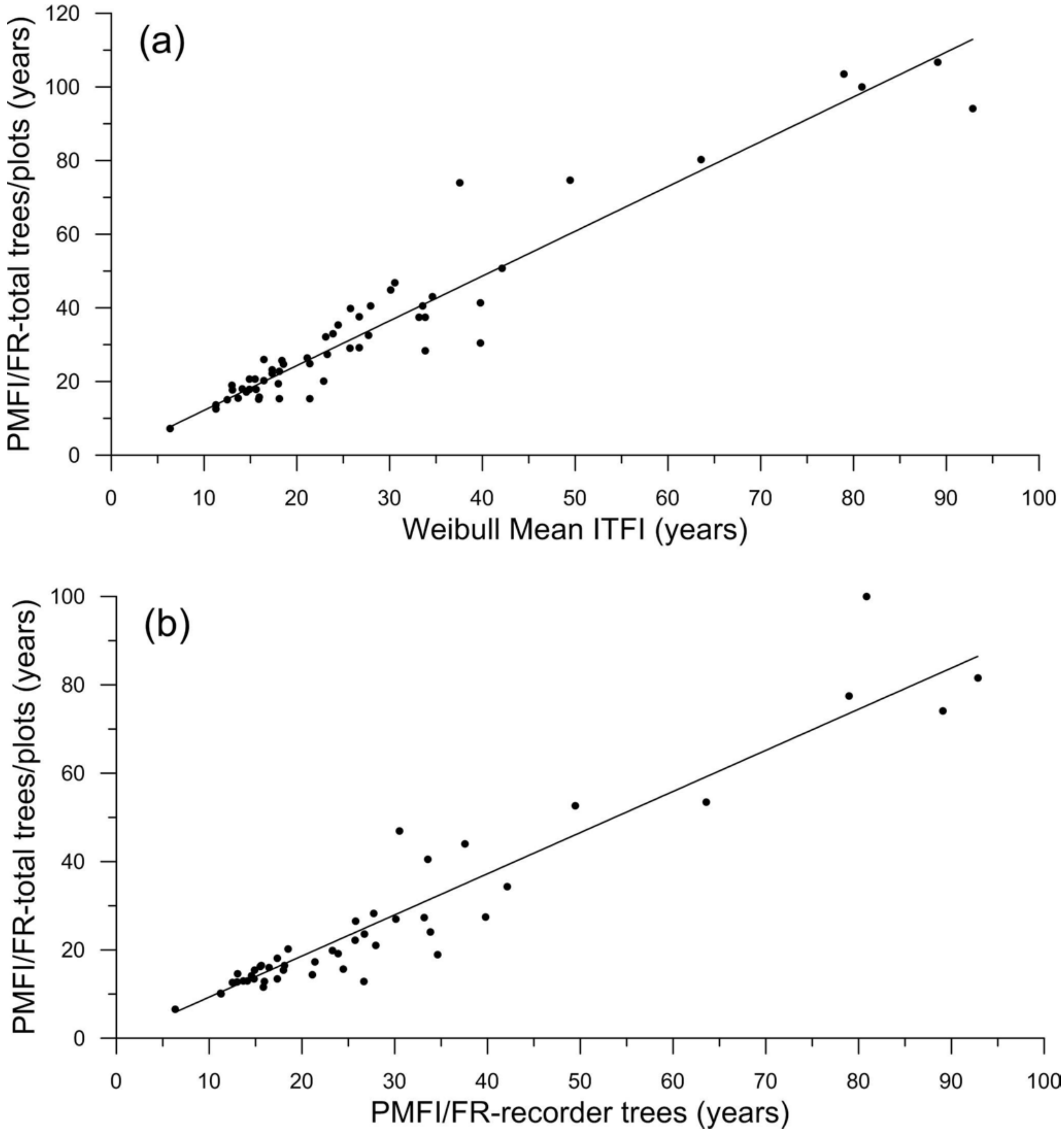

Fig 2. Scatterplots showing the linear relationships between: (a) Weibull mean ITFI and fire rotation-total trees/plots, and (b) Fire rotation-total trees/plots and fire rotation-recorder trees.

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magnitudes, patterns, and trends, with the best still having inaccuracies of 40–50%. ITFI estimators had lower bias and inaccuracy than CFI measures, with bias ranging from -3 to -30% and inaccuracy ranging from 16–33%. Only mean ITFI was unbiased, but still had 30% inaccuracy. FR-recorders also produced significantly biased and inaccurate estimates of FR, averaging 27% too low.

Table 1. Bias and inaccuracy in composite fire interval (CFI) and individual-tree fire interval (ITFI) estimates if used to estimate fire rotation-total trees/plots within the 96-case calibration dataset.

Measure	Test of bias					Test of inaccuracy			
	n	Mean RME (%)	s.e. of mean (%)	t	p	Mean RMAE (%)	s.e. of mean (%)	t	p
Mean CFI—all fires	84	-69.35	2.11	-32.79	<0.001	69.42	2.09	33.27	<0.001
Median CFI—all	76	-70.10	2.62	-26.77	<0.001	70.22	2.57	27.27	<0.001
Weibull Scale CFI—all	58	-69.25	2.28	-30.41	<0.001	69.25	2.28	30.41	<0.001
Weibull Mean CFI—all	58	-72.10	2.03	-35.47	<0.001	72.10	2.03	35.47	<0.001
Weibull Median CFI—all	58	-75.46	1.92	-39.33	<0.001	75.46	1.92	39.33	<0.001
Mean CFI—10% scarred	61	-63.29	2.08	-30.46	<0.001	63.39	2.03	31.27	<0.001
Median CFI—10%	62	-68.69	2.14	-32.11	<0.001	68.79	2.09	32.95	<0.001
Weibull Scale CFI—10%	57	-60.09	2.11	-28.48	<0.001	60.09	2.11	28.48	<0.001
Weibull Mean CFI—10%	57	-63.84	1.86	-34.29	<0.001	63.84	1.86	34.29	<0.001
Weibull Median CFI—10%	57	-68.03	1.85	-36.70	<0.001	68.03	1.85	36.70	<0.001
Mean CFI—25% scarred	71	-42.12	2.28	-18.43	<0.001	43.19	1.98	21.82	<0.001
Median CFI—25%	65	-48.88	2.66	-18.35	<0.001	50.77	2.04	24.91	<0.001
Weibull Scale CFI—25%	56	-38.41	2.64	-14.54	<0.001	40.24	2.09	19.30	<0.001
Weibull Mean CFI—25%	56	-44.66	2.35	-19.00	<0.001	45.92	1.86	24.74	<0.001
Weibull Median CFI—25%	56	-49.11	2.24	-21.90	<0.001	49.88	1.91	26.14	<0.001
Mean ITFI	67	-2.71	6.76	- 0.40	0.689	30.10	5.66	5.32	<0.001
Median ITFI	66	-29.71	2.75	-10.79	<0.001	33.43	1.99	16.78	<0.001
Weibull Scale ITFI	56	-8.50	2.53	-3.35	0.001	16.34	1.69	9.65	<0.001
Weibull Mean ITFI	56	-16.64	2.31	-7.22	<0.001	21.19	1.48	14.33	<0.001
Weibull Median ITFI	56	-28.25	2.10	13.43	<0.001	29.68	1.71	17.37	<0.001
FR–recorders	52	-26.79	2.01	-13.31	<0.001	26.79	2.01	13.33	<0.001

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Prediction error and fit show that the best regression models to predict PMFI/FR-total scarred trees/plots (Table 2) were from ITFI estimators, particularly Weibull mean ITFI (RMSE = 7.52, R^2_{adj} = 0.972), Weibull scale ITFI (RMSE = 8.04, R^2_{adj} = 0.970), and Weibull median ITFI (RMSE = 9.46, R^2_{adj} = 0.958), although the mean ITFI model was also good (RMSE = 10.30, R^2_{adj} = 0.944). Models based on CFI-25% scarred measures had moderately low prediction errors (RMSE from 11.0–13.7) and high R^2_{adj} values of 0.870–0.929. Models using CFI-10% had higher prediction errors and somewhat lower fit (Table 2). The poorest models were from CFI-all measures (Table 2). Weibull mean models consistently had lowest prediction errors and highest R^2_{adj} compared to models based on mean, median, Weibull scale, or Weibull median (Table 2).

Sample-size variables were not significant in most models (Table 3). The few models with significant sample-size variables had R^2_{adj} values generally improved only slightly, averaging higher by only 0.006–0.010 except for the model for mean CFI-all fires, which was 0.086 higher (Tables 2 and 3). Thus, simpler models in Table 2 should suffice for estimating PMFI and FR, except that the sample-size model may be worth using in the case of mean CFI-all fires (Table 3).

Using prediction error (RMSE) as the criterion, supplemented by fit (R^2_{adj}), the best model (Table 2) is based on Weibull mean ITFI, which had the lowest RMSE of 7.52 years and the highest R^2_{adj} of 0.972 (Table 2). The Weibull mean ITFI model was thus used for all PMFI/FR estimation for the 252-site dataset. Given its 7.52 year RMSE, 15–20 year bins are appropriate for reporting estimates, as about 68% of predictions are expected to be within the ± 1 RMSE of 7.52 years. Models other than the Weibull mean ITFI model (Tables 2 and 3) can also be used for deriving estimates from CFI and ITFI estimates, assuming prediction error and fit are acceptable.

Table 2. Linear regression models for estimating PMFI/FR-total scarred trees/plots, based on the 96-case calibration dataset. All slopes (*β*) were significant ($p < 0.001$) at α = 0.05.

Estimator	*β* †	Outliers‡	*n*	R^2_{adj}	*RMSE*§
Mean CFI—all fires	2.440	25, 89	82	0.721	18.14
Median CFI—all fires	2.450	25, 89	74	0.675	18.52
Weibull Scale CFI—all fires	2.655	25, 93	56	0.755	19.05
Weibull Mean CFI—all fires	2.915	25, 93	56	0.762	18.63
Weibull Median CFI—all fires	3.294	25, 93	56	0.730	20.12
Mean CFI—10% scarred	2.467	25, 89	59	0.837	15.65
Median CFI—10% scarred	2.783	25, 89	60	0.812	16.34
Weibull Scale CFI—10% scarred	2.423	25, 93	55	0.856	16.09
Weibull Mean CFI—10% scarred	2.666	25, 93	55	0.865	15.39
Weibull Median CFI—10% scarred	2.992	25, 93	55	0.826	17.66
Mean CFI—25% scarred	1.715	2, 89	69	0.923	11.00
Median CFI—25% scarred	1.834	26, 89	63	0.870	13.67
Weibull Scale CFI—25% scarred	1.597	2	55	0.925	11.96
Weibull Mean CFI—25% scarred	1.749	2	55	0.929	11.36
Weibull Median CFI—25% scarred	1.867	2	55	0.906	13.00
Mean ITFI	1.121	2, 70	65	0.944	10.30
Median ITFI	1.366	24, 26	64	0.896	12.57
Weibull Scale ITFI	1.108	2	55	0.970	8.04
Weibull Mean ITFI	1.216	2	55	0.972	7.52
Weibull Median ITFI	1.361	2	55	0.958	9.46
PMFI/FR-recorders	1.337	None	52	0.961	10.39

† All models have the form: PMFI/FR-total scarred trees/plots = *β* * predictor

‡ Numbers represent row numbers in the 96-case calibration dataset (S1 Table)

§ RMSE = root mean square error, the prediction error, in years, from the 10-fold cross validation

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Estimated historical PMFI/FRs across the 342-site dataset

Overall, estimated historical PMFI/FR across the 342 sites had a mean of about 39 years and a median of about 30 years (Table 4). Mean PMFI/FR did not differ significantly between dry pine forests and dry mixed-conifer forests (Table 4; $t\,(181) = -0.34$, $p = 0.731$). Maps and histograms show that shorter historical PMFI/FRs (< 25 years) were concentrated in Arizona and New Mexico, but also were scattered across parts of all other states, except for few in South Dakota, Wyoming, Colorado, and Mexico (Figs 3 and 4). Historically long PMFI/FR (> 55 years), in contrast, were common only in a band from northern New Mexico to western South

Table 3. Best linear regression models for estimating PMFI/FR-total scarred trees/plots, including estimators in Table 2 plus measures of sample size, based on the 96-case calibration dataset. Only cases where sample-size variables were significant are shown here, otherwise the best models are in Table 2.

Estimator	Best model	*n*	R^2_{adj}
Mean CFI—all fires	1.817 Mean CFI-all + 0.000896 Sample area (ha) + 0.927 Scarred Trees/100 ha	82	0.807
Mean CFI—10% scarred	2.347 Mean CFI-10% scarred + 0.0447 Scarred Trees	59	0.847
Mean ITFI	1.178 Mean ITFI—0.037 Scarred Trees	65	0.951
Median ITFI	1.260 Median ITFI + 0.360 Scarred Trees/100 ha	64	0.902
PMFI/FR-recorders	1.281 FR from recorders + 0.0702 Scarred Trees	52	0.966

doi:10.1371/journal.pone.0172288.t003

Table 4. Overall statistics for historical low-severity PMFI/FR in dry forests and by forest type, based on the merged 342-site dataset. Sample size was 342 overall, 223 in dry pine, 119 in dry mixed conifer.

Statistic	Overall (years)	Dry Pine (years)	Dry Mixed Conifer (years)
Mean	38.62	39.11	37.69
95% confidence interval for mean	35.13–42.10	35.40–42.83	30.42–44.97
Standard deviation	32.75	28.17	40.08
Minimum	7.20	7.20	10.21
1st quartile	19.55	18.80	21.24
Median	29.68	29.95	29.20
95% confidence interval for median	27.01–31.70	26.40–34.63	25.07–31.58
3rd quartile	46.11	50.49	37.62
Maximum	327.16	175.09	327.16

doi:10.1371/journal.pone.0172288.t004

Dakota, and were otherwise only scattered in a few locations in California, Oregon and Washington, with no occurrences in Idaho and Montana (Figs 3 and 4). Variability in historical PMFI/FRs was substantial but generally modest within a state, with coefficients of variation (CV) typically between about 30–60%, although California had a high CV and Arizona had a low CV (Table 5). Minima were typically 7–15 years except 20–30 years in South Dakota, Wyoming, and Mexico. Maxima were not very indicative, as a few long PMFI/FR were not uncommon (Fig 4). However, the 3rd quartile of about 93 years in Colorado, 56 years in Wyoming, and 50 years in South Dakota suggests that long historical PMFI/FRs were common in the southern Rocky Mountains and Black Hills (Table 5, Fig 3). At the state level, Colorado stands out in having the greatest variability and total range in historical PMFI/FRs (Fig 4I), and Arizona stands out as having the lowest variability and total range (Fig 4A).

Another pattern is that in the most mountainous areas with the steepest environmental gradients and topographic diversity, the full range (all four classes) in historical PMFI/FRs often was found in a small area (Fig 3). This high diversity occurred in northeastern Washington, the central Sierra Nevada, northern New Mexico, southwestern Colorado, north-central Colorado, and in western South Dakota, but not in Arizona, Idaho, Montana, or Oregon (Fig 3). However, even in these areas, with the exception of Arizona, some diversity in historical PMFI/FR was found over relatively short distances (Fig 3), suggesting the importance of local factors in addition to the large trends evident across the western USA.

Most studies of low-severity fire in dry western forests were conducted in forest stands that were mostly old forests in the pre-EuroAmerican era (Fig 5). Stands with beginning analysis years before A.D. 1750 were likely generally ≥ 150 years old, and stands with beginning analysis years before A.D. 1700 were likely generally ≥ 200 years old, in A.D. 1900, thus meeting the age criterion for old-growth forests (Fig 5). A history of predominantly low-severity fire in the century before the late-1800s was found across about 34%, on average (ranging from 2.5–62.4%), of eleven dry-forest landscapes across the western USA (Table 6). Thus, estimated historical PMFI/FRs apply primarily to old forests, which were likely concentrated historically in the 34% of overall dry-forest landscapes with a history of predominantly low-severity fire.

Discussion

Limitations of CFIs and ITFIs if used to estimate PMFI/FR

Researchers in the past commonly sampled fire scars and trees to generally increase the length of the fire-history record, minimize physical damage to trees, and maximize efficiency [55]. Unfortunately, these methods also produced CFI and ITFI estimates that are biased and

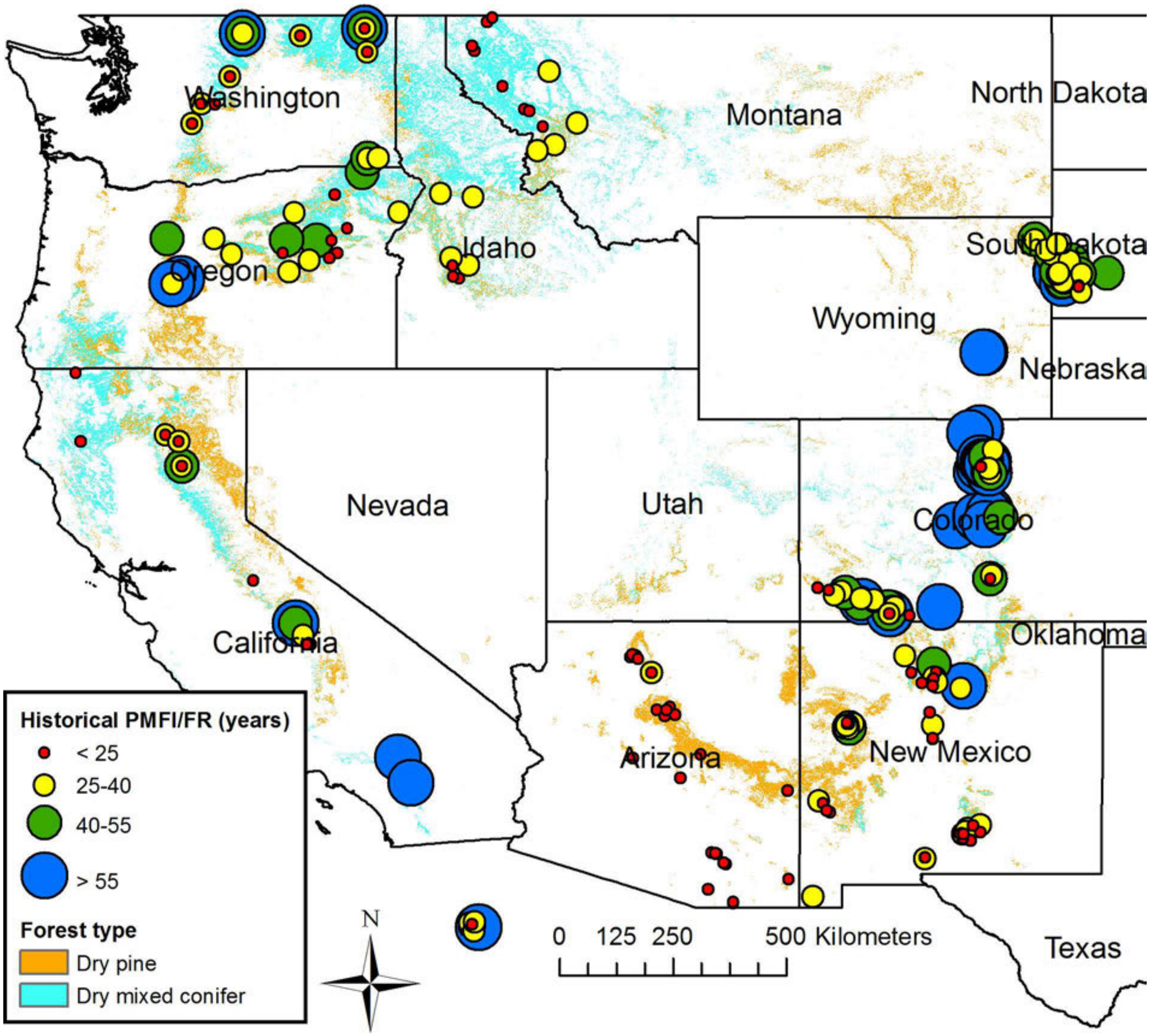
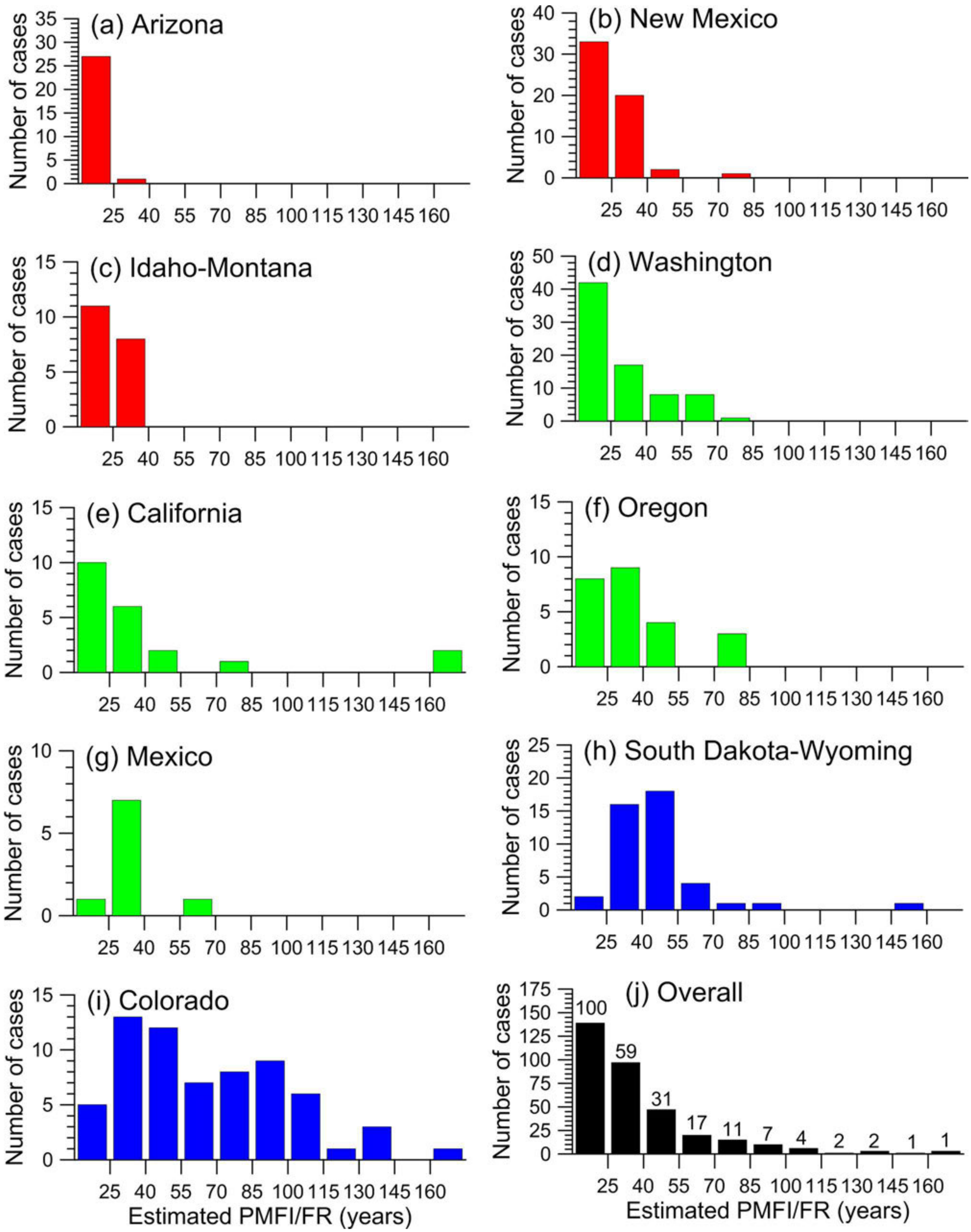


Fig 3. Estimated historical low-severity population mean fire interval/fire rotation (PMFI/FR) for the combined set (n = 342) of calibration cases and prediction sites in dry forests of the western USA.

doi:10.1371/journal.pone.0172288.g003

inaccurate if used to estimate the PMFI/FR (Table 1), as also found from modern calibration [11], and this is now accepted to be an inappropriate use. However, it is possible to estimate PMFI/FR accurately from past CFI and ITFI estimates using linear regression (Table 2).

Further discussion of the limitations of past measures as estimators of the PMFI/FR is thus generally moot, but for those interested, I include further analysis in S1 Text and a summary here. The main factors unique to underestimation of PMFI/FR by CFI measures likely include: (1) overcompensation—sampling and compositing across too large an area, (2) loss of long real fire intervals to the compositing process, and (3) restriction rules that do not omit enough small fires. ITFI measures do not use compositing and have lower bias and inaccuracy, but still are biased and inaccurate (Table 1). Both CFI and ITFI measures must be missing longer intervals from a sampling bias, because their estimates are low relative to PMFI/FR. Major factors likely are targeting trees and sampling areas with the most scars, excluding trees with no or one scar, and censoring intervals at the beginning and end of a tree's record (S1 Text).



(a) Arizona
(b) New Mexico
(c) Idaho-Montana
(d) Washington
(e) California
(f) Oregon
(g) Mexico
(h) South Dakota-Wyoming
(i) Colorado
(j) Overall
Number of cases
100
59
31
17
11
7
4
2
2
1
1
25 40 55 70 85 100 115 130 145 160
Estimated PMFI/FR (years)

Fig 4. Histograms showing the variability in historical PMFI/FRs (342 sites). These are shown among: (a-i) the eleven western states and (j) overall. In (j) the numbers above the bars indicate the percentage of the distribution that exceeds the lower limit of each bin. For example, 59% of the distribution had historical PMFI/FR > 25 years. Idaho and Montana were combined, as were South Dakota and Wyoming, because of insufficient samples and similarity of histograms within these pairs of adjoining states. Colors indicate similar histograms, with the shortest historical PMFI/FRs predominating in Arizona, New Mexico, and Idaho-Montana, intermediate in Washington, California, Oregon, and Mexico, and the longest in South Dakota-Wyoming and Colorado.

doi:10.1371/journal.pone.0172288.g004

Targeting can also reduce estimated PMFI/FR itself (S1 Text), and needs to be avoided in new landscape methods.

Inference space for the PMFI/FR estimates

Studies of new probabilistic landscape methods for reconstructing PMFI/FR encourage ". . . clearly defining the inference space, not extrapolating to unrepresentative areas . . ." ([55]:1030), and this is also important for estimates of PMFI/FR from regression. The dataset of 342 sites spans dry forests in the western USA (Fig 1). The set of published studies corresponding to this dataset (S2 Table) includes many of the studies of low-severity fire in dry forests in the western USA, but other studies exist. This dataset and these other studies likely are not a probabilistic sample of historical dry forests, however, as many studies targeted old forests or forests with concentrations of fire scars (S1 Text) and occurred in forests that likely were old in the pre-EuroAmerican era (Fig 5). Old trees were historically dominant in some dry-forest landscapes and old trees were not uncommon in many forests, but young to middle-aged forests historically dominated most dry-forest landscapes [24, 62, 63]. Based on the GLO reconstructions and early aerial photographs (Table 6), the PMFI/FR estimates here apply most clearly to no more than about 34% of dry-forest landscapes, particularly in old forests. That leaves about 66% of dry-forest landscapes without PMFI/FR estimates. It is possible that some estimates do apply to parts of these other forests, possibly representing the low-severity parts of mixed-severity fire regimes on sites that had not recently burned at high severity. However, it is impossible to determine this from data in FHX files or for the 74% of studies that did not reconstruct fire severity (S1 Text).

Several studies, that targeted old forests to obtain long fire records, indicated that younger forests had few fire scars and, because these studies were focused on long and complete records of fire years, they avoided sampling younger stands. In El Malpais, New Mexico: "The most abundant, best preserved fire-scarred samples were found at sites on the northwestern and western peripheries of the malpais . . . We found no fire-scarred samples on the kipukas in the northern and eastern portions of the malpais, and found few samples in the southern portions. These areas contained ponderosa forests that appeared younger than elsewhere, perhaps due to more recent, intense stand-replacing fires . . ." ([64]:136). Sampling was concentrated in

Table 5. Statistics for historical low-severity PMFI/FR in dry forests by state, based on the merged 342-site dataset. Sample sizes were 28 in AZ, 21 in CA, 65 in CO, 7 in ID, 12 in MT, 56 in NM, 24 in OR, 40 in SD, 76 in WA, 3 in WY, 9 in MX and 1 in BC.

Statistic	AZ (yrs)	CA (yrs)	CO (yrs)	ID (yrs)	MT (yrs)	NM (yrs)	OR (yrs)	SD (yrs)	WA (yrs)	WY (yrs)	MX (yrs)	BC (yrs)
Mean	15.48	54.21	65.70	25.96	21.81	24.59	36.41	46.19	30.60	47.18	35.04	40.49
s.d.	4.26	83.01	35.32	8.28	6.77	11.24	19.09	22.23	16.09	14.92	13.06	-
CV	27.52	153.13	53.76	31.90	31.04	45.71	52.43	48.13	52.58	31.62	37.27	-
Minimum	7.20	8.56	15.20	16.88	13.25	10.21	15.30	21.18	11.00	29.95	23.15	40.49
1st quartile	12.51	18.68	35.05	17.00	14.88	16.25	24.12	35.20	19.73	29.95	28.62	-
Median	15.22	27.20	60.45	27.07	22.47	22.28	29.66	41.84	23.89	55.79	32.28	40.49
3rd quartile	17.98	40.77	92.67	32.95	26.95	30.62	42.33	49.97	38.30	55.79	35.88	-
Maximum	25.70	327.16	175.09	37.37	32.83	74.70	83.25	158.70	81.93	55.79	68.08	40.49

doi:10.1371/journal.pone.0172288.t005

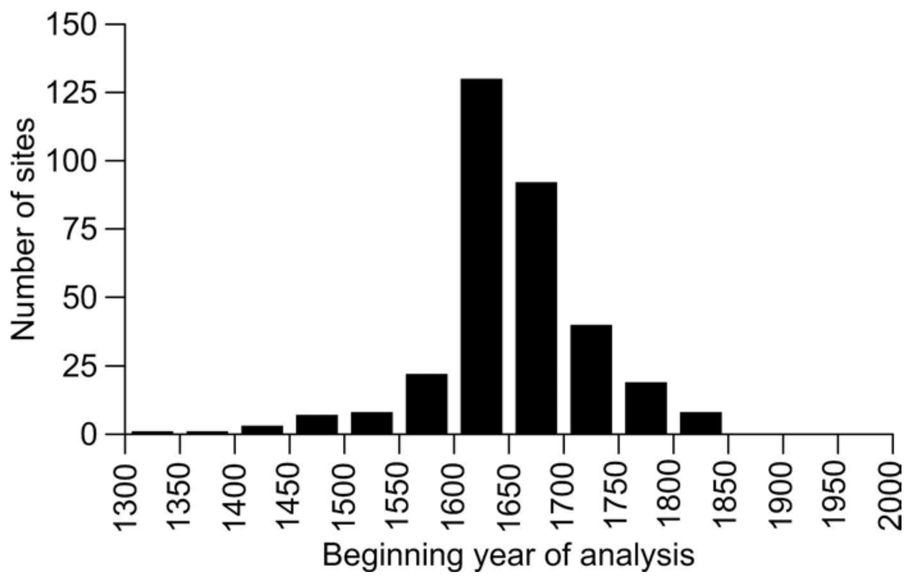


Fig 5. Beginning year of analysis for the 331 sites with available data in the 342-site merged dataset.

doi:10.1371/journal.pone.0172288.g005

areas with abundant fire-scars, but later this targeting was forgotten, and these areas were portrayed as representing the whole El Malpais landscape: "These increased fuel loadings in malpais forests have essentially changed the trajectory of fire behavior to one that now favors the occurrence of high-intensity, stand-replacing fires in contrast to the low-intensity, stand-maintenance fires that occurred prior to Euro-American settlement . . ." ([64]:234). Similarly, no fire scars were found in 5 of 12 transect locations in mixed-conifer forests in northern New Mexico [65]. The study sampled scars on relatively flat ridges nearby, where scars were abundant, and composite fire intervals from these sites were assumed to apply to the whole mixed-

Table 6. Area and percentage of 11 dry-forest landscapes in the western USA that meet the low-severity model, based on GLO surveys and early aerial photographs.

Source/Author(s)	Study area		Fits low-severity model	
	Location	**Area (ha)**	**%**	**Area (ha)**
General Land Office Surveys				
Williams and Baker [53]	Mogollon Plateau, AZ	405,214	62.4	252,854
	Black Mesa, AZ	151,080	12.0	18,130
	Front Range, CO	65,525	2.5	1,638
	Blue Mountains, OR	304,709	40.3	122,798
Baker [59]	North-E Cascades, OR	146,555	32.5	47,630
	Central-E Cascades, OR	147,502	10.4	15,340
	South-E Cascades, OR	104,160	29.4	30,623
Williams and Baker [60]	Coconino Plateau, AZ	41,214	58.8	24,234
Baker [61]	N. Sierra, CA	115,766	12.6	14,587
	S. Sierra, CA	187,085	26.4	49,390
Early Aerial Photographs				
Hessburg et al. 2007	E. WA & E. OR	112,115	21.6	24,200
Symopsis				
Total		1,780,925		601,424
Mean percentage			33.8	

doi:10.1371/journal.pone.0172288.t006

conifer forest [65]. This was also the pattern in northern Colorado: "Most of the 67 fire-scarred trees that were sampled were found on ridges or in open areas (Fig 1). It was uncommon to find scarred trees in dense stands" ([66]:138).

These observations suggest low-severity fire was likely less frequent or even rare in younger and denser historical dry forests, that likely were common in the 66% of dry forests lacking a history of exclusive low-severity fire (Table 6). However, specific studies of rates of low-severity fire are lacking for stands $\leq$ 150–200 years old in the pre-EuroAmerican era (Fig 5), that are the predominant forests today. Because they are not in the inference space for past fire-history studies in dry forests, it is not valid to infer that today's young to middle-aged forests would have been subject to low-severity fires at the historical mean rates in the 342-site dataset or in other comparable published fire-histories for dry forests.

Historical dry forests not predominantly frequent-fire forests

Dry pine and dry mixed-conifer forests have been described as frequent-fire forests, an attribute still supported for only about 14% of overall dry-forest area, with multidecadal low-severity fire likely historically over about 86% of overall dry-forest area in the western USA. Only about 41% of the old, dry forests, which were likely concentrated in about 34% of western USA dry forests (41% of 34% = 14% of overall dry forest), had frequent fire, with a historical PMFI/FR $<$ 25 years (Fig 4J). Old forests with frequent fire were historically concentrated in Arizona and found at scattered sites across the West (Fig 3), particularly in New Mexico, Washington, Idaho, Montana, and California (Fig 4A–4E). In contrast, about 59% of cases in old forests and thus about 20% of dry forests in the western USA (59% of 34% = 20% of overall dry forest) had a historical mean PMFI/FR $\geq$ 25 years (Fig 4J). Low-severity fire was likely even less frequent in the remaining overall 66% of dry-forest landscapes lacking a history of exclusive low-severity fire (Table 6). Altogether roughly 14% of dry forests in the western USA historically had frequent (PMFI/FR $<$ 25 years) low-severity fire and 86% of dry forests in the western USA historically instead had multidecadal low-severity fire.

Even in the 34% of dry-forest landscapes with an exclusive history of low-severity fire, the overall mean PMFI/FR was 39 years, half the cases had PMFI/FR $>$ 30 years, and a quarter of cases had PMFI/FR $>$ 46 years (Table 4). These old forests are better described overall as having diverse rates of low-severity fire, spanning the range from frequent to multidecadal. This diversity in rates varied on two scales, first across large regions from predominantly multidecadal (median $>$ 40 years), in Colorado, South Dakota, and Wyoming, to predominantly frequent in Arizona, New Mexico, and Idaho-Montana, with other states having broader mixtures, ranging from frequent to multidecadal (Figs 3 and 4). Second, individual smaller areas often contained a diversity of rates over short distances, particularly in mountain ranges, often spanning or nearly spanning a broad range from frequent to multidecadal (Fig 3).

Estimated historical PMFI/FR mean rates are relevant, because many ecological processes and structures change across a narrow range in rates. In the roughly 86% forests with PMFI/FR $\geq$ 25 years, fuels that required about 7–25 years to build back up after a low-severity fire [12–14]. would, on average, have been fully recovered for an extended period before the next fire. Shrubs would likely have been able to fully recover and dominate for substantial periods. Small trees that rely on seed (e.g., ponderosa pine) would also have been able to regenerate and become common in forest understories, as documented in several historical dry forests [24]. The role of the forest floor in replenishing soil nutrients and organic matter, enhancing absorption of water, and fostering microbial communities [15] would not have been limited by too-frequent fires. Greater opportunities for trees to regenerate and less mortality from low-severity fire also help to explain dense areas of dry forests that occurred historically across substantial parts of many dry-forest landscapes (e.g. [53, 59]). Natural fuels, less limited by low-severity fire, would have favored higher-severity fires via ladder fuels. Adverse effects on habitat for wildlife that use snags or coarse down wood [15] would be less because of less low-severity fire, and fires of higher intensity would likely increase snags and coarse dead wood.

In contrast, in the roughly 14% of historical dry forests with historical PMFI/FR < 25 years, levels of fuels, including shrubs and small trees, would have been more consistently kept low (Fig 3). Frequent low-severity fires would likely have fostered a diversity of grasses and forbs, but would have limited shrubs and small trees. In these settings, lower-density forests would have been favored and higher-severity fires would have been discouraged, at least by fuel conditions [19, 53]. Potential adverse ecosystem and wildlife effects of frequent low-severity fire [15] would remain a natural historical characteristic of these primarily southwestern frequent-fire forests (Fig 3). However, high local and regional diversity in rates (Fig 3) meant that a diversity of processes, rates, and structures occurred across even the old-forest part of many dry-forest landscapes, within both small areas and across the western USA.

Limitations and error in calibration and prediction PMFI/FRs

The calibration cases (S1 Table) are from larger land areas and include estimates of PMFI/FR that are directly usable as a guide for restoration and management in old, dry forests. The appropriate estimate in S1 Table is FR-YrsTot, which was directly estimated in the study in many cases. Where a direct estimate was not made, I estimated PMFI/FR-YrsTot from PMFI/FR-YrsRec using the equation in Table 2 and Fig 2B.

The 252 prediction cases (part of S2 Table) are from single-plot samples in smaller plot areas, and likely have more error. The estimated prediction error for PMFI/FR in a small plot was a 7.52 year RMSE, which suggests bins about 15-years wide, as in Fig 3, would likely contain about 68% of observations. Bins about 30-years wide would contain about 95% of observations. Smaller plots used at the 252 sites also may not individually provide an adequate sample of a forest area. In an accuracy study, estimates from small plots required averaging across 5–6 plots representing 600–1000 ha to achieve mean relative errors $< 30\%$ in estimating PMFI/FR [11]. The estimated PMFI/FRs from the available set of small plots cannot be pooled to decrease this error, as they are not necessarily samples from one population. The problem for small plots is inherent stochastic variability in realized fire intervals, even from a fixed fire regime in a particular land area [67], and errors in the sample and estimators. Thus, the PMFI/FR estimates are a significant improvement over using CFI and ITFI, but greater accuracy can be expected from larger studies in the calibration dataset and also from future landscape-scale reconstructions.

Most of the 342 estimates are likely low estimates for two reasons. Targeting multi-scarred trees reduces CFI and ITFI estimates, but also reduces estimated PMFI/FR by not sampling trees with one scar or no scar that can indicate areas that did not burn in a particular fire (S1 Text). Thus, the area burned by each fire may be inflated and the PMFI/FR too short. Because 94% of 250 cases with evidence did target multi-scarred trees (Table A in S1 Text), this affects almost all estimates of PMFI/FR. Targeted sampling of individual trees led to PMFI/FR estimates reduced to about 86–95% of estimates from equal-sized probabilistic samples [55]. This would mean that PMFI/FR estimates here need to be multiplied by 1.05–1.18. Also, both calibration and prediction PMFI/FR estimates are low estimates in many cases because PMFI/FR could not be estimated separately for low-severity fires in the 74% of cases where fire-severity was not studied. Even where fire severity was studied, the study did not report separate rates, instead only rates for fire severities combined (S1 Text). Because estimates are for old forests with a history of low-severity fire, the higher-severity component was likely not large, but could affect longer estimates (Table B in S1 Text). Combining these two factors likely would increase estimated PMFI/FR, but more research is needed to narrow and validate the needed corrections before they are applied. In contrast, FR estimates in the calibration are from all trees, not recorders, and regression equations applied to the predicted dataset are from all trees. Estimates from recorders would be lower, but I explained earlier why the truth is likely closer to FR-all trees. Further research is warranted, and could possibly resolve all remaining uncertainties, leading to improved equations and estimates.

PMFI/FRs as a guide to restore and manage low-severity fires

In spite of these limitations, these new PMFI/FR estimates are the best available and usable estimates of historical mean rates of low-severity fire to use as a guide in restoring and managing low-severity fire in dry forests of the western USA. Past CFI and ITFI estimates were not intended to estimate PMFI/FR and would be misapplied, with adverse impacts on biological diversity and ecosystem functioning, if used directly for this purpose, as is shown by their biases, inaccuracies, and needed adjustments using regressions (Tables 1, 2).

Estimated historical PMFI/FRs specify how long, on average, it took to burn across a land area (the FR), and how long the intervals were, on average, between fires at points in the land area (the PMFI). They can be estimated at multiple scales, from small plots to large land areas, although with greater accuracy over larger land areas. Congruent estimates of modern and historical low-severity PMFI/FR can be made, and directly compared. Modern estimates can be made using digital fire maps (e.g., Monitoring Trends in Burn Severity at: http://www.mtbs.gov) or other sources. All that is needed is to add up the areas of fires that burned in a particular landscape of interest at low severity over a particular period, calculate the area of the landscape, and use Eq 2. Temporal and spatial variability in PMFI/FR can be estimated as well, using subareas or sub-periods (e.g. [23]). Comparison of modern and historical rates of low-severity fire facilitates monitoring the effectiveness of restoration and management programs, and analysis of trends in rates of modern relative to historical fires [3].

Fire-size distributions are also important, but those from small plots have inherently limited value. At this point, distributions of annual area burned, which approximate fire sizes, can be shown for some larger study areas in dry forests (Fig 6). I compiled data for these histograms from graphs or tables in the sources. Note that this is area burned at all severities, not just low severity, and is not restricted to old-forest parts of landscapes. Several graphs show that the most fire years were in the smallest size class, with decreasing abundance in larger size classes. Historical fire sizes could reach at times into the 5,000–11,000 ha size classes, at least in three study areas (Fig 6C, 6F and 6G). In many study areas, the maximum area burned reached the size of the study area (Fig 6B, 6C, 6D, 6E and 6F), suggesting fires could have been larger.

In an Arizona dry-forest landscape, 5.1% of total fires, that were the largest fires, contributed 97% of total burned area [1]. This pattern, common in forests [17], also suggests that in dry forests most of the burned area is from infrequent large fires, with frequent small fires not adding much to total burned area. This pattern of variable fire sizes and infrequent large fires is important to mimic, as it fosters diverse times since fire, at any instant, across a landscape, which allows species with different responses to fire to all remain viable across landscapes [16].

Low-severity fires can kill up to about 20% of basal area [27–28], and it is usually expected that this mortality is from torching or passive crown fires that kill individual trees or small groups of trees. However, little is known about the size and distribution of patches of mortality in low-severity fire regimes. Only about 23% of reconstructions of low-severity fires analyzed fire severity and even these provided little information about this topic (S1 Text), as it is difficult to reconstruct the size of mortality patches. Early historical observations provide some

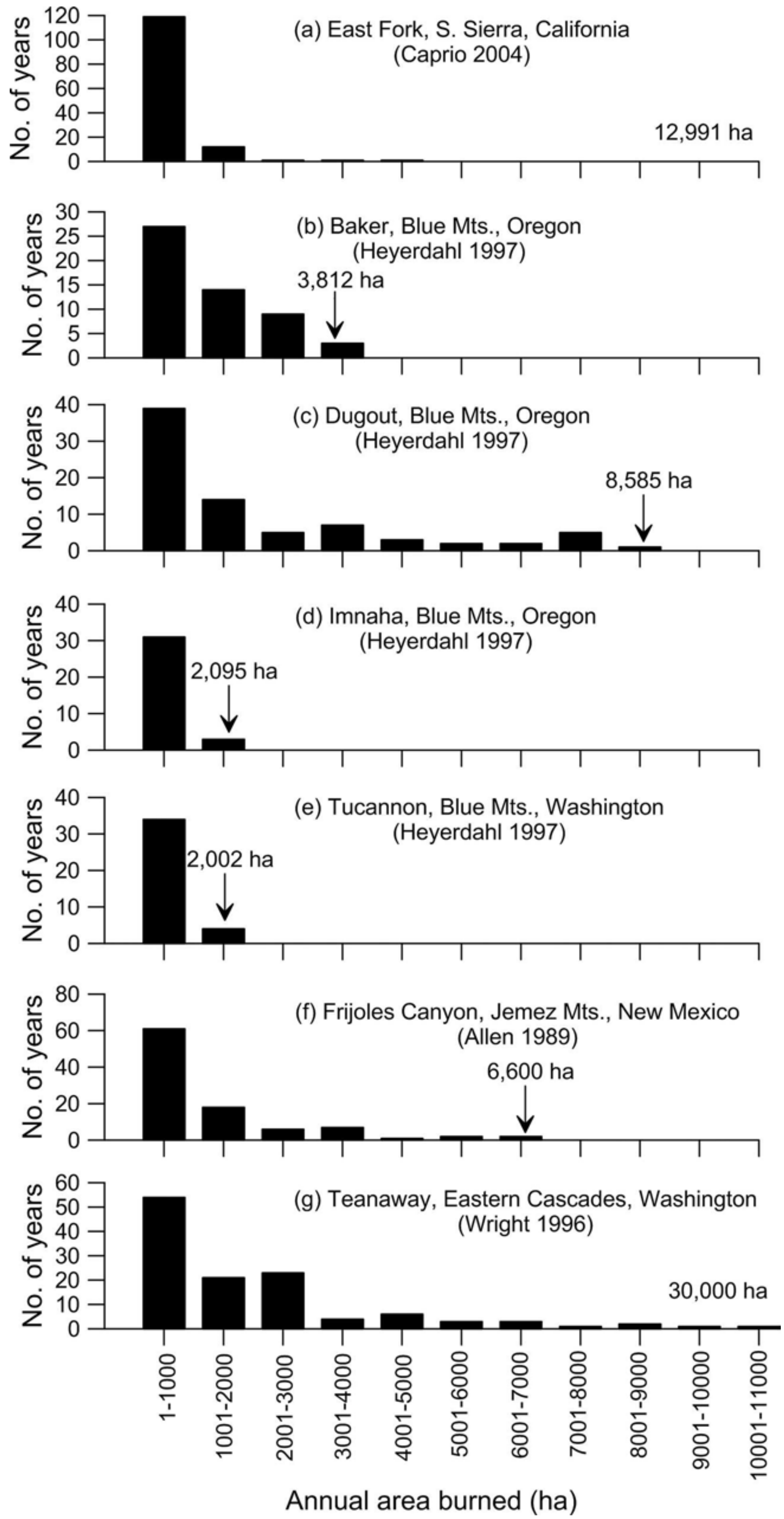


Fig 6. Size distribution for historical annual area burned in seven large study areas in dry forests of the western USA. Study area size is given above arrows or at the right of the x-axis.

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evidence. For example, generally low-severity fires in Sierran mixed conifer forests were observed to also have about 15% high-severity fire in small patches [20]. Small high-severity patches from low-severity fires in these forests were described [68] as mostly < 2–4 ha ([61]: Table A1 Q18-Q25). More research is needed on early historical observations of canopy mortality from low-severity fires, but some is historically congruent and expected from low-severity fires in modern forests.

Unburned areas within the perimeters of fires are also important for biological diversity and natural recovery, as these areas serve as refugia for less fire-tolerant plants or those that regenerate by seed, facilitating survival in these areas and natural recovery within the burned area [69]. This study found 35% average unburned area inside 154 modern fire perimeters in Yosemite National Park, California, which included substantial dry-forest area. Unfortunately, little is known about the extent of unburned area in historical low-severity fires in dry forests. It is known that prescribed burning that fully blackens burn units can reduce spatial heterogeneity in fire that promotes coexistence of multiple species [16]. Also, as reviewed in the introduction, unburned areas historically were locations where tree regeneration to replace tree mortality could survive. Thus, including unburned area within burn units, rather than blackening the whole unit, is ecologically important to restore and maintain tree populations and biological diversity.

The extent of needed burning to restore and manage old dry forests and the rest of dry forests is lower than previously thought. Earlier estimates were largely based on the assumption that reported mean CFI estimates represent PMFI/FR, which they do not (Tables 1, 2) and apply to all dry forests, which they also do not. Estimated historical rates of low-severity fire in dry forests in the U.S. Landfire program, for example, typically incorrectly use reported mean CFI estimates as though they represent PMFI/FR, although some actual PMFI/FR estimates are also used. These are applied to all dry forests, not just old forests. Both misapplications likely have adverse effects on biological diversity and ecosystem functioning. Prescribed burning in U.S. national forests, national parks, and on other public lands, where Landfire or other estimates from mean CFIs have been used as a guide, is likely too much by 1.6–3.3 times, depending on the CFI measure used (See *ß* inTable 2) in the roughly 14% of dry-forest area that was historically old forest with frequent fire (PMFI/FR < 25 years). Mean rates are likely too high by > 1.6–3.3 times in the 86% of the area of dry-forest landscapes that historically had multidecadal low-severity fire.

A need for less low-severity fire in restoration and management of dry forests is good news, because costs of prescribed burning and other restoration treatments are high, effects on invasive species, ecosystem processes, and biological diversity are a concern [15, 70], and the feasibility of restoring and managing low-severity fire is higher with longer rates. Longer rates also mean that completed treatments may have already been sufficient in many old-forest areas, and further management of low-severity fire can be redirected to using managed fire for resource benefit [71]. Where initial treatment is incomplete, one prescribed fire should suffice before a managed-fire program can begin. At that point, managers can monitor low-severity fire using historical mean PMFI/FR rates, fire-size distributions, and other attributes (e.g., unburned area) as a guide.

In locations where managed fire for resource benefit is infeasible, and an ongoing prescribed-burning program must be used, burning at rates longer than the mean PMFI/FRs reported here and using a diversity of rates and patterns of prescribed fires would be congruent with the findings. First, substantially lower rates (longer PMFI/FR) are warranted, if forests are not old forests, because estimated rates here apply mostly to old forests and the prevailing younger forests today likely burned historically at longer PMFI/FR. Second, the rates reported here are likely somewhat too short, as explained in "Limitations and errors. . ." Finally, lower rates would likely reduce the spread of invasive species and adverse effects on ecosystem processes and biological diversity. Also, historical rates varied substantially within small areas, particularly where there was topographic diversity, but also because of natural variability over time. It makes sense to similarly vary prescribed burning rates within local areas, leaving some areas unburned for longer periods. An approximation of the percentage of western USA old-forest parts of landscapes that experienced longer historical rates of fire is in Fig 4J. More local data can be derived from S2 Table, which lists PMFI/FR by state.

Data presented here can generally be used, with other evidence and tools, to create more comprehensive and spatially informative local understanding about mean historical PMFI/FR to guide local restoration and management of low-severity fire in old-forest parts of landscapes. Data in S2 Table have latitude and longitude and other ancillary information, and can be downloaded (S2 Dataset) and used directly or be read into a GIS program, where topography, land ownership and other information can be added for context. As new data are added to the IMPD, an FHX file for each new site can be downloaded and read into FHAES. Weibull mean ITFI can be calculated, which can then be used (Table 2) to estimate historical PMFI/FR, if not already provided in the study. Geographical coordinates, usually in the FHX file, allow new data to be added to the database (S2 Dataset) for use in GIS. Estimates of historical mean CFI and ITFI are available in the published scientific literature for other sites, not in the IMPD, which can also be used to estimate historical PMFI/FR using the equations in Tables 2 and 3, then added to the dataset (S2 Dataset) and input into GIS for local analysis. Of course, these estimates usually apply to only old-forest parts of historical landscapes.

Dry-forest landscapes until recently were thought to have historically been primarily old-growth forests, with a history of frequent low-severity fire, across their extent (e.g. [72]), but this has been refuted by GLO reconstructions and early aerial photographs (Table 6), paleo-ecological evidence [24], and early forest-reserve reports and other evidence [63, 73]. Even in Arizona, which had abundant old forests with frequent fire (Fig 3), denser forests and high-severity fire were extensive at certain times and in certain places, as on Black Mesa and parts of the Mogollon Plateau [60, 73]. It is sensible to restore low-severity fire to its former dominance in the parts of dry-forest landscapes with a history of primarily low-severity fire, historically averaging about 34% of western dry-forest landscapes (Table 6). Estimated mean PMFI/FRs here provide a guide for restoration and management of low-severity fire in extant old-forest parts of landscapes. For most dry-forests today, which are not old, using frequent fire (PMFI/FR $<$ 25 years) in restoration is not supported, and fuels do not need to be substantially reduced, because historical PMFI/FRs naturally allowed historical shrubs and small trees to fully recover after fires. Restoration of low-severity fire is still needed. The most appropriate approach, given likely long but uncertain mean rates of historical low-severity fire, is for most dry forests today to receive at most one prescribed fire, followed by managed fire for resource benefit, with the goal of mimicking mean historical PMFI/FRs and variability in fire (fire-size distributions, unburned area) as forests reach old age.

Supporting information

S1 Dataset.
(XLS)

S2 Dataset.
(XLS)

S1 Dataset metadata.
(PDF)

S2 Dataset metadata.
(PDF)

S1 Table. Authors, locations, and values for CFI and ITFI estimators and PMFI/FR in the 96-case calibration dataset.
(PDF)

S2 Table. Authors, sites, the Weibull mean ITFI estimate, and the calibrated or predicted PMFI/FR for the merged 342-site dataset.
(PDF)

S1 Text. Why CFIs and ITFIs underestimate PMFI/FR.
(PDF)

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Bet-hedging dry-forest resilience to climate-change threats in the western USA based on historical forest structure

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Dry forests are particularly subject to wildfires, insect outbreaks, and droughts that likely will increase with climate change. Efforts to increase resilience of dry forests often focus on removing most small trees to reduce wildfire risk. However, small trees often survive other disturbances and could provide broader forest resilience, but small trees are thought to have been historically rare. We used direct records by land surveyors in the late-1800s along 22,206 km of survey lines in 1.7 million ha of dry forests in the western USA to test this idea. These systematic surveys (45,171 trees) of historical forests reveal that small trees dominated (52–92% of total trees) dry forests. Historical forests also included diverse tree sizes and species, which together provided resilience to several types of disturbances. Current risk to dry forests from insect outbreaks is 5.6 times the risk of higher-severity wildfires, with small trees increasing forest resilience to insect outbreaks. Removal of most small trees to reduce wildfire risk may compromise the bet-hedging resilience, provided by small trees and diverse tree sizes and species, against a broad array of unpredictable future disturbances.

Keywords: dry forests, wildfires, insect outbreaks, droughts, climate change, resilience, land surveys, bet-hedge

INTRODUCTION

Dry forests globally may be particularly vulnerable to climatic change, because their setting is prone to wildfires, insect outbreaks, and droughts; these disturbances may increase, and post-disturbance tree recruitment is often poor. Recruitment limitation in forests is a widespread concern (Clark et al., 1999), particularly where moisture is limiting, as in *Pinus* forests in drier parts of precipitation gradients (Dorman et al., 2013). For example, dry forests of the western USA (Figure S1), which include montane ponderosa pine (*Pinus ponderosa*) forests and dry mixed-conifer forests also with firs (*Abies* spp.) and Douglas-fir (*Pseudotsuga*), can have poor tree recruitment that limits their recovery after fires, insect outbreaks, and droughts. Tree recruitment in dry *P. ponderosa* forests of the western USA over the last century has been poor, concentrated in episodic pluvials (Savage et al., 1996), and spatially variable (Stein, 1988; Roccaforte et al., 2012). Mortality of *P. ponderosa* at their ecotone with lower-elevation woodlands during a 1950s drought (Allen and Breshears, 1998) also indicates vulnerability. Rising temperatures and drought could further reduce tree recruitment in dry forests (Anderson-Teixeira et al., 2013). Climate envelopes of seedlings vs. established trees of *P. ponderosa* suggest general recruitment failure is underway, possibly a precursor to broader range contraction (Bell et al., 2014).

In contrast, paleoecological research shows that dry forests of the western USA persisted for thousands of years in the face of wildfires, insect outbreaks, and droughts (Jenkins et al., 2011), suggesting recruitment was not generally deficient and historical forests were resilient. However, this persistence appears incongruent with the hypothesis that these dry forests historically had low abundance of seedlings, saplings and small trees (Covington and Moore, 1994; Allen et al., 2002). This hypothesis is based in part on tree-ring reconstructions, which show that large trees were historically dominant in most sampled stands (Williams and Baker, 2012a). However, small trees could have been common, but missed in tree-ring reconstructions because small trees had high mortality rates and may decompose by the time of reconstruction (Allen et al., 2002). Also, tree-ring reconstructions are not located systematically across landscapes and plot-level size-class distributions are often averaged, masking variability (Williams and Baker, 2013). Nonetheless, frequent surface fires were thought to have limited small trees, and some early accounts do suggest low abundance of tree recruitment (Leiberg et al., 1904; Covington and Moore, 1994; Allen et al., 2002). Today, large trees are likely less abundant and small trees more abundant than historically (Covington and Moore, 1994), but our focus is only on historical abundance of small trees, not current abundance. The common hypothesis is that low-severity fires historically limited small trees, so they were a low percentage of total trees and were found across a low percentage of land area.

We use a previously untapped historical source, the General Land Office (GLO) land surveys, which provide spatially extensive direct empirical data on historical tree recruitment (seedlings/saplings, small trees). We use seven study areas that span dry forests of the western USA (Figure S1) to test the hypothesis that dry forests historically had little tree recruitment. We formalize this for the two data sources from the GLO surveys and two components of recruitment abundance: H_1: Small trees were <20% of total trees, and H_2: Seedlings and saplings (trees < 10 cm diameter) were present on <20% of forest area.

Past specific estimates of percentages were lacking; we used test values that conservatively represent the hypotheses. Small trees are ≥10 cm dbh, with an upper size limit of 30–50 cm, defined for each study area (Williams and Baker, 2012a). We measured and compared recent risks of higher-severity wildfires and insect outbreaks in dry forests, separated into ponderosa pine forests and dry mixed-conifer forests, across the western USA using government data. We reviewed the role of tree recruitment in recovery after these disturbances. We suggest a strategy to maintain the resilience of dry forests to future disturbances, based on our findings.

MATERIALS AND METHODS

Data from the public land survey system, conducted by the U.S. General Land Office, have been widely used in the USA to reconstruct historical vegetation (Schulte and Mladenoff, 2001). Surveys in the study areas were generally done in the late-1800s before widespread expansion of EuroAmerican land uses. The system consists of 9.6 × 9.6 km townships containing thirty-six 1.6 × 1.6 km sections. Surveyors marked quarter corners at the 0.8 km mark and section corners at the 1.6 km mark along section lines. Surveyors were required to record azimuth, distance, species, and diameter of two bearing trees at quarter corners and four trees at section corners. Here we used surveyors' direct estimates of tree diameters. In an accuracy study, we found surveyors estimated diameters with sufficient accuracy to place trees in 10-cm diameter bins (Williams and Baker, 2010). After applying an empirical correction, diameter distributions from bearing trees were 87–88% similar to distributions from plot data (Williams and Baker, 2011), thus are quite accurate. Bearing trees are a statistically valid sample, as they have low bias and error (Williams and Baker, 2010).

We also used section-line data recorded by surveyors. Surveyors in forests were required to record, in order of abundance, the dominant overstory trees and understory plants, often including small trees (seedlings and saplings) and shrubs (Williams and Baker, 2012a). Surveyors also often recorded qualitative estimates of understory tree density. Not all surveyors followed the instructions, thus we limited analysis to the set of surveyors who did so for at least one section-line. The section-line data represent a statistically valid line-intercept estimate of cover (Butler and McDonald, 1983).

To provide data to test hypothesis H_1, we totaled small and large trees in each of the seven study areas and for the composite (**Table 1**, **Figure 1**). Small trees were defined as ≥10 cm but ≤40 cm, except ≤30 cm in the Colorado Front Range, where tree growth is slower (Williams and Baker, 2012a) and ≤50 cm in the western Sierra, where tree growth is faster (Baker, 2014).

Table 1 | Study areas, corresponding number of trees and section-line length in forested area, and the percentage of forest section line-length with seedlings and saplings.

Hypotheses and variables	Front range, Colorado[a]	Coconino Plateau, Arizona	Mogollon Plateau, Arizona	Black Mesa, Arizona	Blue Mts., Oregon	Eastern Cascades, Oregon	Western Sierra, California	Total or mean
Dry-forest study area (ha)	65,525	41,214	405,214	151,080	304,709	398,346	329,943	1,696,031[b]
H_1: SMALL TREES WERE < 20% OF TOTAL TREES								
Number of trees	1055	1643	10,848	2741	7496	11,856	9532	45,171[b]
Small-tree diameters used (cm)	≤30	≤40	≤40	≤40	≤40	≤40	≤50	≤30 to 50
Small trees (% of total trees)	91.8	69.5	51.8	81.1	62.0	62.4	60.9	61.6[c]
Chi-square test result[d]	$X^2 = 3404$	$X^2 = 2517$	$X^2 = 6859$	$X^2 = 6403$	$X^2 = 8267$	$X^2 = 13,326$	$X^2 = 9976$	$X^2 = 48,772$
	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$
H_2: SEEDLINGS AND SAPLINGS WERE PRESENT ON < 20% OF FOREST AREA								
Section-line length (km)	4004	413	4230	1441	5878	3873	2367	22,206
Seedlings/Saplings present (%)	3.8	43.4	13.3	8.0	34.6	57.4	54.9	29.6
Chi-square test result[f]	$X^2 = 657$	$X^2 = 140$	$X^2 = 119$	$X^2 = 150$	$X^2 = 780$	$X^2 = 3385$	$X^2 = 1780$	$X^2 = 1238$
	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p < 0.001$
Seedlings/Saplings dense (%)	0.2	28.8	1.9	-	22.4	30.3	20.0	14.3
Seedlings/sapling pines[e]	0.9	1.4	9.8	7.9	32.7	51.0	42.3	24.8
Seedlings/Sapling firs[e]	0.5	0.0	0.0	0.0	27.1	27.8	39.7	16.4
Seedling/Sapling oaks[e]	0.5	43.3	8.8	7.1	0.0	0.2	42.4	7.6
Seedling/Sapling other trees[e]	2.5	0.4	1.2	2.0	0.3	2.6	25.1	4.0

[a] *Study areas include the Colorado Front Range (Williams and Baker, 2012a), Coconino Plateau, Arizona (Williams and Baker, 2013), Mogollon Plateau and Black Mesa, Arizona and Blue Mountains, Oregon (Williams and Baker, 2012a), Eastern Cascades of Oregon (Baker, 2012), and western Sierra Nevada, California (Baker, 2014).*

[b] *Total.*

[c] *Percentage for the composite across the seven study areas.*

[d] *Degrees of freedom = 1 and N = the number of trees, for all chi-square tests.*

[e] *Seedling/Sapling pines, firs, oaks, and other trees may be overlapping, as a line can have, for example, both pines and firs.*

[f] *Degrees of freedom = 1 and N = the number of 1-km line-lengths, for all chi-square tests.*

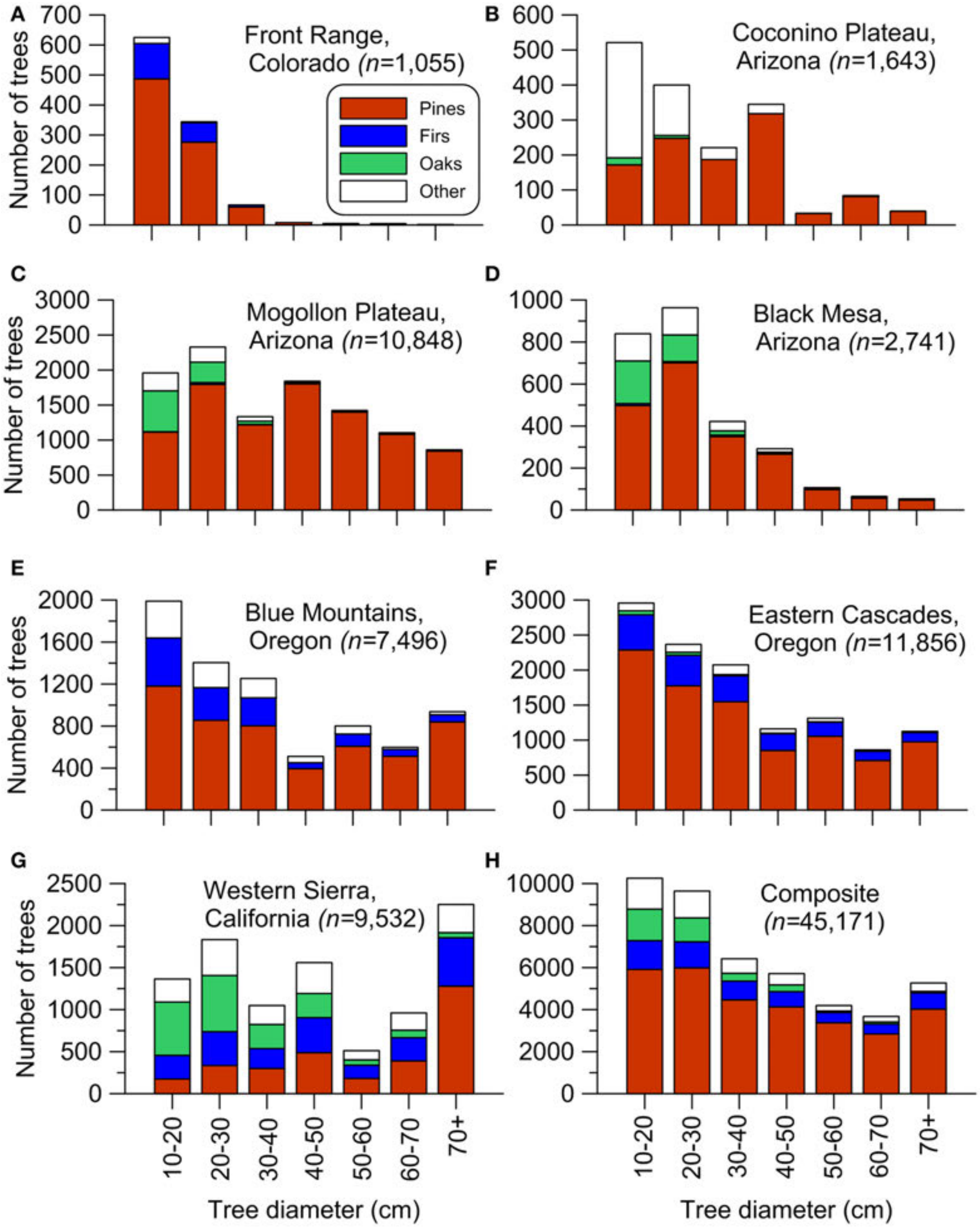

FIGURE 1 | Historical tree size-class distributions for the seven study areas and a composite across all the study areas: (A) Front Range, Colorado, (B) Coconino Plateau, Arizona, (C) Mogollon Plateau, Arizona, (D) Black Mesa, Arizona, (E) Blue Mountains, Oregon, (F) Eastern Cascades, Oregon, (G) Western Sierra, California, (H) The composite of all areas. Distributions use 10-cm bins compatible with the accuracy of diameters measured by the surveyors (Williams and Baker, 2011). Other trees, not found in every area, include *Pinus edulis* and *Juniperus* spp., *Calocedrus decurrens*, *Populus tremuloides*, and *Larix occidentalis*. As in **Table 1**, small trees were defined as trees ≥10 cm but ≤40 cm diameter, except ≤30 cm in Colorado **(A)** and ≤50 cm in California **(G)**.

These diameters generally represent trees that are less than about 140 years old (Bright, 1912; Baker, 2012, 2014; Williams and Baker, 2013). Trees this size today are often thought to have widely established after EuroAmerican settlement because of logging, livestock grazing, and fire exclusion (Covington and Moore, 1994; Allen et al., 2002; Franklin and Johnson, 2012), and thus may be removed in restoration treatments. To test H_1, we used a chi-square goodness-of-fit test of a null hypothesis that small trees were 0.2 of total trees and large trees were 0.8 of total trees. If this null was rejected, we rejected H_1 if small trees were <0.2 of total trees. To control error rates, we Bonferroni-corrected $\alpha = 0.05$, for 8 planned tests, one per study area and one for the composite (**Table 1**, **Figure 1**), to $\alpha = 0.00625$.

To provide data to test H_2, we totaled 1-km section lines for which surveyors recorded understory trees in each of the study areas and for the composite. Similarly, to test H_2, we used a chi-squared goodness-of-fit test of a null hypothesis that the area with seedlings/saplings was 0.2 of the total forested area and the area without seedlings/saplings was 0.8 of the total forested area. If this null was rejected, we then rejected H_2 if seedlings/saplings were found across <0.2 of total forest area. We also Bonferroni-corrected an initial $\alpha = 0.05$ for 8 planned tests.

We used maps of ponderosa pine and dry mixed-conifer forests from Landfire Biophysical Settings (www.landfire.gov). Wildfire area and severity were from raster maps of actual burned area, not fire perimeters, from the Monitoring Trends in Burn Severity (MTBS) program (http://www.mtbs.gov). Insect-caused mortality was from the US Forest Service Forest Health Technology Enterprise Team (http://foresthealth.fs.usda.gov/portal/Flex/IDS). Insect outbreaks were detected using annual aerial surveys. To limit analysis to dry western forests, aerial survey polygons and wildfires were both clipped by the maps of ponderosa pine and dry mixed conifer. The annual sample area varied, but averaged about 9.8 million ha of ponderosa pine and 10.9 million ha of dry mixed-conifer forests (Table S1), about 80% of the 25.8 million ha area of western dry forests.

Comparison of wildfire and insect outbreaks was done for each year both datasets were available. We compared moderate- and high-severity wildfire area, which are the severities with substantial tree mortality, with areas where tree mortality from insects was also substantial, as it was visually detected from aerial surveys. We calculated the rate of wildfire using the fire rotation, which is the number of survey years divided by the fraction of the survey area impacted by fire in those years. The rate of insect outbreaks was determined similarly. Some outbreak areas appeared to overlap in subsequent years and potentially be cumulative. We performed a union and spatial dissolve in GIS to derive a conservative estimate of total area impacted by insect outbreaks over the analysis period. Additional details are in Supplementary Methods.

RESULTS

SMALL TREES HISTORICALLY ABUNDANT AND DOMINANT

Hypothesis H_1 is rejected across all seven study areas and the composite (**Table 1**). Small trees generally dominated historical dry forests, ranging from 51.8 to 91.8% of total trees across the seven study areas and equaling 61.6% of trees in the overall composite (**Table 1**, **Figure 1**). Small trees can be suppressed older individuals, but were predominantly <140 years old (Bright, 1912; Williams and Baker, 2012a). Small trees were somewhat diverse, with pines most abundant, but also firs, oaks and other conifers and hardwoods (**Figure 1**). Hypothesis H_2 is rejected for study areas in California and Oregon, but not in Arizona and Colorado (**Table 1**).

HIGHER RECENT THREAT FROM INSECT OUTBREAKS THAN FROM WILDFIRE

Data from government agencies show that insect outbreaks were recently a more significant threat to dry forests than were moderate- to high-severity wildfires; similar data are not available for droughts. It is conservatively estimated (i.e., consolidating all areas of spatial overlap) that insect outbreaks caused substantial detectable tree mortality in 5,193,752 ha of western dry forests over the 1999–2012 period for which spatial data were available, which is 5.6 times the 934,551 ha impacted by moderate- to high-severity wildfires (Table S1). Mean ratios of insect to fire impact were 4.5 in ponderosa pine and 6.9 in dry mixed-conifer forests (Table S1). At the rates during 1999–2012, it would require 311 years for moderate- to high-severity wildfires to burn once across an area equal to the area of western dry forests, but only 56 years for insect outbreaks to impact this area (Table S1). Rotations for fire varied from 265 years in ponderosa pine to 367 years in dry mixed-conifer forests, and for insects from 53 years in dry mixed-conifer to 59 years in ponderosa pine forests (Table S1).

DISCUSSION

NATURAL DISTURBANCES FOSTERED HISTORICALLY ABUNDANT SMALL TREES AND DIVERSE TREE SIZES

Historical dominance of small trees in dry forests (**Figure 1**) does not support the hypothesis that surface fires generally kept small trees rare. Small trees had successfully recruited and were dominant in all dry-forest areas (**Figure 1**). These small, established trees are given more weight, than smaller, more ephemeral seedlings/saplings, for which evidence is more mixed. Seedlings/saplings were abundant in the majority of areas, except two southwestern landscapes (Black Mesa, Mogollon Plateau) and the Colorado Front Range (**Table 1**). Early scientific sources corroborate limited seedlings/saplings in these areas (Leiberg et al., 1904; Williams and Baker, 2012b). Early foresters emphasized preserving advanced recruitment during logging (Pearson, 1923). Thus, recent high-severity fires do not have unprecedented poor recruitment (Savage and Mast, 2005). Seedling/sapling populations in these landscapes must have fluctuated, since small trees had been able to recruit and dominate all dry forests (**Figure 1**). Particular sequences of fires, droughts, and other disturbances may explain fluctuating seedling/sapling populations (Dugan and Baker, in press), and reinforce the historical role of advanced recruitment.

Dominance of small trees, and even ephemeral seedling/sapling populations in most areas, indicates more imperfect limitation of tree recruitment by historical low-severity fires than previously thought. Other disturbances, including droughts, insect outbreaks, and more severe fires likely killed canopy trees and increased tree recruitment, particularly if followed by pluvials (Savage et al., 1996; Dugan and Baker, in press). The Colorado Front Range and Black Mesa (Williams and Baker, 2012a) had the greatest dominance of small trees (**Figures 1A,D**), and our reconstructions showed these areas had more higher-severity fires (Williams and Baker, 2012a,b). Historical abundance of small trees and importance of higher-severity fires in structuring tree populations across dry-forest landscapes are supported by an independent dataset of tree ages (Odion et al., 2014). Higher-severity fires likely interacted with other disturbances to produce diverse tree sizes that were together more resilient to disturbance than would have been the case if only low-severity fires had occurred and large trees had dominated. Historical dominance by small trees and diverse trees sizes are consistent with long-term persistence and resilience of dry forests after disturbances (Jenkins et al., 2011).

ABUNDANT SMALL TREES AND DIVERSE TREE SIZES CONFER RESILIENCE IN MODERN FORESTS

Modern observations also document key, but contrasting roles for advance recruitment and surviving larger trees in forest resilience after fires, insect outbreaks, and droughts. Higher-severity fires may be followed by variable recruitment, including poor recruitment, lags in recruitment, or abundant recruitment in some areas (Roccaforte et al., 2012), with large, surviving trees and proximity to them important (Bonnet et al., 2005; Haire and McGarigal, 2010).

About a dozen bark-beetles, that kill trees over large areas of dry forests in the western USA, are the major outbreak insects (Bentz et al., 2010; Weed et al., 2013). In this case, larger trees are differentially susceptible, which often leaves smaller surviving trees as the key source of post-outbreak recruitment. Vulnerability of larger trees to bark beetles is related to greater food resources (Raffa et al., 2008). In a 1970s outbreak of mountain pine beetle (*Dendroctonus ponderosae*) in ponderosa pine in Colorado, tree survival was substantially higher for trees <20 cm diameter (McCambridge et al., 1982). Similarly, western pine beetles (*Dendroctonous brevicomi*s) kill relatively few trees <40 cm (Miller and Keen, 1960). However, *Ips* in Arizona preferentially kill smaller trees (Negrón et al., 2009). Nonetheless, advance recruitment generally dominates post-outbreak recruitment. After spruce beetle (DeRose and Long, 2010) and mountain pine beetle outbreaks (Astrup et al., 2008), small trees present before outbreaks dominated post-outbreak recruitment. Since these small trees were more diverse than pre-outbreak canopy trees, post-outbreak forests may have greater resilience to future outbreaks (Diskin et al., 2011; Kayes and Tinker, 2012).

Drought often also differentially kills the largest, oldest trees, with less mortality in small and mid-sized trees (Allen et al., 2010), thus also leaving advance recruitment. Drought effects on tree mortality can be widespread and affect forests for centuries (Allen et al., 2010). Drought also influences the occurrence of wildfires, insect outbreaks, and regional tree mortality (Allen et al., 2010), thus it is difficult to parse the impacts of drought alone.

The upshot is that both small trees and surviving larger trees and a diversity of tree species provide resilience to disturbances. Surviving larger trees are particularly important after higher-severity fires and abundant small trees are particularly important after insect outbreaks and droughts.

RESTORING AND MAINTAINING THE BET-HEDGING RESILIENCE OF HISTORICAL FORESTS

Current restoration strategies that seek to increase forest resilience focus predominately on impacts from severe wildfires, but bark-beetle outbreaks and other insects affected 5.6 times the area of western dry forests impacted by moderate- to high-severity fires over the most recent 14-year period (1999–2012). Current rates of moderate- and high-severity fire, with a combined rotation of 311 years (Table S1), would likely not prevent recovery of old-growth forests in the interlude between fires, but rates of insect outbreaks, with a rotation of 56 years (Table S1), could prevent recovery of most older dry forests. Previous research, using the same data sources, in a more limited and lower-elevation area in the southwestern United States, found that beetle-outbreaks affected 2.5–4 times as much area as moderate- to severe wildfires (Williams et al., 2010). Both wildfires (Dennison et al., 2014) and beetle-outbreaks (Bentz et al., 2010; Weed et al., 2013) are increasing in parts of the western United States. Future outcomes are uncertain and complex, however, as beetle-outbreaks can affect wildfire probability (Simard et al., 2011), and as tree mortality occurs, both beetle outbreaks and wildfires could become self-limited (Williams et al., 2010).

Ecological restoration of public dry forests in the western USA is increasingly a goal, because these forests were altered by unsustainable logging, livestock grazing, and fire exclusion that allowed abundant small trees to recruit (Covington and Moore, 1994). Retaining older trees, while removing most small trees up to ages or sizes of trees recruited since EuroAmerican settlement (Figure S2A), is thus often a restoration focus (Covington and Moore, 1994; Allen et al., 2002; Abella et al., 2006; Franklin and Johnson, 2012). Typical upper tree age and size limits are 120–150 years old or 30–50 cm diameter (Abella et al., 2006; Franklin and Johnson, 2012).

We show here, however, that these small trees were the tree sizes historically dominant in these forests (**Figure 1**, **Table 1**), thus removing most small trees so they are no longer dominant is not ecological restoration. There are also efforts underway to increase resilience of forests to droughts by removing most small trees and lowering stand density. However, stand density does not appear to play a major role in level of tree mortality from drought (Ganey and Vojta, 2011). Thus, strategies to reduce most small trees are neither restorative nor very effective.

We suggest diverse historical tree sizes and abundant and dominant small trees long provided bet hedging in dry-forest landscapes subject to unpredictable disturbances. These forests can be more effectively restored and their resiliency to future disturbances increased by maintaining or restoring the historical abundance, dominance, and diversity of small trees, while also restoring large trees depleted by logging (Figure S2B). This can be achieved with historically congruent diversities of forest structures across landscapes, based on GLO and other spatial reconstructions. This bet-hedging landscape approach to ecological restoration is consistent with long-term persistence of historical forests, the high current threat from insects, and would likely confer more resilience to disturbances, that may all increase in the future, than would just retaining larger or older trees across large areas.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: http://www.frontiersin.org/journal/10.3389/fevo.2014.00088/abstract

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S1 Text. Why CFIs and ITFIs underestimate PMFI/FR

Since both mean composite fire interval (CFI) and mean individual tree fire interval (ITFI) typically underestimate the population mean fire interval or fire rotation (PMFI/FR), it is logical to infer that both methods are missing longer intervals because of biased estimates (Table 1 main text). Compositing alone could explain nearly the whole bias in CFI measures, as explained below, but ITFI measures are not composites and are still biased, although less so. The most likely explanation for bias in ITFI measures, and also a contributor to bias in CFI measures, as estimators of PMFI/FR, is targeted sampling. These potential sources of bias are reviewed in detail here.

Compositing overcompensates, destroys long fire intervals, and restriction rules do not remedy this

Scarring fraction, compositing, and widespread over-compensation

The purpose of compositing is to compensate for the incomplete scar record on individual trees, since trees can often resist scarring even if burned (Baker and Dugan 2013). Scarring fraction (SF) is the fraction of burned live trees that survive a fire but receive a scar. Studies of SF are few (e.g., Collins and Stephens 2007, Stephens et al. 2010). A study of 16 fires in ponderosa pine forests in northern Arizona found a mean SF of 0.375, ranging from 0.121 to 0.728 across 52 plot samples (Baker and Dugan 2013).

Given a particular SF, how many trees must be sampled or composited to compensate for SF < 1.0 (Baker and Dugan 2013)? The minimum is to have a sample size that has a high probability of recording each fire on at least one scarred tree. The probability, P, of at least one tree scarring in a sample of n live trees, for a scarring fraction, SF, is given by:

$$P = 1.0 - (1.0 - \text{SF})^{n} \quad (1)$$

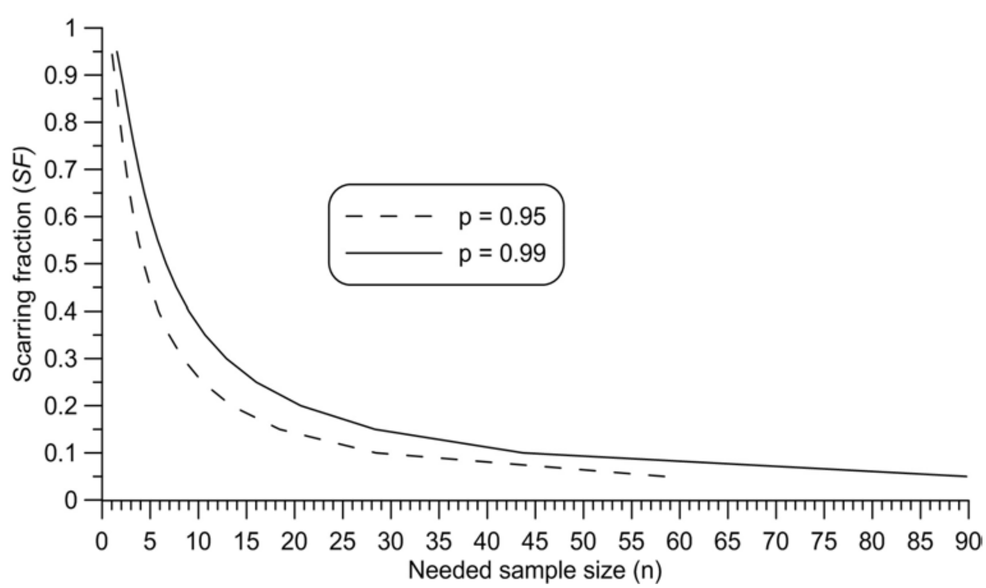
and the corresponding estimate of n, for a particular SF, is given by:

$$n = \log(1.0 - P) / \log(1.0 - \text{SF}) \quad (2)$$

The necessary sample sizes to achieve a probability ≥ 0.95 or 0.99 of detecting a fire are modest, typically < 20 trees, whether scarred or not, to detect fires with SF < 0.25 (Figure A).

However, this equation does not adjust for scar healing. Scars can, but do not always, heal from the sides and disappear under new bark unless subsequent fires occur (Baker and Dugan 2013). However, Fiegener (2002) examined over 8,000 stumps and snags in a Sierran mixed-conifer forest and found only 2% with scars. An empirical study of scar healing after fires in northern Arizona ponderosa pine forests

Figure A. Scarring fraction and its effect on the sample size needed to have either a 0.95 or 0.99 probability of scarring at least one tree.



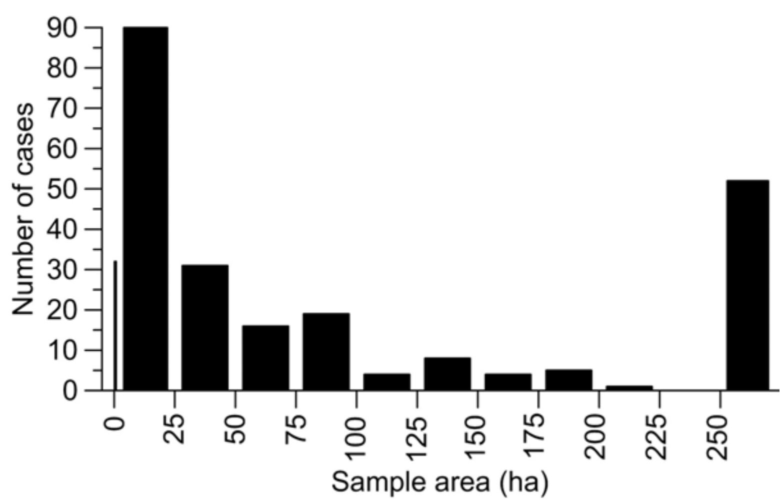
showed that larger initial scars have longer expected healing times and subsequent fires increased healing times of all scars (Baker and Dugan 2013). Healing rates from this study can be used to estimate the needed sample size to find at least one unhealed scar for a fire after 100 years, or another time since fire, using an equation developed from forests differing in time since fire:

$$\text{Effective mean SF} = \text{Initial SF} \times \exp(-0.0125 \times \text{Time since fire}) \qquad (3)$$

For example, if expected mean SF is 0.400, then six sample trees would likely ($P = 0.95$) contain at least one scar (using Eq. 2 or Figure A) from a fire burned recently. In contrast, the effective mean SF if that same fire had burned 100 years ago, and scars had healed since then, would be 0.115, from Eq. 3, thus requiring a sample of about 26 trees each > 100 years old (Eq. 1, Figure A). Similarly, a lower SF of 0.200 would require 51 trees > 100 years old for a fire 100 years ago. These calculations suggest that sufficient trees to detect fires in historical landscapes could likely be obtained from unlogged areas that are on the order of about 1 ha in area or even less.

In contrast, most compositing is from areas far too large for the area and number of sample trees usually needed to compensate for SF < 1.0. In the merged dataset of 342 sites, only 262 reported area sampled. Of those, only 32 (12.2%) were from areas < 1.0 ha (Figure B). One concern is whether SF rates estimated in this study are higher than they would have been in historical forests, because fire exclusion increased fuel loads in modern forests, likely increasing SF. Some effect is likely, but the effect would not change the general pattern of widespread overcompensation. First, if a preceding fire occurred within 30 years, then SF was reduced from a mean of 0.393 to 0.324, only an 18% reduction, in the Baker and Dugan (2013) study. This would have a minor effect of increasing the number of needed sample trees from 26 trees to 31 trees, having almost no effect on the widespread pattern of over-compensation evident in Figure B. Second, even in the extreme case of a 0.05 mean SF in historical forests, assuming a historical fire rotation of < 10 years (Stephens et al. 2010), only 208 trees, whether scarred or not, would be needed after 100 years to achieve a probability ≥0.95 of detecting a fire. This could be obtained in most historical dry forests in < 2-3 ha. Even at this extreme level of SF, only 18.7% of the 262 sites were from areas < 3 ha, thus 81.3% of studies were over-compensating.

Figure B. Number of cases (n = 262 total) by sample area (ha). Many cases did not report area. The first bin is from 0-1 ha, the remainder are 25-ha wide, and the last bin is for areas > 250 ha.



This over-compensation particularly biases CFI estimates toward values that are too short, since mean CFI declines as sample area or number of sample trees increases (Arno and Petersen 1983, Baker and Ehle 2001, Everett 2003, Kou and Baker 2006a, b). ITFI and FR estimates, in contrast, do not systematically decline with larger samples, and may even become more precise. Compositing records across an area or number of trees that is too large could explain why CFI estimates are too short relative to FRs, but cannot explain why ITFI estimates, which do not use compositing, are also too short.

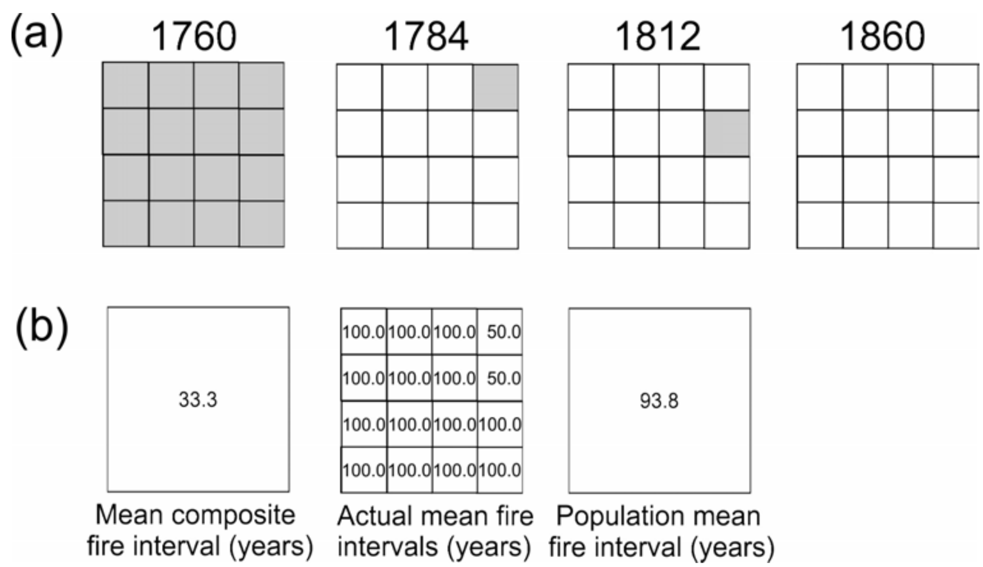
Compositing not only over-compensates, but also destroys long fire intervals that are real

Compositing is a processing step, separate from finding and collecting an adequate sample. Several methods can be used to process sample data, including calculating mean ITFI (Dugan and Baker 2014), or estimating FR, thus the compositing step is not essential. How does the compositing step contribute error if used to estimate PMFI/FR? Most fires are small and only a few are large (Baker and Ehle 2001). When a composite list is created, and intervals are calculated among fires in the list, each small fire year counts the same as a large fire year. Even though some compositing might offset incomplete evidence, at the same time it destroys other evidence. Longer fire-free intervals that are real occur in unburned parts of landscapes adjacent to where small fires occurred, and some long intervals that are false because scarring is incomplete also occur. However, all these long intervals, whether real or false, are erased across the whole sample area when a composite list is created, rather than disappearing only in the area where a small fire occurred (Figure C). Since longer intervals, some of which are real, are all lost to compositing, this in part explains why mean CFI underestimates PMFI/FR.

CFI restriction rules are ad hoc, inconsistent, and likely insufficient in excluding small fires

Some suggest that there is only a problem with mean CFI and its use if it is presented without omitting spot fires: "...this becomes a problem only if the fire chronology is presented with all fires, even the smaller spot fires, and is interpreted by the reader as if the chronology indicates how often the entire stand burned" (Stephens et al. 2003 p. 1091).

Figure C. How compositing destroys long fire intervals that are real: (a) a landscape that burned (shaded area) in each of three fire years between 1760-1860, and (b) the actual mean fire intervals (center panel), the mean composite fire interval for the landscape (left panel) calculated from a composite list of fires (1760, 1784, 1812, ending in 1860), thus three intervals in 100 years = 33.3 years, and the PMFI (right panel), which is the grand mean of the actual mean fire intervals (1500 years/16 = 93.8 years). Note that the reason that the mean CFI underestimates the PMFI is that compositing treats the 1784 and 1812 fire years as though they burned the same land area as the 1760 fire year, which also eliminates the 100.0-year fire intervals that occurred over most of the landscape.



Restriction rules are traditionally applied to filter out fires, like spot fires, that are small, using the number of fire scars or the percentage of total scarred trees that record a fire year (e.g., 10%, 25%). However, no way is known to objectively identify a spot fire or other small fire that should be omitted, since fire-size distributions are typically nearly linear on a log-log plot and have no natural breaks (Kou and Baker 2006b). Also, distributions vary in slope among forest types and environments (Kou and Baker 2006b), so imposing a particular filter (e.g., 25%) has varying effects. This means that restriction rules are ad hoc and inherently inconsistent in their effects.

Moreover, 10% and 25% filters that are typically applied, may be insufficient to limit the fires that should be included in a composite list, if the goal is that mean intervals between fires in the list estimate the PMFI/FR. In a spatial reconstruction of fire sizes in dry forests, Farris et al. (2010) found that 414 total fires occurred in their study area from 1937-2000, but only 21 fires (5.1% of total fires) accounted for 97% of total burned area. This suggests a restriction rule would have to exclude 95% of fires to limit a composite to the fire years that account for most of the total burned area, which would more likely accurately estimate PMFI/FR. Together, the ad hoc, inconsistent, and insufficient extent of traditional restriction rules in part explain why mean CFI underestimates PMFI/FR.

Censoring incomplete fire intervals leaves out long intervals in both CFI and ITFI estimates

Fire-history data contain incomplete intervals at the beginning and end of a period of record unless those periods begin and end with fires (Polakow and Dunne 1999). Incomplete intervals can be included or omitted ("censored") in analysis of fire-interval data (Polakow and Dunne 1999). Censoring (i.e., using only scar-to-scar intervals) biases both mean CFI and ITFI by omitting incomplete intervals at the beginning or end of a tree's record. Incomplete fire intervals occur on most trees, but longer intervals have more chance, than shorter intervals, of appearing as incomplete intervals, indicated by no scars or one scar on a tree (Kou and Baker 2006a). Simulation has shown that in a landscape subject to low-severity fires at modest intervals (e.g., 50 years), actual intervals at some locations may be several times longer (Kou and Baker 2006a) and even up to an order of magnitude longer than the mean interval (Parsons et al. 2007). These are real intervals that occur by chance, not an artifact of incomplete scarring. Censoring is biased against these expected long fire intervals and leads to estimates of PMFI/FR that are too short and have reduced variability, since longer intervals are omitted (Kou and Baker 2006a). These effects from censoring were also found in two studies in Mediterranean shrublands, in which censoring reduced the scale parameter (indicator of length of fire intervals) of a Weibull fire-interval distribution and also reduced estimated variability in fire intervals (Polakow and Dunne 1999, Moritz et al. 2009).

These censoring effects have ecological implications in dry forests subject to periodic fires, since most composited fire-scar records, which are traditionally censored, lack evidence of the long intervals needed for tree regeneration and survival of fire-intolerant species. We suggested that the interval before the first fire scar (origin-to-scar interval--OS) on individual trees may record the fire-free period needed for trees to successfully regenerate (Baker and Ehle 2001), since both wide-area and local processes producing long intervals should be recorded as OS intervals. Mean OS intervals are, in fact, usually much longer than mean scar-to-scar intervals in the same stands, and many are sufficiently long to allow tree regeneration (Baker and Ehle 2001). Mean OS intervals in ponderosa pine forests were 51 years in the Black Hills (Brown et al. 2008), 55.4 years in Rocky Mountain National Park (Baker and Ehle 2003), 81 years across five studies (Baker and Ehle 2001), and 101.5 years in one case in northern Arizona (Van Horne and Fulé 2006). Arguments can be made for and against including the OS interval in CFI estimates (e.g., Baker and Ehle 2003, Van Horne and Fulé 2006, Stephens et al. 2010). However, long intervals that are real do occur and are directly censored by traditional use of only scar-to-scar intervals in CFI and ITFI estimates, contributing to underestimation of PMFI/FR by CFI measures.

Targeted sampling likely a significant source of underestimates of PMFI/FR by ITFI, as well as by CFI

<u>Why researchers target fire-history evidence and why it remains a concern for estimating PMFI/FR</u>

Researchers target fire-history evidence to increase the length of record and maximize the data obtained with minimal physical effort and damage to trees (Farris et al. 2013). If only 50 scarred trees can be sampled, more fire years per scarred tree and a longer mean length of record will nearly always be obtained from 50 trees selected by targeting than from a random sample.

Unfortunately, targeting fire-history evidence at the scale of individual trees, sampling areas, and landscapes produces biased estimates of fire history (Lorimer 1985, Johnson and Gutsell 1994, Baker and Ehle 2001). The consequences are generally that estimates of historical PMFI/FR are too short and fire-severity is underestimated. The magnitude of targeting and its effects is now better known. Targeting remains common in fire-history studies, as illustrated in Table A, which shows that targeting of individual trees, particularly multi-scarred trees and old trees, was widespread, almost universal for multi-scarred trees, and almost 1/3 of studies placed study plots where there were concentrations of scarred trees and old trees.

Table A. Percentage of 342 sites in which various types of targeting sampling were used.

Targeting type and measures	**Yes**	**No**	**No explanation**
1. Target trees to get best information or longest record of fires?			
Number of cases	114	68	160
Percentage of yes/no (%)	62.6	37.4	-
2. Target multi-scarred trees?			
Number of cases	235	15	92
Percentage of yes/no (%)	94.0	6.0	-
3. Target clusters of scarred trees?			
Number of cases	37	9	296
Percentage of yes/no (%)	80.4	19.6	
4. Target scars on dead wood?			
Number of cases	270	27	45
Percentage of yes/no (%)	90.9	9.1	
5. Target tree species thought to better record fire			
Number of cases	12	13	317
Percentage of yes/no (%)	48.0	52.0	
6. Target plot locations in old forests and concentrations of scars			
Number of cases	73	153	116
Percentage of yes/no (%)	32.3	67.7	
7. Target study areas in old forests and concentrations of scars			
Number of cases	18	168	156
Percentage of yes/no (%)	9.7	90.3	

Specific studies of some of these types of targeting are now available (Baker and Ehle 2003, Van Horne and Fulé 2006, Kou and Baker 2006a, Brown et al. 2008, Farris et al. 2010, 2013), but the most significant types are less studied. Studies whose findings supported targeted sampling (e.g., Van Horne and Fulé 2006) for some purposes did not study using targeted sampling for estimating PMFI/FR, the focus here, thus targeted sampling has not been supported for this purpose.

Targeting individual trees

Targeting individual trees typically includes a bias component and a non-random sampling component. The bias component is from omitting trees with no scars or one scar and preferentially or exclusively using trees with multiple scars. The non-random sampling component comes from purposely choosing particular multi-scarred trees rather than randomly sampling them.

Significant bias is likely from omission of trees with no or single scars, which are traditionally omitted because only scar-to-scar intervals provide estimates of complete fire intervals. However, no scars or single scars on a tree may be false, because fires do not scar every tree that burns, but no or single-scarred trees also include real but incomplete fire intervals. No or single scars that represent real, incomplete long fire intervals are more likely where fire intervals also are longer (Kou and Baker 2006a). More long intervals and more of the length of long fire intervals are inherently present on unscarred or single-scarred trees, assuming tree ages are similar to those of multiple-scarred trees. Since longer fire intervals are more likely to be omitted by individual-tree targeting, all types of estimators (i.e., CFI, ITFI) from multi-scarred trees are biased toward being too short (Kou and Baker 2006a). FR estimates are also biased toward being too short if only multi-scarred trees are sampled, because trees with no scar or one scar can indicate places where a fire did not burn, and these omissions inflate area burned for that fire year, and shorten the estimated FR.

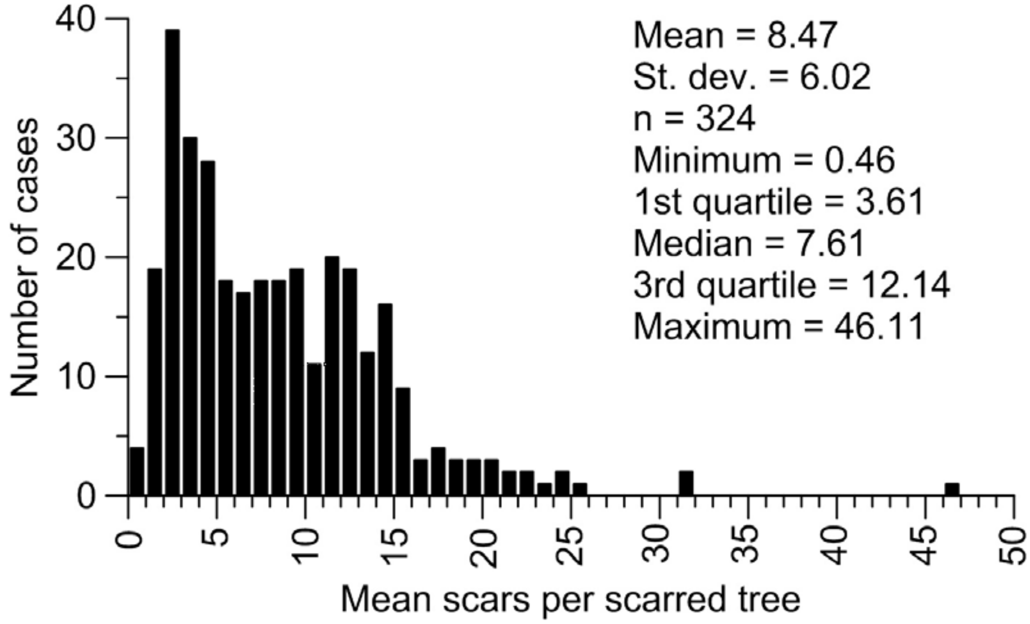
How does targeting trees with more than one scar (multi-scarred trees, recorder trees, and open-scarred trees) lead to CFI and ITFI values that underestimate PMFI/FR? Trees have visible, open scars because they are the trees that have had fires often enough to prevent healing. We found that the time for a fire scar to heal had a median of 38 years and was <100 years for 89% of scars (Baker and Dugan 2013). Longer fire intervals, that are real, have a high probability of not being selected by targeting trees with > 1 scar, because longer intervals often are expressed as no scars or one scar. Of course, long intervals can be an artifact of incomplete scarring, so that including all of them would lead to bias, but excluding all of them does too. Targeting trees with > 1 scar omits trees most likely to have long real fire intervals and selects trees with short fire intervals.

The substantial numerical dominance of unscarred and single-scarred trees in dry forests suggests omission of longer real fire intervals by individual-tree targeting of trees with > 1 scar could be among the most significant sources of bias in CFI estimates and possibly the main source of bias in ITFI estimates. In a sample of 906 pre-EuroAmerican trees we collected on 8 transects in northern Arizona, near Flagstaff and in Grand Canyon National Park, 779 trees had no scar (86%), 111 had one scar (12%), and only 16 trees had two or more scars (2%). In a mixed-conifer forest in the western Sierra, 98% of nearly 8,000 stumps and snags examined for scars did not have any scars, only 13 (0.2%) had one scar, and 48 (0.6%) had two or more scars (Fiegener 2002). Multi-scarred trees are rare in modern landscapes.

The magnitude of effects of omitting trees with no or one scar is unstudied, but within the set of multi-scarred trees with ≥2 scars, the effect of restricting fire history to increasing levels of multiple-scarring was studied (Fiegener 2002). To gauge how relevant this study is to multi-scarred sets of trees actually used in fire histories, I analyzed the number of scars per tree found by studies in the merged dataset, although data were available for only 324 cases. First, I calculated mean number of scarred trees, over each site's sample period, which is less than the total number of sample trees, since trees usually each cover only part of the sample period. Then, I calculated mean number of scars per scarred tree as total number of scars/mean number of scarred trees from the summary table for the FHX file in FHAES.

A histogram of mean scars per scarred tree had a mean of 8.47 scars/sample tree and a median of 7.61 scars/sample tree (Figure D). Fiegener (2002) found that restriction to ≥3 scars reduced ITFI from 17.4 years to 16.8 years (to 96.6%). This is above the minimum of 0.46 scars/tree in the distribution (Figure D). Restriction to ≥4 scars, just above the 1st quartile in the distribution, reduced ITFI to 15.8

Figure D. Histogram of the ratio of mean scars per sample tree in the 342-case merged dataset, and parameters of the distribution. Mean scars could not be calculated for 18 cases.



years (to 90.8%), restriction to ≥7 scars, just below the median, reduced ITFI to 14.6 years (to 83.9%), and restriction to ≥10 scars, below the 3rd quartile, reduced ITFI to 13.2 years (to 75.9%). These responses show that the more scars on a multi-scarred tree, the shorter is the mean ITFI estimate.

Roughly the median level of multi-scar targeting (≥7 scars), which reduced ITFI to 83.9%, or by 16.1%, closely matches the -16.64% bias in Weibull mean ITFI relative to PMFI/FR-total scarred trees/plots (Main text--Table 1). Other ITFI measures have biases of -2.71 to -29.71 (Main text-Table 1), so the close match with the Weibull Mean ITFI could possibly be a coincidence. Mean CFI-10% scarred also declined from 6.7 years to 5.7 years (85.1%) with restriction to ≥3 scars, but fluctuated or increased with higher levels of restriction (Fiegener 2002). Thus, the response of ITFI to targeting multi-scarred trees could explain much of why ITFI underestimates PMFI/FR, but the response of CFI estimators was inconsistent, suggesting it is possible too, but also may not be a main effect for CFI measures.

Mean CFI, ITFI, and FR estimates are further biased and shortened by non-random sampling of multi-scarred trees. Van Horne and Fulé (2006) found a statistical difference, using 95% confidence intervals, between mean CFI for an individual-tree targeted sample and a large census. Comparison of a random sample and a targeted sample, each of 40 trees, shows that mean CFI in the targeted sample was 79.1% (2.23/2.82) of the mean CFI in the random sample for all fires, 98.4% (3.00/3.05) for mean CFI-10%-scarred, and 86.9% (5.43/6.25) for mean CFI-25%-scarred. Farris et al. (2013) re-analyzed the Van Horne and Fulé (2006) dataset and added two other datasets, which together showed targeted samples had a mean CFI-all fires that was 78.9-112.5%, a mean CFI-10% scarred that was 93.5-131.4%, and a mean CFI-25% scarred that was 80.0-96.1%, of the corresponding mean CFI from a probabilistic sample. In Brown et al. (2008), a target-supplemented sample (their Figure 4d) had a mean CFI that was 88.9% (24/27) of that from a systematic plot sample (their Figure 4c). ITFI and FR estimates from recorders are similarly affected. Van Horne and Fulé (2006) found that mean ITFI in a targeted sample was 83.3% of mean ITFI in a random sample. Everett (2003) sampled fire-scarred trees using a grid at two sites and chose the closest fire-scarred tree, thus a probabilistic sample without targeting multi-scarred trees. No comparable estimate from non-random sampling and a targeted sample was made, but Everett's estimated ITFIs were in the 3rd and 4th quartiles of the distribution of estimated ITFIs in the 96-case calibration dataset, consistent with the possibility that ITFIs were long because of lack of targeting.

Farris et al. (2013) showed that individual-tree targeting and non-random sampling even led to ratio-based estimates of FR at three sites that were reduced to 85.5%, 88.3%, and 94.8% of FR estimates from equal-sized probabilistic samples. As suggested earlier, this may be because places with long fire intervals that are real are omitted. These omissions may be places that particular fires did not burn, thus fire size for those fire years is inflated, leading to FR estimates that are too short.

Another impact of individual-tree targeting is reduced completeness of the fire record and over-representation by small, low-severity fires. Fiegener (2002) found that targeting trees with ≥ 5 scars reduced detected fires from 76 to 68 (to about 89.5%), but reduced detection of larger fires to 77%, thus increasing the proportion of small fires in the sample. Baker and Ehle (2003) also found that targeting multi-scarred trees identified and emphasized more small, low-severity fires, including one-tree fires that are another central source of bias in CFI estimates (Baker and Ehle 2001). A non-targeted sample did as well or better at identifying large, low-severity and mixed-severity fires (Baker and Ehle 2003). Also, 18% of 60 total fires and 30% of the most ancient fires (pre-1700), including a significant high-severity fire, found in a non-targeted sample would have been missed if only trees with ≥4 scars were sampled (Baker and Ehle 2003). Targeting multi-scarred trees thus leads to an incomplete fire record, missing significant fires, and a bias toward small fires that produce CFI and ITFI estimates that are too short.

A related type of individual-tree targeting focuses only on "recorder" trees with at least one previous fire scar (thus ≥2 fire scars), which are thought to preferentially record fires, leading to a more complete fire record. To have increased the probability of receiving a subsequent scar, these trees had to have been effectively open, with a scar lacking bark, at the time of the next fire. Previously scarred trees do have a much higher probability of receiving a new scar than do unscarred trees (Baker and Dugan 2013). However, they are much less common than unscarred trees, and unscarred trees appear to typically be scarred at a sufficient rate in a fire to outnumber scars on recorder trees. For example, in a single fire, 73% of scarred trees were first scars and only 27% were recorders that had a previous scar (Stephens et al. 2010), suggesting previously scarred trees were poorer recorders of the fire, in terms of number of scars per unit area, even though scarred at a higher rate. In a larger Rocky Mountain National Park (RMNP) study (Baker and Ehle 2003), for 24 fires that showed up both as first scars and on recorders, 62% of the scars documenting these 24 fires were not on recorders, while 38% were on recorders, a significant difference ($\chi^2 = 4.76$, $p = 0.029$) and lower rate per unit area for recorders, just as in the Stephens et al. study. Moreover, we found 60 total fires, and 32% of these fires showed up only as first scars while 28% of the 60 fires showed up only on recorders, suggesting neither source alone provides a complete fire history. However, there was not much difference in the ability of more numerous unscarred and less numerous recorder trees to record complete histories of fire. Moreover, recorders have the same additional biases, as estimators of PMFI/FR, as do other multi-scarred trees, as reviewed above.

Targeting open-scarred trees often aims at trees with a cat-face or deep semicircular wound, which typically also means they are multi-scarred trees and qualify as recorders. In a study of a single fire in a California Sequoia grove, 68% of open-scarred trees were scarred in a 1797 fire, but only 20% of intact trees were scarred (Kilgore and Taylor 1979). Across many fires, in a California study, a significantly greater mean fraction (0.22) of oaks with open scars at the time of a fire had scars from the fire than did intact trees (0.09), but twice as many intact trees on the sites had scars since there were 5.5 times as many intact trees as trees with open scars (McClaran 1988). This pattern is similar to that of recorders. Mean CFI did not differ between open-scarred and intact trees at one site, but open-scarred trees had 27% fewer fire dates (McClaran 1988). Thus, targeting open-scarred trees thought to be better recorders of fires also leads to omission of fires and the other biases of multi-scarred trees.

Species targeting focuses on particular tree species thought to be better recorders of fire. For example, one might obtain fire scars from ponderosa pine trees on the edge of piñon-juniper woodlands, because the ponderosa are thought to have a better record, from a higher SF (e.g., Miller and Rose 1999). However, fires that burned the ponderosa likely did not penetrate into the woodlands much, if at all (Huffman et al. 2008), thus the apparent difference in SF may reflect real differences in burning rates. To avoid a targeting effect from assuming that the tree species with more scars has a more complete record, data can be acquired from piñon-juniper woodlands and adjusted for their lower scarring fraction. This is what the ATFI method allows, a separate SF for differing trees on the same site (Kou and Baker 2006a).

Individual-tree targeting of older trees for sampling occurs because older trees have a potentially longer record (Farris et al. 2013). This type of targeting may also occur if trees with multiple scars are targeted, since trees generally must get older before they have multiple scars. By definition, individual trees with long fire-scar records have a history of only low-severity fires at that tree, thus a targeted sample of only old trees is certain to indicate a long history of low-severity fire. When fire is moderate- to high-severity, evidence of fire severity on surviving older trees underestimates fire severity in the stand (Hessburg et al. 2007). A targeted sample of old trees in a landscape with trees of other ages thus provides strongly biased evidence about the fire severities that affected the stand.

<u>Targeting sampling areas in landscapes</u>

Targeting particular landscapes or parts of landscapes also leads to bias, generally toward CFI and ITFI estimates that are too short relative to PMFI/FR, since the methods of individual-tree targeting are also used at the landscape scale. Researchers seeking to reconstruct pre-EuroAmerican fire regimes may select parts of landscapes with concentrations of multi-scarred trees, recorders, open-scarred trees or catfaces, and old trees or old-growth forests. In almost 1/3 of the cases where targeting or lack of it was reported, researchers located plots specifically in these areas and in about 10% of cases researchers chose study areas with these concentrations (Table A). These plot locations and study areas may contain long fire records and many fire scars, and are attractive to researchers seeking long fire records (Farris et al. 2013). However, these parts of landscapes also are forests that had a predominance of low-severity fire and little to no mixed- or high-severity fire for hundreds of years, as most trees would otherwise be younger. As explained in the main text, researchers may target areas with abundant fire scars and omit or reduce sampling in areas that lack or have few scars, then also may inappropriately assume that fire history in areas with abundant fire scars also applies to areas with few or no fire scars.

In contrast, probabilistic sampling areas, particularly if appropriately small (e.g., 1 ha) may commonly lack scarred trees or have few. Heyerdahl (1997) sampled using plots located in a grid, thus without targeting sampling areas in landscapes, and found that scarred trees were lacking in more than half the plots at three study sites. These areas could in part have had few scars because of a low scarring fraction, but could also have been areas that really did not burn for a long period. If the latter, then omitting these long intervals, that are real, would bias results toward underestimating fire severity and bias estimated rates of low-severity fire toward shorter intervals. This kind of targeting is not clearly rejected by supporters of targeting (Farris et al. 2013 p. 1030), although they encourage "...clearly defining the inference space, not extrapolating to unrepresentative areas..." This kind of targeting did clearly include extrapolating to unrepresentative areas in past fire histories that are the subject of this paper. I am not singling out particular authors, as most used sampling methods that were common practice at the time, largely aimed at finding and sampling the best evidence (Farris et al. 2013).

However, targeting of old forests, that inherently have a history of low-severity fire, likely explains the unexpected findings of landscape analyses of fire history that did not use targeting. When an objective, large sample (303,156 ha) of historical dry forests was studied in the Pacific Northwest using early aerial photography, middle-aged forests resulting from mixed- and high-severity fires were found to have dominated historical landscapes and old, park-like forests, exclusively with low-severity fire, were found to have been comparatively uncommon (Hessburg et al. 2007 p. 7): "Moreover, old, park-like or similar ponderosa pine stand structures did not dominate the landscapes, and this was particularly perplexing because this was to be the signature outcome of frequent low severity fires." Similarly, spatially extensive reconstruction across landscapes using the early land surveys, found evidence of abundant denser and younger forests from mixed- and high-severity fire across dry forests in northern Arizona, the Colorado Front Range, and the Blue Mountains in Oregon, where previous fire-history studies had found predominantly low-severity fire and open, low-density old forests (Williams and Baker 2012). Finally, stand age data from the Forest Inventory and Analysis Program also showed that young and middle-aged forests, not park-like old forests, were most common historically in ponderosa pine and mixed-conifer forests across relatively undisturbed parts of the western USA (Odion et al. 2014). These studies with probabilistic sampling at the landscape scale show that targeting parts of landscapes containing old forests with abundant fire scars led to very substantial over-estimation of the historical extent of low-density old forests that predominantly had low-severity fire.

Unstudied fire severity in dry forests also inflates low-severity fire rates

Estimates from CFIs, ITFIs, and FRs likely often included all fire severities, not just low severity, and the low-severity rates alone are thus likely longer. Fire severity has been relatively infrequently studied in dry forests. Baker and Ehle (2003) found that only about 25% of fire-scar studies also collected the age-structure data needed to determine whether higher-severity fires occurred historically. Of the 335 cases in the merged dataset with data, 254 (74.3%) did not study fire severity, 80 (23.4%) did study fire severity, and 8 (2.3%) did not explain whether they studied fire severity. Most studies that did analyze fire severity did not distinguish fire severities when they reported fire rates (e.g., Taylor and Skinner 1998). Where fire severity was studied, some mixed- or high-severity fires were nearly always found, but few studies estimated PMFI or FR for the higher-severity fires. Thus, most fire-history studies provide estimates of rates for all fires combined, including low, moderate- and high-severity fires.

The potential effect of combined fire severities on estimated rates for low-severity fire can be illustrated by subtracting, using partitioning (Baker 2009), reported rates of moderate- to high-severity fire from rates of low-severity fire. Odion et al. (2014) reported historical rates of combined moderate- to high-severity fire ranged from 115-128 years in the eastern Cascades of Oregon to 319 years on the Mogollon Plateau. I used the full range of 115-319 years to remove the moderate- to high-severity component, and found that 10-year combined PMFI/FRs would have a 10.3-11.0 low-severity component, but 50-year combined PMFI/FRs would have a 59.3-88.5-year low-severity PMFI/FR after removing the moderate- to high-severity component (Table B). I did not apply an adjustment, for this fire-severity issue, to estimated PMFI/FRs because the adjustments are imprecise and have a large range, and because sites where fire severity was unstudied did not necessarily have higher-severity fires. Nonetheless, this finding illustrates the limitation of unstudied fire severity, and shows that many estimates of low-severity PMFI/FR are likely low estimates.

Table B. Partitioning combined fire rotations (FR) into components for low- versus moderate and high-severity fire for three example levels of combined fire rotations.

	10-year combined-severity PMFI/FR	25-year combined-severity PMFI/FR	50-year combined-severity PMFI/FR
a. Combined annual probability of fire (1/FR)	0.10000	0.04000	0.02000
UPPER LIMIT OF LOW-SEVERITY RANGE			
b. Annual probability of fire for moderate-high component of 115 years†	0.00870	0.00870	0.00870
c. Annual probability of fire for low-severity component, from a - b.	0.09130	0.03130	0.01130
d. Net fire rotation for low-severity component, from 1 / c.	10.95 years	31.95 years	88.50 years
LOWER LIMIT OF LOW-SEVERITY RANGE			

e. Annual probability of fire for moderate-high component of 319 years†	0.00313	0.00313	0.00313
f. Annual probability of fire for low-severity component, from a - e.	0.09687	0.03687	0.01687
g. Net fire rotation for low-severity component, from 1 / g.	10.32 years	27.12 years	59.28 years
NET ESTIMATED LOW-SEVERITY RANGE	10.32-10.95 years	27.12-31.95 years	59.28-88.50 years

Notes

† The 115-319 year range for moderate- to high-severity fire rotation in dry forests is from Odion et al. (2014)

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Human-started wildfires expand the fire niche across the United States

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The economic and ecological costs of wildfire in the United States have risen substantially in recent decades. Although climate change has likely enabled a portion of the increase in wildfire activity, the direct role of people in increasing wildfire activity has been largely overlooked. We evaluate over 1.5 million government records of wildfires that had to be extinguished or managed by state or federal agencies from 1992 to 2012, and examined geographic and seasonal extents of human-ignited wildfires relative to lightning-ignited wildfires. Humans have vastly expanded the spatial and seasonal "fire niche" in the coterminous United States, accounting for 84% of all wildfires and 44% of total area burned. During the 21-y time period, the human-caused fire season was three times longer than the lightning-caused fire season and added an average of 40,000 wildfires per year across the United States. Human-started wildfires disproportionally occurred where fuel moisture was higher than lightning-started fires, thereby helping expand the geographic and seasonal niche of wildfire. Human-started wildfires were dominant (>80% of ignitions) in over 5.1 million km^2, the vast majority of the United States, whereas lightning-started fires were dominant in only 0.7 million km^2, primarily in sparsely populated areas of the mountainous western United States. Ignitions caused by human activities are a substantial driver of overall fire risk to ecosystems and economies. Actions to raise awareness and increase management in regions prone to human-started wildfires should be a focus of United States policy to reduce fire risk and associated hazards.

anthropogenic wildfires | fire starts | ignitions | modern fire regimes | wildfire causes

The United States has experienced some of the largest wildfire years this decade, with over 36,000 km^2 burned in 2006, 2007, 2012, and 2015 (1). There is national and global concern over how fire regimes have changed in the past few decades and how they will change in the future (2–4). In the western United States, there is strong evidence that regional warming and drying, including that directly attributed to anthropogenic climate change, are linked to increased fire frequency and size and longer fire seasons (5–9). However, the role that humans play in starting these fires and the direct role of human-ignitions on recent increases in wildfire activity have been overlooked in public and scientific discourse because of the difficulty in ascribing a cause, either human- or lightning-started (10). Humans primarily alter fire regimes in three ways: changing the distribution and density of ignitions, shifting the seasonality of burning, or altering available fuels (2, 3). Geographic variability in regional and continental-scale fire activity in the United States is strongly tied to proxies for these human-caused changes, including population and road density, and different land-use and development patterns (10–15). Although changing climate and fuels also influence fire regimes across the United States (10, 16, 17), there can be no fire without an ignition source. Here, we explore the role that human-started wildfires play in modern United States fire regimes.

Ignitions are often presumed to be saturated (18, 19), and therefore have limited ability to predict fire activity. However, several studies suggest that humans play an important role in redistributing ignitions (20–22), particularly where lightning rarely occurs or where lightning is not concurrent with dry conditions (23). The human–fire connection in the modern era appears strongest at intermediate levels of development, as fires become less likely in the landscape beyond a certain population density, level of urbanization, or dependence on fossil fuels (11, 13, 24). Overall, humans expand the spatial and temporal "fire niche" by introducing ignitions into landscapes when fuels are sufficiently dry enough to ignite and carry fire, but when lightning is rare. Human ignitions are therefore a critical force acting to expand how the fire niche is realized across United States ecoregions.

National-scale analysis of human alteration of the fire niche is critical given that the annual expense of fighting wildfires has exceeded $2 billion in recent years, and the accrued direct and indirect impacts of wildfire on infrastructure and communities could be 30 times that amount (25). Policies that govern wildfire management and response are also directed at the national level, demanding analysis at a national scale (10, 22, 26). Although recent human influence on fire regimes has been studied at local (13) to regional scales (14), human influence nationally remains poorly understood (10). National policies can strongly influence fire regimes (27) and, with sufficient information on human ignitions, policy directives could target human behavior in ways that remediate increasing trends in wildfire risk.

Here, we ask how human ignitions have altered the spatial extents, seasonality, and temporal trends in wildfire across the coterminous United States. We analyze over 1.5 million records of both human- and lightning-started fires in the United States from

Significance

Fighting wildfires in the United States costs billions of dollars annually. Public dialog and ongoing research have focused on increasing wildfire risk because of climate warming, overlooking the direct role that people play in igniting wildfires and increasing fire activity. Our analysis of two decades of government agency wildfire records highlights the fundamental role of human ignitions. Human-started wildfires accounted for 84% of all wildfires, tripled the length of the fire season, dominated an area seven times greater than that affected by lightning fires, and were responsible for nearly half of all area burned. National and regional policy efforts to mitigate wildfire-related hazards would benefit from focusing on reducing the human expansion of the fire niche.

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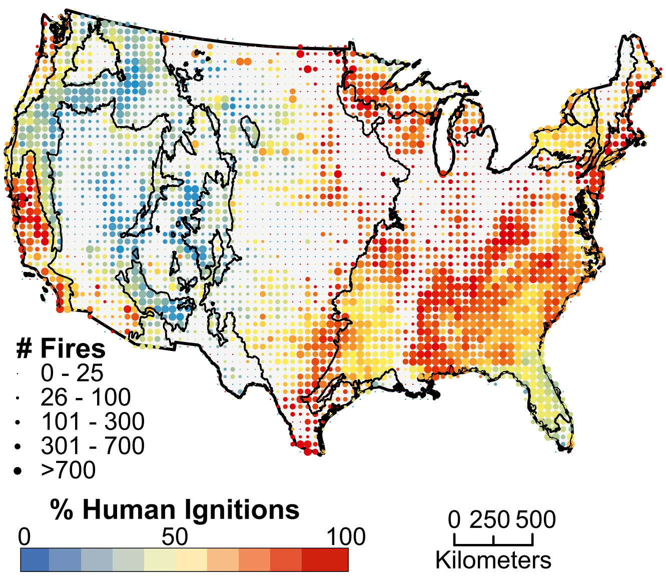

Fig. 1. The total number of wildfires (dot size) and the proportion started by humans (dot color: red indicating greater number of human started fires) within each 50 km × 50-km grid cell across the coterminous United States from 1992 to 2012. Black lines are ecoregion boundaries, as defined in the text.

1992 to 2012 (28). All of these wildfires necessitated an agency response to manage or suppress them, and therefore posed a threat to ecosystems or infrastructure; this record does not include intentionally set prescribed burns or managed agricultural fires. To our knowledge, this is the most comprehensive assessment of the role of human-started wildfires across the United States over the past two decades. We compare: (*i*) the spatial extents of human- vs. lightning-started wildfires, (*ii*) the seasonality of human vs. lightning wildfires, (*iii*) the climate niche for human- vs. lightning-started wildfires, and (*iv*) 21-y trends in large human vs. lightning wildfires. Our analysis documents the pronounced expansion of wildfire extent, seasonality of wildfires, and increasing numbers of large wildfires through time as a result of human-related ignitions across the United States.

Human-Related Ignitions Vastly Expanded the Extent of Wildfire

Human-started wildfires represented 84% of the 1.5 million wildfires included in this analysis (n = 245,446 lightning-started fires; n = 1,272,076 human-started wildfires). The eastern United States and western coastal areas were dominated by human-started wildfires, whereas lightning-started fires dominated the mountainous regions of the western United States (Fig. 1, Table 1 and Table S1). Here we define a fire regime as dominated by either human or lighting ignitions when one cause accounts for more than 80% of the number of fires in a given 50 × 50-km grid cell. Based on this definition, 5.1 million km^2, or 60% of the total land area of the coterminous United States, was dominated by human-started wildfires, whereas only 0.7 million km^2, or 8% of the area, was dominated by lightning-started fires. In addition to expanding the numbers of fires, humans also expanded the total area burned. Human-started wildfires burned a total of 160,274 km^2, or ~44% of the total area burned from 1992 to 2012 (Table 1).

Human-Related Ignitions More Than Tripled the Length of the Wildfire Season

Human ignitions dramatically expanded the wildfire season in the United States, particularly during spring. The length of the human-started wildfire season [defined as the interquartile range (IQR) of human-ignited fires] was 154 d, more than triple that of the lightning wildfire season (IQR = 46 d) (Fig. 2 and Table 1). This national-scale expansion is driven by earlier (spring) human-started fires in eastern ecoregions coupled with later (late summer or fall) human-started fires in western ecoregions (Table S2). The median discovery date for human-started fires was over 2-mo (May 20th) earlier than lightning-started fires (July 25th). Summed across the 21-y record, the most common day for human-started fires by far was July 4th, US Independence Day, with 7,762 fires starting that day over the course of the record (Fig. 2), whereas, the most common day for lightning-started fires was July 22nd. Of all lightning-ignited fires, 78% occurred in the summer (June–August), 9% in the spring (March–May), and 12% in the fall (September–November). In contrast, human-ignited wildfires were more evenly distributed throughout the year, with 24% in summer, 38% in spring, 19% in fall, and 19% in winter. This pronounced expansion of the wildfire season was also evident spatially (Fig. 3), with human-ignited wildfires occurring predominantly in spring in the eastern United States and in the fall and winter in Texas and the Gulf states. See Table S1 for state-level analysis. When lightning-started fires were rare (<5% and >95% quantile; i.e., before May 13th or after September 16th), humans ignited 842,289 wildfires, effectively increasing the number of wildfires 35-fold compared with the 24,081 lightning-ignited wildfires during these spring, fall, and winter seasons.

Table 1. The number of wildfires, total burned area (ha), and fire season length (IQR, in days), by ecoregion (ordered by percent human-caused fires) and within the coterminous United States from 1992 to 2012

	No. of fires			Area burned (ha)			Length (IQR, days)		
Ecoregion	Human	Light	Human caused (%)	Human	Light	Human caused (%)	Human	Light	Human expansion (%)
MC	87,274	2,855	97	2,143,282	253,210	89	85	45	189
NF	61,673	2,574	96	302,561	82,721	79	51	79	N/A
ETF	815,499	44,859	95	3,827,045	829,293	82	167	66	253
MWCF	14,586	925	94	19,251	27,291	41	67	52	129
GP	134,944	17,586	88	3,992,557	2,564,955	61	148	47	315
SSH	7,504	2,167	78	340,873	254,418	57	55	41	134
TWF	4,832	1,917	72	357,150	350,477	50	98	52	188
NAD	55,422	52,044	52	2,394,677	8,880,691	21	92	40	230
NFM	76,735	94,017	45	1,895,622	5,731,733	25	75	36	208
TS	13,607	26,502	34	754,393	1,152,064	40	85	39	218
CONUS	1,272,076	245,446	84	16,027,412	20,126,852	44	154	46	335

CONUS, Coterminous United States; ETF, Eastern Temperate Forests; GP, Great Plains; MC, Mediterranean California; MWCF, Marine West Coast Forests; NAD, North American Desert; NF, Northern Forests; NFM, Northwest Forested Mountains; SSH, Southern Semiarid Highlands; TWF, Tropical Wet Forests; TS, Temperate Sierras.

ENVIRONMENTAL SCIENCES

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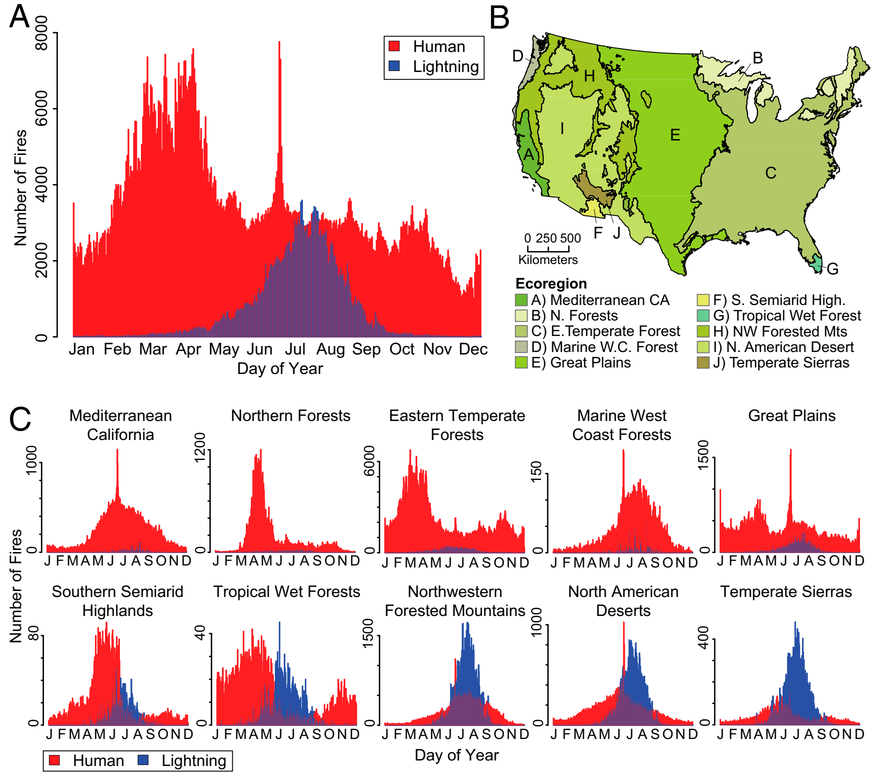

Fig. 2. Frequency distributions of human and lightning-caused wildfires by Julian day of year. (*A*) Frequency distribution of wildfires across the coterminous United States from 1992 to 2012 (n = 1.5 million); (*B*) map of United States ecoregions; (*C*) frequency distributions of wildfires by ecoregions, ordered by decreasing human dominance.

Human-Driven Expansion of the Fire Niche

Humans greatly expanded the natural fire niche (Fig. 4), which we calculated as the co-occurrence of the average monthly lightning density and 1,000-h dead fuel moisture. Regions and seasons of moderate to high lightning-started fire density (>0.4 fires per 1,000 km^2 per month) had a median lightning-strike density of 0.19 (IQR: 0.065–0.57) strikes per square kilometer per month and a median 1,000-h fuel moisture of 11.9% (IQR: 9.25–15.6%) (Fig. 4*A*). In contrast, regions and seasons of moderate to high human-started fire density (>0.4 fires per 1,000 km^2 per month) had a median lightning-strike density of only 0.11 (IQR: 0.025–0.39) strikes per square kilometer per month and a median 1,000-h fuel moisture of 17.8% (IQR: 15.95–19.25%) (Fig. 4*B*). The median fuel moisture and lightning conditions when human-started wildfires occurred were significantly different from those values for lightning-started fires (P < 0.0001). Areas and months of moderate to high human-caused fire density had approximately 40% fewer lightning strikes, and nearly 50% higher fuel moisture levels (based on median values) than for moderate to high lightning-caused fire density. Additional exploration of the fire niche for human-started and lightning-started fires relative to lightning density, fuel moisture, and net primary production (NPP), a proxy for fuels, is provided in Figs. S1 and S2.

Increasing Trends in Large Human-Started Wildfires

During the 21-y time period, there were significant increasing trends in large wildfires ignited by both lightning (n = 4,312; Theil-Sen estimated slope = 12.2; P = 0.001) and humans (n = 4,143; Theil-Sen estimated slope = 3.6; P = 0.004) (Fig. S3). There was a strong dichotomy in human vs. lightning trends seasonally (Fig. 5). Overall trends in lightning-caused fires were primarily driven by increasing numbers of large summer fires (Fig. 5*B*), whereas overall trends in human-caused fires were primarily driven by increasing numbers of large spring fires (Fig. 5*D*). Spatially, lightning-caused fires increased the most in the Northwest Forested Mountains ecoregion (Fig. S4*A*), whereas human-caused wildfires increased the most in the Great Plains ecoregion (Fig. S4*B*).

Discussion

Humans, the keystone fire species (29), play a primary role in spatially and temporally redistributing ignitions and resulting wildfires. We document that over 84% of the government-recorded

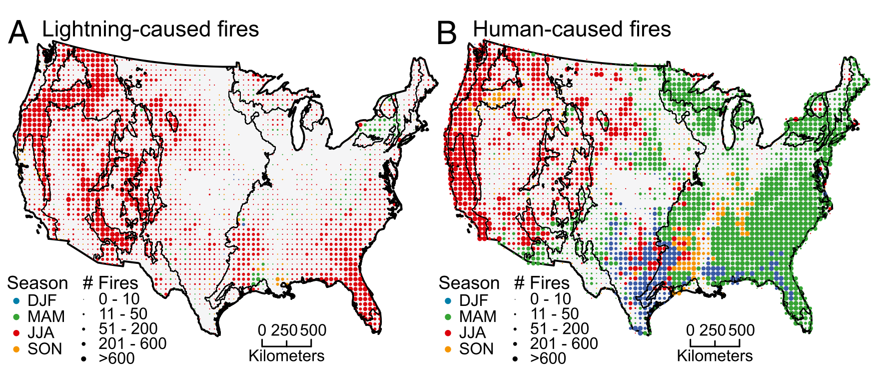

Fig. 3. Comparison of seasonality for (*A*) lightning- vs. (*B*) human-ignited wildfires. Human ignitions expand the seasonal fire niche considerably into spring and fall months. Colors show the season with the maximum ignitions caused by lightning and human within each 50 km × 50-km grid cell. Size of dot indicates the number of unique lightning and human fires between 1992 and 2012. Ecoregion boundaries are overlaid for visualization.

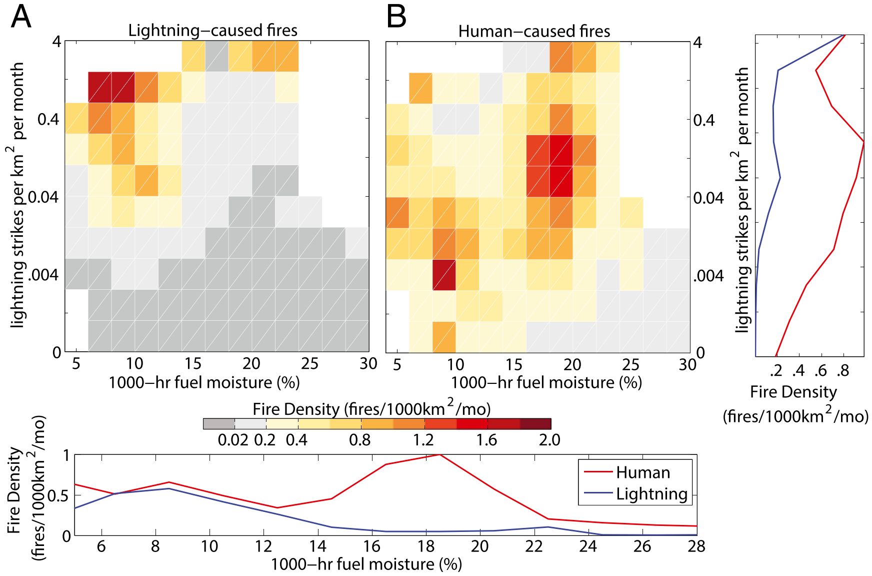

Fig. 4. Human vs. lightning fire niche relative to fuel moisture and lightning density, with greatest resulting wildfire density represented by dark red. (*A*) Lightning-started fires occur in areas with high lightning-strike density and dry fuels. (*B*) Human-started wildfires expand the fire niche to include areas with low lightning-strike density as well as areas with higher fuel moisture. Graphs on the bottom and far right show histograms of 1,000-h dead fuel moisture and lightning strikes, respectively, for human- and lightning-started fires.

wildfires were started by people from 1992 to 2012. Sixty percent of the total land area of the coterminous United States was dominated by human-started wildfires, whereas only 8% of the area was dominated by lightning fires. Humans tripled the length of the wildfire season, extending burning into the spring, fall, and winter months. During the spring, fall, and winter, people added more than 840,000 wildfires, a 35-fold increase over the number of lightning-started fires in those seasons. This expansion of the fire-niche was caused by human-related ignitions under higher fuel moisture conditions, compared with lightning-started fires. Moreover, during this 21-y record, large human-started wildfires increased significantly.

There was a strong national east–west dichotomy in the spatial distribution of human-started wildfires. Although human-started wildfires were pervasive across the United States (Fig. 1), the expansion of human-started wildfires relative to lightning-started fires was most dramatic in the eastern United States and central and southern California (Figs. 1 and 2*C*). Recent work for California confirms the important role of humans, with anthropogenic variables explaining half of the variability in fire probability over the past four decades (30). In contrast, lightning-started fires were found primarily in the intermountain west and Florida and occurred predominantly in the summer, reflecting national lightning strike patterns (31) (Fig. 2*C*). This finding supports other studies of human vs. lightning ignition sources that have found an important distinction between eastern and western United States fire patterns (10, 21) and drivers (32). Some explanations for this distinction include higher population and housing densities, lower proportions of public land, and more extensive land use and development in the eastern United States (33, 34), all of which could lead to more sources of anthropogenic ignitions. Synchrony between lightning activity and the seasonal nadir of fuel moisture in the western United States also likely contributes to these geographic differences. However, even with a projected increase in the number of lightning strikes as a result of anthropogenic climate change (50% by 2100) (35), humans would still remain the dominant ignition source across the majority of the United States land area. The majority of the wildfires requiring agency suppression in the east can be attributed to escaped fires from debris burning occurring in the spring months (or winter in Texas and the Gulf Coast) (Fig. 3). Between 1992 and 2012, wildfires caused by debris burning tended to be small (median

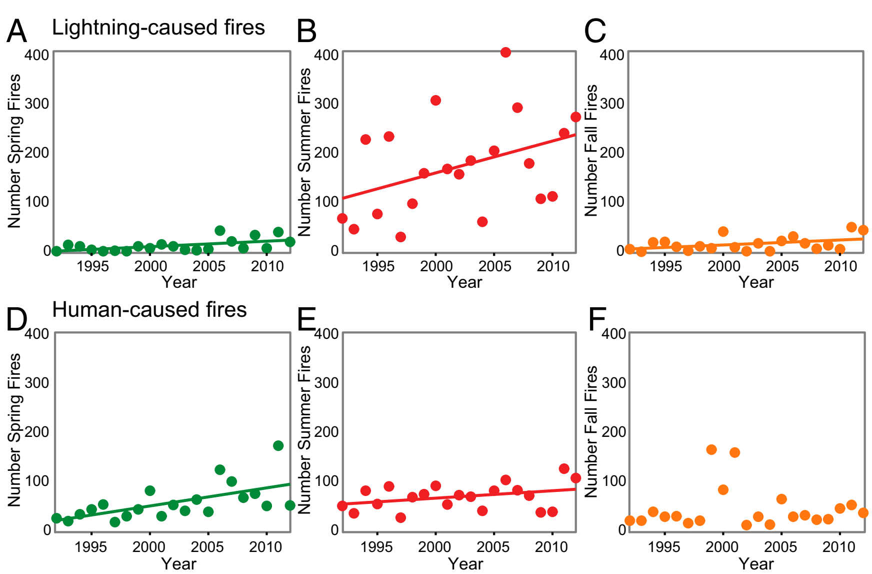

Fig. 5. Trends in the number of large wildfires verified by MTBS records from 1992 to 2012 for lightning-started fires (*A*–*C*) vs. human-started fires (*D*–*F*) in the spring (green: *A* and *D*), summer (red: *B* and *E*), and fall (orange: *C* and *F*). Where trend lines are shown, Theil-Sen estimated slopes are significantly different from zero ($P < 0.05$).

fire size 0.4 ha, IQR: 0.14–1.62 ha), but still an important source of risk to surrounding ecosystems. At finer scales, there are also notable patterns in human- vs. lightning-started wildfires (Fig. S5). Increased wildfires can follow road networks (36), the wildland–urban interface (13), and boundaries between agricultural and forested areas (37), highlighting just a few examples of how human activities and cultural drivers provide ignitions that substantially change the distribution of fire across the United States (38).

Our findings reinforce the strong imprint of people on fire regimes through changes in wildfire seasonality, which has been documented globally (39). In the past few decades, early onset of warmer and drier conditions has promoted greater fire activity across the western United States (6, 7, 40). However, our study highlights the equally important role of human ignitions in changing modern fire regimes by increasing the fire season length to encompass the entire year. The vast majority (78%) of lightning-started fires occurred during the summer months, whereas 76% of human-started fires occurred during the spring, fall, and winter months. Moreover, this trend varies substantially by ecoregion, reflecting again the principle dichotomy between the eastern and western United States (Fig. 3). Human-started fires extend the fire season earlier in the east, and later in the west (Fig. 3 and Table S2). Observations suggest that climate change has extended the duration of the fire weather season across most of the globe, including parts of the United States by a couple of weeks over the past three decades (5, 9), whereas we show that human ignitions in the United States increased the length of the fire season by more than three mo. There was also a notable mark of American culture on the distribution of wildfires, with the peak day of wildfires occurring on July 4th, concurrent with Independence Day fireworks displays (Fig. 2). Indeed, Americans start over twice as many wildfires on July 4th as any other summer day. A similar cultural mark has also been demonstrated globally with a marked decline in wildfires on Sunday compared with other weekdays (41).

Thus, at the national scale, human ignitions dramatically expand the spatial and seasonal niche of fire. The key components that define the fire niche are ignition sources, fuel mass, and desiccation. By exploring the fire niche along these axes, our results show that lightning fires are primarily constrained to areas with a lightning-strike density of greater than 100 strikes per grid cell per month (0.04 strikes/km^2 per month) and are concurrent with drier fuels (< 15% fuel moisture) (Fig. 4). Human ignitions expand fires into regions with higher fuel moisture (Fig. 4) and higher NPP (Figs. S1 and S2), suggesting that humans create sufficient ignition pressure for wetter fuels to burn. As a consequence, human ignitions have expanded the fire niche into areas with historically low lightning-strike density, such as Mediterranean California, or low concurrence of lightning and dry conditions, such as Eastern Temperate Forests (Fig. 1).

Over the past two decades, there was a significant increase across the United States for both human- and lightning-caused large fires (Fig. S3). The significant increase in large lightning fires is driven primarily by fires in summer months (Fig. 5) in the Northwest Forested Mountains ecoregion of the western United States (Fig. S4). This finding is consistent with other studies that have demonstrated an increase in large fires across the western United States (6, 7, 40), likely as a consequence of changes in climate and fuels rather than ignitions. In contrast, the significant trend in human-caused fires is primarily driven by an increase in large fires during spring months (Fig. 5) in the Great Plains ecoregion of the United States (Fig. S4). This increasing trend suggests that earlier springs as a result of climate change (42, 43) may be interacting with human ignition sources to increase the risk of large fires in the central United States.

The strong year-to-year variability in human ignitions (Fig. S3 and S4) may reflect the degree to which human choices can affect fire regimes. However, interannual climate variability also influences fuel moisture, NPP, and short-term weather conditions that enable the spread of human-ignited wildfires (44). There was a significant temporal correlation between large human- and lightning-started fires (R = 0.75). This pattern has been observed previously in the western United States (23) and suggests that large-scale climate drivers affect the frequency of both human- and lightning-caused fires. It is unknown how human actions will be affected by hotter and drier conditions, potentially increasing or decreasing ignitions from land use, recreation, and other sources. Increased public awareness and focused policy and management, particularly in years with elevated fire risk associated with climatic anomalies, are needed to reduce the number of human-caused ignitions.

In conclusion, we demonstrate the remarkable influence that humans have on modern United States wildfire regimes through changes in the spatial and seasonal distribution of ignitions. Although considerable fire research in the United States has rightly focused on increased fire activity (e.g., larger fires and more area burned) because of climate change, we demonstrate that the expanded fire niche as a result of human-related ignitions is equally profound. Moreover, the convergence of warming trends and expanded ignition pressure from people is increasing the number of large human-caused wildfires (Fig. 5). Currently, humans are extending the fire niche into conditions that are less conducive to fire activity, including regions and seasons with wetter fuels and higher biomass (Figs. 3 and 4). Land-use practices, such as clearing and logging, may also be creating an abundance of drier fuels, potentially leading to larger fires even under historically wetter conditions. Additionally, projected climate warming is expected to lower fuel moisture and create more frequent weather conditions conducive to fire ignition and spread (45), and earlier springs attributed to climate change are leading to accelerated phenology (42). Although plant physiological responses to rising CO_2 may reduce some drought stress (46), climate change will likely lead to faster desiccation of fuels and increased risk in areas where human ignitions are prevalent.

Uncertainty remains regarding how anthropogenic climate change will alter wildfire activity geographically and seasonally (47, 48), particularly in areas where human-caused fires dominate. Moreover, the current wildland–urban interface, where houses intermingle with natural areas, constitutes 9% of the United States total land area (33) but is projected to double by 2030, predominantly in the intermountain West (49). This expected development expansion will increase not only ignition pressure, but also the vulnerability of new infrastructure. Human-driven expansion of the spatial and temporal distribution of ignitions makes national- and regional-scale policy interventions and increased public awareness critical for reducing national wildfire risk.

Materials and Methods

For this analysis, we used the publically available US Forest Service Fire Program Analysis-Fire Occurrence Database (FPA-FOD) (28). This comprehensive dataset includes United States federal, state, and local records of wildfires (both on public and private lands) that were suppressed from 1992 to 2012, a total of ~1.6 million records. Previous studies have focused on the western United States (20), federal lands (22), or records from just one agency (21). Each entry includes at minimum the location, discovery date, and cause of the wildfire. We excluded 114,191 wildfires with an unknown cause and analyzed the spatial, seasonal, and temporal patterns of human- vs. lightning-started wildfires. In total, 1,517,522 wildfires were included in the analysis. Human-started wildfires were caused by a variety of sources, including the US Forest Service-designated categories of equipment use, smoking, campfire, railroad, arson, debris burning, children, fireworks, power line, structure, and miscellaneous fires (28). Spatially, we calculated the proportion of human- vs. lightning-caused wildfires within equal-area 50 × 50-km grid cells across the coterminous United States. This grid size corresponds roughly to the size of an average United States county. For each grid cell, we calculated the season (winter, spring, summer, or fall) when the majority of human-caused and lightning-caused wildfires were started. All spatial analyses were conducted in the Albers-Conical equal-area projection. To determine the seasonal distribution of wildfires, we plotted the distribution of human- and lightning-started fires by the day of year for the coterminous United States and for individual ecoregions. We used the level 1 ecological regions of North America, developed by the Commission for Environmental Cooperation (50). We calculated the length of the human- and lightning-caused fire seasons as the IQR of the Julian day of recorded fire ignition: that is, the difference between the first and third quartiles.

We determined how humans expanded the fire niche by comparing the lightning-strike density (i.e., natural ignition pressure) and fuel-moisture conditions under which actual human- and lightning-started fire events occurred. We obtained daily 1,000-h dead fuel moisture data from the surface meteorological data (51) on a 4-km grid from 1992 to 2012, and computed monthly averages across the 21-y study period. We obtained 4-km gridded monthly lightning-strike data from the Vaisala National Lightning Detection Network (https://www.ncdc.noaa.gov/data-access/severe-weather/lightning-products-and-services) and averaged the data over the 21-y study period. To account for fuel limitations, we also explored the fire niche as a function of fuel amount (approximated by NPP). We used MODIS mean annual NPP data (1-km resolution, from 2002 to 2015) (52) for this purpose. These three datasets were aggregated to the common 50×50-km grid cell. We calculated the number of human- and lightning-started fires by grid cell using the FPA-FOD dataset (28). We excluded any grid cells from subsequent analyses that did not report at least one lightning-caused or human-caused wildfire over the period of record. We tested whether fire niche expansion (as determined by fuel moisture and lightning-strike density) caused by human ignitions was significant based on Mann–Whitney tests between human- vs. lightning-started fires.

To assess trends in human- vs. lightning-caused wildfires through time, we used only large fires that were independently verified by the Monitoring Trends in Burn Severity (MTBS) project (53). We specifically focused on these large fires (>400 ha in the west, >200 ha in the east; n = 8,455) for comparability with previous research, which has examined temporal trends in the western United States and the link to climate warming (6, 7, 40), but has not investigated the relative contribution of human-started fires at a national scale. In addition to overall temporal trends, we tested for significant trends by ignition source versus season (spring, summer, fall) and versus ecoregion based on the level I ecological regions of North America (50). We explored a similar analysis using all available FPA-FOD data, but changes in reporting frequency through time for some states precluded a robust temporal analysis. We tested for trends in wildfire numbers through time using the nonparametric Theil-Sen estimator (54) and tested for trend significance using nonparametric Mann–Kendall tests (55).

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Adapting to Climate Change on Western Public Lands: Addressing the Ecological Effects of Domestic, Wild, and Feral Ungulates

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Abstract Climate change affects public land ecosystems and services throughout the American West and these effects are projected to intensify. Even if greenhouse gas emissions are reduced, adaptation strategies for public lands are needed to reduce anthropogenic stressors of terrestrial and aquatic ecosystems and to help native species and ecosystems survive in an altered environment. Historical and contemporary livestock production—the most widespread and long-running commercial use of public lands—can alter vegetation, soils, hydrology, and wildlife species composition and abundances in ways that exacerbate the effects of climate change on these resources. Excess abundance of native ungulates (e.g., deer or elk) and feral horses and burros add to these impacts. Although many of these consequences have been studied for decades, the ongoing and impending effects of ungulates in a changing climate require new management strategies for limiting their threats to the long-term supply of ecosystem services on public lands. Removing or reducing livestock across large areas of public land would alleviate a widely recognized and long-term stressor and make these lands less susceptible to the effects of climate change. Where livestock use continues, or where significant densities of wild or feral ungulates occur, management should carefully document the ecological, social, and economic consequences (both costs and benefits) to better ensure management that minimizes ungulate impacts to plant and animal communities, soils, and water resources. Reestablishing apex predators in large, contiguous areas of public land may help mitigate any adverse ecological effects of wild ungulates.

Keywords Ungulates · Climate change · Ecosystems · Public lands · Biodiversity · Restoration

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Introduction

During the 20th century, the average global surface temperature increased at a rate greater than in any of the previous nine centuries; future increases in the United States (US) are likely to exceed the global average (IPCC 2007a; Karl and others 2009). In the western US, where most public lands are found, climate change is predicted to intensify even if greenhouse gas emissions are reduced dramatically (IPCC 2007b). Climate-related changes can not only affect public-land ecosystems directly, but may exacerbate the aggregate effects of non-climatic stressors, such as habitat modification and pollution caused by logging, mining, grazing, roads, water diversions, and recreation (Root and others 2003; CEQ 2010; Barnosky and others 2012).

One effective means of ameliorating the effects of climate change on ecosystems is to reduce environmental stressors under management control, such as land and water uses (Julius and others 2008; Heller and Zavaleta 2009; Prato 2011). Public lands in the American West provide important opportunities to implement such a strategy for three reasons: (1) despite a history of degradation, public lands still offer the best available opportunities for ecosystem restoration (CWWR 1996; FS and BLM 1997; Karr 2004); (2) two-thirds of the runoff in the West originates on public lands (Coggins and others 2007); and (3) ecosystem protection and restoration are consistent with laws governing public lands. To be effective, restoration measures should address management practices that prevent public lands from providing the full array of ecosystem services and/or are likely to accentuate the effects of climate change (Hunter and others 2010). Although federal land managers have recently begun considering how to adapt to and mitigate potential climate-related impacts (e.g., GAO 2007; Furniss and others 2009; CEQ 2010; Peterson and others 2011), they have not addressed the combined effects of climate change and ungulates (hooved mammals) on ecosystems.

Climate change and ungulates, singly and in concert, influence ecosystems at the most fundamental levels by affecting soils and hydrologic processes. These effects, in turn, influence many other ecosystem components and processes—nutrient and energy cycles; reproduction, survival, and abundance of terrestrial and aquatic species; and community structure and composition. Moreover, by altering so many factors crucial to ecosystem functioning, the combined effects of a changing climate and ungulate use can affect biodiversity at scales ranging from species to ecosystems (FS 2007) and limit the capability of large areas to supply ecosystem services (Christensen and others 1996; MEA 2005b).

In this paper, we explore the likely ecological consequences of climate change and ungulate use, individually and in combination, on public lands in the American West. Three general categories of large herbivores are considered: livestock (largely cattle [*Bos taurus*] and sheep [*Ovis aries*]), native ungulates (deer [*Odocoileus* spp.] and elk [*Cervus* spp.]), and feral ungulates (horses [*Equus caballus*] and burros [*E. asinus*]). Based on this assessment, we propose first-order recommendations to decrease these consequences by reducing ungulate effects that can be directly managed.

Climate Change in the Western US

Anticipated changes in atmospheric carbon dioxide (CO_2), temperature, and precipitation (IPCC 2007a) are likely to have major repercussions for upland plant communities in western ecosystems (e.g., Backlund and others 2008), eventually affecting the distribution of major vegetation types. Deserts in the southwestern US, for example, will expand to the north and east, and in elevation (Karl and others 2009). Studies in southeastern Arizona have already attributed dramatic shifts in species composition and plant and animal populations to climate-driven changes (Brown and others 1997). Thus, climate-induced changes are already accelerating the ongoing loss of biodiversity in the American West (Thomas and others 2004).

Future decreases in soil moisture and vegetative cover due to elevated temperatures will reduce soil stability (Karl and others 2009). Wind erosion is likely to increase dramatically in some ecosystems such as the Colorado Plateau (Munson and others 2011) because biological soil crusts—a complex mosaic of algae, lichens, mosses, microfungi, cyanobacteria, and other bacteria—may be less drought tolerant than many desert vascular plant species (Belnap and others 2006). Higher air temperatures may also lead to elevated surface-level concentrations of ozone (Karl and others 2009), which can reduce the capacity of vegetation to grow under elevated CO_2 levels and sequester carbon (Karnosky and others 2003).

Air temperature increases and altered precipitation regimes will affect wildfire behavior and interact with insect outbreaks (Joyce and others 2009). In recent decades, climate change appears to have increased the length of the fire season and the area annually burned in some western forest types (Westerling and others 2006; ITF 2011). Climate induced increases in wildfire occurrence may aggravate the expansion of cheatgrass (*Bromus tectorum*), an exotic annual that has invaded millions of hectares of sagebrush (*Artemisia* spp.) steppe, a widespread yet threatened ecosystem. In turn, elevated wildfire occurrence facilitates the conversion of sagebrush and other native shrub-perennial grass communities to those dominated by alien grasses (D'Antonio and Vitousek 1992; Brooks 2008), resulting in habitat loss for imperiled greater sage-grouse (*Centrocercus urophasianus*) and other sagebrush-dependent species (Welch 2005). The US Fish and Wildlife Service (FWS 2010) recently concluded climate change effects can exacerbate many of the multiple threats to sagebrush habitats, including wildfire, invasive plants, and heavy ungulate use. In addition, the combined effects of increased air temperatures, more frequent fires, and elevated CO_2 levels apparently provide some invasive species with a competitive advantage (Karl and others 2009).

By the mid-21st century, Bates and others (2008) indicate that warming in western mountains is very likely to cause large decreases in snowpack, earlier snowmelt, more winter rain events, increased peak winter flows and flooding, and reduced summer flows. Annual runoff is predicted to decrease by 10–30 % in mid-latitude western North America by 2050 (Milly and others 2005) and up to 40 % in Arizona (Milly and others 2008; ITF 2011). Drought periods are expected to become more frequent and longer throughout the West (Bates and others 2008). Summertime decreases in streamflow (Luce and Holden 2009) and increased water temperatures already have been documented for some western rivers (Kaushal and others 2010; Isaak and others 2012).

Snowmelt supplies about 60–80 % of the water in major western river basins (the Columbia, Missouri, and Colorado Rivers) and is the primary water supply for about 70 million people (Pederson and others 2011). Contemporary and future declines in snow accumulations and runoff (Mote and others 2005; Pederson and others 2011) are an important concern because current water supplies, particularly during low-flow periods, are already inadequate to satisfy demands over much of the western US (Piechota and others 2004; Bates and others 2008).

High water temperatures, acknowledged as one of the most prevalent water quality problems in the West, will likely be further elevated and may render one-third of the current coldwater fish habitat in the Pacific Northwest unsuitable by this century's end (Karl and others 2009). Resulting impacts on salmonids include increases in virulence of disease, loss of suitable habitat, and mortality as well as increased competition and predation by warmwater species (EPA 1999). Increased water temperatures and changes in snowmelt timing can also affect amphibians adversely (Field and others 2007). In sum, climate change will have increasingly significant effects on public-land terrestrial and aquatic ecosystems, including plant and animal communities, soils, hydrologic processes, and water quality.

Ungulate Effects and Climate Change Synergies

Climate change in the western US is expected to amplify "combinations of biotic and abiotic stresses that compromise the vigor of ecosystems—leading to increased extent and severity of disturbances" (Joyce and others 2008, p. 16). Of the various land management stressors affecting western public lands, ungulate use is the most widespread (Fig. 1). Domestic livestock annually utilize over 70 % of lands managed by the Bureau of Land Management (BLM) and US Forest Service (FS). Many public lands are also used by wild ungulates and/or feral horses and burros, which are at high densities in some areas. Because ungulate groups can have different effects, we discuss them individually.

Livestock

History and Current Status

Livestock were introduced to North America in the mid-sixteenth century, with a massive influx from the mid-1800s through early 1900s (Worster 1992). The deleterious effects of livestock—including herbivory of both herbaceous and woody plants and trampling of vegetation, soils, and streambanks—prompted federal regulation of grazing on western national forests beginning in the 1890s (Fleischner 2010). Later, the 1934 Taylor Grazing Act was enacted "to stop injury to the public grazing lands by preventing overgrazing and soil deterioration" on lands subsequently administered by the BLM.

Total livestock use of federal lands in eleven contiguous western states today is nearly 9 million animal unit months (AUMs, where one AUM represents forage use by a cow and calf pair, one horse, or five sheep for one month) (Fig. 2a). Permitted livestock use occurs on nearly one million square kilometers of public land annually, including 560,000 km^2 managed by the BLM, 370,000 km^2 by the FS, 6,000 km^2 by the National Park Service (NPS), and 3,000 km^2 by the US Fish and Wildlife Service (FWS).

Livestock use affects a far greater proportion of BLM and FS lands than do roads, timber harvest, and wildfires combined (Fig. 3). Yet attempts to mitigate the pervasive effects of livestock have been minor compared with those aimed at reducing threats to ecosystem diversity and productivity that these other land uses pose. For example, much effort is often directed at preventing and controlling wildfires since they can cause significant property damage and social impacts. On an annual basis, however, wildfires affect a much smaller portion of public land than livestock grazing (Fig. 3) and they can also result in ecosystem benefits (Rhodes and Baker 2008; Swanson and others 2011).

The site-specific impacts of livestock use vary as a function of many factors (e.g., livestock species and density, periods of rest or non-use, local plant communities, soil conditions). Nevertheless, extensive reviews of published research generally indicate that livestock have had numerous and widespread negative effects to western ecosystems (Love 1959; Blackburn 1984; Fleischner 1994; Belsky and others 1999; Kauffman and Pyke 2001; Asner

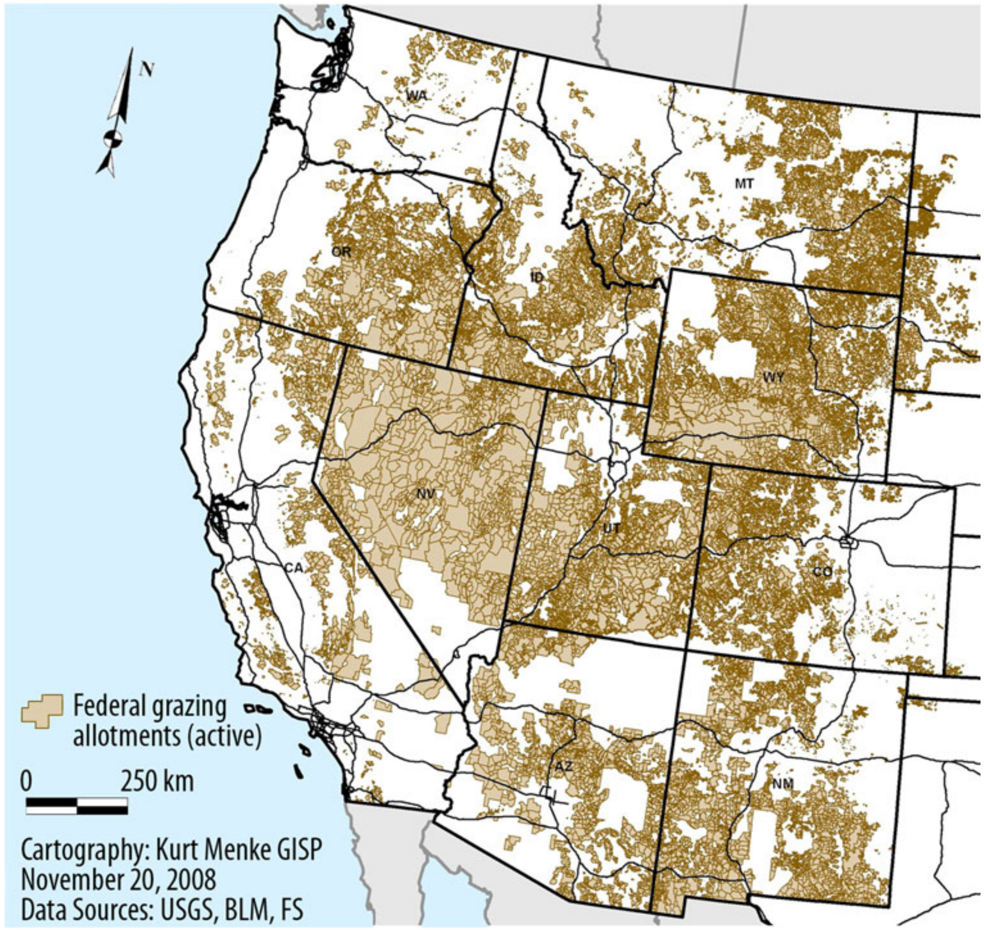

Fig. 1 Areas of public-lands livestock grazing managed by federal agencies in the western US (adapted from Salvo 2009)

and others 2004; Steinfeld and others 2006; Thornton and Herrero 2010). Moreover, public-land range conditions have generally worsened in recent decades (CWWR 1996, Donahue 2007), perhaps due to the reduced productivity of these lands caused by past grazing in conjunction with a changing climate (FWS 2010, p. 13,941, citing Knick and Hanser 2011).

Plant and Animal Communities

Livestock use effects, exacerbated by climate change, often have severe impacts on upland plant communities. For example, many former grasslands in the Southwest are now dominated by one or a few woody shrub species, such as creosote bush (*Larrea tridentata*) and mesquite (*Prosopis glandulosa*), with little herbaceous cover (Grover and Musick 1990; Asner and others 2004; but see Allington and Valone 2010). Other areas severely affected include the northern Great Basin and interior Columbia River Basin (Middleton and Thomas 1997). Livestock effects have also contributed to severe degradation of sagebrush-grass ecosystems (Connelly and others 2004; FWS 2010) and widespread desertification, particularly in the Southwest (Asner and others 2004; Karl and others 2009). Even absent desertification, light to moderate grazing intensities can promote woody species encroachment in semiarid and mesic environments (Asner and others 2004, p. 287). Nearly two decades ago, many public-land ecosystems, including native shrub steppe in Oregon and Washington, sagebrush steppe in the Intermountain West, and riparian plant communities, were considered threatened, endangered, or critically endangered (Noss and others 1995).

Simplified plant communities combine with loss of vegetation mosaics across landscapes to affect pollinators, birds, small mammals, amphibians, wild ungulates, and other native wildlife (Bock and others 1993; Fleischner 1994; Saab and others 1995; Ohmart 1996). Ohmart and Anderson (1986) suggested that livestock grazing may be the major factor negatively affecting wildlife in eleven western states. Such effects will compound the problems of adaptation of these ecosystems to the dynamics of climate change (Joyce and others 2008, 2009). Currently, the widespread and ongoing declines of many North American bird populations that use grassland and grass–shrub habitats affected by grazing are “on track to become a prominent wildlife conservation crisis of the 21st century” (Brennan and Kuvlesky 2005, p. 1).

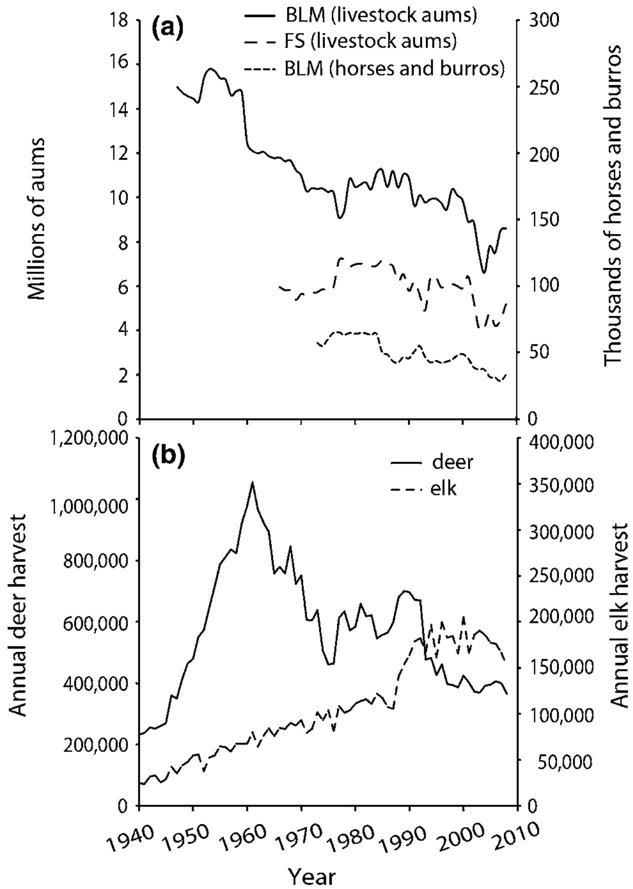

Fig. 2 **a** Bureau of Land Management (BLM) and US Forest Service (FS) grazing use in animal unit months (AUMs) and number of feral horses and burros on BLM lands, and **b** annual harvest of deer and elk by hunters, for eleven western states. *Data sources* **a** BLM grazing and number of horses and burros reported annually in Public Land Statistics; FS grazing reported annually in Grazing Statistical Summary; **b** deer and elk harvest records from individual state wildlife management agencies

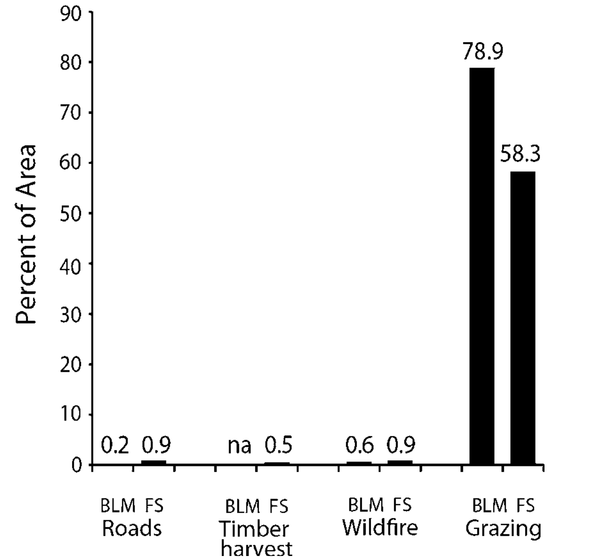

Fig. 3 Percent of Bureau of Land Management (BLM) and US Forest Service (FS) lands in eleven western states that are occupied by roads or are affected annually by timber harvest, wildfire, and grazing. *Data sources* Roads, BLM (2009) and FS, Washington Office; Timber harvest (2003–09), FS, Washington Office; Wildfire (2003–09), National Interagency Fire Center, Missoula, Montana; Grazing, BLM (2009) and GAO (2005). “na” = not available

Soils and Biological Soil Crusts

Livestock grazing and trampling can damage or eliminate biological soil crusts characteristic of many arid and semiarid regions (Belnap and Lange 2003; Asner and others 2004). These complex crusts are important for fertility, soil stability, and hydrology (Belnap and Lange 2003). In arid and semiarid regions they provide the major barrier against wind erosion and dust emission (Munson and others 2011). Currently, the majority of dust emissions in North America originate in the Great Basin, Colorado Plateau, and Mojave and Sonoran Deserts, areas that are predominantly public lands and have been grazed for nearly 150 years. Elevated sedimentation in western alpine lakes over this period has also been linked to increased aeolian deposition stemming from land uses, particularly those associated with livestock grazing (Neff and others 2008).

If livestock use on public lands continues at current levels, its interaction with anticipated changes in climate will likely worsen soil erosion, dust generation, and stream pollution. Soils whose moisture retention capacity has been reduced will undergo further drying by warming temperatures and/or drought and become even more susceptible to wind erosion (Sankey and others 2009). Increased aeolian deposition on snowpack will hasten runoff, accentuating climate-induced hydrological changes on many public lands (Neff and others 2008). Warmer temperatures will likely trigger increased fire occurrence, causing further reductions in cover and composition of biological soil crusts (Belnap and others 2006), as well as vascular plants (Munson and others 2011). In some forest types, where livestock grazing has contributed to altered fire regimes and forest structure (Belsky and Blumenthal 1997; Fleischner 2010), climate change will likely worsen these effects.

Water and Riparian Resources

Although riparian areas occupy only 1–2 % of the West’s diverse landscapes, they are highly productive and ecologically valuable due to the vital terrestrial habitats they provide and their importance to aquatic ecosystems (Kauffman and others 2001; NRC 2002; Fleischner 2010). Healthy riparian plant communities provide important corridors for the movement of plant and animal species

(Peterson and others 2011). Such communities are also crucial for maintaining water quality, food webs, and channel morphology vital to high-quality habitats for fish and other aquatic organisms in the face of climate change. For example, well-vegetated streambanks not only shade streams but also help to maintain relatively narrow and stable channels, attributes essential for preventing increased stream temperatures that negatively affect salmonids and other aquatic organisms (Sedell and Beschta 1991; Kondolf and others 1996; Beschta 1997); maintaining cool stream temperatures is becoming even more important with climate change (Isaak and others 2012). Riparian vegetation is also crucial for providing seasonal fluxes of organic matter and invertebrates to streams (Baxter and others 2005). Nevertheless, in 1994 the BLM and FS reported that western riparian areas were in their worst condition in history, and livestock use—typically concentrated in these areas—was the chief cause (BLM and FS 1994).

Livestock grazing has numerous consequences for hydrologic processes and water resources. Livestock can have profound effects on soils, including their productivity, infiltration, and water storage, and these properties drive many other ecosystem changes. Soil compaction from livestock has been identified as an extensive problem on public lands (CWWR 1996; FS and BLM 1997). Such compaction is inevitable because the hoof of a 450-kg cow exerts more than five times the pressure of heavy earth-moving machinery (Cowley 2002). Soil compaction significantly reduces infiltration rates and the ability of soils to store water, both of which affect runoff processes (Branson and others 1981; Blackburn 1984). Compaction of wet meadow soils by livestock can significantly decrease soil water storage (Kauffman and others 2004), thus contributing to reduced summer base flows. Concomitantly, decreases in infiltration and soil water storage of compacted soils during periods of high-intensity rainfall contribute to increased surface runoff and soil erosion (Branson and others 1981). These fundamental alterations in hydrologic processes from livestock use are likely to be exacerbated by climate change.

The combined effects of elevated soil loss and compaction caused by grazing reduce soil productivity, further compromising the capability of grazed areas to support native plant communities (CWWR 1996; FS and BLM 1997). Erosion triggered by livestock use continues to represent a major source of sediment, nutrients, and pathogens in western streams (WSWC 1989; EPA 2009). Conversely, the absence of grazing results in increased litter accumulation, which can reduce runoff and erosion and retard desertification (Asner and others 2004).

Historical and contemporary effects of livestock grazing and trampling along stream channels can destabilize streambanks, thus contributing to widened and/or incised channels (NRC 2002). Accelerated streambank erosion and channel incision are pervasive on western public lands used by livestock (Fig. 4). Stream incision contributes to desiccation of floodplains and wet meadows, loss of floodwater detention storage, and reductions in baseflow (Ponce and Lindquist 1990; Trimble and Mendel 1995). Grazing and trampling of riparian plant communities also contribute to elevated water temperatures—directly, by reducing stream shading and, indirectly, by damaging streambanks and increasing channel widths (NRC 2002). Livestock use of riparian plant communities can also decrease the availability of food and construction materials for keystone species such as beaver (*Castor canadensis*).

Livestock effects and climate change can interact in various ways with often negative consequences for aquatic species and their habitats. In the eleven ecoregions encompassing western public lands (excluding coastal regions and Alaska), about 175 taxa of freshwater fish are considered imperiled (threatened, endangered, vulnerable, possibly extinct, or extinct) due to habitat-related causes (Jelks and others 2008, p. 377; GS and AFS 2011). Increased sedimentation and warmer stream temperatures associated with livestock grazing have contributed significantly to the long-term decline in abundance and distribution and loss of native salmonids, which are imperiled throughout the West (Rhodes and others 1994; Jelks and others 2008).

Water developments and diversions for livestock are common on public lands (Connelly and others 2004). For example, approximately 3,700 km of pipeline and 2,300 water developments were installed on just 17 % of the BLM's land base from 1961 to 1999 in support of livestock operations (Rich and others 2005). Such developments can reduce streamflows thus contributing to warmer stream temperatures and reduced fish habitat, both serious problems for native coldwater fish (Platts 1991; Richter and others 1997). Reduced flows and higher temperatures are also risk factors for many terrestrial and aquatic vertebrates (Wilcove and others 1998). Water developments can also create mosquito (e.g., *Culex tarsalis*) breeding habitat, potentially facilitating the spread of West Nile virus, which poses a significant threat to sage grouse (FWS 2010). Such developments also tend to concentrate livestock and other ungulate use, thus locally intensifying grazing and trampling impacts.

Greenhouse Gas Emissions and Energy Balances

Livestock production impacts energy and carbon cycles and globally contributes an estimated 18 % to the total anthropogenic greenhouse gas (GHG) emissions (Steinfeld and others 2006). How public-land livestock contribute to

Fig. 4 Examples of long-term grazing impacts from livestock, unless otherwise noted: **a** bare soil, loss of understory vegetation, and lack of aspen recruitment (i.e., growth of seedlings/sprouts into tall saplings and trees) (Bureau of Land Management, Idaho), **b** bare soil, lack of ground cover, lack of aspen recruitment and channel incision (US Forest Service, Idaho), **c** conversion of a perennial stream to an intermittent stream due to grazing of riparian vegetation and subsequent channel incision; channel continues to erode during runoff events (Bureau of Land Management, Utah), **d** incised and widening stream due to loss of streamside vegetation and bank collapse from trampling (Bureau of Land Management, Wyoming), **e** incised and widening stream due to loss of streamside vegetation and bank collapse from trampling (US Forest Service, Oregon), and **f** actively eroding streambank from the loss of streamside vegetation due to several decades of excessive herbivory by elk and, more recently, bison (National Park Service, Wyoming). *Photographs* **a** J Carter, **b** G Wuerthner, **c** and **d** J Carter, **e** and **f** R Beschta

these effects has received little study. Nevertheless, livestock grazing and trampling can reduce the capacity of rangeland vegetation and soils to sequester carbon and contribute to the loss of above- and below-ground carbon pools (e.g., Lal 2001b; Bowker and others 2012). Lal (2001a) indicated that heavy grazing over the long-term may have adverse impacts on soil organic carbon content, especially for soils of low inherent fertility. Although Gill (2007) found that grazing over 100 years or longer in subalpine areas on the Wasatch Plateau in central

Utah had no significant impacts on total soil carbon, results of the study suggest that "if temperatures warm and summer precipitation increases as is anticipated, [soils in grazed areas] may become net sources of CO_2 to the atmosphere" (Gill 2007, p. 88). Furthermore, limited soil aeration in soils compacted by livestock can stimulate production of methane, and emissions of nitrous oxide under shrub canopies may be twice the levels in nearby grasslands (Asner and others 2004). Both of these are potent GHGs.

Reduced plant and litter cover from livestock use can increase the albedo (reflectance) of land surfaces, thereby altering radiation energy balances (Balling and others 1998). In addition, widespread airborne dust generated by livestock is likely to increase with the drying effects of climate change. Air-borne dust influences atmospheric radiation balances as well as accelerating melt rates when deposited on seasonal snowpacks and glaciers (Neff and others 2008).

Other Livestock Effects

Livestock urine and feces add nitrogen to soils, which may favor nonnative species (BLM 2005), and can lead to loss of both organic and inorganic nitrogen in increased runoff (Asner and others 2004). Organic nitrogen is also lost via increased trace-gas flux and vegetation removal by grazers (Asner and others 2004). Reduced soil nitrogen is problematic in western landscapes because nitrogen is an important limiting nutrient in most arid-land soils (Fleischner 2010).

Managing livestock on public lands also involves extensive fence systems. Between 1962 and 1997, over 51,000 km of fence were constructed on BLM lands with resident sage-grouse populations (FWS 2010). Such fences can significantly impact this wildlife species. For example, 146 sage-grouse died in less than three years from collisions with fences along a 7.6-km BLM range fence in Wyoming (FWS 2010). Fences can also restrict the movements of wild ungulates and increase the risk of injury and death by entanglement or impalement (Harrington and Conover 2006; FWS 2010). Fences and roads for livestock access can fragment and isolate segments of natural ecological mosaics thus influencing the capability of wildlife to adapt to a changing climate.

Some have posited that managed cattle grazing might play a role in maintaining ecosystem structure in shortgrass steppe ecosystems of the US, if it can mimic grazing by native bison (*Bison bison*) (Milchunas and others 1998). But most public lands lie to the west of the Great Plains, where bison distribution and effects were limited or nonexistent; livestock use (particularly cattle) on these lands exert disturbances without evolutionary parallel (Milchunas and Lauenroth 1993; MEA 2005a).

Feral Horses and Burros

Feral horses and burros occupy large areas of public land in the western US. For example, feral horses are found in ten western states and feral burros occur in five of these states, largely in the Mojave and Sonoran Deserts and the Great Basin (Abella 2008; FWS 2010). About half of these horses and burros are in Nevada (Coggins and others 2007), of which 90 % are on BLM lands. Horse numbers peaked at perhaps two million in the early 1900s, but had plummeted to about 17,000 by 1971, when protective legislation (Wild, Free-Ranging Horses and Burros Act [WFRHBA]) was passed (Coggins and others 2007). Protection resulted in increased populations and today some 40,000 feral horses and burros utilize $\sim$ 130,000 km^2 of BLM and FS lands (DOI-OIG 2010; Gorte and others 2010). Currently, feral horse numbers are doubling every four years (DOI-OIG 2010); burro populations can also increase rapidly (Abella 2008). Unlike wild ungulates, feral equines cannot be hunted and, unlike livestock, they are not regulated by permit. Nor are their numbers controlled effectively by existing predators. Accordingly, the BLM periodically removes animals from herd areas; the NPS also has undertaken burro control efforts (Abella 2008).

In sage grouse habitat, high numbers of feral horses reduce vegetative cover and plant diversity, fragment shrub canopies, alter soil characteristics, and increase the abundance of invasive species, thus reducing the quality and quantity of habitat (Beever and others 2003; FWS 2010). Horses can crop plants close to the ground, impeding the recovery of affected vegetation. Feral burros also have had a substantial impact on Sonoran Desert vegetation, reducing the density and canopy cover of nearly all species (Hanley and Brady 1977). Although burro impacts in the Mojave Desert may not be as clear, perennial grasses and other preferred forage species likely require protection from grazing in burro-inhabited areas if revegetation efforts are to be successful (Abella 2008).

Wild Ungulates

Extensive harvesting of wild (native) ungulates, such as elk and deer, and the decimation of large predator populations (e.g., gray wolf [*Canis lupus*], grizzly bear [*Ursus arctos*], and cougar [*Puma concolor*]) was common during early EuroAmerican settlement of the western US. With continued predator control in the early 1900s and increased protection of game species by state agencies, however, wild ungulate populations began to increase in many areas. Although only 70,000 elk inhabited the western US in the early 1900s (Graves and Nelson 1919), annual harvest data indicate that elk abundance has increased greatly since the about the 1940s (Fig. 2b), due in part to the loss of apex predators (Allen 1974; Mackie and others 1998). Today, approximately one million elk (Karnopp 2008) and unknown numbers of deer inhabit the western US where they often share public lands with livestock.

Because wild ungulates typically occur more diffusely across a landscape than livestock, their presence might be expected to cause minimal long-term impacts to vegetation. Where wild ungulates are concentrated, however, their browsing can have substantial impacts. For example, sagebrush vigor can be reduced resulting in decreased cover or mortality (FWS 2010). Heavy browsing effects have also been documented on other palatable woody shrubs, as well as deciduous trees such as aspen (*Populus tremuloides*), cottonwood (*Populus* spp.), and maple (*Acer* sp.) (Beschta and Ripple 2009).

Predator control practices that intensified following the introduction of domestic livestock in the western US resulted in the extirpation of apex predators or reduced their numbers below ecologically effective densities (Soulé and others 2003, 2005), causing important cascading effects in western ecosystems (Beschta and Ripple 2009). Following removal of large predators on the Kaibab Plateau in the early 20th century, for example, an irruption of mule deer (*O. hemionus*) led to extensive over-browsing of aspen, other deciduous woody plants, and conifers; deterioration of range conditions; and the eventual crash of the deer population (Binkley and others 2006). In the absence of apex predators, wild ungulate populations can significantly limit recruitment of woody browse species, contribute to shifts in abundance and distribution of many wildlife species (Berger and others 2001; Weisberg and Coughenour 2003), and can alter streambanks and riparian communities that strongly influence channel morphology and aquatic conditions (Beschta and Ripple 2012). Numerous studies support the conclusion that disruptions of trophic cascades due to the decline of apex predators constitute a threat to biodiversity for which the best management solution is likely the restoration of effective predation regimes (Estes and others 2011).

Ungulate Herbivory and Disturbance Regimes

Across the western US, ecosystems evolved with and were sustained by local and regional disturbances, such as fluctuating weather patterns, fire, disease, insect infestation, herbivory by wild ungulates and other organisms, and hunting by apex predators. Chronic disturbances with relatively transient effects, such as frequent, low-severity fires and seasonal moisture regime fluctuations, helped maintain native plant community composition and structure. Relatively abrupt, or acute, natural disturbances, such as insect outbreaks or severe fires were also important for the maintenance of ecosystems and native species diversity (Beschta and others 2004; Swanson and others 2011). Livestock use and/or an overabundance of feral or wild ungulates can, however, greatly alter ecosystem response to disturbance and can degrade affected systems. For example, high levels of herbivory over a period of years, by either domestic or wild ungulates, can effectively prevent aspen sprouts from growing into tall saplings or trees as well as reduce the diversity of understory species (Shepperd and others 2001; Dwire and others 2007; Beschta and Ripple 2009).

Natural floods provide another illustration of how ungulates can alter the ecological role of disturbances. High flows are normally important for maintaining riparian plant communities through the deposition of nutrients, organic matter, and sediment on streambanks and floodplains, and for enhancing habitat diversity of aquatic and riparian ecosystems (CWWR 1996). Ungulate effects on the structure and composition of riparian plant communities (e.g., Platts 1991; Chadde and Kay 1996), however, can drastically alter the outcome of these hydrologic disturbances by diminishing streambank stability and severing linkages between high flows and the maintenance of streamside plant communities. As a result, accelerated erosion of streambanks and floodplains, channel incision, and the occurrence of high instream sediment loads may become increasingly common during periods of high flows (Trimble and Mendel 1995). Similar effects have been found in systems where large predators have been displaced or extirpated (Beschta and Ripple 2012). In general, high levels of ungulate use can essentially uncouple typical ecosystem responses to chronic or acute disturbances, thus greatly limiting the capacity of these systems to provide a full array of ecosystem services during a changing climate.

The combined effects of ungulates (domestic, wild, and feral) and a changing climate present a pervasive set of stressors on public lands, which are significantly different from those encountered during the evolutionary history of the region's native species. The intersection of these stressors is setting the stage for fundamental and unprecedented changes to forest, arid, and semi-arid landscapes in the western US (Table 1) and increasing the likelihood of alternative states. Thus, public-land management needs to focus on restoring and maintaining structure, function, and integrity of ecosystems to improve their resilience to climate change (Rieman and Isaak 2010).

Federal Law and Policy

Federal laws guide the use and management of public-land resources. Some laws are specific to a given agency (e.g., the BLM's Taylor Grazing Act of 1934 and the FS's

Table 1 Generalized climate change effects, heavy ungulate use effects, and their combined effects as stressors to terrestrial and aquatic ecosystems in the western United States

Climate change effects	Ungulate use effects	Combined effects
Increased drought frequency and duration	Altered upland plant and animal communities	Reduced habitat and food-web support; loss of mesic and hydric plants, reduced biodiversity
Increased air temperatures, decreased snowpack accumulation, earlier snowmelt	Compacted soils, decreased infiltration, increased surface runoff	Reduced soil moisture for plants, reduced productivity, reductions in summer low flows, degraded aquatic habitat
Increased variability in timing and magnitude of precipitation events	Decreased biotic crusts and litter cover, increased surface erosion	Accelerated soil and nutrient loss, increased sedimentation
Warmer and drier in the summer	Reduced riparian vegetation, loss of shade, increased stream width	Increased stream temperatures, increased stress on cold-water fish and aquatic organisms
Increased variability in runoff	Reduced root strength of riparian plants, trampled streambanks, streambank erosion	Accelerated streambank erosion and increased sedimentation, degraded water quality and aquatic habitats
Increased variability in runoff	Incised stream channels	Degraded aquatic habitats, hydrologically disconnected floodplains, reduced low flows

National Forest Management Act [NFMA] of 1976), whereas others cross agency boundaries (e.g., Endangered Species Act [ESA] of 1973; Clean Water Act [CWA] of 1972). A common mission of federal land management agencies is “to sustain the health, diversity, and productivity of public lands” (GAO 2007, p. 12). Further, each of these agencies has ample authority and responsibility to adjust management to respond to climate change (GAO 2007) and other stressors.

The FS and BLM are directed to maintain and improve the condition of the public rangelands so that they become as productive as feasible for all rangeland values. As defined, “range condition” encompasses factors such as soil quality, forage values, wildlife habitat, watershed and plant communities, and the present state of vegetation of a range site in relation to the potential plant community for that site (Public Rangelands Improvement Act of 1978). BLM lands and national forests must be managed for sustained yield of a wide array of multiple uses, values, and ecosystem services, including wildlife and fish, watershed, recreation, timber, and range. Relevant statutes call for management that meets societal needs, without impairing the productivity of the land or the quality of the environment, and which considers the “relative values” of the various resources, not necessarily the combination of uses that will give the greatest economic return or the greatest unit output (Multiple-Use Sustained-Yield Act of 1960; Federal Land Policy and Management Act of 1976 [FLPMA]).

FLPMA directs the BLM to “take any action necessary to prevent unnecessary or undue degradation” of the public lands. Under NFMA, FS management must provide for diversity of plant and animal communities based on the suitability and capability of the specific land area. FLMPA also authorizes both agencies to “cancel, suspend, or modify” grazing permits and to determine that “grazing uses should be discontinued (either temporarily or permanently) on certain lands.” FLPMA explicitly recognizes the BLM’s authority (with congressional oversight) to “totally eliminate” grazing from large areas (> 405 km^2) of public lands. These authorities are reinforced by law providing that grazing permits are not property rights (*Public Lands Council v. Babbitt* 2000).

While federal agencies have primary authority to manage federal public lands and thus wildlife *habitats* on these lands, states retain primary management authority over resident *wildlife*, unless preempted, as by the WFRHBA or ESA (*Kleppe v. New Mexico* 1976). Under WFRHBA, wild, free-roaming horses and burros (i.e., feral) by law have been declared “wildlife” and an integral part of the natural system of the public lands where they are to be managed in a manner that is designed to achieve and maintain a thriving natural ecological balance.

Restoring Ungulate-Altered Ecosystems

Because livestock use is so widespread on public lands in the American West, management actions directed at ecological restoration (e.g., livestock removal, substantial reductions in numbers or length of season, extended or regular periods of rest) need to be accomplished at landscape scales. Such approaches, often referred to as passive restoration, are generally the most ecologically effective and economically efficient for recovering altered ecosystems because they address the root causes of degradation and allow natural recovery processes to operate (Kauffman and others 1997; Rieman and Isaak 2010). Furthermore, reducing the impact of current stressors is a “no regrets” adaptation strategy that could be taken now to help enhance

Fig. 5 Examples of riparian and stream recovery in the western United States after the removal of livestock grazing: Hart Mountain National Antelope Refuge, Oregon, in **a** October 1989 and **b** September 2010 after 18 years of livestock removal; Strawberry River, Utah, in **c** August 2002 after 13 years of livestock removal and **d** July 2003 illustrating improved streambank protection and riparian productivity as beaver reoccupy this river system; and San Pedro River, Arizona in **e** June 1987 and **f** June 1991 after 4 years of livestock removal. *Photographs* **a** Fish and Wildlife Service, Hart Mountain National Antelope Refuge, **b** J Rhodes, **c** and **d** US Forest Service, Uintah National Forest, **e** and **f** Bureau of Land Management, San Pedro Riparian National Conservation Area

ecosystem resilience to climate change (Joyce and others 2008). This strategy is especially relevant to western ecosystems because removing or significantly reducing the cause of degradation (e.g., excessive ungulate use) is likely to be considerably more effective over the long term, in both costs and approach, than active treatments aimed at specific ecosystem components (e.g., controlling invasive plants) (BLM 2005). Furthermore, the possibility that passive restoration measures may not accomplish all ecological goals is an insufficient reason for *not* removing or reducing stressors at landscape scales.

For many areas of the American West, particularly riparian areas and other areas of high biodiversity, significantly reducing or eliminating ungulate stressors should, over time, result in the recovery of self-sustaining and ecologically robust ecosystems (Kauffman and others 1997; Floyd and others 2003; Allington and Valone 2010; Fig. 5). Indeed, various studies and reviews have concluded that the most effective way to restore riparian areas and aquatic systems is to exclude livestock either temporarily (with subsequent changed management) or long-term (e.g., Platts 1991;BLM and FS 1994; Dobkin and others

1998; NRC 2002; Seavy and others 2009: Fleischner 2010). Recovering channel form and riparian soils and vegetation by reducing ungulate impacts is also a viable management tool for increasing summer baseflows (Ponce and Lindquist 1990; Rhodes and others 1994).

In severely degraded areas, initiating recovery may require active measures in addition to the removal/reduction of stressors. For example, where native seed banks have been depleted, reestablishing missing species may require planting seeds or propagules from adjacent areas or refugia (e.g., Welch 2005). While active restoration approaches in herbivory-degraded landscapes may have some utility, such projects are often small in scope, expensive, and unlikely to be self-sustaining; some can cause unanticipated negative effects (Kauffman and others 1997). Furthermore, if ungulate grazing effects continue, any benefits from active restoration are likely to be transient and limited. Therefore, addressing the underlying causes of degradation should be the first priority for effectively restoring altered public-land ecosystems.

The ecological effectiveness and low cost of wide-scale reduction in ungulate use for restoring public-land ecosystems, coupled with the scarcity of restoration resources, provide a forceful case for minimizing ungulate impacts. Other conservation measures are unlikely to make as great a contribution to ameliorating landscape-scale effects from climate change or to do so at such a low fiscal cost. As Isaak and others (2012, p. 514) noted with regard to the impacts of climate change on widely-imperiled salmonids: "…conservation projects are likely to greatly exceed available resources, so strategic prioritization schemes are essential."

Although restoration of desertified lands was once thought unlikely, recovery in the form of significant increases in perennial grass cover has recently been reported at several such sites around the world where livestock have been absent for more than 20 years (Floyd and others 2003; Allington and Valone 2010; Peters and others 2011). At a desertified site in Arizona that had been ungrazed for 39 years, infiltration rates were significantly (24 %) higher (compared to grazed areas) and nutrient levels were elevated in the bare ground, inter-shrub areas (Allington and Valone 2010). The change in vegetative structure also affected other taxa (e.g., increased small mammal diversity) where grazing had been excluded (Valone and others 2002). The notion that regime shifts caused by grazing are irreversible (e.g., Bestelmeyer and others 2004) may be due to the relative paucity of large-scale, ungulate-degraded systems where grazing has been halted for sufficiently long periods for recovery to occur.

Removing domestic livestock from large areas of public lands, or otherwise significantly reducing their impacts, is consistent with six of the seven approaches recommended for ecosystem adaptation to climate change (Julius and others 2008, pp. 1-3). Specifically, removing livestock would (1) protect key ecosystem features (e.g., soil properties, riparian areas); (2) reduce anthropogenic stressors; (3) ensure representation (i.e., protect a variety of forms of a species or ecosystem); (4) ensure replication (i.e., protect more than one example of each ecosystem or population); (5) help restore ecosystems; and (6) protect refugia (i.e., areas that can serve as sources of "seed" for recovery or as destinations for climate-sensitive migrants). Although improved livestock management practices are being adopted on some public lands, such efforts have not been widely implemented. Public land managers have rarely used their authority to implement landscape-scale rest from livestock use, lowered frequency of use, or multi-stakeholder planning for innovative grazing systems to reduce impacts.

While our findings are largely focused on adaptation strategies for western landscapes, reducing ungulate impacts and restoring degraded plant and soil systems may also assist in mitigating any ongoing or future changes in regional energy and carbon cycles that contribute to global climate change. Simply removing livestock can increase soil carbon sequestration since grasslands with the greatest potential for increasing soil carbon storage are those that have been depleted in the past by poor management (Wu and others 2008, citing Jones and Donnelly 2004). Riparian area restoration can also enhance carbon sequestration (Flynn and others 2009).

Socioeconomic Considerations

A comprehensive assessment of the socioeconomic effects of changes in ungulate management on public lands is beyond the scope of this paper. However, herein we identify a few of the *general* costs and benefits associated with implementing our recommendations (see next section), particularly with regard to domestic livestock grazing. The socioeconomic effects of altering ungulate management on public lands will ultimately depend on the type, magnitude, and location of changes undertaken by federal and state agencies.

Ranching is a contemporary and historically significant aspect of the rural West's social fabric. Yet, ranchers' stated preferences in response to grazing policy changes are as diverse as the ranchers themselves, and include intensifying, extensifying, diversifying, or selling their operations (Genter and Tanaka 2002). Surveys indicate that most ranchers are motivated more by amenity and lifestyle attributes than by profits (Torell and others 2001, Genter and Tanaka 2002). Indeed, economic returns from ranching are lower than any other investments with similar risk

(Torrell and others 2001) and public-land grazing's contributions to income and jobs in the West are relatively small fractions of the region's totals (BLM and FS 1994; Power 1996).

If livestock grazing on public lands were discontinued or curtailed significantly, some operations would see reduced incomes and ranch values, some rural communities would experience negative economic impacts, and the social fabric of those communities could be altered (Genter and Tanaka 2002). But for most rural economies, and the West in general, the economic impacts of managing public lands to emphasize environmental amenities would be relatively minor to modestly positive (Mathews and others 2002). Other economic effects could include savings to the US Treasury because federal grazing fees on BLM and FS lands cover only about one-sixth of the agencies' administration costs (Vincent 2012). Most significantly, improved ecosystem function would lead to enhanced ecosystem services, with broad economic benefits. Various studies have documented that the economic values of other public-land resources (e.g., water, timber, recreation, and wilderness) are many times larger than that of grazing (Haynes and others 1997; Laitos and Carr 1999; Patterson and Coelho 2009).

Facilitating adaptation to climate change will require changes in the management of public-land ecosystems impacted by ungulates. *How* ungulate management policy changes should be accomplished is a matter for the agencies, the public, and others. The recommendations and conclusions presented in the following section are based solely on ecological considerations and the federal agencies' legal authority and obligations.

Recommendations

We propose that large areas of BLM and FS lands should become free of use by livestock and feral ungulates (Table 2) to help initiate and speed the recovery of affected ecosystems as well as provide benchmarks or controls for assessing the effects of "grazing versus no-grazing" at significant spatial scales under a changing climate. Further, large areas of livestock exclusion allow for understanding potential recovery foregone in areas where livestock grazing is continued (Bock and others 1993).

While lowering grazing pressure rather than discontinuing use might be effective in some circumstances, public land managers need to rigorously assess whether such use is compatible with the maintenance or recovery of ecosystem attributes such as soils, watershed hydrology, and native plant and animal communities. In such cases, the contemporary status of at least some of the key attributes and their rates of change should be carefully monitored to ascertain whether continued use is consistent with ecological recovery, particularly as the climate shifts (e.g., Karr and Rossano 2001, Karr 2004; LaPaix and others 2009). To the extent possible, assessments of recovering areas should be compared to similar measurements in reference areas (i.e., areas exhibiting high ecological integrity) or areas where ungulate impacts had earlier been removed or minimized (Angermeier and Karr 1994; Dobkin and others 1998). Such comparisons are crucial if scientists and managers are to confirm whether managed systems are attaining restoration goals and to determine needs for intervention, such as reintroducing previously extirpated species. Unfortunately, testing for impacts of livestock use at landscape scales is hampered by the lack of large, ungrazed areas in the western US (e.g., Floyd and others 2003; FWS 2010).

Shifting the burden of proof for continuing, rather than significantly reducing or eliminating ungulate grazing is warranted due to the extensive body of evidence on ecosystem impacts caused by ungulates (i.e., consumers) and the added ecosystem stress caused by climate change. As Estes and others (2011, p. 306) recommended: "[T]he burden of proof [should] be shifted to show, for any ecosystem, that consumers do (or did) not exert strong cascading effects" (see also Henjum and others 1994; Kondolf 1994; Rhodes and others 1994). Current livestock or feral

Table 2 Priority areas for permanently removing livestock and feral ungulates from Bureau of Land Management and US Forest Service lands to reduce or eliminate their detrimental ecological effects

Watersheds and other large areas that contain a variety of ecotypes to ensure that major ecological and societal benefits of more resilient and healthy ecosystems on public lands will occur in the face of climate change
Areas where ungulate effects extend beyond the immediate site (e.g., wetlands and riparian areas impact many wildlife species and ecosystem services with cascading implications beyond the area grazed)
Localized areas that are easily damaged by ungulates, either inherently (e.g., biological crusts or erodible soils) or as the result of a temporary condition (e.g., recent fire or flood disturbances, or degraded from previous management and thus fragile during a recovery period).
Rare ecosystem types (e.g., perched wetlands) or locations with imperiled species (e.g., aspen stands and understory plant communities, endemic species with limited range), including fish and wildlife species adversely affected by grazing and at-risk and/or listed under the ESA
Non-use areas (i.e., ungrazed by livestock) or exclosures embedded within larger areas where livestock grazing continues. Such non-use areas should be located in representative ecotypes so that actual rates of recovery (in the absence of grazing impacts) can be assessed relative to resource trend and condition data in adjacent areas that continue to be grazed
Areas where the combined effects of livestock, wild ungulates, and feral ungulates are causing significant ecological impacts

ungulate use should continue only where stocking rates, frequency, and timing can be demonstrated, in comparison with landscape-scale reference areas, exclosures, or other appropriate non-use areas, to be compatible with maintaining or recovering key ecological functions and native species complexes. Furthermore, such use should be allowed only when monitoring is adequate to determine the effects of continued grazing in comparison to areas without grazing.

Where wild native ungulates, such as elk or deer, have degraded plant communities through excessive herbivory (e.g., long-term suppression of woody browse species [Weisberg and Coughenour 2003; Beschta and Ripple 2009; Ripple and others 2010]), state wildlife agencies and federal land managers need to cooperate in controlling or reducing those impacts. A potentially important tool for restoring ecosystems degraded by excessive ungulate herbivory is reintroduction or recolonization of apex predators. In areas of public land that are sufficiently large and contain suitable habitat, allowing apex predators to become established at ecologically effective densities (Soulé and others 2003, 2005) could help regulate the behavior and density of wild ungulate populations, aiding the recovery of degraded ecosystems (Miller and others 2001; Ripple and others 2010; Estes and others 2011). Ending government predator control programs and reintroducing predators will have fewer conflicts with livestock grazing where the latter has been discontinued in large, contiguous public-land areas. However, the extent to which large predators might also help control populations of feral horses and burros is not known.

Additionally, we recommend removing livestock and feral ungulates from national parks, monuments, wilderness areas, and wildlife refuges wherever possible and managing wild ungulates to minimize their potential to adversely affect soil, water, vegetation, and wildlife populations or impair ecological processes. Where key large predators are absent or unable to attain ecologically functional densities, federal agencies should coordinate with state wildlife agencies in managing wild ungulate populations to prevent excessive effects of these large herbivores on native plant and animal communities.

Conclusions

Average global temperatures are increasing and precipitation regimes changing at greater rates than at any time in recent centuries. Contemporary trends are expected to continue and intensify for decades, even if comprehensive mitigations regarding climate change are implemented immediately. The inevitability of these trends requires adaptation to climate change as a central planning goal on federal lands.

Historical and on-going ungulate use has affected soils, vegetation, wildlife, and water resources on vast expanses of public forests, shrublands, and grasslands across the American West in ways that are likely to accentuate any climate impacts on these resources. Although the effects of ungulate use vary across landscapes, this variability is more a matter of degree than type.

If effective adaptations to the adverse effects of climate change are to be accomplished on western public lands, large-scale reductions or cessation of ecosystem stressors associated with ungulate use are crucial. Federal and state land management agencies should seek and make wide use of opportunities to reduce significant ungulate impacts in order to facilitate ecosystem recovery and improve resiliency. Such actions represent the most effective and extensive means for helping maintain or improve the ecological integrity of western landscapes and for the continued provision of valuable ecosystem services during a changing climate.

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Abstract

Surface fire intensity (kilowatts per metre) and crown fire initiation were predicted using Rothermel's 1972 and Van Wagner's 1977 fire models with fuel data from 47 upland subalpine conifer stands varying in age from 22-258 yr and 35 yr of daily weather data (fuel moisture and wind speeds). Rothermel's intensity model was divided into a fuel component variable and weather component variable, which were then used to examine the relative roles of fuel and weather on surface fire intensity (kilowatts per metre). Similar variables were defined in the crown fire initiation model of Van Wagner. Both surface fire intensity and crown fire initiation were strongly related to the weather components and weakly related to the fuel components, due to much greater variability in weather than fuel, and stronger relationship to the fire behavior mechanisms for weather than for fuel. Fire intensity was correlated to annual area burned; large area burned years had higher fire intensity predictions than smaller area burned years. The reason for this difference was attributed directly to the weather variable frequency distribution, which was shifted towards more extreme values in years in which large areas burned. During extreme weather conditions, the relative importance of fuels diminishes since all stands achieve the threshold required to permit crown fire development. This is important since most of the area burned in subalpine forests has historically occurred during very extreme weather (i.e., drought coupled to high winds). The fire behavior relationships predicted in the models support the concept that forest fire behavior is determined primarily by weather variation among years rather than fuel variation associated with stand age.

S2 Table. Authors, sites, the Weibull mean ITFI estimate, and the calibrated or predicted PMFI/FR for the merged 342-site dataset.

State/Author(s)†	Sites	State	Weibull Mean ITFI (years)	Calibrated or predicted	Calibrated/ Predicted PMFI/FR (years)
ARIZONA					
Dieterich and Hibbert (1990)	Battle Flat	AZ	6.31	Calibrated	7.20
Kaib and Swetnam, no publ.	Mt. Ord	AZ	8.72	Predicted	10.60
Dieterich (1980)	Chimney Springs	AZ	8.80	Predicted	10.70
Swetnam and Baisan (1996)	Walnut Canyon	AZ	8.81	Predicted	10.71
Swetnam et al. (2001)	Palisades	AZ	9.21	Predicted	11.20
Farris et al. (2013)	Centennial Forest	AZ	-	Predicted	12.03
Fulé et al. (2003a)	Galahad Point	AZ	11.26	Calibrated	12.50
Seklecki et al. (1996)	Rustler Park	AZ	10.32	Predicted	12.55
Farris et al. (2013)	Mica Mountain	AZ	-	Predicted	12.57
Baisan et al. (1998)	Rose Canyon Lower	AZ	10.80	Predicted	13.13
Danzer (1998)	Sawmill Canyon	AZ	10.92	Predicted	13.28
Fulé et al. (2003b)	Fire Point	AZ	11.25	Calibrated	13.60
Baisan and Swetnam (1990)	Mica Mountain	AZ	12.48	Calibrated	15.00
Baisan et al. (1998)	Mount Lemmon	AZ	12.36	Predicted	15.03
Fulé et al. (2003b)	Powell Plateau	AZ	13.68	Calibrated	15.40
Baisan et al. (1998)	Rose Canyon Upper	AZ	12.74	Predicted	15.49
Swetnam and Baisan (1996)	Josephine Saddle	AZ	12.90	Predicted	15.69
Baisan et al. (1998)	Rose Canyon East	AZ	13.40	Predicted	16.29
Fulé et al. (2003b)	Swamp Ridge	AZ	14.56	Calibrated	17.10
Danzer (1998)	Pat Scott Peak	AZ	13.06	Calibrated	17.70
Fulé et al. (2003b)	Grandview	AZ	14.89	Calibrated	17.90
Fulé et al. (2003b)	Rainbow Plateau	AZ	14.10	Calibrated	18.00
Fulé et al. (1997)	Camp Navajo	AZ	13.02	Calibrated	19.00
Huffman et al. (2015)	Mogollon Rim	AZ	-	Predicted	19.25
Heinlein et al. (2005)	San Francisco Peaks West	AZ	15.50	Calibrated	20.60
Dieterick (1983)	Thomas Creek	AZ	17.31	Calibrated	22.10
Heinlein et al. (2005)	San Francisco Peaks East	AZ	17.32	Calibrated	23.20
Fulé et al. (2003b; Dugan and Baker (2014)	Grandview	AZ	18.40	Calibrated	25.70
CALIFORNIA					
Caprio and Swetnam (1995)	Ash Peak Ridge	CA	7.04	Predicted	8.56

Taylor and Skinner (1998)	Thompson Ridge: 1850-1904	CA	-	Calibrated	12.30
Scholl and Taylor (2010)	Tuolomne River	CA	-	Calibrated	13.00
Beaty and Taylor (2001)	South-facing	CA	-	Calibrated	17.40
Caprio and Swetnam (1995)	Bobcat Point Pine	CA	15.09	Predicted	18.35
Taylor and Skinner (1998)	Thompson Ridge: 1626-1849	CA	-	Calibrated	19.00
Taylor and Skinner (2003)	Hayfork: 1628-1849	CA	22.86	Calibrated	20.00
Bekker and Taylor (2001)	White fir-Jeffrey pine	CA	-	Calibrated	21.50
Caprio and Swetnam (1995)	High Sierra Ridge Pine	CA	18.75	Predicted	22.80
Taylor (2000)	Prospect Peak: Jeffrey Pine	CA	-	Calibrated	24.50
Beaty and Taylor (2001)	Northern headwaters	CA	-	Calibrated	27.20
Beaty and Taylor (2001)	Combined study areas	CA	-	Calibrated	28.20
Taylor (2000)	Prospect Peak: Jeffrey Pine-White fir	CA	-	Calibrated	31.30
Bekker and Taylor (2001)	White fir-Sugar pine	CA	-	Calibrated	33.70
Beaty and Taylor (2001)	Southern headwaters	CA	-	Calibrated	37.20
Swetnam et al., no publication	Buck Rock Flat	CA	32.11	Predicted	39.05
Beaty and Taylor (2001)	North-facing	CA	-	Calibrated	42.50
Fiegener (2002)	Teakettle	CA	-	Predicted	49.87
Fiegener (2002)	Teakettle	CA	-	Predicted	75.31
Everett (2003)	Black Mountain	CA	-	Predicted	269.41
Everett (2003)	Big Pine Flat	CA	-	Predicted	327.16
COLORADO					
Grissino-Mayer et al. (2004)	Plateau	CO	15.86	Calibrated	15.20
Grissino-Mayer et al. (2004)	Five Pine Canyon	CO	15.96	Calibrated	15.80
Veblen et al. (2000)	BM34	CO	14.31	Predicted	17.40
Brown and Wu (2005)	Archuleta Mesa Plot A05	CO	15.19	Predicted	18.47
Grissino-Mayer et al. (2004)	Benson Creek	CO	14.86	Calibrated	20.70
Grissino-Mayer et al. (2004)	Hermosa Creek	CO	20.97	Predicted	25.50
Brown and Wu (2005)	Archuleta Mesa Plot A1	CO	21.49	Predicted	26.13
Grissino-Mayer et al. (2004)	Turkey Springs	CO	21.14	Calibrated	26.40
Veblen et al. (2000)	BM31	CO	23.68	Predicted	28.79
Grissino-Mayer et al. (2004)	Smoothing Iron	CO	25.74	Calibrated	29.00
Grissino-Mayer et al. (2004)	Taylor Creek	CO	26.72	Calibrated	29.20
Brown and Shepperd (2001)	Wet Mountains South	CO	25.85	Predicted	31.43
Brown and Wu (2005)	Archuleta Mesa	CO	23.13	Calibrated	32.10
Veblen et al. (2000)	BM14	CO	26.79	Predicted	32.58

Brown and Shepperd (2001)	M Kaufmanns Cabin	CO	26.80	Predicted	32.59
Bigio et al. (2010)	Vallecito Country Market	CO	27.75	Calibrated	32.60
Grissino-Mayer et al. (2004)	Monument	CO	33.16	Calibrated	37.50
Bigio (2013)	Marina Basin	CO	30.95	Predicted	37.64
Grissino-Mayer et al. (2004)	Burnette Canyon	CO	33.83	Predicted	41.14
Brown and Shepperd (2001)	Black Mountain	CO	35.28	Predicted	42.90
Brown and Wu (2005)	Archuleta Mesa Plot AA1	CO	36.00	Predicted	43.78
Veblen et al. (2000)	BM15	CO	30.12	Calibrated	44.92
Fulé et al. (2009)	Lower Middle Mountain	CO	30.53	Calibrated	46.80
Veblen et al. (2000)	BM28	CO	39.34	Predicted	47.84
Brown and Wu (2005)	Archuleta Mesa Plot B2	CO	40.02	Predicted	48.66
Veblen et al. (2000)	BM24	CO	40.26	Predicted	48.96
Brown and Wu (2005)	Archuleta Mesa Plot C2	CO	41.52	Predicted	50.49
Bigio et al. (2010)	Haflin Canyon	CO	42.11	Calibrated	50.80
Veblen et al. (2000)	BM11	CO	42.27	Predicted	51.40
Brown and Shepperd (2001)	Manitou Demo Plot	CO	42.34	Predicted	51.49
Veblen et al. (2000)	BM8	CO	47.23	Predicted	57.43
Veblen et al. (2000)	BM9	CO	48.61	Predicted	59.11
Brown and Shepperd (2001)	Mica Mine	CO	49.71	Predicted	60.45
Brown and Wu (2005)	Archuleta Mesa Plot AA15	CO	50.41	Predicted	61.30
Veblen et al. (2000)	BM22	CO	-	Calibrated	61.90
Brown and Shepperd (2001)	Parachute Hill	CO	51.75	Predicted	62.93
Veblen et al. (2000)	BM20	CO	53.57	Predicted	65.14
Veblen et al. (2000)	BM32	CO	58.07	Predicted	70.61
Brown et al. (2000)	Hot Creek	CO	58.44	Predicted	71.06
Brown and Wu (2005)	Archuleta Mesa Plot B3	CO	58.97	Predicted	71.71
Veblen et al. (2000)	BM13	CO	59.80	Predicted	72.72
Bigio (2013)	Steven’s Canyon	CO	37.56	Calibrated	74.00
Brown and Wu (2005)	Archuleta Mesa Plot C5	CO	64.07	Predicted	77.91
Veblen et al. (2000)	BM23	CO	63.59	Calibrated	80.30
Brown and Shepperd (2001)	Left Hand Canyon	CO	67.75	Predicted	82.38
Veblen et al. (2000)	BM5	CO	71.75	Predicted	87.25
Veblen et al. (2000)	BM12	CO	72.73	Predicted	88.44
Brown and Wu (2005)	Archuleta Mesa Plot A15	CO	73.80	Predicted	89.74
Donnegan et al. (2001)	BSA Shortcut	CO	75.54	Predicted	91.86
Brown and Shepperd (2001)	Cheesman Lake South	CO	76.87	Predicted	93.47

Donnegan et al. (2001)	Badger Mountain	CO	92.84	Calibrated	94.10
Brown and Shepperd (2001)	Washout Gulch Burn	CO	77.56	Predicted	94.31
Brown and Shepperd (2001)	Cheesman Lake North	CO	78.62	Predicted	95.60
Veblen et al. (2000)	BM6	CO	80.88	Calibrated	100.00
Veblen et al. (2000)	BM18	CO	78.98	Calibrated	103.50
Brown and Shepperd (2001)	Old Tree Cluster	CO	86.85	Predicted	105.61
Donnegan et al. (2001)	Salt Creek	CO	89.07	Calibrated	106.70
Brown and Shepperd (2001)	Lone Pine	CO	88.59	Predicted	107.73
Veblen et al. (2000)	BM39	CO	88.81	Predicted	107.99
Veblen et al. (2000)	BM10	CO	-	Calibrated	112.60
Veblen et al. (2000)	BM21	CO	99.22	Predicted	120.65
Brown and Shepperd (2001)	Lone Pine Upper	CO	107.72	Predicted	130.99
Donnegan et al. (2001)	China Wall	CO	-	Calibrated	138.30
Veblen et al. (2000)	BM19	CO	114.59	Predicted	139.34
Veblen et al. (2000)	BM17	CO	143.99	Predicted	175.09
IDAHO					
Heyerdahl et al. (2008)	Warm Springs Ridge	ID	13.88	Predicted	16.88
Heyerdahl et al. (2008)	Bannock Creek	ID	13.98	Predicted	17.00
Heyerdahl et al. (2008)	Wash Creek	ID	15.98	Predicted	19.43
Heyerdahl et al. (2008)	Keating Ridge	ID	22.26	Predicted	27.07
Heyerdahl et al. (2008)	Cove Mountain	ID	25.53	Predicted	31.04
Shapiro-Miller et al. (2007)	Powderhouse	ID	23.89	Calibrated	32.95
Heyerdahl et al. (2008)	Lowman RNA	ID	30.73	Predicted	37.37
MONTANA					
Heyerdahl et al. (2008)	Sophie Lake	MT	10.90	Predicted	13.25
Heyerdahl et al. (2008)	Sheldon Flats	MT	11.05	Predicted	13.44
Heyerdahl et al. (2008)	Butler Creek	MT	12.22	Predicted	14.86
Heyerdahl et al. (2008)	Blue Mountain	MT	12.28	Predicted	14.93
Heyerdahl et al. (2008)	McCormick Creek	MT	18.00	Calibrated	19.40
Heyerdahl et al. (2008)	McMillan Mountain	MT	17.71	Predicted	21.54
Heyerdahl et al. (2008)	Corona Road	MT	19.25	Predicted	23.41
Heyerdahl et al. (2008)	Hunter Point	MT	19.84	Predicted	24.13
Heyerdahl et al. (2008)	Sheafman Creek	MT	21.06	Predicted	25.61
Jones (2005)	Lubrecht	MT	23.26	Calibrated	27.40
Heyerdahl et al. (2008)	Crane Lookout	MT	25.47	Predicted	30.97
Heyerdahl et al. (2008)	Sawmill Creek RNA	MT	27.00	Predicted	32.83

NEW MEXICO					
Brown et al. (2001)	Pines at Sunspot	NM	8.40	Predicted	10.21
Swetnam and Dieterich (1985)	Langstroth Mesa	NM	8.62	Predicted	10.48
Kaye and Swetnam (1999)	Lower San Andreas	NM	9.74	Predicted	11.84
Morino (1996)	Upper Fillmore West	NM	10.55	Predicted	12.83
Grissino-Mayer & Swetnam (1997)	Cerro Bandera North	NM	10.71	Predicted	13.02
Swetnam and Dieterich (1985)	Gilita Ridge	NM	10.81	Predicted	13.14
Swetnam and Dieterich (1985)	McKenna Park	NM	11.05	Predicted	13.44
Kaye and Swetnam (1999)	Lower Pine Spring	NM	11.56	Predicted	14.06
Farris et al. (2013)	Monument Canyon	NM	-	Predicted	14.31
Brown et al. (2001)	James Ridge	NM	12.11	Predicted	14.73
Swetnam et al., no publication	Cerro Balitas	NM	12.32	Predicted	14.98
Morino (1996)	Upper Fillmore Side Cany. 1	NM	12.48	Predicted	15.18
Kaye and Swetnam (1999)	Upper San Andreas	NM	13.02	Predicted	15.83
Grissino-Mayer & Swetnam (1997)	Cerro Bandera East	NM	13.10	Predicted	15.93
Morino (1996)	Snag Saddle	NM	14.14	Predicted	17.19
Baisan and Swetnam (1997)	Capilla Peak Campground	NM	14.45	Predicted	17.57
Morino (1996)	Fillmore Side Canyon 2	NM	14.69	Predicted	17.86
Touchan et al. (1996)	Clear Creek Campground	NM	15.61	Calibrated	17.90
Grissino-Mayer & Swetnam (1997)	Candelaria	NM	14.97	Predicted	18.20
Grissino-Mayer & Swetnam (1997)	La Marchanita	NM	14.97	Predicted	18.20
Kaye and Swetnam (1999)	Cherry Canyon	NM	15.23	Predicted	18.52
Swetnam et al. (2001)	Black Mountain	NM	15.91	Predicted	19.35
Baisan and Swetnam (1997)	Canon de Turrieta	NM	16.02	Predicted	19.48
Morino (1996)	Rock House Spring	NM	16.10	Predicted	19.58
Brown et al. (2001)	Monument Canyon	NM	16.17	Predicted	19.66
Morino (1996)	Narrows	NM	16.42	Predicted	19.97
Kaye and Swetnam (1999)	Upper Pine Spring	NM	18.11	Predicted	22.02
Morino (1996)	Fillmore Side Canyon	NM	18.20	Predicted	22.13
Morino (1996)	Ledge Site	NM	18.44	Predicted	22.42
Brown et al. (2001)	Monument Canyon Upper	NM	18.66	Predicted	22.69
Touchan et al. (1996)	Pajarito Mountain Ridge	NM	19.04	Predicted	23.15
Baisan and Swetnam (1997)	La Luz Trail	NM	20.29	Predicted	24.67
Touch an et al. (1996)	Gallina Mesa	NM	18.54	Calibrated	24.70
Swetnam and Baisan (1996)	Ice Canyon	NM	21.58	Predicted	26.24
Swetnam (1990)	Bear Wallow	NM	21.74	Predicted	26.44

Swetnam and Baisan (1996)	Continental Divide Peak	NM	21.86	Predicted	26.58
Grissino-Mayer & Swetnam (1997)	Cerro Rendija	NM	22.19	Predicted	26.98
Grissino-Mayer & Swetnam (1997)	Mesita Blanca	NM	23.15	Predicted	28.15
Grissino-Mayer & Swetnam (1997)	Lost Woman	NM	23.50	Predicted	28.58
Swetnam et al., no publication	Laguna Garule	NM	23.79	Predicted	28.93
Grissino-Mayer & Swetnam (1997)	Hoya de Cibola Lava Flow	NM	24.03	Predicted	29.22
Swetnam and Baisan (1996)	El Calderon	NM	24.54	Predicted	29.84
Allen (1989)	Frijoles Canyon	NM	-	Predicted	30.88
Brown et al. (2001)	Delworth	NM	25.73	Predicted	31.29
Brown et al. (2001)	Fir Campground	NM	25.85	Predicted	31.43
Brown et al. (2001)	Peake Canyon	NM	28.63	Predicted	34.81
Touchan et al. (1996)	Camp May East	NM	28.91	Predicted	35.15
Brown et al. (2001)	Cosmic Ray Obs	NM	29.79	Predicted	36.22
Swetnam and Baisan (1996)	Continental Divide Saddle	NM	29.81	Predicted	36.25
Brown et al. (2001)	Sunspot	NM	29.89	Predicted	36.35
Baisan et al., no publication	Bonita Canyon	NM	30.70	Predicted	37.33
Touchan et al. (1996)	Canada Bonita South	NM	31.53	Predicted	38.34
Margolis and Balmat (2009)	Santa Fe Watershed Ponderosa Pine	NM	25.78	Calibrated	39.80
Touchan et al. (1996)	Cerro Pedernal	NM	33.56	Predicted	40.81
Grissino-Mayer & Swetnam (1997)	Hidden Kipuka	NM	38.90	Predicted	47.30
Margolis and Balmat (2009)	Santa Fe Watershed Dry Mixed Conifer	NM	49.46	Calibrated	74.70
OREGON					
Heyerdahl (1997), Heyerdahl et al. (2001)	Baker City	OR	18.11	Calibrated	15.30
Heyerdahl (1997), Heyerdahl et al. (2001)	Dugout	OR	21.39	Calibrated	15.30
Maruoka (1994)	Spring Mountain (12)	OR	16.40	Predicted	19.94
Heyerdahl (1997), Heyerdahl et al. (2001)	Baker City	OR	18.11	Calibrated	22.70
Maruoka (1994)	Seed Orchard (4)	OR	18.69	Predicted	22.73
Maruoka (1994)	Widow's Creek (1)	OR	19.72	Predicted	23.98
Maruoka (1994)	East Camp Creek (5)	OR	20.17	Predicted	24.53
Heyerdahl (1997), Heyerdahl et al. (2001)	Dugout	OR	21.39	Calibrated	24.80
Maruoka (1994)	Smoothing Iron Ridge (15)	OR	21.93	Predicted	26.67
Maruoka (1994)	Little Bear Burn (7)	OR	23.19	Predicted	28.20

Heyerdahl (1997), Heyerdahl et al. (2001)	Imnaha	OR	33.82	Calibrated	28.40
Maruoka (1994)	Five Mile Creek (6)	OR	24.11	Predicted	29.32
Maruoka (1994)	West Myrtle Creek (8)	OR	24.67	Predicted	30.00
Bork (1984)	Pringle Butte	OR	-	Calibrated	31.00
Heyerdahl, no publication	McKay Creek	OR	24.47	Calibrated	35.30
Heyerdahl (1997), Heyerdahl et al. (2001)	Imnaha	OR	33.82	Calibrated	37.50
Heyerdahl, no publication	Lytle Creek	OR	26.70	Calibrated	37.57
Maruoka (1994)	Raddue (2)	OR	33.27	Predicted	40.46
Heyerdahl, no publication	Green Ridge	OR	34.62	Calibrated	42.96
Maruoka (1994)	Troy (14)	OR	36.84	Predicted	44.80
Maruoka (1994)	Dixie Butte (3)	OR	43.64	Predicted	53.07
Bork (1984)	Lookout Mountain	OR	-	Calibrated	77.00
Bork (1984)	Cabin Lake	OR	-	Calibrated	79.00
Arabas et al. (2006)	Lava Cast Forest	OR	37.00	Calibrated	83.25
SOUTH DAKOTA					
Brown and Sieg (1999)	Pigtail Bridge	SD	17.42	Predicted	21.18
Brown and Sieg (1999)	Wind Cave North	SD	19.44	Predicted	23.64
Wienk et al. (2004)	Badger Game Prod. Area	SD	22.24	Predicted	27.04
Brown et al. (2008)	Mount Rushmore	SD	-	Calibrated	30.00
Brown (2003, 2006)	Black Hills Plot 105	SD	26.30	Predicted	31.98
Brown (2003, 2006)	Black Hills Plot 111	SD	26.85	Predicted	32.65
Brown and Sieg (1996)	Jewel Cave South	SD	27.20	Predicted	33.08
Brown (2003, 2006)	Black Hills Plot 204	SD	27.89	Predicted	33.91
Brown (2003)	Bear Lodge Central	SD	28.87	Predicted	35.11
Brown (2003, 2006)	Black Hills Plot 210	SD	28.95	Predicted	35.20
Brown (2003)	Reynold's Prairie	SD	28.95	Predicted	35.20
Brown (2003, 2006)	Black Hills Plot 213	SD	30.05	Predicted	36.54
Brown and Sieg (1996)	Jewel Cave North	SD	30.08	Predicted	36.58
Brown and Sieg (1999)	Gobbler Ridge	SD	31.02	Predicted	37.72
Brown (2003, 2006)	Black Hills Plot 207	SD	32.25	Predicted	39.22
Brown (2003)	Riflepit Gulch West	SD	32.72	Predicted	39.79
Brown (2003, 2006)	Black Hills Plot 202	SD	32.72	Predicted	39.79
Brown (2003, 2006)	Black Hills Plot 109	SD	33.02	Predicted	40.15
Brown and Sieg (1996)	Jewel Cave East	SD	33.58	Calibrated	40.52
Brown (2003, 2006)	Black Hills Plot 209	SD	34.04	Predicted	41.39

Brown et al. (2000)	Upper Pine Mid-Basin	SD	34.78	Predicted	42.29
Brown (2003)	Black Hills Exp. Forest	SD	35.22	Predicted	42.83
Brown (2003, 2006)	Black Hills Plot 112	SD	36.02	Predicted	43.80
Brown and Sieg (1996)	Jewel Cave Central	SD	36.47	Predicted	44.35
Brown (2003)	Bear Lodge North	SD	37.75	Predicted	45.90
Brown (2003, 2006)	Black Hills Plot 205	SD	38.42	Predicted	46.72
Brown (2003, 2006)	Black Hills Plot 106	SD	38.84	Predicted	47.23
Brown (2003, 2006)	Black Hills Plot 208	SD	40.05	Predicted	48.70
Brown (2003, 2006)	Black Hills Plot 113	SD	40.11	Predicted	48.77
Brown (2003, 2006)	Black Hills Plot 114	SD	40.75	Predicted	49.55
Brown (2003, 2006)	Black Hills Plot 203	SD	41.21	Predicted	50.11
Brown (2003)	Riflepit Gulch East	SD	41.35	Predicted	50.28
Brown (2003)	Riflepit Gulch North	SD	42.56	Predicted	51.75
Brown (2003, 2006)	Black Hills Plot 101	SD	42.75	Predicted	51.98
Brown (2003, 2006)	Black Hills Plot 206	SD	44.86	Predicted	54.55
Brown (2003, 2006)	Black Hills Plot 201	SD	46.10	Predicted	56.06
Brown (2003, 2006)	Black Hills Plot 110	SD	46.16	Predicted	56.13
Brown (2003, 2006)	Black Hills Plot 108	SD	63.33	Predicted	77.01
Brown (2003, 2006)	Black Hills Plot 103	SD	36.95 §	Predicted	90.16
Brown (2003, 2006)	Black Hills Plot 104	SD	64.33 ¶	Predicted	158.70
WASHINGTON					
Everett et al. (2000)	Entiat Mud Creek overall	WA	-	Calibrated	11.00
Everett et al. (2000)	Nile Creek overall	WA	-	Calibrated	12.20
Kernan and Hessl (2010)	Entiat	WA	-	Calibrated	13.10
Everett et al. (2000)	Entiat Mud Creek 165	WA	11.92	Predicted	14.49
Everett et al. (2000)	Entiat Mud Creek 230	WA	11.98	Predicted	14.57
Everett et al. (2000)	Entiat Mud Creek 201	WA	12.41	Predicted	15.09
Everett et al. (2000)	Entiat Mud Creek 205	WA	12.50	Predicted	15.20
Kernan and Hessl (2010)	Swauk	WA	-	Calibrated	15.80
Everett et al. (2000)	Entiat Mud Creek 199	WA	13.08	Predicted	15.91
Everett et al. (2000)	Entiat Mud Creek 202	WA	13.70	Predicted	16.66
Kernan and Hessl (2010)	Nile Creek	WA	-	Calibrated	17.00
Everett et al. (2000)	Nile Creek 10	WA	14.21	Predicted	17.28
Everett et al. (2000)	Entiat Mud Creek 196	WA	14.29	Predicted	17.38
Everett et al. (2000)	Entiat Mud Creek 207	WA	15.07	Predicted	18.33
Everett et al. (2000)	Entiat Mud Creek 203	WA	15.32	Predicted	18.63

Everett et al. (2000)	Nile Creek 5	WA	15.46	Predicted	18.80
Everett et al. (2000)	Quartzite 1	WA	15.54	Predicted	18.90
Everett et al. (2000)	Entiat Mud Creek 208	WA	16.02	Predicted	19.48
Everett et al. (2000)	Entiat Mud Creek 167	WA	16.17	Predicted	19.66
Everett et al. (2000)	Frosty 8	WA	16.38	Predicted	19.92
Wright (1996); Wright and Agee (2004)	Teanaway Demonstration Area	WA	16.43	Calibrated	20.20
Everett et al. (2000)	Quartzite 8	WA	16.73	Predicted	20.34
Everett et al. (2000)	Entiat Mud Creek 200	WA	16.84	Predicted	20.48
Everett et al. (2000)	Quartzite 6	WA	16.93	Predicted	20.59
Everett et al. (2000)	Nile Creek 3	WA	17.22	Predicted	20.94
Everett et al. (2000)	Quartzite 3	WA	17.40	Predicted	21.16
Everett et al. (2000)	Frosty 7	WA	17.47	Predicted	21.24
Everett et al. (2000)	Nile Creek 9	WA	17.66	Predicted	21.47
Everett et al. (2000)	Entiat Mud Creek 206	WA	17.78	Predicted	21.62
Everett et al. (2000)	Quartzite 2	WA	18.10	Predicted	22.01
Everett et al. (2000)	Nile Creek 4	WA	18.14	Predicted	22.06
Everett et al. (2000)	Frosty 4	WA	18.51	Predicted	22.51
Everett et al. (2000)	Frosty 3	WA	18.68	Predicted	22.71
Everett et al. (2000)	Frosty 2	WA	19.00	Predicted	23.10
Everett et al. (2000)	South Deep 1	WA	19.01	Predicted	23.12
Everett et al. (2000)	Quartzite 4	WA	19.17	Predicted	23.31
Everett et al. (2000)	Nile Creek 8	WA	19.36	Predicted	23.54
Everett et al. (2000)	Quartzite 5	WA	19.53	Predicted	23.75
Everett et al. (2000)	Entiat Mud Creek 204	WA	19.76	Predicted	24.03
Everett et al. (2000)	Nile Creek 2	WA	20.28	Predicted	24.66
Everett et al. (2000)	Nile Creek 6	WA	20.36	Predicted	24.76
Everett et al. (2000)	Frosty 1	WA	20.38	Predicted	24.78
Everett et al. (2000)	Nile Creek 1	WA	21.09	Predicted	25.65
Wright (1996); Wright and Agee (2004)	Teanaway Demonstration Area	WA	16.43	Calibrated	26.00
Everett et al. (2000)	Entiat Mud Creek 211	WA	21.71	Predicted	26.40
Everett et al. (2000)	Twenty Mile 3	WA	22.85	Predicted	27.79
Everett et al. (2000)	Frosty 6	WA	23.20	Predicted	28.21
Everett et al. (2000)	Quartzite 7	WA	24.28	Predicted	29.52
Everett et al. (2000)	Nile Creek 11	WA	24.65	Predicted	29.97

Heyerdahl (1997), Heyerdahl et al. (2001)	Tucannon	WA	39.80	Calibrated	30.50
Everett et al. (2000)	Twenty Mile 4	WA	25.60	Predicted	31.13
Everett et al. (2000)	Twenty Mile 1	WA	26.03	Predicted	31.65
Everett et al. (2000)	Frosty 5	WA	26.10	Predicted	31.74
Everett et al. (2000)	Twenty Mile 2	WA	26.68	Predicted	32.44
Everett et al. (2000)	Nile Creek 7	WA	29.34	Predicted	35.68
Everett et al. (2000)	Twenty Mile 6	WA	29.54	Predicted	35.92
Everett et al. (2000)	Twenty Mile 7	WA	31.24	Predicted	37.99
Everett et al. (2000)	South Deep 6	WA	31.58	Predicted	38.40
Everett et al. (2000)	Twenty Mile 8	WA	32.42	Predicted	39.42
Everett et al. (2000)	Twenty Mile 13	WA	33.75	Predicted	41.04
Heyerdahl (1997), Heyerdahl et al. (2001)	Tucannon	WA	39.80	Calibrated	41.40
Kernan and Hessl (2010)	South Deep	WA	-	Calibrated	45.30
Everett et al. (2000)	Twenty Mile 9	WA	40.17	Predicted	48.85
Everett et al. (2000)	Twenty Mile 12	WA	40.20	Predicted	48.88
Everett et al. (2000)	South Deep 7	WA	41.91	Predicted	50.96
Everett et al. (2000)	South Deep 3	WA	43.33	Predicted	52.69
Everett et al. (2000)	South Deep 5	WA	44.13	Predicted	53.66
Everett et al. (2000)	South Deep 9	WA	47.74	Predicted	58.05
Everett et al. (2000)	Twenty Mile 10	WA	48.96	Predicted	59.54
Everett et al. (2000)	South Deep 11a	WA	51.65	Predicted	62.81
Everett et al. (2000)	South Deep 11b	WA	51.65	Predicted	62.81
Everett et al. (2000)	South Deep 12	WA	51.65	Predicted	62.81
Everett et al. (2000)	South Deep 14	WA	51.82	Predicted	63.01
Everett et al. (2000)	South Deep 4	WA	53.37	Predicted	64.90
Everett et al. (2000)	South Deep 10	WA	55.09	Predicted	66.99
Everett et al. (2000)	Twenty Mile 11	WA	67.38	Predicted	81.93
WYOMING					
Brown (2003)	Cold Springs Creek	WY	24.63	Predicted	29.95
Brown et al. (2000)	Ashenfelder Lower	WY	45.88	Predicted	55.79
Brown et al. (2000)	Ashenfelder Upper	WY	45.88	Predicted	55.79
MEXICO					
Skinner et al. (2008)	PINO (San Pedro Martir)	MX	19.04	Predicted	23.15
Skinner et al. (2008)	BLAN (San Pedro Martir)	MX	22.18	Predicted	26.97
Skinner et al. (2008)	PYRA (San Pedro Martir)	MX	24.89	Predicted	30.27

Skinner et al. (2008)	WEST (San Pedro Martir)	MX	25.02	Predicted	30.42
Skinner et al. (2008)	TASA (San Pedro Martir)	MX	26.55	Predicted	32.28
Skinner et al. (2008)	VALL (San Pedro Martir)	MX	26.67	Predicted	32.43
Skinner et al. (2008)	PUER (San Pedro Martir)	MX	28.07	Predicted	34.13
Skinner et al. (2008)	CORO (San Pedro Martir)	MX	30.94	Predicted	37.62
Skinner et al. (2008)	AZUL (San Pedro Martir)	MX	55.99	Predicted	68.08
CANADA-BRITISH COLUMBIA					
Heyerdahl et al. (2012)	Middle Stein River Valley	BC	27.93	Calibrated	40.49

Notes
† Observations are in increasing order of calibrated/predicted PMFI/FR within each state
‡ Missing observations in this column occur because some calibration cases did not have an FHX file and did not report this statistic in the publication.
§ Mean ITFI could not be estimated, but mean CFI-all could be, is reported here, and was used to estimate PMFI/FR
¶ Mean ITFI could not be estimated, but mean CFI-10% could be, is reported here, and was used to estimate PMFI/FR

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Do Bark Beetle Outbreaks Increase Wildfire Risks in the Central U.S. Rocky Mountains? Implications from Recent Research

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CONSERVATION ISSUES

Do Bark Beetle Outbreaks Increase Wildfire Risks in the Central U.S. Rocky Mountains? Implications from Recent Research

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ABSTRACT: Appropriate response to recent, widespread bark beetle (*Dendroctonus* spp.) outbreaks in the western United States has been the subject of much debate in scientific and policy circles. Among the proposed responses have been landscape-level mechanical treatments to prevent the further spread of outbreaks and to reduce the fire risk that is believed to be associated with insect-killed trees. We review the literature on the efficacy of silvicutural practices to control outbreaks and on fire risk following bark beetle outbreaks in several forest types. While research is ongoing and important questions remain unresolved, to date most available evidence indicates that bark beetle outbreaks do not substantially increase the risk of active crown fire in lodgepole pine (*Pinus contorta*) and spruce (*Picea engelmannii*)-fir (*Abies* spp.) forests under most conditions. Instead, active crown fires in these forest types are primarily contingent on dry conditions rather than variations in stand structure, such as those brought about by outbreaks. Preemptive thinning may reduce susceptibility to small outbreaks but is unlikely to reduce susceptibility to large, landscape-scale epidemics. Once beetle populations reach widespread epidemic levels, silvicultural strategies aimed at stopping them are not likely to reduce forest susceptibility to outbreaks. Furthermore, such silvicultural treatments could have substantial, unintended short- and long-term ecological costs associated with road access and an overall degradation of natural areas.

Index terms: bark beetles, *Dendroctonus*, forest health, forest management, wildfire

INTRODUCTION

Forests in the western United States are being affected by the largest outbreaks of bark beetles in at least a century, which has caused concern about forest health and wildfire risk and led to proposals for tree removal in natural areas such as roadless forests. Such proposals stem in part from the rationale that bark beetle outbreaks increase wildfire risks due to increased dead fuels and that widespread treatment in beetle-affected forests is needed to lower such risks. Here, we review available peer-reviewed literature to determine if: (1) bark beetle outbreaks are associated with a higher incidence of wildfires in forest types in the central Rockies; and (2) if silvicultural treatments are effective at lowering beetle-associated tree mortality before, during, and after outbreaks. We briefly review the impacts that additional logging roads associated with broad-scale tree removal may have on the ecology of roadless natural areas. Our results may have broader policy implications in western forests as concerns over insect outbreaks have led to proposals to reduce environmental protections in favor of widespread thinning and post-disturbance tree removal.

INTERACTIONS AMONG FOREST INSECTS AND FIRES

We examined the long-standing belief that insect outbreaks lead to increased risk of fire (USDA Forest Service 2011). A large body of literature indicates that the occurrence of large, severe fires in subalpine, lodgepole pine (*Pinus contorta*), and spruce (*Picea engelmannii)*-fir (*Abies* spp.) forests is strongly contingent on climatic conditions, especially drought (e.g., Kipfmueller and Baker 2000; Romme et al. 2006; Sibold and Veblen 2006; Schoennagel et al. 2007; Jenkins et al. 2008; Simard et al. 2008, 2011).

The debate on how outbreaks affect fire risk and hazard is ongoing, but recent work emphasizes that the effect of outbreaks on subsequent fire risk is complex and is contingent on time since last outbreak and on biophysical setting. To date, the majority of studies have found no increase in fire occurrence, extent, or severity following outbreaks of spruce beetle (*Dendroctonus rufipennis*) and mountain pine beetle (*Dendroctonus ponderosae*) in Colorado, Wyoming, and other areas (Bebi et al. 2003; Kulakowski et al. 2003; Bigler et al. 2005; Kulakowski and Veblen 2007; Jenkins et al. 2008; Simard et al. 2008, 2011).

Theoretically, the effect of outbreaks on subsequent fires may vary with the time since the outbreak occurred (Romme et al. 2006). For example, it is reasonable to expect that foliar moisture in trees killed by beetles will decrease and canopy density will be reduced during and immediately after an outbreak. In subsequent years, canopy density may be further reduced as dead needles and small branches fall from killed trees reducing canopy bulk density, but increasing surface fire hazard (i.e., the type, volume, and arrangement of fuels that determines the ease of ignition and resistance to control regardless of the fuel type's weather-influenced moisture content). Although such a relationship may theoretically increase the risk of surface fires, studies on the influence of outbreaks on subsequent stand-replacing fires, over a range of years since outbreak, have found little or no increase in surface or canopy fire occurrence, extent, or severity (Bebi et al. 2003; Kulakowski et al. 2003; Bigler et al. 2005; Kulakowski and Veblen 2007; Jenkins et al. 2008; Simard et al. 2008, 2011) (Table 1).

Fire and Mountain Pine Beetle Outbreaks in Lodgepole Pine Forests

Although outbreaks of mountain pine beetle do alter fuel structure (Page and Jenkins 2007; Klutsch et al. 2009; Simard et al. 2011), the actual effects of these changes in fuels on subsequent fire risk (i.e., the chance that a fire might start based on all causative agents such as fuel hazard, ignition source, and weather) are complex, contradictory, and appear counterintuitive. For instance, lodgepole stands in which > 50 % of susceptible trees were killed by beetles in the 5 to 15 years preceding the 1988 Yellowstone fires had a higher incidence of crown fire than stands in which mortality was not as high (Turner et al. 1999). In contrast, stands with low to moderate beetle mortality (< 50% tree kill) had a lower incidence of high-severity crown fires. However, it is unclear whether these differences in fire behavior were primarily the result of the outbreak or of pre-outbreak stand structure (Simard et al. 2008), because beetle mortality occurred preferentially in older stands that were, in turn, inherently more likely to burn at high severity than younger stands because of differences in fuel structures even in the absence of beetle activity (Renkin and Despain 1992).

Other studies have found that beetle-kill may have decreased the hazard of high-severity crown fire by reducing the continuity of the canopy. For example, beetle-killed lodgepole pine stands, characterized by lower stand density, were affected by

Table 1. Forest types and relations between fire and insects in the Rocky Mountains.

Forest Type	Location	Insect-fire link	Citation
Lodgepole	Yellowstone	Beetle killed stands had significantly lower fire severity.	Omi 1997
Lodgepole	Yellowstone	Stands with higher mortally from bark beetles had higher incidence of crown fires. Stands with low to moderate beetle mortality had a lower incidence of crown fires.	Turner et al. 1999
Lodgepole	Yellowstone	Stands affected by outbreak in 1972-1975 were associated with a slightly higher probability of fire. Stands affected by outbreak in 1980-1983 were not more likely to burn.	Lynch et al. 2006
Lodgepole	Yellowstone	The probability of active crown fire in stands recently affected by beetles was significantly lower than in stands not affected by beetles.	Simard et al. 2011
Lodgepole and spruce	Colorado	Bark beetle outbreak did not affect the extent or severity of fire.	Kulakowski and Veblen 2007
Lodgepole and spruce	Intermountain west	Modeling study predicted a reduced risk of active crown fire 5 to 60 years after outbreaks.	Jenkins et al. 2008
Spruce	Colorado	Bark beetles caused no increase in the numbers of fires.	Bebi et al. 2003
Spruce	Colorado	Beetle-affected stands were not more susceptible to a low-severity fire.	Kulakowski et al. 2003
Spruce	Colorado	Previous bark beetle outbreaks had only a minor influence on fire severity.	Bigler et al. 2005
Spruce	Central Rocky Mountains	Modeling studies predicted reductions in the probability of active crown fire after bark beetle outbreaks.	DeRose and Long 2009

significantly lower fire severity compared to adjacent burned areas that had not been affected by beetles in the 3400-hectare Robinson Fire that burned in Yellowstone National Park in 1994 (Omi 1997). Lynch et al. (2006) also examined the influence of previous beetle activity on the 1988 Yellowstone fires by testing whether beetle-affected stands were more likely to have burned than those stands not affected by beetles. Stands affected by outbreak from 1972 to 1975 had a higher probability of burning, but the increase was relatively minor (about 11% greater compared to areas unaffected by beetles). In contrast, stands that were affected by outbreak from 1980 to 1983 were not more likely to burn in comparison to unaffected stands (Lynch et al. 2006).

It has been hypothesized that the risk of fire may increase only during and immediately after outbreaks of bark beetles when the dry red needles are still on the trees (Romme et al. 2006). However, Kulakowski and Veblen (2007) found that ongoing outbreaks of mountain pine beetle (and spruce beetle) did not affect the extent or severity of fire and suggested that changes in fuels brought about by outbreaks may be overridden by climatic conditions. Simard et al. (2011) examined fuel conditions for 35 years following outbreaks of mountain pine beetle in Yellowstone National Park. They documented reduced canopy moisture content after an outbreak, which was coupled with reduced canopy bulk density. In simulation models of fire behavior, under intermediate wind conditions (40 to 60 kilometers per hour), the probability of active crown fire in stands recently affected by beetles was significantly lower than in stands not affected by beetles (Simard et al. 2011). In addition, if winds were below 40 kph or above 60 kph, stand structure had little effect on fire behavior. Thus, although the canopy was drier immediately after an outbreak, no increase in fire risk was observed, likely because of the more important effect of reductions in canopy bulk density. Other modeling studies also have predicted a reduced risk of active crown fire 5 to 60 years after outbreaks, due to decreased canopy bulk density (Jenkins et al. 2008). In sum, outbreaks of bark beetles in lodgepole pine may have little or no effect on subsequent fires and may in some cases actually reduce the risk of fire.

Fire and Spruce Beetle in Subalpine Spruce-Fir Forests

There is increasing evidence that spruce beetle outbreaks have little or no affect on the occurrence or severity of fires in spruce-fir forests (Simard et al. 2008). It is well established that in this forest type, extensive fires are highly dependent on infrequent, severe droughts (e.g., Schoennagel et al. 2007). Under such extreme drought conditions, increased dead fuels from bark beetle outbreaks appear to play only a minor role, if any, in increasing fire risk. For instance, after a 1940s spruce beetle outbreak that resulted in dead-standing trees over thousands of hectares of subalpine forests in the White River National Forest of western Colorado, there was no increase in the numbers of fires compared to unaffected subalpine forests (Bebi et al. 2003). Beetle-affected stands were not more susceptible to a low-severity fire that spread through adjacent forests several years after the outbreak subsided (Kulakowski et al. 2003). During the extreme drought of 2002, large fires affected extensive areas of Colorado, including some spruce-fir stands that were previously affected by the 1940s outbreak of spruce beetle. Despite the expectation that these outbreaks would have led to an increased risk of severe fires, they had only a minor influence on fire severity (Bigler et al. 2005). Likewise, ongoing outbreaks of spruce beetle (and mountain pine beetle) had no detectable effect on the extent or severity of fires in 2002 (Kulakowski and Veblen 2007). These empirical findings are consistent with modeling studies that predict reductions in the probability of active crown fire for one to two decades after high-severity bark beetle outbreaks in pure stands of Engelmann spruce (*Picea engelmannii)* (DeRose and Long 2009). Other modeling studies have likewise predicted a reduced risk of active crown fire 5 to 60 years after outbreaks, due to decreased canopy bulk density (Jenkins et al. 2008).

The emerging view is that for lodgepole pine and spruce-fir forests: (1) the effect of bark beetle outbreaks on fuels is complex; and (2) weather and climate are more important in influencing fire risk and behavior the effects of insect outbreaks. When evaluating the influence of bark beetle outbreaks, it is important to recognize that outbreaks not only reduce foliar moisture content and increase the volume of dead wood, which can increase fire hazard, but that outbreaks also reduce canopy density, which can decrease fire risk (Simard et al. 2011). Therefore, when assessing the risk of wildfires following outbreaks, it is essential to recognize the relative importance of weather and climate to overall fire risk.

EFFICACY OF BARK BEETLE CONTAINMENT MEASURES

Prior to Outbreaks

The effectiveness of thinning to reduce forest susceptibility to bark beetles is believed to be related to tree vigor (Fettig et al. 2007); which may increase as moisture stress is decreased, and which in turn may make trees less susceptible to insect infestation. The premise is that if the trees are healthy and vigorous, they may be able to "pitch out" the attacking beetles, essentially flooding the entrance site with resin that can push out or drown the beetle (Figure 1).

Some studies have suggested that competition for light and water may reduce the vigor of surviving trees and increase susceptibility to bark beetle attacks (Fettig et al. 2007) and that thinning may, therefore, improve outbreak resistance. For instance, low-vigor ponderosa pine (*Pinus ponderosa)* in central Oregon was more often attacked by beetles than high-vigor trees during early stages of outbreaks (Larsson et al. 1983). Similarly, beetle activity has been associated with high tree densities in ponderosa pine and Douglas-fir *(Pseudotsuga* sp.) stands (Negrón et al. 2001; Negrón and Popp 2004). Ponderosa pine study plots in Colorado's Front Range infested by mountain pine beetle had significantly higher tree basal area and density (Negrón and Popp 2004). Douglas-fir beetles (*D. pseudotsugae*) more often attacked stands

Figure 1. Mountain pine beetle being pitched out. Photo taken by Whitney Cranshaw, Colorado State University, Bugwood.org.

containing a high percentage of basal area represented by high densities of Douglas-fir and slow growth during the five years prior to attack in Colorado's Front Range (Negrón et al. 2001).

Studies that have looked directly at thinning and its effects on tree vigor in Western forests have shown mixed results. While some studies have found that thinning reduces stand susceptibility in some circumstances (Fettig et al. 2007), other research has found bark beetles do not preferentially infest trees with declining growth. For example, Sánchez-Martínez and Wagner (2002) found that ponderosa pine forests of northern Arizona growing in dense stands were not more likely to be colonized by bark beetles.
Under some circumstances, thinning may alleviate tree stress at the stand level but is unlikely to be effective at mitigating susceptibility against extensive or severe outbreaks (Safranyik and Carroll 2006). Preisler and Mitchell (1993) found that thinned plots of lodgepole pine in Oregon were initially unattractive to mountain pine beetles; but when large numbers of attacks occurred, colonization rates were similar to those in unthinned plots. Similarly, Amman et al. (1988) studied the effects of spacing and diameter of trees and concluded that tree mortality was reduced as basal area was lowered. However, if the stand was in the path of an ongoing mountain pine beetle epidemic, spacing and density of trees had little effect (Amman et al. 1988).

While thinning has the potential to reduce tree stress, which can reduce susceptibility to insect attack, it also has the potential to bring about other conditions that can increase susceptibility. For example, thinning may injure surviving trees and their roots, which can provide entry points for pathogens and ultimately reduce tree resistance to other organisms (Hagle and Schmitz 1993; Paine and Baker 1993; Goyer et al. 1998). Although thinning can be effective in maintaining adequate growing space and resources, there is accumulating evidence to suggest that tree injury, soil compaction, and temporary stress due to changed environmental conditions caused by thinning may increase susceptibility of trees to bark beetles and pathogens (Hagle and Schmitz 1993).

From an adaptive management standpoint, it is most prudent to implement thinning in appropriate settings (e.g., already degraded areas in need of restoration) with sufficient controls that would lead to an improved understanding of the efficacy of these approaches, particularly under a range of climatic conditions. It is also important to consider how such strategies may alter normal stand structure. For example, thinning in Engelmann spruce forests is likely to create novel conditions that would be atypical for these ecosystems due to their naturally high tree densities (Daubenmire 1943). Further, thinning forest stands before epidemics is not likely to prevent major outbreaks, due to the inherent difficulties of manipulating stand structure over large enough areas and the overriding influence of climatic stress in driving outbreaks.

During Outbreaks

There is general agreement that silvicultural treatments cannot effectively stop outbreaks once a large-scale insect infestation has started. Citing multiple sources, Hughes and Drever (2001) found that most control efforts have had little effect on the final size of outbreaks. In another review, Romme et al. (2006) point out that once an extensive outbreak has started, timber management is unlikely to stop it. Control of such outbreaks is theoretically possible, but it would require treatment of almost all of the infected trees (Hughes and Drever 2001). Amman and Logan (1998) point to failed attempts to use direct control measures, such as pesticides and logging, after an infestation starts. They suggest that by the early 1970s, it was apparent that attempts to control the extensive mountain pine beetle outbreaks that were occurring in the northern Rockies, by directly killing the beetles, were not working.

If a bark beetle infestation is relatively restricted and concentrated in a limited area, it may be feasible to reduce the impact of that outbreak by removing infested trees from a forest stand, or by thinning a stand to reduce stress of trees competing for limited nutrients, sunlight, and moisture. However, specific climatic conditions are believed to be required for beetle populations to reach epidemic levels. As such, a small population of beetles is not sufficient for an outbreak to occur and would not necessarily lead to an outbreak. Conversely, under climatic conditions favorable for an outbreak, bark beetle outbreaks can erupt simultaneously in numerous, dispersed stands across the landscape. Thus, even if a growing population of beetles is successfully removed from one stand or the stand is thinned to increase vigor, under climatic conditions suitable for outbreaks, beetles from other stands are likely to spread over a landscape. Given that climate typically favors beetle populations and stresses trees over very large areas, successfully identifying and treating stands over a large enough region to have a significant impact on the overall infestation is impractical and costly.

Following Outbreaks

Post-disturbance harvest is common practice on forest lands and is designed to remove trees or other biomass in order to produce timber or other resources. This type of resource extraction has the potential to inadvertently lead to heightened insect activity (Nebeker 1989; Hughes and Drever 2001; Romme et al. 2006). In particular, snags and fallen logs contribute to the protection of soils and water quality and provide habitat for numerous cavity- and snag-dependent species (Romme et al. 2006), many of which prey on bark beetles and other economically destructive insects. Therefore, outbreaks could be prolonged because of a reduction in the beetle's natural enemies (Nebeker 1989), including both insects and bird species that feed on mountain pine beetles (Koplin and Baldwin 1970; Shook and Baldwin 1970; Otvos 1979). Furthermore, post-disturbance harvest can damage soil and roots by compacting them (Lindenmayer et al. 2008) leading to greater water stress in trees, which may reduce conifer regeneration by increasing sapling mortality (Donato et al. 2006) and, in general, may cause more damage to forests than that caused by natural disturbance events (DellaSala et al. 2006).

ROAD BUILDING FOR BARK BEETLE CONTROL

A broad scale program to treat forests that have been affected by bark beetle will require an extensive road system, which will likely have significant impacts to forest and aquatic ecosystems.

In general, the major physical results of roads on the terrestrial environment are increases in forest fragmentation and disruption of the movement of organisms and flow of ecological processes across the landscape (Lindenmayer and Fisher 2006). Aquatic systems have been impacted through the disruption of natural infiltration of water into the soil and increased runoff to streams (Forman and Alexander 1998). These effects have been particularly pronounced in mountainous regions, especially on high gradient streams and headwaters (Ziegler et al. 2001). Increased sediment input to streams can result in changes to channel morphology and channel substrate, as well as the creation of shallow pools (Beschta 1978). These changes to stream structure, an indirect effect of road construction, often adversely affect native fish habitat. Thus, any road network constructed to thin or harvest insect-infested stands will have to be carefully engineered to prevent increased sedimentation rates or alteration of hill slope processes (Beschta 1978). While proper engineering can help mitigate some negative effects, it does not mitigate the overall impact of roads on hydrologic processes, water flow, and fragmentation of wildlife habitat.

CONCLUSIONS AND RECOMMENDATIONS

Climate change and other factors are leading to unprecedented changes in western forest ecosystems (Logan et al. 2003; Carroll et al. 2004; Breshears et al. 2005; Bentz et al. 2009). One consequence of recent and predicted climate change is increased bark beetle activity leading to tree mortality over large areas (Logan and Powell 2001; Williams and Liebhold 2002; Carroll et al. 2004). Such ongoing outbreaks have led to widespread public concern about increased fire risk; however, outbreaks of mountain pine beetle and spruce beetle do not appear to substantially increase the risk of subsequent fire under most conditions. Instead, fire risk in spruce-fir and lodgepole pine is strongly tied to warm and dry conditions, such as those of recent decades. Insect containment measures have yielded mixed results and may pose significant risks to forested ecosystems. We recommend that priority be given to removing hazardous trees, which were killed by fire or insects and that might fall across roads or in campgrounds in areas of high human use to limit damages and potential loss of life. Moreover, in order to reduce existing and future risks of fire, it would be prudent to concentrate fuel reduction measures in the wildland-urban interface by creating defensible space, as the 40-meter zone around homes and structures has been shown to be critical to a home's ignitability (Cohen 1999). Thus, to be effective at reducing fire hazard to communities, tree-cutting can be directed at removing all flammable material (not just economically valuable timber) in the immediate vicinity of homes and settlements.

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Influence of Pre-Fire Tree Mortality on Fire Severity in Conifer Forests of the San Bernardino Mountains, California

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Abstract: High tree mortality due to drought and insects often is assumed to increase fire severity once ignition occurs. In 2002-2003, coniferous forests in the San Bernardino Mountains, California experienced a significant tree mortality event due to drought and an outbreak of western pine beetles (*Dendroctonus brevicomis*). In October 2003, fire burned approximately 5,860 ha of conifer forest types in many beetle- and drought-affected stands where most pre-fire dead trees had retained needles. We used pre- and post-fire GIS data to examine how fire severity was affected by pre-fire tree mortality, vegetation characteristics, and topography. We found no evidence that pre-fire tree mortality influenced fire severity. These results indicate that widespread removal of dead trees may not effectively reduce higher-severity fire in southern California's conifer forests. We found that sample locations dominated by the largest size class of trees (≥61 cm diameter at breast height (dbh)) burned at lower severities than locations dominated by trees 28-60 cm dbh. This result suggests that harvesting larger-sized trees for fire-severity reduction purposes is likely to be ineffective and possibly counter-productive.

INTRODUCTION

Tree mortality due to drought and insect attacks is common in western coniferous forests [1], but may be increasing in recent years in some areas [2, 3]. Bark beetles (Coleoptera: Scolytidae) are common native insects that kill firs and pines, and are capable of large-scale population increases following disturbances such as droughts [4, 5]. Dense forests are considered relatively more susceptible to insect mortality [6, 7], and recent studies have concentrated on how prescribed fire and thinning affect susceptibility of trees to insects [5, 8, 9]. However, few data are available on the influence of tree mortality on fire behavior.

Stands with high tree mortality due to drought and insects often are presumed to burn at higher severity during fires, increasing the mortality of dominant overstory vegetation in the stand [10, 11]. This assumption is based on expectations of greater dead fine and coarse fuel loads, including canopy fuels, resulting from pre-fire mortality [11]. The hypothesis that insect-caused tree mortality increases fire severity has relied upon two principal assumptions: (1) dead needles remaining on trees could increase the amount and vertical continuity of fine, dry fuels [11, 12]; and (2) tree mortality could open the canopy and intensify seasonal desiccation of understory fuels [12]. However, the few empirical studies testing this hypothesis have not found support for it. A widespread low-severity fire in subalpine forest in the White River National Forest, Colorado did not burn any stands affected by spruce beetle (*Dendroctonus rufipennis*) outbreaks that occurred several decades prior to the fire [13]. Furthermore, a regional analysis of 303 fires in the White River National Forest found that beetle-affected stands did not burn at higher severities than unaffected stands in fires occurring several decades after the outbreak [12].

The hypothesis that stands with recent high tree mortality due to drought or insects have an elevated probability of burning at higher severity when a fire occurs has never been empirically tested. We examined whether fire severity in two large fires that occurred in the midst of a tree mortality event was influenced by the number of trees killed by drought and insects. Specifically, we investigated whether pre-fire tree mortality increased fire severity in stands after ignition occurred. We did not examine the probability of fire igniting in a stand over broad spatial and temporal scales [e.g., 12, 14].

Beginning in rainfall year 1998-1999, southern California entered a period of major drought and higher temperatures. In 2000, the San Bernardino National Forest began to document unusually high mortality of incense-cedars (*Calocedrus decurrens*), and in 2001 slightly increased mortality was witnessed in ponderosa (*Pinus ponderosa*), Coulter (*P. coulteri*) and Jeffrey (*P. jeffreyi*) pines (L. Merrill, USDA Forest Service, unpublished data 2003).[1] In 2002, an outbreak of western pine beetles (*D. brevicomis*)

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resulted in what the USDA Forest Service identified as 'above background' mortality levels of ponderosa and Coulter pines, and many other conifer tree species were dying from drought alone (L. Merrill, USDA Forest Service, unpublished data 2003). In the first half of 2003, both western pine beetles and mountain pine beetles (*D. ponderosae*) were actively colonizing and killing thousands of conifer trees. By April 2003, the San Bernardino National Forest had mapped approximately 70,000 ha with elevated levels of conifer mortality (L. Merrill, USDA Forest Service, unpublished data 2003).

In late October 2003, one year after the beginning of the beetle population outbreak, two large human-ignited fires merged together in the San Bernardino Mountains and burned 5,863 ha of conifer and conifer-hardwood forest types, including stands with high levels of tree mortality due to drought and insects (Fig. **1**). The Old and Grand-Prix fires were driven by hot Santa Ana winds which typically sweep through southern California during the fall [15]. No widespread harvest of the beetle- and drought-killed trees had occurred at the time of the fires.

MATERIALS AND METHODOLOGY

We selected the San Bernardino Mountains study area because of the existence of Geographic Information System (GIS) layers depicting structural characteristics of vegetation, topography, pre-fire tree mortality immediately prior to the fires, and fire severity, allowing us to investigate the influence of numbers of recently dead trees on fire severity. We simultaneously investigated the effects of topography (slope and aspect), tree size, and canopy cover on fire severity in burned stands, because these factors also are known to influence fire behavior [16, 17].

Conifer forests in the San Bernardino Mountains consist of mixed-evergreen forests [18] below 1,500 m, and ponderosa pine, Jeffrey pine, Coulter pine, white fir (*Abies concolor*)–sugar pine (*P. lambertiana*), and bigcone Douglas-fir (*Pseudotsuga macrocarpa*) stands above 1,500 m [19, 20]. Various combinations of white fir, Jeffrey pine, ponderosa pine, Coulter pine, sugar pine, incense-cedar, and black oak *(Quercus kelloggii*) occur at higher elevations, and canyon live oak (*Q. chrysolepis*) and bigcone Douglas-fir

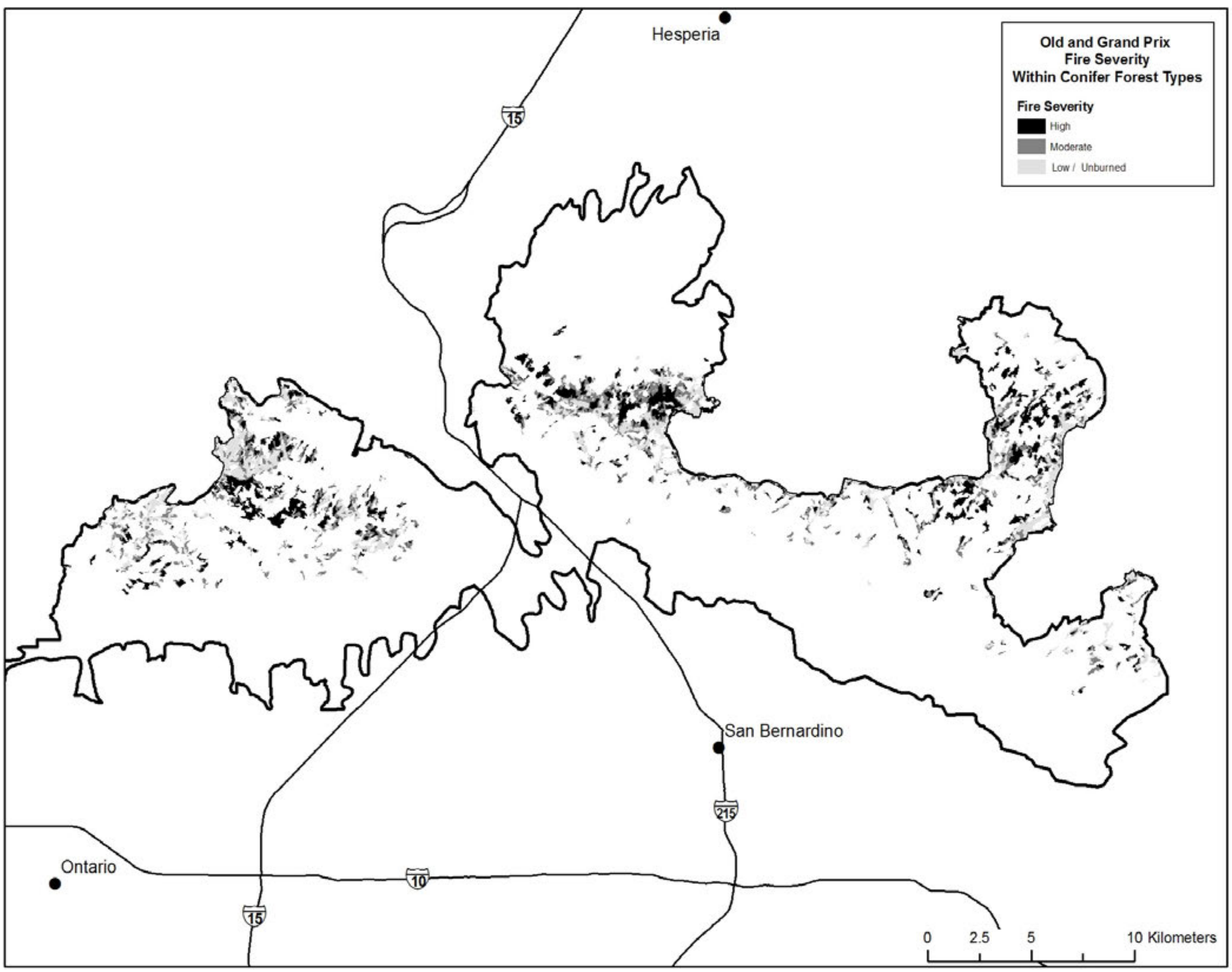

Fig. (1). Perimeters of the 2003 Old and Grand-Prix fires and RdNBR fire severity (low/unburned, moderate, moderate/high) within conifer forest types (Jeffrey Pine, Sierra Mixed Conifer, Montane Hardwood-Conifer). White areas within the fire perimeter are non-conifer vegetation.

dominate at lower elevations [21]. Historic fire return intervals in these forests were variable, with some forest types exhibiting relatively longer fire-free intervals associated with mixed-severity fire effects [20].

We acquired GIS data on vegetation type and structure [22] and pre-fire tree mortality [23] from the USDA Forest Service and GIS data on fire severity [24] and topography [25] from the US Geological Survey. The detailed methodology used by the agencies to create these GIS maps was explained in the metadata for the layers, and is summarized here. Our variables of interest were vegetation type, size of dominant trees, canopy cover, slope, aspect, number of dead trees per ha prior to the fire, and fire severity.

Vegetation type, size class of dominant trees, and canopy cover were derived from a map of existing vegetation from 2002-2003 (EVEG Tiles) [22]. The vegetation layer was generated using a combination of automated systematic procedures, remote-sensing classification, and photo editing and ground surveys to reduce bias while mapping large areas. Minimum mapping size for contrasting vegetation conditions based on cover type, vegetation type, tree cover, and diameter class was 1 ha and pixel size was 30 m.

Cover types were delineated using Landsat Thematic Mapper imagery into the following broad classes: (1) Conifer = >10% conifer cover as dominant type; (2) Mix = >10% tree cover and 20-90% hardwood cover; (3) Hardwood = >10% hardwood cover as dominant type; (4) Shrub = >10% shrub cover as dominant type; (5) Grass = >10% grass cover as dominant type; (6) Barren = <10% cover of any natural vegetation; (7) Agriculture; (8) Urban; and (9) Ice/snow. Attributes including tree cover from above and overstory tree diameter interpreted from aerial photography and satellite imagery were then mapped within the cover type classes and used to develop additional classifications. We used California Wildlife Habitat Relations (WHR) [26] to describe specific vegetation types, canopy cover, and tree size-class. "*WHR vegetation type,*" is derived primarily from CALVEG cover type and relative cover of conifer and hardwood trees for mixed vegetation types. For our study area, the *WHR vegetation types* consisted of Jeffrey Pine, Sierra Mixed Conifer, Montane Hardwood-Conifer, Eastside Pine, and Closed-cone Pine-Cypress. "*WHR density*" is a measure of tree density indexed by percent canopy cover and included: Sparse (10.0-24.9%), Poor (25.0-39.9%), Moderate (40.0-59.9%), and Dense ($\geq$60%). "*WHR size*" identified size classes of overstory trees. *WHR size* included the following three classes: WHR size 3 = dominated by trees 15-27 cm dbh; WHR size 4 = dominated by trees 28-60 cm dbh; WHR size 5 = dominated by trees $\geq$61 cm dbh.

The GIS layer depicting tree mortality was created from annual aerial surveys conducted by the USDA Forest Service. Current-year tree mortality from 2001-2003 was sketch-mapped by an aerial observer who quantified the number of yellow to reddish brown trees. Polygons were categorized by mortality type (drought or insect kill) and number of trees affected per acre (we converted acres to hectares for this study). Generally, areas with <1 tree per acre of mortality were considered to have background levels of mortality and were not usually mapped during the flight. The resulting layer is a vector data set of polygons each associated with a level of tree mortality for that year. Each year's layer was non-cumulative with respect to numbers of dead trees; however, we used only the 2003 GIS map in our analyses because (1) prior to 2003, few polygons showed above-background levels of mortality within the fire perimeter and (2) we were interested only in very recent mortality since these trees were most likely to have retained dead needles to potentially contribute to fire severity. Therefore, the actual number of all dead trees in a given polygon was likely higher than reported herein.

The fire severity GIS data of the 2003 Old and Grand Prix fires were derived from Landsat Thematic Mapper data. Pre-fire and post-fire data were used to create a Relative delta Normalized Burn Ratio (RdNBR) image, which portrays fire severity to vegetation within a fire while accounting for variation in pre-fire live tree cover, as described in Miller and Thode [27]. Because we were interested in ascertaining whether pre-fire tree mortality influenced fire severity, we used a relative rather than absolute index. Absolute dNBR measures how much vegetation was killed by the fire, while RdNBR measures the amount of vegetation killed in relation to the amount of pre-fire vegetation [27]. Miller and Thode [27] found that RdNBR more accurately classified high-severity fire effects than dNBR in heterogeneous landscapes with variable amounts of pre-fire vegetation, such as our study area in the San Bernardino Mountains. Higher RdNBR values are correlated with more severe burning of vegetation. The RdNBR image was classified into 4 classes of fire severity based on cutoff thresholds informed by field data collected on understory, midstory, and overstory vegetation one year post-fire on several fires from 2001 through 2004 using Composite Burn Index (CBI) protocols [27]. We used CBI classifications because they provide information about fire effects on all vegetation strata from the forest floor to the upper canopy, and are a useful and easily understood measurement for managers.

The fire severity map identified 4 classes of fire severity. "*Unchanged*" included areas in which conditions one year after the fire were indistinguishable from pre-fire conditions. "*Low Severity*" represented areas of surface fire with little change in cover and little mortality of the dominant vegetation. "*Moderate Severity*" was between low and high and represented a mixture of effects on the dominant vegetation. "*High Severity*" represented areas where the dominant vegetation had high to complete mortality of canopy foliage due to the fire. We used this classification system to represent the severity of fire in the forest canopy in our analyses. For areas mapped as high severity using RdNBR, we categorized these as "moderate/high severity" because RdNBR measures fire-induced mortality of canopy foliage, rather than tree mortality. The RdNBR high-severity mapping category has a lower threshold of 80% canopy mortality, which equates to 65% tree mortality for trees >20 cm dbh [28]. Basal area mortality would likely be somewhat lower than 65%, since the larger trees that dominate in terms of basal area are less fire-susceptible than the abundant small trees that dominate in terms of tree density [29].

The GIS layers of vegetation type and structure, pre-fire tree mortality, and a Digital Elevation Model [25] were clipped to the Old and Grand Prix fire perimeters. We selected conifer and mixed hardwood-conifer type polygons from within the vegetation layer for analyses. We generated 500 randomly located points throughout the conifer and mixed hardwood-conifer forest types to create a table of sample stand locations. At each sample location we determined the values of the variables: (1) *slope* [%]; (2) *aspect* [degrees]; (3) *mortality* [drought and beetle killed only] expressed as the number of dead trees per ha from year 2003; (4) *WHR vegetation type*; (5) *WHR size*; (6) *WHR density*; and (7) *fire severity*.

We removed 31 locations from our sample due to small sample sizes within specific categories, including: (1) the 5 locations where *WHR size* = 3; (2) 8 locations of various *WHR vegetation types* that had <5 samples in categories; and (3) 18 locations in the Closed-Cone Pine-Cypress type. Final sample size was 469 random points in *WHR types* Jeffrey Pine, Sierra Mixed Conifer, and Montane Hardwood-Conifer (hereafter conifer forest), in *WHR size* classes 4 and 5. *WHR densities* were modified from categorical variables to the mean value of each category (17.5%, 32.5%, 50%, and 80%). Pre-fire tree mortality data was expressed in terms of total number of dead trees per ha in 2003 (immediately prior to the October fires).

We analyzed how fire severity was affected by pre-fire insect and drought mortality along with topography, tree size, and canopy cover variables using two model structures best suited to categorical response variables: binomial and rank-ordered logistic. For the binomial method, we created a generalized linear model (GLM) using a binomial error structure and a logit link function to examine the effects of explanatory variables on the probability that each randomly selected location experienced moderate/high severity fire. The binomial response variable was moderate/high severity burn = 1; and unchanged, low, or moderate severity burn = 0. For the rank-ordered method we performed ordered logistic regression (OLR) to fit an ordered logit model examining how explanatory variables affected the probability that each randomly selected location burned at low, moderate, or moderate/high severity. Our response variable, *fire severity*, was treated as ordinal under the assumption that the levels of *fire severity* have a natural ordering (low to moderate/high), but the distances between adjacent levels are unspecified. All analyses were performed using Stata 8.2 (Stata Corp. 2004, College Station, Texas 77845).

We generated binomial categorical variables for *aspect* (south, east, and west), *WHR type* (Sierra Mixed Conifer and Montane Hardwood-Conifer), and *WHR size* 5, conditioning the model on north-facing slopes of Jeffrey Pine dominated by trees 28-60 cm dbh (*WHR size* 4). *WHR density* was included to control for variation in stand density (canopy cover) within mortality polygons and across the landscape. *Slope*, *aspect*, *WHR size*, and *WHR vegetation type* variables were included because all of these factors can influence fire behavior [16, 17].

We used trend surface analysis to model broadscale spatial pattern in the burn-severity data as a control for spatial autocorrelation. This methodology has two primary aims [30, 31]: (1) to guard against false correlations between fire severity and explanatory variables, as may arise when an unmeasured environmental factor causes a common spatial structure in fire severity and in the measured explanatory variables; and (2) to determine if there is a substantial amount of broadscale spatially structured variation in the fire-severity data that is unexplained by the measured explanatory variables. We fitted a trend surface to fire severity by including variables for x and y spatial coordinates of each sample location, polynomial terms up to the third-degree, and interactions. Prior to analysis, x and y were centered on their respective means to reduce collinearity with higher-order terms [31] and standardized to unit variance. Nonsignificant ($P > 0.05$) trend surface terms were removed by stepwise selection.

RESULTS

Fire severity in the Old and Grand Prix fires was highly variable, as is typical of forest fires, leaving patches of unburned and lightly burned areas intermixed with moderate and moderate/high severity patches (Fig. **1**). Throughout conifer forest, the fires burned 1,882 ha (32%) at moderate/high severity; 2,010 ha (34%) at moderate severity; 1,385 ha (24%) at low severity; and 586 ha (10%) remained unchanged. The distribution of fire severity categories of our sample locations closely matched the distribution of fire severities in conifer forest throughout the study area (32% at moderate/high severity; 34% at moderate severity; 23% at low severity; and 12% remained unchanged). Tree mortality due to drought and beetle infestation prior to the fire ranged from an average of 0 to 21.83 dead trees per ha in each polygon. In smaller patches within a polygon the density of dead trees may have been much higher. Fifty percent of our sample locations had no pre-fire tree mortality above background level. Of the remaining 50% of our sample locations with above-background tree mortality levels, most observations were evenly distributed among four categories: (1) < 2.47; (2) 7.41-12.35; (3) 14.83; and (4) 19.77-22.24 dead trees per ha. The original data were reported in these categories and were expressed in terms of dead trees per acre. We converted acres to hectares to derive our dead tree density values.

The GLM indicated that pre-fire tree mortality due to drought and beetle infestation did not significantly affect the probability that a location within the fire burned at moderate/high severity ($P = 0.88$; Table **1**), while controlling for the effects of topography and vegetation characteristics. Burned locations in Montane Hardwood-Conifer vegetation were significantly more likely ($P = 0.04$) to burn at moderate/high severity than locations in Sierra Mixed Conifer or Jeffrey Pine vegetation. Western aspect decreased the probability of moderate/high severity fire ($P < 0.10$; Table **1**). The pseudo r^2 value was 0.067, indicating that 7% of the variation in probability of high-severity fire was explained by our model.

Similarly, the OLR indicated that pre-fire tree mortality did not increase the probability that a location within the fire area burned at higher severity ($P = 0.53$; Table **2**). Montane Hardwood-Conifer vegetation significantly increased the probability that a location burned at higher severity than Sierra Mixed Conifer or Jeffrey Pine vegetation ($P < 0.001$; Table **2**). Sample locations with western aspect and those dominated by trees $\geq$61 cm dbh were more likely ($P < 0.10$) to burn at lower severities relative to locations with north

Table 1. Table of Coefficients from a Binomial Generalized Linear Model (GLM) Examining Effects of Pre-fire Tree Mortality, Slope, Aspect, Vegetation Type, Tree Size Class, and Canopy Cover Class on Probability of Moderate/High Severity Fire in Conifer Types within the October 2003 Old and Grand Prix Fires in the San Bernardino National Forest, California (*n* = 469).

Variable	Coeff.	SE	z	P>|z|	95% CI	
Insect/drought mortality	0.005	0.035	0.15	0.882	-0.064	0.074
Slope	0.001	0.002	0.35	0.726	-0.002	0.003
East	-0.062	0.263	-0.24	0.812	-0.577	0.452
South	-0.293	0.345	-0.85	0.395	-0.970	0.383
West **	-0.585	0.300	-1.95	0.051	-1.173	0.003
WHR size 5	-0.361	0.228	-1.58	0.114	-0.808	0.087
WHR type MHC *	0.575	0.283	2.03	0.043	0.019	1.130
WHR type SMC	-0.637	0.698	-0.91	0.362	-2.004	0.731
WHR density	0.011	0.007	1.55	0.120	-0.003	0.024
Y-coordinate *	1.3E-04	4.2E-05	3.17	0.002	5.2E-05	2.2E-04
X-coordinate^2 *	2.5E-09	7.5E-10	3.24	0.001	9.7E-10	3.9E-09
Intercept	-2.247	0.701	-3.21	0.001	-3.620	-0.874

* = P < 0.05.
** = 0.05 < P < 0.10.

Table 2. Table of Coefficients from Ordered Logistic Regression (OLR) Examining Effects of Pre-fire Tree Mortality, Slope, Aspect, Vegetation Type, Tree Size Class, and Canopy Cover Class on Fire Severity (Low, Moderate, Moderate/High) in Conifer Types within the October 2003 Old and Grand Prix Fires in the San Bernardino National Forest, California (*n* = 415).

Variable	Coeff.	SE	z	P>|z|	95% CI	
Insect/drought mortality	0.020	0.032	0.63	0.532	-0.043	0.083
Slope	0.000	0.001	-0.16	0.874	-0.003	0.002
East	-0.191	0.240	-0.80	0.425	-0.662	0.279
South	-0.051	0.311	-0.16	0.870	-0.659	0.558
West **	-0.455	0.254	-1.79	0.073	-0.953	0.043
WHR size 5 **	-0.343	0.206	-1.67	0.095	-0.746	0.060
WHR type MHC *	0.915	0.244	3.75	<0.001	0.437	1.394
WHR type SMC	-0.302	0.577	-0.52	0.601	-1.433	0.829
WHR density	0.001	0.006	0.14	0.891	-0.010	0.012
XY *	5.8E-09	1.9E-09	3.03	0.002	2.1E-09	9.6E-09
Cutpoint 1	-0.634	0.521				
Cutpoint 2	1.109	0.524				

* = P < 0.05.
** = 0.05 < P < 0.10.

aspect or those dominated by trees 28-60 cm dbh (Table **2**). The pseudo r^2 value of 0.04 suggested that 4% of the variation in fire severity among locations was explained by our model.

DISCUSSION

We found that stands with recent high pre-fire tree mortality due to drought and insects did not burn at higher severity in coniferous forests of the San Bernardino Mountains, southern California, in the two fires we examined. Pollet and Omi [32] reported anecdotally that stands of lodgepole pine (*P. contorta*) that experienced an insect epidemic in the 1940s in Yellowstone National Park burned at lower severities compared to adjacent burned areas in the 1994 Robinson Fire. A widespread low-severity fire in subalpine forests in the White River National Forest, Colorado did not burn any beetle-affected stands [13]. Further, Bebi *et al.* [12] found that stands of Engelmann spruce (*Picea engelmannii*) and subalpine fir (*A. lasiocarpa*) in the White River National Forest influenced by a spruce beetle outbreak in the 1940s did not show higher susceptibility to 303 subsequent forest fires that burned after 1950. Our study area differed from these previous sites because most of the trees killed by insects and drought just prior to the fires in the San Bernardino Mountains were still standing and had retained needles. Despite differences in sites and forest types, previous studies and our results provide compelling evidence that when fire does occur, stands with considerable tree mortality due to drought and insects will not burn at higher severity than stands without significant tree mortality, either in the short or long term.

While pre-fire tree mortality had no effect on fire severity in burned stands, we found that sample locations dominated by the largest size class of trees ($\geq$61 cm dbh) burned at lower severities than locations dominated by trees 28-60 cm dbh (Table **2**). This result suggests that harvesting larger-sized trees for fire-severity reduction purposes is likely to be ineffective, and possibly counter-productive. These findings corroborate other recently published studies indicating that retention of the largest trees is likely to maintain normative fire behavior [33-35]. The smallest tree-size classes were not included in our analyses due to low sample sizes, so we could not determine the effects of still smaller tree-size classes on fire severity. An additional limitation on the potential effectiveness of fuel treatments to reduce fire severity in stands with high pre-fire mortality is the low likelihood that such stands will be affected by fire [14].

Weather conditions can supersede the influence of stand structure and fuels on fire behavior in mixed-severity fire regimes [36], which probably accounts for the low r^2 values of our models. We included topographical and stand structure variables, but we had no variables for wind speed, air temperature, and fuel and air moisture levels, for example. Odion and Hanson [36] analyzed the spatial patterns of fire severity for conifer forests in the three largest fires in the Sierra Nevada Mountains, California since 2000, and found that high-severity fire ranged from 10.9 to 28.9% of total area burned. Overall, we documented that 32% of conifer and mixed hardwood-conifer types burned at moderate/high severity in the 2003 Old and Grand Prix fires. The Old and Grand Prix fires may have had relatively high proportions of moderate/high severity due to the extreme fire weather resulting from Santa Ana winds, the lack of large-tree components due to past harvest, or some combination thereof.

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Chapter 4

c0020 ~~Mammal Habitat Selection~~

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s0010 4.1 INTRODUCTION

p0010 Mammals are ecologically and economically important members of the landscapes in which they live. Large herbivores like deer (*Odocoileus* spp.) and elk (*Cervus elaphus*), and predators like bears (*Ursus* spp.) and wolves (*Canis lupus*), are highly conspicuous and well-known "flagship" mammal species, whereas rodents, bats, and mustelids are cryptic but no less important in their ecosystems. Many species have developed broad ecological tolerance from exposure to environmental variation and natural disturbances over long time periods (Lawler, 2003). However, widespread hunting and excessive habitat fragmentation of landscapes by modern-day humans are qualitatively and quantitatively different from the natural disturbances to which these mammals were exposed in the past (Spies and Turner, 1999), and they have resulted in contraction of historical ranges and population declines. In North America alone notable population declines include elk, grizzly bears (*Ursus arctos*), gray wolves, Canadian lynx (*Lynx canadensis*), bighorn sheep (*Ovis canadensis*), beaver (*Castor canadensis*), the larger species of forest mustelids, and several herteromyid rodents.

p0015 Mixed- and high-severity wildfire is a natural disturbance in many vegetation systems of North America, the Mediterranean, Australia, and Africa (see Chapters 1, 2, 8~~, and 9~~). The effects of severe fire on organisms vary spatially and temporally, by habitat type, and by species, but how do these disturbances specifically impact mammals? As with any natural disturbance, some species are adversely affected ("fire-averse" species), others benefit ("fire-loving" or pyrophilous species), and still others have a neutral response to fires.

p0020 The dynamics of populations and communities of mammals after severe fire depend on factors such as the degree of ecological change, time since fire, size and spatial configuration of burned and unburned areas, extent of edge, isolation of habitat patches by urbanization and roads, and invasion of nonnative species (Smith, 2000; Shaffer and Laudenslayer, 2006; Arthur et al., 2012; Diffendorfer et al., 2012; Fontaine and Kennedy, 2012). In theory, mammalian populations

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should be stable and resilient across the landscape wherever prefire populations and critical habitats are not greatly reduced and/or fragmented by human activities, and where severe fires occur in a spatial and temporal pattern in which a species has evolved (Shaffer and Laudenslayer, 2006). The capability of fire-loving individuals to utilize severely burned areas or for fire-averse populations to recover after fire, however, can be compromised when prefire habitat fragmentation has resulted in small and/or isolated populations and where postfire management actions, such as logging of burned trees and use of herbicides and pesticides, adversely influence population dynamics and habitat use.

p0025 In this chapter I provide an overview of published studies about mammalian responses to mixed- and high-severity fires in forests, woodlands, shrublands, deserts, and grasslands around the world. I describe research on the effects of severe fire on four major taxonomic groups of mammals: bats, small mammals, carnivores, and ungulates. I emphasized peer-reviewed publications, particularly those with robust methodologies and analyses, because these are the accepted standard in science. I also used non-peer-reviewed data when necessary to supplement information from the peer-reviewed literature. I do not cite every published study but instead provide a balanced overview of severe-fire effects on these taxa. I encourage readers to investigate further the scientific literature on habitat use and population responses of mammals to severe fire because the state of the science is constantly evolving.

p0030 Few studies have documented direct effects of fire on wildlife (e.g., mortality from asphyxiation, heat stress, burning, or physiological stress; however, see Singer et al., 1989), wildlife biologists generally agree that direct mortality from fire is typically low and does not significantly influence populations (Smith, 2000). Thus I focus here on the indirect responses ~~of~~ severe fire, such as postfire occupancy, abundance or density, survival, reproduction, and use of habitat (e.g., breeding, resting, foraging). I define "significant effects" according to the generally accepted scientific definition of statistical significance (i.e., at the 0.05 probability level). I exclude studies that simulated or modeled fires, choosing instead to focus on observations of real systems responding to severe wildfire.

p0035 Appendix 4.1 is a summary of published studies by mammalian taxa and directional response to severe wildfire (negative, neutral, positive) over three time periods after fire. I present results from studies comparing unburned habitats with high-severity burn from wildfire (rather than prescribed fire) and without the confounding effect of postfire logging. For small mammals, only species with enough detections to determine directional response were included in the appendix.

s0015 4.2 BATS

p0040 Bats perform unique and critical ecosystem services by consuming vast quantities of insects, thereby transferring nutrients, most notably nitrogen, from foraging to roosting areas via their feces (Gruver and Keinath, 2006). Bats are predators of adult mosquitoes and thus play an important role in controlling

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mosquito populations and reducing disease transmission (Reiskind and [illegible]l, 2009). Further, nectar-feeding bats are primary pollinators of many ~~plants~~ species throughout the world (Molina-Freaner and Eguiarte, 2003).

p0045 The current literature on the effects of fire on bats strongly suggests that mixed- and high-severity fires are explicitly beneficial. In a study comparing the relative activity of six phonic groups of mostly rare and sensitive bat species across unburned and moderate- and high-severity burned mixed-conifer stands 1 year after fire in the southern Sierra Nevada, bat activity in burned areas was equivalent to or greater than activity in unburned areas for all groups based on echolocation frequencies (Buchalski et al., 2013). Indeed, two of the phonic groups showed a positive response to high-severity fire but a neutral response to moderate-severity fire, demonstrating the importance of severity-specific responses. The positive response to mixed- and high-severity fire by bats mirrors findings for a range of bird species (see Chapter 3) and provides evidence of a long evolutionary relationship between bats and severe fire.

p0050 Several studies have documented how roosting bats use basal hollows of large trees (Gellman and Zielinski, 1996; Zielinski and Gellman, 1999; Fellers and Pierson, 2002; Mazurek, 2004). (Figure 4.1) Basal hollows are cavities formed by repeated fire scarring and healing (Zielinski and Gellman, 1999). For bats that roost in basal hollows of large trees, high-severity fire may destroy or reduce the longevity of existing roost trees, but it also creates new roost trees. In addition, fire creates gaps in the canopy that increase the amount of solar radiation reaching the subcanopy where bats roost. These warmer temperatures may facilitate thermoregulation (Brigham et al., 1997; Boyles and Aubrey, 2006) and are particularly beneficial to [illegible]ductive females because increased temperatures are associated with increased fetal and neonate growth (Brigham et al., 1997; Johnson et al., 2009). Finally, high-severity fire c[illegible] a "pulse" of insect prey (e.g., aquatic insects (Malison and Baxter, 2010) ~~and~~ moths, beetles, and flies (Schwab, 2006)), as well as new natural edge habitat that provides novel foraging opportunities (Fellers and Pierson, 2002).

p0055 Comparisons of food web components between unburned watersheds [illegible] areas of low- and high-severity fires 5 years after fire in Douglas fir (*Pseudotsuga menziesii*) and ponderosa pine (*Pinus ponderosa*) forests in central Idaho showed high insect biomass in heavily burned areas and correspondingly high bat detection rates (Malison and Baxter, 2010). Notably, high-severity sites had almost five times more biomass of zoobenthic insects and more than three times the number of emerging adult aquatic insects than low-severity sites (and twice as many as unburned areas). The frequency of bat echolocation calls also was significantly greater at high-severity sites than at unburned sites, because aquatic insects emerging from streams into the terrestrial envir[illegible]t are an important food source for bats. In a review of the responses of [illegible]n benthic macroinvertebrates to fire, Minshall (2003) concluded that "[r]esults for macroinvertebrates generally support the belief that fire and similar natural disturbance events are not detrimental to the sustained

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f0010 **FIGURE 4.1** Basal hollow in large trees are created by periodic fire scarring and healing, creating important roost sites for bats. A Townsend's big-eared bat (*Corynorhinus townsendii*) roost tree in a coast redwood (*Sequoia sempervirens*) in Grizzly Creek State Park, northern California. (Photo by M.J. Mazurek (2015)).

maintenance of diverse and productive aquatic ecosystems (i.e., those found in undisturbed forests)" (p. 159). While individual taxa respond differently to the physical changes in stream structure and short-term and long-term postfire changes in vegetation, Minshall noted that streams are inherently unstable and dynamic environments in which disturbance, including high-severity fire, is a regular occurrence, and many species are opportunistic and can shift food resources in response to fire.

p0060 In mid-elevation forests burned at mixed and high severity in western Montana, Schwab (2006) characterized roost sites and sampled potential prey sources for two forest-dwelling, insectivorous bat species, the little brown bat (*Myotis lucifugus*) and the long-eared myotis (*Myotis evotis*). These species roosted in larger-diameter snags (standing dead trees) in high-density stands of fire-killed trees. Proximity to perennial streams also was important in roost site selection for these two species in burned forests. Wildland fire apparently

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created an abundance of roosting sites and insect prey for bats. Although the abundance of Lepidoptera (moths) and Trichoptera (caddis flies) was similar in burned and unburned forests, the abundance of Diptera (flies) and Coleoptera (beetles) was significantly higher in burned forests. Overall, the median capture rate of all insects in the burn was 1.78 times higher than the median capture rate in unburned forests, ~~although~~ there was considerable variability in the composition and abundance of particular species. Eight of the 11 orders of insects were more abundant in burned sites. In addition, beetles, flies, and caddis flies were significantly more abundant in burned than unburned sites in the first year after fire, although they decreased significantly the second year after fire. Thus, retention of burned trees the first year is important for insectivorous bats. In fact, removing burned trees decreased mammalian (and avian) predation on the abundance of insects that occurred 1 year after fire. Snags in unburned forests can be recruited from existing green trees, but in severely burned forests postfire logging eliminates both existing and future snags for nearly a century because few trees are available for snag recruitment until large-diameter trees have regrown (Schwab, 2006).

p0065 As with many bird species, mixed- and high-severity fire in forest ecosystems likely enhances foraging opportunities for bats (Buchalski et al., 2013). Many insect species inhabiting coniferous forests are highly evolved to exploit severely burned forests and are aptly termed "pyrophilous." Certain beetle species in particular are strongly attracted to highly burned forests. Saint-Germain et al. (2004) noted that, "[s]ome insect groups have adapted to recurrent forest fires by evolving sensory organs and life strategies that allow them to exploit these high quality habitats efficiently. Pyrophilous Buprestids of the genera *Oxypteris* and *Merimna* and the Cerambycid *Arhopalus tristis* (F.) have been shown to respond physiologically to smoke and/or heat generated by fire, and use them as signals leading toward the newly created habitat . . . Several other Coleoptera species uncommon in mature forests congregate in exceptionally high densities in burned stands" (p. 583).

p0070 In a study of fire-loving beetle communities in a large fire that burned boreal black spruce (*Picea mariana*) forest in Quebec, Canada, more than half of the 86 taxa captured were restricted to burned stands (Saint-Germain et al., 2004). Moreover, total captures and species richness were higher in burned stands, especially the oldest severely burned forests. Captures were significantly lower the second year after the fire for all burned stands, indicating that the utility of burned forests for these beetles is greatest in the first year following fire.

p0075 Insects utilizing dead trees occur at much lower abundances in low-severity sites, which by definition have far fewer fire-killed trees than high-severity sites. Malison and Baxter (2010) stated that, "[o]ur results suggest that high severity fires do not play the same ecological role as low severity fires and allowing high severity fires to burn (rather than suppressing them) in certain forest types could be important in maintaining ecosystem function" (p. 577). Similarly, in his severely burned study site, Schwab (2006) noted, "26% of all [insect] families captured were restricted to sites within the burn suggesting

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b0010 **BOX 4.1**

o0010 **(1)** Bats preferentially roost and forage in burned forests.
o0015 **(2)** High-severity fire creates a superabundance of native insect prey.
o0020 **(3)** Bats select denser stands of fire-killed trees for roosting in burned forests and Au7 forage significantly more in forests burned by high-severity fire than in unburned and low-severity fire-affected forests.
o0025 **(4)** Large burned trees for roosting have significant positive benefits for bats.
o0030 **(5)** Postfire logging removes roost trees, reduces the abundance of prey, and reduces habitat suitability for bats.

a unique environment created only after fire." Thus, ecological changes caused by mixed- and high-severity fires cannot be mimicked by low-severity prescribed burns (also see Chapter 13 for similar discussion) (Box 4.1).

s0020 4.3 SMALL MAMMALS

p0110 Small mammals are critically important to ecosystems because they can influence vegetation structure and composition by dispersing seeds and ectomycorrhizal fungi and by aerating soils (Maser et al., 1978). They also provide an essential prey base for carnivores, and the distribution of small mammals can affect the use of space and the habitat selection of their predators (Carey et al., 1992; Ward et al., 1998). Small mammals have comparatively small home ranges and therefore are quite sensitive to habitat change, making them good biological indicators (Haim and Izhaki, 1994). Small mammal assemblages include rodents and insectivores of the families Soricidae (shrews), Talpidae (moles), Aplodontidae (mountain beavers), Sciuridae (squirrels, chipmunks, and marmots), Geomyidae (gophers), Heteromyidae (pocket mice and kangaroo rats), superfamily Muroidea (voles, mice, and woodrats), and Dipodidae (jumping mice). Larger-bodied small mammals include rodents in the Castoridae (beaver) and Erethizontidae (porcupine) families, as well as lagomorphs (pika, hares, and rabbits), and Australian and American marsupials (Marsupialia).

p0115 The occupation of severely burned areas by small mammals is related to regrowth of the vegetation structure with which various species are associated (Torre and Díaz, 2004; Lee and Tietje, 2005; Vamstad and Rotenberry, 2010; Diffendorfer et al., 2012; Kelly et al., 2012; Borchert and Borchert, 2013), as well as with seed and insect production and availability (Coppeto et al., 2006), and cavities created by woodpeckers in snags (Tarbill, 2010). I discuss fire effects on small mammals according to habitat type but give special attention to the deer mouse (*Peromyscus maniculatus*)—an exceptionally "fire-loving" species—in its own section. (Figure 4.2)

f0015 **FIGURE 4.2** Deer mice increase after severe fire in a variety of habita[illegible]er mouse captured two years after forest dominated by Douglas-fir with some lodgepole pine, western larch, and ponderosa pine burned severely in the 2005 Tarkio Fire, Montana. (Photo by Rafal Zwolak (2005)).

s0025 Chaparral and Coastal Sage Scrub

p0120 The chaparral and coastal sage scrub vegetation types in central and southern California support an exceptionally rich diversity of rodents that are well-adapted to a regime of periodic, very-high-intensity fire (see Chapter 7). Many studies have examined small-mammal communities after both prescribed and wildfire in these vegetative types. During intense fires, some individuals among small, less vagile animals may suffer mortality, but many others survive in rock crevices, riparian areas, large downed logs, and underground burrows where temperatures remain cool and the air clean (Chew et al., 1959; Quinn, 1979; Lawrence, 1966; Wirtz, 1995; Smith, 2000). Following fire, small-mammal communities change over time (Diffendorfer et al., 2012; Arthur et al., 2012; Borchert and Borchert, 2013) and space (Schwilk and Keeley, 1998), depending on the vegetation associations of the various species. Species preferring open habitat, including pocket mice (*Chaetodipus* spp.), California voles (*Microtus californicus*), harvest mice (*Reithrodontomys megalotis*), and, especially, kangaroo rats (*Dipodomys* spp.) and deer mice can increase quite dramatically and quickly after severe shrubland fire. Over a period of several years, as shrubs resprout and grow denser and as different food sources become available, small-mammal species preferring a shrubby overstory, including woodrats (*Neotoma* spp.), California mice (*Peromyscus californicus*), brush mice (*Peromyscus boylii*), and cactus mice (*Peromyscus eremicus*), increase in number (Cook, 1959; Wirtz, 1977; Price and Waser, 1984; Brehme et al., 2011; Borchert and Borchert, 2013). Compared with unburned chaparral and grassland, severely burned chaparral had the highest rodent diversity 4 years after a high-intensity wildfire near Mount Laguna in San Diego County (Lillywhite, 1977). Published data are not currently available for lagomorphs in chaparral wildfires, but prescribed burning of chamise (*Adenostoma fasciculatum*) chaparral in northern California increased black-tailed jackrabbit (*Lepus californicus*) densities by 500-1000% the year following fire (Howard, 1995).

s0030 Forests

p0125 Forests offer important habitats for small mammals, especially shrews, mice, tree voles, and squirrels. Mixed- and high-severity fire in forested habitats can have pronounced effects on small-mammal populations by creating or transforming habitat structures such as live and dead trees, shrubs, and coarse woody debris. While some studies have shown that severely burned conifer forests in North America support fewer individuals of some rodents and insectivores immediately after fire compared with adjacent unburned sites (e.g., pinyon mice [*Peromyscus truei*; Borchert et al., 2014] and masked shrews (Sorex cinereus) and southern red-backed voles [*Myodes gapperi*; Zwolak and Forsman, 2007]), numbers begin to rebound several years after fire, often by individuals surviving in unburned refuges within the larger burn perimeter. Northern red-backed voles (*Myodes rutilus*), considered old-growth specialists, began repopulating an intense burn in boreal Alaska from surrounding unburned forest and started reproducing 3 years thereafter (West, 1982).

p0130 Unburned refuges and vegetation changes over time also mediate postfire mammal population dynamics in other forests types, notably *Eucalyptus* forests in Australia. Numbers of bush rats (*Rattus fuscipes*) and agile antechinus (*Antechinus agilis*) were reduced compared with populations in adjacent unburned forests 6 months after severe fire in a mountain ash (*Eucalyptus regnans*) forest, but the population in the burned area was composed of residual animals that had survived the fire rather than animals recolonizing from adjacent forests (Banks et al., 2011). Long-term studies are especially useful because responses relative to time since fire can be quantified. One study examined marsupial population dynamics over a 28-year period following severe wildfire in a southeastern Australia *Eucalyptus* forest reserve (Arthur et al., 2012). Bandicoots (*Isoodon obesulus* and *Perameles nasuta*) increased immediately following the fire, peaked 15 years later, and then declined, associated with an increase and decline of shrub cover. The potoroo (*Potorous tridactylus*) population was similar before and immediately after the fire but began to increase a decade later as tree cover increased. Wombats (*Vombatus ursinus*) exhibited a stable population trend for the first decade after the fire, then slowly declined along with a decline in ground litter cover. Finally, larger macropods (eastern gray kangaroo [*Macropus giganteus*], red-necked wallaby [*Macropus rufogriseus*], and swamp wallaby [*Wallabia biocolor*]) remained at high densities after the fire then declined a decade later as vegetation cover increased.

p0135 Rabbits and hares are associated with shrubs and small conifers that provide cover (Ream, 1981; Howard, 1995). Severe fire temporarily eliminates this habitat structure, but it quickly returns as the vegetation regrows, stimulated by intense fire. Snowshoe hares (*Lepus americanus*) in a boreal forest in Alberta, Canada, moved out of intensely burned sites to surrounding habitat immediately after fire but returned the second summer after the fire when shrubs resprouted, and the postfire population trajectory increased above prefire numbers (Keith and Surrendi, 1971).

p0140 Tree squirrels, including Douglas squirrels (*Tamiasciurus douglasii*) and northern flying squirrels (Glaucomys sabrinus), typically are associated with late-successional coniferous forests in California and the Pacific Northwest in the United States (Carey, 2000); thus they may be adversely affected by intense fire (Zwolak and Forsman, 2007), but few data currently are available to refute or support this hypothesis. Chipmunks and ground squirrels can occupy forests after severe fire where shrubs provide cover and food (Borchert et al., 2014). Townsend's chipmunks (*Neotamias townsendii*) were abundant in early seral forests with dense shrub cover (Campbell and Donato, 2014). Gray-collared chipmunks (*Tamias cinereicollis*) and least chipmunks (*Tamias minimus*) showed no significant response to wildfire in ponderosa pine forests of the southwestern United States (Converse et al., 2006), and the proportion and composition of two chipmunk species, *Tamias amoenus* and *Tamias ruficaudus*, did not differ between severely burned and unburned conifer forest in Montana (Zwolak and Forsman, 2007).

p0145 The increase in the availability, amount, and quality of forage for herbivorous small mammals is an important determinant of the post-severe-fire community. In plots recently burned by large, intense wildfires in a Mediterranean pine-oak woodland in Spain, the abundance of small mammals—mostly mice and shrews—was higher than expected based on vegetation characteristics alone (Torre and Díaz, 2004). The authors attributed small-mammal increases to large quantities of seeds and seedlings in burned sites.

s0035 Deserts

p0150 The role of severe fire and its effects on small mammals in desert grasslands is somewhat controversial (Killgore et al., 2009; Vamstad and Rotenberry, 2010). Most desert systems are not adapted to frequent fire because many species of long-lived perennial desert plants have low recruitment rates and long life spans and lack the ability to resprout. Fire size and frequency in some areas has increased recently because of the invasion of exotic grasses from livestock grazing (Brooks, 2000) and other causes (Burbidge and McKenzie, 1989). In general, most research shows a lack of significant long-term effects of intense fire on the abundance of desert small mammals, although fire can alter community composition. Similar to shrub types in southern California, rodents in the family Heteromyidae increased following a large, intense wildfire in a perennial grassland in southeastern Arizona, whereas species in the family Cricetidae declined immediately after fire, began increasing 4 years after fire, and returned to prefire levels by the sixth year (Bock et al., 2011). Rodent abundance and species richness we[illegible] different between burned and unburned plots after wildfires in Joshua tree woodlands of the Mojave Desert in the American Southwest (Vamstad and Rotenberry, 2010). Merriam's kangaroo rat (*Dipodomys merriami*) dominated the burned sites. As postfire vegetation changed from annuals to sub-shrubs and then to long-lived perennials, however, the composition of rodent species changed and the diversity of rodents increased over time.

p0155 Habitat type is important to fire effects in deserts. In Australia, wildfires in stony desert habitats with sparse grasses have less effect on habitat structure and small mammals than wildfires in sandy desert habitats with denser hummock grass spinifex (*Triodia* spp.) (Pastro et al., 2014). For example, an intense wildfire did not affect the total abundance and species richness of small mammals in the stony (gibber) desert in central Australia, although some species increased and others decreased immediately following fire (Letnic et al., 2013). By contrast, 9 months after intense wildfire in a spinifex grassland in the same region, small-mammal diversity declined compared with before the fire and with prescribed burned areas, although the abundance of animals captured was similar (Pastro et al., 2011). Data were unavailable from wildfires, but hare (*Lepus* spp.) abundance increased by 300% after prescribed burning in East African savanna grasslands (Ogen-Odoi and Dilworth, 1984).

s0040 Deer Mice

p0160 In North America, generalist deer mice are often the most abundant rodent after severe fire in a variety of vegetation types (Borchert et al., 2014). This species responds strongly and positively to high-intensity fire in both shrubland and conifer forests. Deer mice increased significantly over time in moderately and severely burned mixed-conifer forests in the San Bernardino Mountains of southern California over a 5-year period after fire (Borchert et al., 2014). During 2 years subsequent to intense fire, deer mice were invariably the most numerous species in burned study sites in a Douglas-fir-Western larch ~~(*Larix occidentalis* Nutt.)~~ forest in Montana (Zwolak and Forsman, 2008). Converse et al. (2006) attributed increased abundance of deer mice after wildfire in southwestern United States ponderosa pine forests to ~~increased~~ seed production or ~~greater~~ detectability of seeds after fire.

p0165 Dramatic increases in deer mice in severely burned conifer forests were not simply a result of colonization of the burn by animals from surrounding unburned forests. When population densities were low, the vast majority of individually ear-tagged deer mice were found in forest areas after severe fire, and mice appeared regularly in unburned forests only when population densities were high (Zwolak and Forsman, 2008). This finding indicated that severely burned forest was *preferred* deer mouse habitat and that the postfire population increase was intrinsic to the burn; thus the burn itself was a source habitat.

b0015 **BOX 4.2**

o0035 **(1)** After intense wildfire, small-mammal communities are dynamic and associated with vegetation structure at different successional stages.

o0040 **(2)** Intense fire may increase the availability and abundance of seeds and seedlings for herbivorous small mammals.

o0045 **(3)** Unburned refuges and time since fire are important determinants of small-mammal communities following intense fire.

o0050 **(4)** The richness and abundance of small-mammal species is high following intense fire in chaparral and coastal sage scrub communities of southern California. Heteromyid rodents and deer mice often dominate severely burned shrublands, and heteromyids dominate postburn desert grasslands.

o0055 **(5)** Some small-mammal species decrease shortly after intense fire in North American conifer forests, but they can recover to prefire levels within 1 to several years after fire. Deer mice dramatically increase following intense fire.

p0170 Overall, these observations from small-mammal studies in mixed- and severely burned shrublands, forests, and grasslands underscore the important roles played by high-severity fire patches, unburned refuges within a fire area, and the time since fire in population dynamics after severe fire (Box 4.2).

s0045 4.4 CARNIVORES

p0205 Carnivores are critically important "top-down" regulators of ecosystem processes. Elimination of top carnivores unleashes a cascade of adverse effects, including relaxation of predation as a selective force on prey species, spread of disease, explosions of herbivore populations, and subsequent reproductive failure and local extinction of some plants, birds, herptiles, and rodents (Crooks and Soule, 1999; Terborgh et al., 2001). ~~Soulé~~ Large carnivores include ursids (bears), canids (wolves), and larger felids (puma, lions, and jaguars). Medium-sized carnivores, or "mesocarnivores," include canids (coyotes and foxes), Procyonidae (ringtails and raccoons), mustelids (wolverine, marten, fisher, weasels, mink, and badgers), Mephitidae (skunks), and smaller felids (lynx and bobcats). Currently published research on carnivores in mixed and severe wildfires is limited primarily to forested habitats.

s0050 Mesocarnivores and Large Cats

p0210 Many mesocarnivores are associated with forested habitats. Some are habitat generalists, whereas others are forest specialists, riparian associates, or semi-aquatic (Buskirk and Zielinski, 2003). Martens (*Martes* spp.) occur in dense coniferous or deciduous forests across the northern hemisphere. They also regularly use severely burned habitats. Some evidence suggests martens use burns only when postfire trees are not logged. For instance, stone marten (*Martes foina*) were not detected in an intensely burned but extensively postfire-logged Aleppo pine (*Pinus halepensis*) forest in Greece the second and third years after wildfire and logging (Birtsas et al., 2012). These martens were found only in Turkish red pine (*Pinus brutia*) forests burned by wildfire 9 years earlier and not in nearby unburned forests (Soyumert et al., 2010). In coniferous forests of the Alaskan taiga, resident and transient American martens (*Martes americanus*) were captured in a 6-year-old unlogged burn more often than in an island of unburned mature forest surrounded by the burn (Paragi et al., 1996). The authors did not quantify burn severity in their study area but described fire-affected sites as having portions of "severe" burn, and most of the vegetation was in early to mid-seral stages, with dead, fire-scarred trees still standing, consistent with mixed- and high-severity fire. There was no age difference between martens trapped live in the mature forests ~~versus~~ and those trapped in the burn, and marten foraging intensity was greatest in the recently burned area (Paragi et al., 1996). Conversely, martens avoided stands of boreal forests burned from 2 to 20 years prior (Gosse et al., 2005), but the study did not quantify or describe burn severity nor specify whether the burned forest was logged.

p0215 Larger cousins to the marten, fisher (*Martes pennanti* or *Pekania pennanti*) are rare mesocarnivores associated with dense, mature, boreal and mixed conifer-hardwood forests of North America (Powell and Zielinski, 1994). A recent study in the southern Sierra Nevada, however, used scat sampling to detect fisher habitat preferences and demonstrated that the species used denser, mature forests that had experienced moderate- and high-severity fire 10 and 12 years prior and that were not logged after fire (Hanson, 2013) (Figure 4.3). It is likely that both martens and fishers use severely burned forests for foraging rather than denning. These results provide intriguing evidence that even old-forest specialist species are adapted to and can exploit postfire conditions in regions where mixed- and high-severity fire is natural (see Chapter 3, Box 3.1: spotted owls).

p0220 Foxes apparently prefer severely burned forest areas over unburned areas, but they may be less tied to forest structure than martens and fishers and thus less sensitive to postfire logging. Red fox (*Vulpes vulpes*) in Turkish red pine forests were detected more often in the 9-year-old unlogged wildfire area (Soyumert et al., 2010); in postfire-logged Aleppo pine forests in Greece, red foxes were detected most often in severely burned areas, rather than moderately and unburned areas (Birtsas et al., 2012). In 3 of 4 years after intense wildfire in mixed-conifer forests of the San Bernardino Mountains in southern California, gray foxes (*Urocyon cinereoargenteus*) were detected more often in mixed-severity burned over unburned areas, and in two of the years no foxes at all were captured in the unburned area, but coyote (*Canis latrans*) were detected more often in unburned forests (Borchert, 2012). Both gray fox and coyote scats were more numerous in areas burned by intense wildfire than in unburned areas

f0020 **FIGURE 4.3** Representative foraging ~~detection~~ location based upon global positioning system coordinates for a confirmed female Pacific fisher scat detection site several hundred meters into the interior of the largest high-severity fire patch (>5000 ha) in the McNally Fire of 2002, Sequoia National Forest, California. *(Photo by Chad Hanson (2014).)*

2 years after fire in interior chaparral, Madrean evergreen woodland, and ponderosa pine forest in Arizona (Cunningham et al., 2006).

p0225 Striped skunk (*Mephitis mephitis*), ringtail (*Bassariscus astutus*), and raccoon (*Procyon lotor*) were photocaptured only in mixed-conifer forests in southern California burned by high-intensity fire, but each were photographed only once (Borchert, 2012). Bobcat (*Lynx rufus*) were photocaptured in similar numbers in severely burned and unburned forest, but captures in the burned area decreased over time over the 4 years of the study. Finally, mountain lion (*Puma concolor*) were photocaptured more often in severely burned forest, but the overall sample was small (four lion in burned areas, one lion in unburned areas).

s0055 Bears

p0230 Although grizzly bears are flexible in the habitats they use, in British Columbia, Canada, radio-collared grizzly bears strongly selected open forest burned by wildfires 50-70 years earlier at high elevations because these sites supported prolific huckleberries (McLellan and Hovey, 2001). Wildfire also promotes the regeneration of whitebark pine (*Pinus albicaulis*) seeds, another important food source for bears (Kunkel, 2003). Wildfire is not equivalent to logging, as regenerating timber harvests were rarely used by bears in any season (McLellan and Hovey, 2001).

p0235 One study compared the demographics and physiology of black bears (*Ursus americanus*) occupying burns of two ages, 13 and 35 years old, in spruce

(*Picea* spp.) and aspen (*Populus tremuloides*) forests of the Kenai Peninsula of Alaska (Schwartz and Franzmann, 1991). The authors did not specify burn intensity, but they noted that 5% of the older burn was logged after fire for "improvement" of moose (*Alces alces*) habitat, and they pointed out that the more recent fire burned at a greater intensity than the older fire. The density of bears and the percentage of cubs born were similar between the two sites, but all age groups of bears were significantly larger in the recent burn area. Bears in the older burn area consumed more cranberries, whereas the number of moose calves consumed per bear was much larger in the recent burn area, likely explaining the larger size of the bears. Females in the recent burn area also produced litters at a younger age and had a shorter interval between weaning of yearlings than females in the older burn area. Moreover, cub survival was significantly higher in the recent burn area. The vigor of black bear populations was associated with moose abundance, which was significantly enhanced in the 13-year-old fire area.

p0240 Another study compared the demography of a population of black bears in interior chaparral, Madrean evergreen woodland, and ponderosa pine forest, burned by high-intensity wildfire for 3 years after fire using (1) the population in a nearby unburned site for 3 years and (2) results from earlier demographic research on the fire site from 20 years earlier, conducted over a 6-year period (Cunningham and Ballard, 2004). The sex ratio at the 3-year-old burned site was more skewed toward males than in either the unburned reference site or 20 years before the burn. The authors presumed that the fire had reduced the adult female population; however, it is also possible that the female population already had been reduced in the 20 years before the fire occurred, when the population was not monitored. Indeed, an alternative scenario could be that the population of both adult females and males had been declining at Four Peaks before fire, and the fire actually attracted males to the site, who have larger home ranges, thus skewing the sex ratio.

p0245 The above study reported complete reproductive failure in the 3 years after fire at the burned site compared with 36% of cubs surviving to 1 year of age on the unburned control site (Cunningham and Ballard, 2004). More cubs had survived to year 1 at the burned site 20 years before the fire. During the 1970s, however, complete reproductive failure also occurred in the absence of fire during 3 of the 6 years of study. Thus years of complete reproductive failure in that study area were not unusual. Overall, reproductive success was lowest in the burned forest compared with the same site 20 years before fire and an unburned reference site, suggesting the possibility of negative short-term effects of high-intensity fire on black bear reproduction. The mortality of adult bears from hunting, however, was 2.5 times higher in the fire area than in the unburned area (Cunningham et al., 2001), which would be expected to influence cub survival, potentially confounding results. The overall density of black bears in the fire area was higher than prefire densities in the area (Cunningham et al., 2001) (Box ~~4.3 and~~ 4.4).

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b0020 **BOX 4.3 Seed Dispersal by Carnivores**

p0250 Fleshy fruits are an important component of the diet of many carnivores, especially during certain seasons when other resources are scarce. Indeed, the germination of many seeds is facilitated by passage through the carnivore gut because it removes the fruit pericarp and scarifies the seed coat (Herrera, 1989). Carnivores are important dispersers of seeds because they have relatively large home ranges and long gut retention times, thus spreading the seeds far from the parent plant. This may be an important mechanism whereby early seral habitats are seeded. For example, in experimental and field tests in severely burned Aleppo pine forest in Spain, Rost et al. (2012) demonstrated that carnivores, including red fox, stone marten, and European badger (*Meles meles*), were important dispersers of Mediterranean hackberry (*Celtis australis*) seeds into the burned areas. These carnivores traveled long distances into the fire area, dispersing seeds more than 1 km from the parent plant. Moreover, seeds collected from scat (i.e., that had passed through the gut) in the burned study area had a significantly greater germination rate than unscarified seeds, both in the greenhouse and in the field.

b0025 **BOX 4.4**

o0060 **(1)** Grizzly bears use areas burned by intense wildfire because of increases in berry production, although results from studies of the effects of intense fire on black bear demographics are equivocal.

o0065 **(2)** Martens and fisher are mesocarnivores that are dense, mature forest specialists for denning and resting but use severely burned forests that were not logged after fire, most likely for foraging.

o0070 **(3)** Foxes regularly use severely burned forests (regardless of postfire logging for one Mediterranean species), but results from research on coyotes are equivocal.

o0075 **(4)** Carnivores are important dispersers of seeds deep into severely burned forest areas.

s0060 4.5 UNGULATES

p0280 As major herbivorous components of ecosystems, ungulates can act as keystone species with profound effects on vegetation development and productivity in forests, woodlands, and grassland ecosystems throughout the world (Hobbs, 1996; Wisdom et al., 2006). Hobbs (1996) stated, “ungulates are not merely outputs of ecosystems, they may also serve as important regulators of ecosystem processes at several scales of time and space” (p. 695). Ungulates, Hobbs further noted, are “important agents of environmental change, acting to create spatial heterogeneity, accelerate successional processes, and control the switching of

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ecosystems between alternative states." Ungulates regulate nitrogen cycling and influence plant size and morphology (Singer et al., 2003). Because grazing and browsing by ungulates affects the biomass, structure, and type of vegetation available to burn, these animals can actually regulate the dynamics of fire (Hobbs, 1996; Wisdom et al., 2006).

p0285 Episodic disturbance agents such as fire strongly interact with ungulate herbivory over space and time. For example, removal of fine fuels by ungulate grazers may reduce the frequency of ground fires but can increase crown fires by enhancing the development of ladder trees, especially when combined with a relatively long absence of fire (Hobbs, 1996). Further, postfire plant regeneration provides forage species that are highly palatable to ungulates, which attracts ungulates to burned areas, where they influence vegetation regrowth after fire (Canon et al., 1987; Wan et al, 2014). Moose rapidly immigrated to burned areas after a large wildfire in mixed coniferous-deciduous forests of northern Minnesota (Peek, 1974). In fact, fire size can moderate the adverse effects of ungulate herbivory on vegetation recovery. Compared with small fires, large fires "swamp" the effects of ungulate herbivory, for example, by providing sufficient new growth production to offset browsing, and enabling woody species such as aspen to grow to tree height (Biggs et al., 2010). In intensively burned ponderosa pine, mixed-conifer, and spruce-fir forests of northern New Mexico, elk selectively foraged on grasses over shrubs (Biggs et al., 2010). In 25 wildfires throughout five national forests in Utah, larger areas of aspen forest that burned with greater severity had the highest growth potential for aspen regeneration, and these high burn-severity conditions stimulated defensive chemicals in plants that lowered the levels of damage done by ungulate browsing (Wan et al., 2014). Wan et al. noted that "this effect may be particularly strong if amplified over large post-fire landscapes by saturating the browse capacity of the ungulate community." (See Box 4.5).

p0290 Positive effects of high-severity fire on ungulates likely are most pronounced in vegetation types that are most adapted to high-intensity fires, such as aspen forests and shrublands. Mountain or bighorn sheep selected intensely burned shrublands up to 15 years after fire in Montana (DeCesare and Pletscher, 2006) and in southern California mountains (Bleich et al., 2008). Wildfire increased the carrying capacity of southern California ~~mountain~~ sheep (*Ovis canadensis nelsoni*) in the San Gabriel Mountains, dramatically increasing the number of animals in this endangered population (Holl et al., 2004). A large natural fire on the eastern slopes of the Sierra Nevada mountains in California improved the winter range of Sierra bighorn sheep (*Ovis canadensis sierrae*) by increasing green forage availability, shifting diet composition to include more forbs, and possibly decreasing predation risk from mountain lions by increasing visibility (Greene et al., 2012). Overall, large, high-severity fire in bighorn sheep shrubland/forest habitats increases forage quality and availability as well as visual openness, which is critical because several populations are listed as endangered.

p0295 Studies investigating the impact of fire on mule deer (*Odocoileus hemionus*), a common herbivore in the western United States, indicate that populations tend to increase after severe fire, especially in chaparral communities. In a review of the literature on ungulate responses to fire, Smith (2000) reported mule deer density in intensely burned chaparral was more than twice as high as that in mature chaparral in California, and it increased 400% the first year after high-intensity fire in chamise chaparral. Density then decreased each year afterward until preburn levels were reached 5-12 years later. Chamise chaparral burned by a large wildfire in California had more deer use per square mile than unburned chamise chaparral (Bendell, 1974). In northern coastal California, mule deer densities in chaparral burned by high-intensity wildfire the year before were four times greater than in unburned chaparral (Taber and Dasmann, 1957). Because the fire described in this study was relatively small, deer may have moved from one area to another rather than actually increasing the population via higher birth rates. Similarly, ~~black-tailed~~ deer in central coastal California strongly preferred burned habitat, with a 400% increase in the density of deer in prescribe-burned chaparral near oak woodlands, relative to preburn density, by the second growing season (Klinger et al., 1989). Here the increase in the use of burned chaparral was attributed to movements of deer from adjacent oak woodlands rather than an intrinsic increase in population size. Heavy use of prescribe-burned chamise chaparral by mule deer was reported in the San Jacinto Mountains of southern California (Roberts and Tiller, 1985).

p0300 Other studies documented postfire increases in the number of mule deer in conifer forests. Visual observations of 543 mule deer indicated a preference for burned over unburned Douglas fir/ninebark and burned ponderosa pine/bluebunch wheatgrass habitat types during winter and spring in the Selway-Bitterroot Wilderness of Idaho, although the authors did not specifically define the burn severity of sites used by deer (Keay and Peek, 1980). Two other studies that documented increases in mule deer in burned forests hypothesized that postfire logging removes protective cover, a critical habitat element for mule deer. Significantly more deer droppings were located in pinyon-juniper woodlands of Arizona burned by high-intensity fire 13 years earlier than in adjacent unburned areas (McCulloch, 1969). The author surmised that the standing forest of dead trees and fallen trunks provided some cover for deer from predators. Both mule deer and elk used intensely burned lodgepole pine ~~(*Pinus contorta*)~~ forests at two sites in Wyoming significantly more than paired clearcut sites of the same ages (9 and 5 years old) based on fecal pellet counts (Davis, 1977). Davis (1977, p. 787) stated: "[D]eer and elk use was greater in burned areas with standing dead timber than in clearcut areas without it. In the Sierra Madre study area, the burned and clearcut plots both had the same number of plant species present, and they both had standing dead timber. However, the burned plot with much more standing dead timber had more deer and elk use. Fire opened up the canopy allowing light to enter, stimulating growth of forage plants, while the dead trees left standing provided good protective cover" (see Figure 4.4).

f0025 **FIGURE 4.4** Mule deer respond positively to high-severity fire in forests. In this photo, mule deer forage on fresh vegetation growing in the first post-fire year following the Rim fire of 2013 on the Stanislaus National Forest, central Sierra Nevada. *(Photo by Chad Hanson (2014).)*

p0305 Available studies generally report increases in the reproductive rates and body condition of female mule deer in burned habitats. The reproductive rate was 1.32 fawns per doe in the first year after wildfire in northern coastal California, compared with 0.77 fawns per doe in unburned chaparral (Taber and Dasmann, 1957). After 3 years, the reproductive rate of deer at the burned site declined to that of deer in the unburned site. Chamise chaparral burned by a large wildfire produced heavier deer, and does had a higher frequency of ovulation, gave birth to more fawns, and wintered in better condition than does in dense, unburned chamise (Bendell, 1974). Another study, however, documented no difference in fawn-to-doe ratios between burned and unburned chaparral interspersed with oak woodlands in central California (Klinger et al., 1989).

p0310 Foraging studies indicate that mule deer populations in chaparral habitats burned by high-intensity fire often increase as a result of the increased availability of browse. *Ceanothus*—a high-quality food for ungulates (Hobbs, 1996)—is abundant after fire because it reproduces from seed that is scarified by burning (Smith, 2000). Thimbleberry (*Rubus parviflorus*) also generally increases after fire (Smith, 2000). Moreover, fire can increase the palatability of foliage for deer as well as the crude protein content (Smith, 2000). The improved quantity and quality of browse may be related to the fire-caused increase in available nutrients in the soil. As such, deer populations often benefit from the increased food production and nutritional value of their food in recently burned areas. Length and surface enlargement factor of papillae (the surface area within the intestine for absorbing nutrients) of necropsied mule deer were greater in those from high-intensity burned than unburned ponderosa pine habitat in the southern Black Hills of South Dakota (Zimmerman et al., 2006). These

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physiological factors indicate higher forage quality, such as greater concentration of volatile fatty acids. The authors concluded that fire was beneficial at the mucosal level for mule deer: the increase in forage quality from burning caused a rapid change in papillary morphology, allowing the deer to take up more nutrients.

p0315 Lichens in boreal habitats are preferred winter forage for caribou (*Rangifer tarandus*), yet large wildfires that depleted lichens had no effect on home-range size, range fidelity, or the survival and fecundity of woodland caribou (*Rangifer tarandus caribou*) in Alberta, Canada (Dalerum et al., 2007). Caribou avoided foraging in burned compared with unburned areas (Dalerum et al., 2007; Joly et al., 2010), although burn severity was not quantified, and some of the fires occurred 50 years before study. Lichens are significantly reduced by wildfire and take decades to recover to prefire abundance (Joly et al., 2010) (Box 4.5).

Au20 b0030 **BOX 4.5**

o0080 **(1)** Ungulates interact strongly with episodic disturbances. Many are attracted to severely burned areas because of increased forage palatability and availability, where in turn they influence vegetation regrowth.

o0085 **(2)** Elk, bighorn sheep, and mule deer generally increase after intense fire in shrublands and forests.

o0090 **(3)** The larger the area of high-severity fire, the lower the adverse impact on regrowth of aspen forests from ungulate herbivory.

o0095 **(4)** Caribou may be adversely affected when intense fire reduces lichen used for winter forage.

s0065 4.6 MANAGEMENT AND CONSERVATION RELEVANCE

p0345 The abundance of certain mammal species after fire has direct benefits to land managers in the form of irreplaceable ecosystem and economic services. Bats are voracious predators of insects—many of them consume crop and forest pests—and as such are important regulators of insect populations, including disease-carrying mosquitoes (Reiskind and Wund, 2009). Bats are also critical pollinators of many plants (Molina-Freaner and Eguiarte, 2003). The loss of bats in North America could cost the economy $3.7 billion per year in agricultural losses alone (Boyles et al., 2011). Small mammals aerate the soil and, along with many carnivores, are important dispersers of seeds and fungi (Maser et al., 1978; Rost et al., 2012). Large carnivores are top-down regulators of smaller carnivores and ungulates and are vital to the health and function of natural ecosystems. Ungulates help to cycle nitrogen and provide big-game hunting opportunities and food for humans. Indeed, in 2001 alone, hunting of ungulates and large carnivores in the United States contributed to approximately $25 billion in retail sales and $17 billion in salaries and wages and

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employed ~~of~~ 575,000 people (IAFWA, 2002). These animals include mule deer, bighorn sheep, moose, elk, and bear, all of which use or thrive within heavily burned habitats.

p0350 As described here, a great many mammals benefit from mixed- and high-severity fire and play essential roles in postfire ecosystem dynamics. Land managers rarely weigh these benefits when evaluating the impacts of large fires of mixed- and high-severity, however, thus undervaluing their ecological and economic importance. The vital ecosystem services of mammals in postfire areas should be quantified and carefully considered when planning potentially harmful management activities such as postfire logging and common management activities following postfire logging, such as the application of herbicides and rodenticides.

s0070 4.7 CONCLUSIONS

p0355 The extraordinary abundance and diversity of mammals using (e.g., American marten, Pacific fisher, grizzly bear) and even thriving (e.g., deer mice, kangaroo rats, bats, mule deer, elk, bighorn sheep) in severely burned grassland, shrubland, and forested habitats is an important indicator of the high habitat suitability of these areas. Prescribed burning does not provide the expected gains in biological diversity for a range of mammal, reptile, bird, and plant taxa (Pastro et al., 2014). Only large, severe wildfires create significant ecological changes associated with increases in fire-loving species, and, as demonstrated herein, only larger fires can "swamp" the effects of ungulate herbivory on postfire vegetation. ~~Mixed-severity and severe~~ fires globally have unique ecological value that must be weighed against the dominant paradigm that such natural disturbance events are "catastrophic" (Zwolak and Foresman, 2008; also see Chapters 1, 2, and 13). Mammals and other wildlife using intensely burned forests provide myriad ecological services that benefit people and ecosystems alike.

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s0075

APPENDIX 4.1 THE NUMBER OF STUDIES BY TAXA SHOWING DIRECTIONAL RESPONSE (NEGATIVE, NEUTRAL, OR POSITIVE) TO SEVERE WILDFIRE OVER THREE TIME PERIODS FOLLOWING FIRE. STUDIES CITED INCLUDE UNBURNED AREAS COMPARED ~~WITH~~ SEVERELY BURNED WITH NO POST-FIRE LOGGING, AND EXCLUDED PRESCRIBED BURNS. FOR SMALL MAMMALS, ONLY SPECIES WITH ENOUGH DETECTIONS TO DETERMINE DIRECTIONAL RESPONSE WERE REPORTED

t9010

	1-5 yr post-fire			6-10 yr post-fire			>10 yr post-fire		
	Negative	*Neutral*	*Positive*	*Negative*	*Neutral*	*Positive*	*Negative*	*Neutral*	*Positive*
Bats[1]		1	3						
Small Mammals[2]									
Masked shrew									
White-toothed shrew			1						1
Tamias spp.		4							
Pacific kangaroo rat		1	2			1			
Dulzura kangaroo rat			2						
Merriam's kangaroo rat			1			1			1
California pocket mouse		1	1						
San Diego pocket mouse	1	2							
Bush rat	1								
Long-haired rat	1								

Continued

	1-5 yr post-fire			6-10 yr post-fire			>10 yr post-fire		
	Negative	*Neutral*	*Positive*	*Negative*	*Neutral*	*Positive*	*Negative*	*Neutral*	*Positive*
Red-backed vole	2								
California vole		1	2						
Canyon mouse	1			1			1		
Brush mouse		1	1						
Deer mouse		2	5		1				
California mouse	3		1			1			
Cactus mouse	1	1	1						
Pinyon mouse	1		1	1					
Harvest mouse	2	1	1						
Desert woodrat	2								
Big-eared woodrat	1	1							
Snowshoe hare			1						
Antechinus	1								
Potoroo		1				1			1
Bandicoot			1			1	1		
Wombat		1			1		1		
Macrocarps (3 spp)		1		1			1		

Carnivores[3] American marten									
Stone marten						1			
Pacific fisher						1			
Gray fox			2						
Red fox						1			
Black bear	1								1
Grizzly bear									1
Ungulates[4] Caribou				2			2		
Moose			1						
Bighorn sheep			3			2			2
Elk			1						
Mule deer			4			1			1

np9010 [1]*Bat citations: Schwab 2006, Malison and Baxter 2010, Buchalski et al. 2013. Bats are categorized by phonic groups: of 6 phonic groups in 3 studies, 5 phonic groups showed positive response and 1 showed neutral response.*

np9015 [2]*Small mammal citations: Keith and Surrendi 1971, Cook 1959, Wirtz 1977, West 1982, Price and Waser 1984, Torre and Díaz 2004, Converse et al. 2006, Zwolak and Forsman 2007, Zwolak and Forsman 2007, Vamstad and Rotenberry 2010, Brehme et al. 2011, Banks et al. 2011, Arthur et al. 2012, Borchert and Borchert 2013, Letnic et al. 2013, Borchert et al. 2014*

np9020 [3]*Carnivore citations: Paragi et al. 1996, McLellan and Hovey 2001, Cunningham and Ballard 2004, Cunningham et al. 2006, Soyumert et al. 2010, Borchert 2012, Hanson 2013*

np9025 [4]*Ungulate citations: Taber and Dasmann 1957, McCulloch 1969, Bendell 1974, Peek 1974, Davis 1977, Keay and Peek 1980, Smith 2000, Holl et al. 2004, Dalerum et al. 2007, Bleich et al. 2008, Biggs et al. 2010, Joly et al. 2010, Greene et al. 2012* p0365 Studies cited include unburned areas compared with severely burned areas with no postfire logging; they exclude prescribed burns. For small mammals, only species with enough detections to determine directional response are reported.

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Non-Print Items

Abstract

Effects of mixed and severe fire on mammals vary spatially and temporally, by habitat type, and by species. Tree voles, masked shrews and some mice decrease, at least temporarily, after severe forest fire, but most bats and ungulates and many small mammals—especially deer mice and kangaroo rats—are strongly attracted to severely burned habitats due to novel foraging opportunities. In heavily burned forests, more insect prey is available for bats, and seeds and sprouting plants feed small mammals. Vegetation re-growth after intense fire produces highly palatable browse for elk, mule deer, and bighorn sheep. Standing dead trees provide cover for deer in severely burned forests, whereas bighorn sheep can more easily perceive predators in heavily burned chaparral. Mesocarnivores, including foxes, martens, and fishers, often are detected in forests that burned intensely. Unburned refugia within larger severe burns, and the time-since-fire, are especially important factors for recolonization by small mammals.

Keywords: Severe fire; Mammal; Bat; Rodent; Lagomorph; Carnivore; Ungulate; Forage.

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RESEARCH ARTICLE

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Key Points:

- Ten years postfire, 85% of fire-killed necromass remain in the forest
- Ten years postfire, fire-killed trees emit 0.6 Mg C ha^{-1} yr^{-1}
- While decomposition from fire-killed trees last decades, their contribution to NEP is small

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Carbon emissions from decomposition of fire-killed trees following a large wildfire in Oregon, United States

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Abstract A key uncertainty concerning the effect of wildfire on carbon dynamics is the rate at which fire-killed biomass (e.g., dead trees) decays and emits carbon to the atmosphere. We used a ground-based approach to compute decomposition of forest biomass killed, but not combusted, in the Biscuit Fire of 2002, an exceptionally large wildfire that burned over 200,000 ha of mixed conifer forest in southwestern Oregon, USA. A combination of federal inventory data and supplementary ground measurements afforded the estimation of fire-caused mortality and subsequent 10 year decomposition for several functionally distinct carbon pools at 180 independent locations in the burn area. Decomposition was highest for fire-killed leaves and fine roots and lowest for large-diameter wood. Decomposition rates varied somewhat among tree species and were only 35% lower for trees still standing than for trees fallen at the time of the fire. We estimate a total of 4.7 Tg C was killed but not combusted in the Biscuit Fire, 85% of which remains 10 years after. Biogenic carbon emissions from fire-killed necromass were estimated to be 1.0, 0.6, and 0.4 Mg C ha^{-1} yr^{-1} at 1, 10, and 50 years after the fire, respectively; compared to the one-time pyrogenic emission of nearly 17 Mg C ha^{-1}.

1. Introduction

Forest fires have long been recognized as an important component of the global carbon cycle. Among natural processes, combustion ranks second after metabolic respiration in mineralizing terrestrial biomass to the atmosphere, fire mortality ranks second after litter production in transferring live aggrading biomass to decomposing necromass, and the pyrolysis of biomass by forest fires feeds a global pool of black carbon which is largely isolated from the biological cycle [*Singh et al.*, 2012]. The role of forest fire in the carbon cycle is especially important in today's changing climate, not only because of its direct contribution to greenhouse gas emissions but also because a warming climate is expected to increase frequency and intensity of wildfires [*Flannigan et al.*, 2000, 2009; *Moritz et al.*, 2012], pushing the terrestrial biosphere toward a new equilibrium wherein less carbon resides in forest biomass and more resides in the atmosphere. Furthermore, because forest fire behavior is viewed by many as manageable, its control is regularly included as part of comprehensive climate change mitigation strategies [*Campbell et al.*, 2012; *Bradstock et al.*, 2012].

Characterizing and quantifying the effects of fire on the flux of carbon from forests into the atmosphere requires an understanding of both pyrogenic emissions due to immediate combustion and the prolonged biogenic emissions due to the decomposition (heterotrophic mineralization of carbon) by fire-killed necromass. A recent wealth of empirical studies aimed at quantifying combustion across a range of forest fires has allowed us to both constrain estimates of pyrogenic emissions and predict how this flux may change under alternate fire regimes (see reviews by *Sommers et al.* [2014] and *Urbanski* [2014]). By comparison, less attention had been paid to the protracted loss of terrestrial carbon to the atmosphere through the decomposition of fire-killed trees and how this flux is expected vary in relation to fire behavior or change under alternate fire regimes [*Harmon et al.*, 2011a, 2011b; *Ghimire et al.*, 2012].

Carbon emissions via the decomposition of fire-killed trees differ from pyrogenic emissions in several important ways. First, we expect that pyrogenic emissions to be lower in magnitude and less tightly coupled to fire behavior than subsequent carbon emissions via decomposition of fire-killed trees. Since combustion of aboveground biomass in forest fires is typically confined to dead surface fuels and live foliage, pyrogenic carbon emissions in any given fire tend not to exceed 15% of a forest's live and dead biomass [*Campbell et al.*, 2007; *Urbanski*, 2014]. Moreover, since the majority of surface fuels are consumed in nearly all fire conditions, while standing biomass experiences little combustion even in a crown fire, it is difficult for a high-mortality fire to combust much more than twice the amount of carbon than does a low-mortality fire. By contrast, subsequent carbon emissions through decomposition of biomass killed in the fire but not consumed may range from none (e.g., low-severity fires when no trees are killed) to all of the prefire biomass (e.g., high-severity fires when all trees are killed). For this simple reason, cumulative carbon emissions through decomposition of fire-killed trees may exceed pyrogenic emissions and are more dependent on fire behavior than are pyrogenic emissions.

Emissions through decomposition of fire-killed biomass also differ from pyrogenic emissions in their influence on Net Ecosystem Production (NEP). While pyrogenic emissions necessarily contribute to net ecosystem carbon balance, the flux itself is concentrated in time. By contrast, the protracted decomposition of fire-killed trees can contribute to disequilibrium in stand-level NEP for decades [*Bond-Lamberty and Gower*, 2008; *Harmon et al.,* 2011a; *Ghimire et al.,* 2012]. Theoretically, fire-induced disequilibrium in NEP will balance out to zero over sufficiently long time frames or spatial extents (after all, no tree ever escapes death and mineralization, fire only aggregates this inevitable emission in time). However, like many natural disturbances, the majority of area subject to high-mortality forest fire is the result of relatively few, very large events [*Malamud et al.,* 1998; *Reed and McKelvey,* 2002]. As such, the extent required for spatial neutrality in NEP to emerge may easily exceed any meaningful geographic boundary, and the time frame required for neutrality in NEP to emerge may easily exceed the meaningful continuity of any fire regime. Consequently, assessing the effects fire on the carbon exchange between forests and the atmosphere demands not only a mechanistic understanding of combustion, mortality, and decomposition (which we largely have) but also the ability to quantify these processes with enough context specificity to accurately account for individual fire events.

In this study, we evaluate the current and future carbon emissions attributable to the decomposition of trees killed but not combusted in the 2002 Biscuit Fire. This exceptionally large wildfire burned over 200,000 ha of mixed-conifer forest in southwest Oregon. Due to its diversity of forest types, forest age-classes, and severity of fire effects, the Biscuit Fire has served as a valuable case study for evaluating the effects of wildfire on carbon dynamics, including the following: pyrogenic emissions [*Campbell et al.,* 2007], export of soil carbon through erosion [*Bormann et al.,* 2008], and charcoal formation [*Donato et al.,* 2009a; *Heckman et al.,* 2013]. In *Campbell et al.* [2007] we reported biomass combustion for 25 functionally distinct carbon pools. Then, using measures of prefire biomass and fire effects on 180 one hectare inventory plots, we estimated fire-wide pyrogenic emissions. In this current companion study, we report the 10 year decay status of various biomass pools killed, but not combusted, by the Biscuit Fire. Then, using measures of fire mortality on the same 180 inventory plots as before, we estimate current and future fire-wide emissions resulting from the decomposition of fire-killed trees. Our specific objectives are as follows:

1. Quantify mortality, dead tree fall rate, and decomposition rates specific to different species, parts (e.g., root, bole, and branch), physical setting, prefire stand history, and fire effects.
2. Using these stratified parameters, calculate the current cumulative flux of carbon from fire-killed trees into the atmosphere and model its attenuation into the future.
3. Evaluate the current and future carbon emissions from fire-killed trees in the context of commensurate forest regrowth and other regional carbon fluxes, including the pyrogenic emissions from the same fire.

2. Methods

2.1. Study Site

The Biscuit Fire burned at a mix of severities across 200,000 ha of forest in the Siskiyou Mountains of southwestern Oregon and northern California in the summer of 2002, making it the largest contiguous forest fire on record for Oregon (Figure 1). The Siskiyou Mountains are characterized by a wide variety of forest types, from Douglas fir/western hemlock/bigleaf maple communities on mesic sites, to Douglas fir/tanoak on drier sites, to Jeffrey pine on ultramafic substrates *Whittaker* [1960]. A general description of the Biscuit Fire and the forests it affected can be found in *Halofsky et al.* [2011].

2.2. Decomposition of Fire-Killed Trees

As illustrated in Figure 2, decomposition of fire-killed trees was computed as the collective mass loss to the atmosphere, over a specified period, from three primary pools representing different physical orientations:

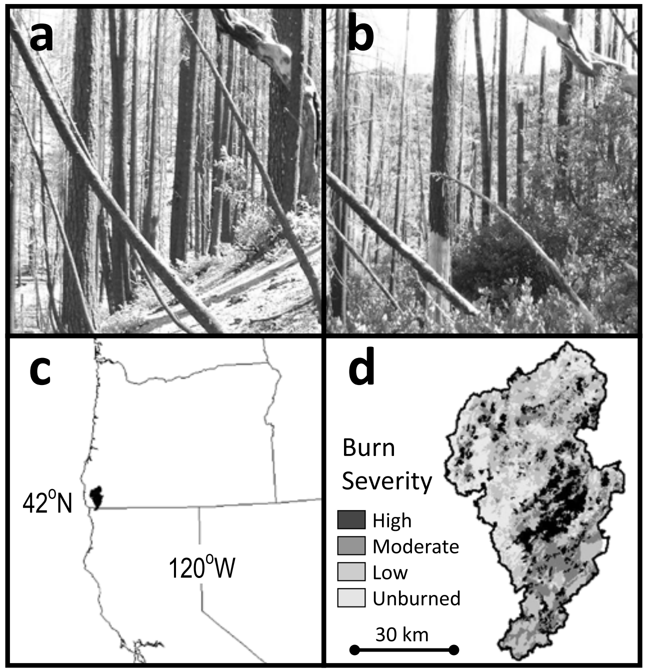

Figure 1. The 2002 Biscuit Fire showing (a) representative fire effects in 2004, (b) the same location in 2012, (c) location of the fire in the U.S. Pacific states of North America, and (d) remotely detected fire severity distribution. High = >90% overstory mortality, unburned = no overstory mortality but typically experiences surface fire.

standing necromass, fallen necromass, and buried necromass (i.e., dead root mass). Three separate rate constants defined mass loss to the atmosphere from standing, fallen, and buried necromass pools, respectively. Two additional rate constants defined transfer of mass from the standing to fallen pool by fragmentation and whole-tree fall, respectively. This three-pool, five-flux model was further stratified by tree part, namely, bole, branch, bark, and foliage (in the standing and fallen pools), and coarse root and fine root (in the buried pool). Boles were further stratified into three diameter classes, and all pools were stratified into three species groups (i.e., pines, non-pine conifers, and hardwoods) and three climatic zones (representing potentially different decomposition regimes) defined by aggregate plant association group and nominally corresponding to mesic, dry, and higher-elevation regions within the Biscuit Fire [*Donato et al.*, 2009b].

To estimate flux rates, we fit empirical observations of mass loss over time to a single-exponential model *Olsen* [1963] of the form:

$$M_t = M_0\left(e^{-kt}\right) \quad (1)$$

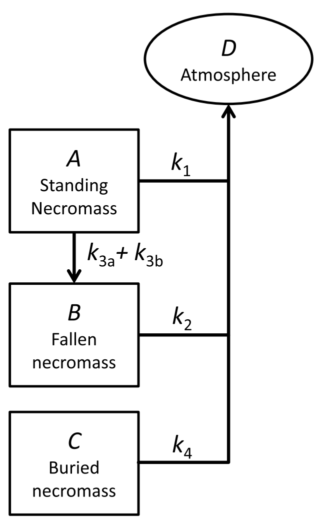

Figure 2. Approach to computing biogenic decomposition of fire-killed necromass. Decomposition was calculated separately for each plant tissue class according to five first-order exponential rate constants. The constant k_1 is the decomposition of necromass in its standing state; k_2 is the decomposition of necromass in its fallen state; k_{3a} and k_{3b} are the transfers between standing and fallen states, via whole-tree fall and fragmentation, respectively; and k_4 is the decomposition of buried roots.

where M_t is the mass of a specified necromass pool at time t, M_0 is the mass of the same pool immediately following its death by fire and any assessed combustion, and t is the elapsed time since the fire (~10 years in this study). In this way, the rate constant k not only describes the cumulative mass loss at year t but can also be used to extrapolate mass loss into the future. The accuracy of such extrapolation does, however, depend on the assumption that loss rates remain constant over time, which may be violated if either the environment in which decay is occurring changes or if discriminating decay renders mixed substrates more recalcitrant over time. Extrapolation of our decay model does not account for climate-driven changes in the decay environment, but our model does account for important changes in decay that occur after wood

Table 1. Methodology and Sampling Design for Determination of Rate Constants[a]

Aerial Decay: $k_1 = \ln(D_0/D_t)/t$, where D_0 = live tree part density, D_t = density of standing fire-killed tree part circa 2012, and t = elapsed time since fire	
Bole	D_t measured for 198 trees, stratified by species group (Douglas fir, pine species, and pacific madrone), diameter class (range 7 to 146 cm DBH), and climatic zone (defined by aggregate plant association group, nominally corresponding to mesic, dry, and higher-elevation regions within the Biscuit fire). Tree-average density was calculated as the average density of three transverse samples (cookies) collected from the lower, middle, and upper third of each tree, weighted by a factor of 0.60, 0.36, and 0.04, respectively, to account for volume proportion by height (derived from the taper equations of *Arney* [2009]). D_0 assumed to be 0.39, 0.45, and 0.58 g cm^{-3} for sugar pine, Douglas fir, and pacific madrone, according to *Maeglin and Wahlgren* [1972], *USFS* [1965], and Wood Data Base, respectively.
Branch	D_t measured for 259 branches stratified by diameter (range 1 to 56 mm) collected from the 198 standing dead trees described above. D_0 measured for 55, similarly stratified live tree branches samples.
Bark	Bark density loss was not directly measured in this study. Based on *Allison and Murphy* [1963], we crudely assumed bark to decompose at one half the rate of bole wood of the same species. Anecdotally, bark from fire-killed trees in this study regularly showed evidence of charring and fragmentation but not any apparent density loss.
Foliage	Aerial decay rates of fire-killed foliage are computationally inconsequential, not only because fire mortality on the Biscuit most often entailed full foliage combustion [*Campbell et al.,* 2007] but also because fall rates of fire-killed foliage approach totality within the first year after mortality such that nearly all decay occurs on the ground. As such, foliage aerial decay rates were arbitrarily set to 0.5 year^{-1}.
Surface Decay: $k_2 = \ln(D_0/D_t)/t$, where D_0 = live tree part density, D_t = density of fire-killed tree part having fallen to ground shortly after fire, and t = elapsed time since fire	
Bole	D_t measured on 60 fallen logs, deduced to have been killed in the Biscuit Fire (by presence of surface charring) and fell within the next year (saw cuts datable to known salvage operations); stratified by species group (see above), diameter class (range 7 to 146 cm DBH), and climatic zone (see above). Density was determined from a single transverse sample (cookie) taken from the center of each log. D_0 as described above for areal bole decay.
Branch	D_t measured for 86 branch samples, stratified by diameter (range 1 to 72 mm) collected from the 60 fallen logs described above. D_0 as described above for areal branch decay
Bark	Crudely assumed to be one half the rate of fallen bole wood (see above for aerial bark decay).
Foliage	Given the short residence times of leaf litter (relative to wood and bark), and the exceptionally small portion of fire-filled biomass represented by uncombusted foliage [*Campbell et al.,* 2007], we chose to avoid the hazard of false accuracy and simply assign foliage decomposition rates the arbitrarily rapid rate of 0.5 year^{-1}.
Whole-tree Fall Rate: $k_{3a} = \ln(C_0/C_t)/t$, where C_0 = count of standing dead trees circa 2004, C_t = count of standing dead trees ca 2013, and t = elapsed time between samples	
Whole tree	Before-and-after stem surveys conducted at 44 independent and dispersed study plots, including a total sample size of >3000 fire-killed trees ranging in size from 2 to 198 cm DBH.
Fragmented Fall Rate: $k_{3b} = \ln(M_0/M_t)/t$, where M_0 = mass of standing tree parts circa 2004, M_t = mass of standing tree parts circa 2012, and t = elapsed time between samples	
Bole	M_0 allometrically modeled from DBH with the assumption that each tree was live and entire. M_t is the same value, corrected to account height loss due to observed breakage. Assessed for each of the 3000 fire-killed trees described above.
Branch	M_0 allometrically modeled from DBH with the assumption that each tree was live and entire. Each fire-killed tree surveyed in 2014 was binned into one of four fragmentation classes through ocular assessment, corresponding to an M_t of 0.05 M_0, 0.15 M_0, 0.60 M_0, and 1.0 M_0, respectively.
Bark	M_0 allometrically modeled from DBH with the assumption that each tree was entire. Each fire-killed tree surveyed in 2014 was binned into one of four fragmentation classes through ocular assessment, corresponding to an M_t of 0.0, 0.25 M_0, 0.75 M_0, and 1.0 M_0, respectively.
Foliage	Practically all uncombusted foliage retained on fire-killed trees fell to the ground within the first year after the fire. To account for this in our decomposition model (constructed only of first-order exponential rate constants) we set the rate constant describing dead foliage fall to 5.0 year^{-1}.
Buried Decay: k_4 = first-order exponential decay constants according to named authors	
Coarse root	k = 0.02 year^{-1} according to *Janisch et al.* [2005] assessment of Douglas fir roots > 1.0 cm diameter.
fine root	k = 0.20 year^{-1} according to *Chen et al.* [2002] and *Fogel and Hunt* [1979] for various tree roots < 1.0 cm diameter.

[a]Dead wood density was determined after oven drying at 95°C to constant mass; an 8% downward correction was then applied to account for oven shrinkage and afford direct comparison with published green tree densities [*Glass and Zelinka*, 2010].

transitions from the aerial to surface environment. Furthermore, by disaggregating our necromass pools (i.e., into bole, branch, bark, foliage, root, species group, and size class) our model minimizes the changes in recalcitrance that any one pool may experience over time [*Freschet*, 2012]. The specific sampling methods used to determine M_t and M_0 for each necromass category are detailed in Table 1. Note that while the form of equation (1) was used in computing all flux rates, at times, density, volume, or count was operationally substituted for mass.

2.3. Initial Fire-Killed Biomass

Within the perimeter of the Biscuit Fire there are 180 regularly spaced permanent federal inventory plots, all of which received postfire measurements in 2003 or 2004 [*Azuma et al.,* 2004]. It is well established that injury caused by fire can sometimes contribute to tree death several years after being burned [*Filip et al.,* 2007].

Our assessment operationally defines fire mortality as trees which died within 1–2 years after the fire. Any subsequent mortality and ensuing decomposition, though perhaps related to fire, was not in this study directly attributed to the Biscuit Fire.

For each tree identified in the inventory plots as having been killed in the Biscuit Fire, we estimated the mass of its fine roots, coarse roots, bole, branch, bark, and foliage as if it were alive and whole. From each of these parts, we then subtracted the proportion estimated to have been combusted in the fire according to *Campbell et al.* [2007] to yield a tree-specific M_0 for each of its component parts. Bole mass was estimated using species- and site-specific allometric equations relating stem diameter to volume and species-specific wood density values [*van Tuyl et al.,* 2005]; foliage and bark mass were estimated directly from species- and site-specific allometric equations [*Means et al.,* 1994]; coarse root mass was assumed to be 0.31 times the bole mass (an average of regionally representative, plot-level ratios, allometrically estimated by *Campbell et al.* [2004a]); and fine root mass was assumed to be 0.16 times the bole mass (an average of regionally representative, plot-level ratios directly sampled by *Campbell et al.* [2004b]). Total biomass was converted to carbon mass assuming a carbon concentration of 0.5 for all woody parts and 0.45 for foliage. These tree-level values for M_0 were then summed across each inventory plot as to be expressed in carbon mass per unit ground area.

2.4. Fire Severity and Scaling Across the Fire

For evaluating the direct effects of fire severity on subsequent carbon emissions, fire severity was calculated, for each of the 180 inventory plots, as the fraction of initial live basal area (including all woody stems $>=$ 2.5 cm diameter breast high (DBH)) killed in the Biscuit Fire. For the purpose of scaling plot-level measurements to the entire Biscuit Fire it was necessary to use a mapped assessment of fire severity. Specifically, plot-level estimates of decomposition were scaled-up to the entire Biscuit Fire according to mapped fire severity classification and whether or not a site had burned in the Silver Fire (a major fire which burned 13 years prior to the Biscuit Fire). Such strata accounted only for variation in M_0 (tree mass killed in the Biscuit Fire), as the rate constants k were assumed to be the same among plots. We employed the same BAER (Burned Area Emergency Response) severity classification map used earlier by *Campbell et al.* [2007]. Since this time, improved maps of Biscuit Fire severity have been built [*Thompson and Spies,* 2009], but we felt it was more important to maintain consistency between our pyrogenic and biogenic accounting. Moreover, since the 180 inventory plots are distributed widely in space and randomly with respect to actual fire effects, misclassification by BAER, or any other severity map, does not bias fire-wide estimates of carbon flux.

2.5. Uncertainty Propagation

For this study, we assumed the inventory-based estimates of fire-killed necromass to be largely accurate and limited our uncertainty analysis to that associated with decomposition rates. To account for this uncertainty, we computed alternate estimates of total carbon emissions using an upper and lower values for the rate constants defining mass loss to the atmosphere. Uncertainty in mass loss from standing and fallen necromass pools (k_1 and k_2 in Figure 2) were based on the upper and lower 95% confidence intervals in dead wood density (among samples collected in 10 years after death). Since we relied on crude literature values for root decay, uncertainty in mass loss from buried necromass pools (k_4 in Figure 2) was generously set to plus and minus 20% density loss at 10 years after death.

3. Results

3.1. Fire Mortality

Prefire live aboveground and belowground biomass among the 180 inventory plots ranged from 1 to 502 (median = 161) Mg C ha^{-1} depending somewhat on site quality but largely disturbance history (i.e., whether sites had experienced late twentieth century fire). Fractional tree mortality, which was largely independent of prefire biomass, ranged from zero to totality. As a result the necromass killed but not combusted among the 180 inventory plots ranged from 0 to 352 (median = 24) Mg C ha^{-1} . Despite smaller trees being more abundant, more often killed, and only somewhat more combusted than larger trees, fire mortality in the form of large-diameter (>30 cm DBH) boles and their associate coarse roots made up greater than 40% of all other fire-killed biomass combined. The remaining uncombusted fire mortality is composed of smaller diameter wood, bark, fine roots, and foliage in that order (Table 2). Overall the Biscuit Fire killed and left uncombusted a total of 10.4 Tg C (an average of 51 Mg C ha^{-1}).

Table 2. Biomass Killed But Not Combusted in Biscuit Fire (kg C ha^{-1})

	Biscuit Fire Severity[a]							
	Not Burned 15 years Earlier in Silver Fire				Also Burned 15 years Earlier in Silver Fire			
Necromass Pool	High	Moderate	Low	Unburned Very Low	High	Moderate	Low	Unburned Very Low
				Foliage				
Small conifers	19	62	58	26	0	0	4	2
Small hardwoods	31	23	57	99	2	42	29	67
Medium conifers	135	367	232	131	0	1	32	29
Medium hardwoods	292	77	354	606	3	151	289	763
Large conifers	180	384	409	162	0	242	67	190
Large hardwoods	52	7	67	162	0	37	98	46
				Branch				
Small conifers	130	115	88	34	0	13	6	3
Small hardwoods	144	37	106	159	146	142	50	120
Medium conifers	1207	981	501	247	0	83	78	60
Medium hardwoods	1778	183	1026	1407	53	1130	806	2202
Large conifers	3279	1837	1438	523	0	2598	350	683
Large hardwoods	835	23	280	610	0	2540	387	211
				Bark				
Small conifers	111	109	86	34	0	12	6	3
Small hardwoods	76	22	65	100	75	83	31	76
Medium conifers	1284	1184	607	314	0	95	95	78
Medium hardwoods	1314	135	861	1207	44	955	701	1917
Large conifers	5019	3097	2641	962	0	4760	639	1281
Large hardwoods	877	23	318	748	0	2944	446	237
				Bole				
Small conifers	537	409	328	146	0	69	28	13
Small hardwoods	1220	250	876	1348	1333	1272	416	974
Medium conifers	7058	5254	2734	1425	0	555	462	401
Medium hardwoods	16733	1559	9027	12772	557	10730	7152	15988
Large conifers	31100	16967	13599	4950	0	28981	3632	6380
Large hardwoods	6885	186	2206	4947	0	21250	3576	1461
				Roots				
Small conifers	193	147	118	52	0	25	10	5
Small hardwoods	439	90	315	485	479	457	150	350
Medium conifers	2538	1890	983	512	0	199	166	144
Medium hardwoods	6017	561	3246	4593	200	3858	2572	5749
Large conifers	11184	6101	4890	1780	0	10422	1306	2294
Large hardwoods	2476	67	793	1779	0	7642	1286	525

[a]As determined by remotely sensed BAER severity classification. Values are the average of 24, 36, 42, and 34 inventory plots for high, moderate, low, and unburned very low severity plots not burned prior in the Silver fire, respectively; and the average of 1, 2, 14, and 5 inventory plots for high, moderate, low, and unburned very low severity plots burned prior in the Silver fire, respectively. Small trees are <10 cm DBH, medium trees are 10–20 cm DBH, and large trees are >20 cm DBH. For our decomposition calculations, conifers were further partitioned into pine and nonpine species (data not shown here), and roots were partitioned into coarse roots and fine roots, consistently computed as 0.66 and 0.34 total root mass, respectively.

3.2. Decomposition Rates

The measured densities of standing and fallen fire-killed wood, from which decomposition rates were calculated, are shown in Figure 3. An analysis of variance performed on the decomposition rates calculated for over 198 sampled tree boles revealed significant effects of species (with Douglas fir decomposing only slightly faster than pine species and pacific madrone) and condition (fallen logs decomposing only slightly faster than standing snags), but nonsignificant effects of geographic zone (mesic, dry, or high elevation) or size (diameter class). The single-exponent decomposition constants (fit to a single 10 year data point and used to subsequently model carbon emissions) are shown in Table 3.

3.3. Tree Fall Rates

As shown in Table 4, a greater fraction of fire-killed biomass fell from the canopy to the ground in 10 years through whole-tree fall than through fragmentation. The proportion of whole trees having fallen after

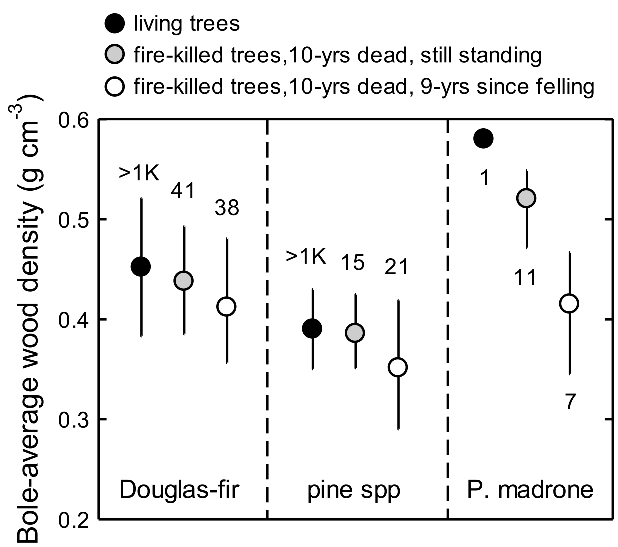

Figure 3. Wood density of green trees (live), fire-killed trees still standing 10 years after death (snags), and fire-killed trees 10 years after death and near immediate falling (logs). Sample size (shown near each symbol) is the number of independent trees sampled, with the density of each being determined as the taper-weighted average density of three cross-sectional subsamples taken along the length of each tree. Variability in wood density among trees is shown as the standard deviation (upper and lower error bars are the average positive and negative residuals of the mean, respectively; except for green trees where only a single symmetrical standard deviation was available from source literature, and green madrone where no variance was reported). Live wood densities are from *Maeglin and Wahlgren* [1972], *US Forest Service* [1965], and *Wood Data Base*, for pine species, Douglas fir, and Pacific madrone, respectively. Dead wood densities are those measured in the present study.

10 years was 20 times greater for smaller diameter trees (<20 cm DBH) than for larger diameter trees. Neither whole-tree fall rate nor fragmentation rate varied according to community type (used here as a proxy for decomposition regime). Across species, size class, and location, 57% of the trees killed in the Biscuit Fire are still standing 10 years after their death and have on average lost only 26% of their postcombustion necromass via fragmentation.

3.4. Biogenic Emissions

The amount of carbon released through the decomposition of fire-killed trees in the first 10 years following the Biscuit Fire is estimated to be 1.3 to 1.6 Tg C (or 6.5 to 7.8 Mg C ha^{-1}). As shown in Table 5, the largest contributing pools were those with the largest initial mass (i.e., bole wood and coarse root), not those with the highest decomposition rates (foliage and fine roots). Extrapolating our 10 year estimates of fall rates and decomposition rates back to the first year following fire and forward to 100 years after fire reveals several emergent patterns. Partitioning emission rates among necromass pools (Figure 4a) illustrates not only differential decay rates (responsible for the inflection point in collective emissions) but also an important 10 year lag in peak emissions from bole, branch, and bark, which results from a particular combination of aerial decay rates, fall rates, and surface decay rates. Total emissions from fire-killed necromass over time exhibit a distinct inflection point approximately five years following the fire (Figure 4b). Such inflection points are indicative of mixed substrate decay and in this case occur when the more labile foliage and fine root pools have become largely exhausted leaving the more recalcitrant wood and coarse roots. Overall, half of the Biscuit-killed necromass will still remain 50 years after the fire, at which time emissions from this single mortality cohort will be approximately 25 Mg C ha^{-1} yr^{-1} (Figure 4c).

The total amount of fire-killed necromass explained 99% of the variation in post fire decomposition among the 180 study plots (Figure 5a), indicating that variation in prefire species composition and tree size class was of

Table 3. Decomposition Constants for Fire-Killed Necromass[a]

	Decomposition Constant k (year^{-1})	
Necromass Pool	Aerial Decay(Standing Snags)	Surface Decay(Fallen Logs and Debris)
	Bole	
Nonpine conifers	0.010 (0.008–0.012)	0.016 (0.013–0.019)
Pines	0.001 (0.001–0.004)	0.010 (0.005–0.014)
Hardwoods	0.010 (0.008–0.012)	0.016 (0.014–0.018)
	Branch	
All species	0.014 (0.013–0.015)	0.010 (0.008–0.012)

[a]Decomposition constant $k = \ln(\mathrm{Density}_{live}/\mathrm{Density}_{11\ years\ dead})/11$ years. Upper and lower estimates shown in parentheses were computed using standard error of the mean $\mathrm{Density}_{11\ years\ dead}$. See Table 1 for assumptions regarding decomposition of other fire-killed necromass pools such as foliage, bark, and roots.

Table 4. Fall Rate of Fire-Killed Necromass[a]

		Fraction Fallen After 10 years		Fall Rate k (year^{-1})	
Necromass Pool	Number of Trees Sampled	Via Whole-Tree Fall	Via Fragmented Fall	Via Whole-Tree Fall	Via Fragmented Fall
		Bole			
Conifers (small)	156	0.86		0.177	
Conifers (medium)	407	0.36		0.041	
Conifers (large)	805	0.03		0.003	
Hardwoods (all sizes)	229	0.03	0.35	0.003	0.043
Nonpine conifers (all sizes)	1075		0.16		0.017
Pines (all sizes)	137		0.14		0.016
		Branch			
Conifers (small)	156	0.86		0.177	
Conifers (medium)	407	0.36		0.041	
Conifers (large)	805	0.03		0.003	
Hardwoods (all sizes)	229	0.03	0.41	0.003	0.053
Nonpine conifers (all sizes)	1075		0.42		0.054
Pines (all sizes)	137		0.50		0.070
		Bark			
Conifers (small)	156	0.86		0.177	
Conifers (medium)	407	0.36		0.041	
Conifers (large)	805	0.03		0.003	
Hardwoods (all sizes)	229	0.03	0.51	0.003	0.070
Nonpine conifers (all sizes)	1075		0.48		0.065
Pines (all sizes)	137		0.57		0.085

[a]Fall rate $k = \ln(\text{standing necromass}_{2004}/\text{standing necromass}_{2014})/10$ years. Small trees are <10 cm DBH, medium trees are 10-20 cm DBH, and large trees are >20 cm DBH.

little importance in dictating postfire decomposition. Moreover, since low-biomass stands often experienced high-fractional mortality and high biomass often experienced low-fractional mortality, fire severity (as assessed by fractional basal area mortality) was, by itself, an imprecise predictor postfire carbon emissions (Figure 5b).

4. Discussion

4.1. Fire Mortality

The necromass generated in high-severity portions of the Biscuit Fire (about 103 Mg C ha^{-1}) corresponds well to the 130–200 Mg C ha^{-1} biomass held in mature and old-growth forests of the Klamath ecoregion according to the regional assessment of *Hudiburg et al.* [2009]. Between the ages of 50 and 100, these particular forests are estimated to experience tree mortality rates of just over one-half percent annually [*Hudiburg et al.*, 2009]. As such, when mature forests burned at high severity in the Biscuit, somewhere between 100 and 200 years of future mortality was compressed into a single event. When individual fires of this size and severity occur in high biomass forests, like those of western Oregon, the generation of decomposing necromass is

Table 5. Biogenic Emissions From Fire-Killed Necromass by Carbon Pool and Burn Severity Class

	Carbon Released (kg C ha^{-1} After 10 years)				Fire-Wide Emissions[b] (Tg C Across 202,642 ha, After 10 years)
Necromass Pool	High Severity[a]	Moderate Severity[a]	Low Severity[a]	Unburned Very Low Severity[a]	
Foliage	676	887	949	1163	0.19
Branch	850	365	307	324	0.08
Bark	405	219	173	168	0.04
Bole	5450	2125	2095	2362	0.54
Roots	6040	2385	2164	2277	0.58
Total	13421	5982	5683	6294	1.44 (1.31–1.59)[c]

[a]As determined by remotely sensed BAER severity classification.
[b]Fire-wide emissions calculated by weighting the emissions from each burn class by the area of that burn class over the fire perimeter.
[c]Upper and lower estimates based on propagated uncertainty in woody decomposition rate constants.

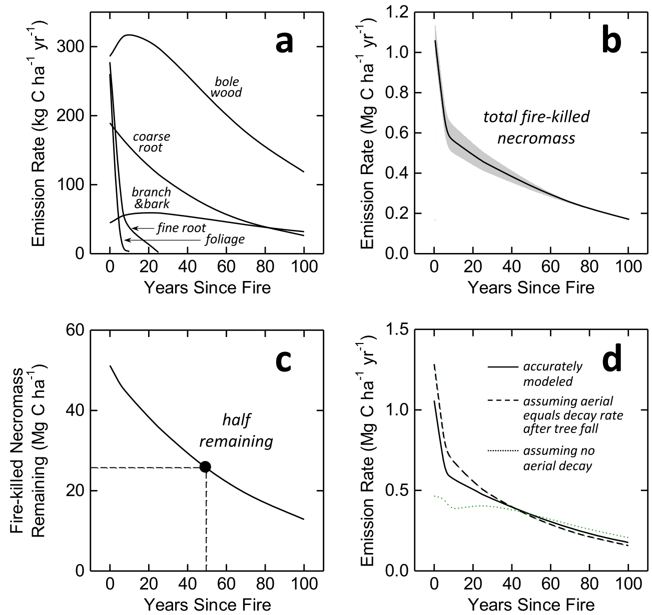

Figure 4. Temporal patterns of carbon emissions from fire-killed necromass, (a) partitioned by pool; (b) total, illustrating inflection point around year 5 and propagated uncertainty in decomposition rates (shaded band = 95% confidence interval); (c) approximate 50 year half-life; and (d) consequences of recognizing differential aerial and surface (fallen) decay rates.

notable at regional and even continental scales. The total amount of carbon transferred by the Biscuit Fire from aggrading living pools into decomposing dead pools was approximately three-quarters the average amount killed annually by wildfire throughout the entire western US (6 Tg C yr^{-1}) [*Hicke et al.,* 2013]. The distribution of fire mortality among different pools (Table 2) is a simple reflection of within-tree allometric proportions sans foliage which is commonly combusted in fire-killed trees. Understandably then, large-diameter wood made up the largest fire-generated necromass pool, more so in forests not recently burned where an even greater proportion of biomass was in the form of bole wood.

Due largely to the wide range of prefire biomass, fractional fire mortality (whether inferred through remote imagery, or direct ground measurement) was a poor predictor of absolute mortality and subsequent carbon emissions. While both intuitive and expected, this observation reminds us of the importance of accurately assessing preburn biomass in mapping and modeling fire effects on carbon dynamics.

4.2. Decay Rates

The wood density decomposition rates reported here fall comfortably within the range reported by other studies in the Pacific Northwest [*Sollins,* 1982; *Harmon et al.,* 1986; *Janisch et al.,* 2005; *Harmon et al.,* 2011b; *Dunn and Bailey,* 2012], which both validates our assessment and brings into question the need for additional field studies, at least those using single-exponent decay models fit to mass loss over a single time interval. In reality, necromass decay over time is expected to exhibit some initial lag (as substrates await decomposer colonization or fragmentation) and a decreasing proportional loss over time (as mixed substrates are reduced to their more recalcitrant fractions). By measuring mass loss across a chronosequence of dead wood, *Harmon et al.* [2000] demonstrated that dead wood decay can, in fact, exhibit such lags and tails in mass loss over time. Still, provided necromass pools are appropriately disaggregated (i.e., relatively recalcitrant and labile substrates assigned their own loss rate constants), single-exponent models like those used in this study fit empirical data just as well as multiparameter models [*Freschet,* 2012].

Given the recognized effects of moisture and temperature on decomposition, our inability to detect site effects on decomposition rate was likely a combination of measurement error (driven largely by our use of a single-species-specific green tree wood density in assessing mass loss for all wood fragments) and a wide variation in realized decay environments within the crude climate zones we recognized (Table 1). Given our samples were so widely distributed across our study area, our mean decomposition rates remain good estimates for our particular study. However, caution should be taken in applying these or any other landscape-average decomposition rates to any particular site, as decay rates of common substrates may vary across forest microenvironments by as much as 10 times, more so even than across large-scale climate gradients [*Vanderhoof*, 2013; *Bradford et al.,* 2014].

4.3. Fall Rates

The fall rates of standing necromass by fragmentation and whole-tree fall pertain to carbon emissions only to the degree that decomposition rates are different between the aerial and surface environments. It is commonly

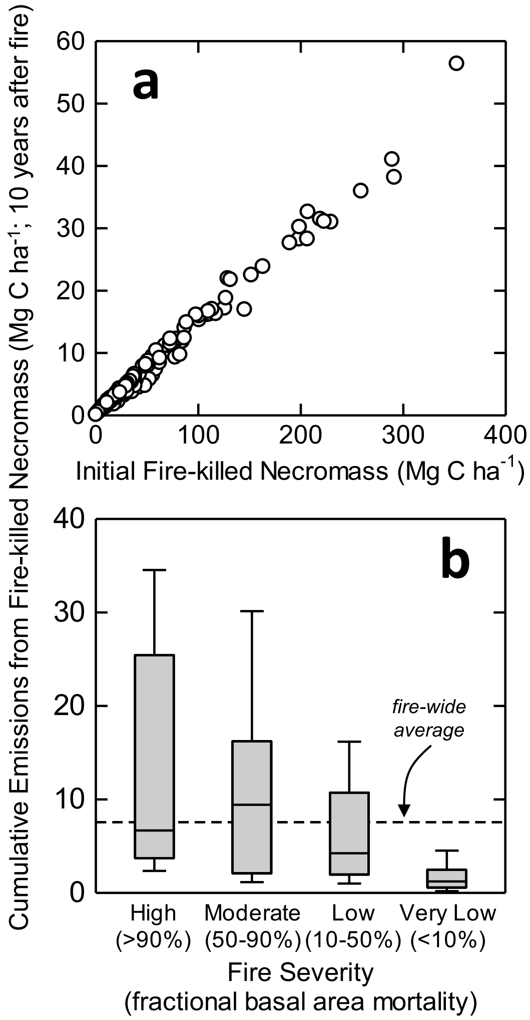

Figure 5. Carbon emissions from fire-killed necromass as a function of (a) absolute mortality and (b) fire severity among 180 inventory plots regularly stratified across the Biscuit Fire. Centerline, box, and whiskers, represent median, 25th percentiles, and range up to three-halves end quartiles (i.e., range excluding outliers), respectively. Fire severity (fractional tree basal area mortality) was directly determined for each plot (not remotely sensed).

assumed and consistently observed that decay rates of wood are slower in the drier aerial environment than in the moister surface environment [*Harmon et al.*, 2011b; *Yatskov et al.*, 2003; *Dunn and Bailey*, 2012]. Ecosystem models which apply the more commonly available surface decay rates to all fire mortality, without considering the decades many dead trees may spend in a standing condition, will inevitable overestimate initial emission rates and underestimate their duration. Similarly, models which assume negligible wood decay until a dead tree falls are prone to an inverse bias. The significance of tree fall rates in the timing of postfire carbon emissions is apparent in Figure 4a where peak emissions from branch, bark, and bole wood occur not immediately following the fire (when pool sizes are necessarily largest), but rather 10–20 years following the fire (after a requisite portion of the pool has fallen to the ground where it decays quicker). To further evaluate the relevance of tree fall on carbon emissions following the Biscuit Fire, we compared our fully parametrized model to others with alternate assumptions regarding fall rate and differential decay. As illustrated in Figure 4d, the largest bias occurred in the model which assumed wood remained undecayed until it fell to the ground. Applying a single surface decay rate to all wood did overestimate the near-term emission rates, but not as much as purported for other disturbed forests where both fall rates and the disparity between aerial and surface decay were determined to be higher than we observed in the Biscuit Fire [*Harmon et al.*, 2011b]. Moreover, once combined with the consistently attenuating emission from fire-killed roots and foliage, the fall-mediated lag in emissions from bole, branch, and bark did not produce a bimodal or "double-humped" emission pattern as it might have [*Harmon et al.*, 2011a].

Some authors have reported a brief (2 to 3 year) delay between tree mortality and the onset of measurable fall (see review by *Cluck and Smith* [2005]), suggesting that fall rates sometimes accelerate after passing some threshold in declining stability (e.g., root or basal decay). Since snag fall in this study is evaluated using stem attrition measured only at one-time point (10 years after death), we cannot resolve any early changes in fall rate. However, as a general rule, snag attrition measured over decades in prior studies conforms well to a first-order decay function as we have done here [*Everett*, 1999; *Cluck and Smith*, 2005]. Necromass decay over time is expected to exhibit some initial lag (as substrates await decomposer colonization or fragmentation) and a decreasing proportional loss over time (as mixed substrates are reduced to their more recalcitrant fractions).

4.4. Emission Rates

It is expected that dead wood dynamics operate over longer time scales in the Pacific Northwest than they do in other forests where environmental conditions or disturbance frequency prevent individual trees from growing as large. The analysis by *Spies and Franklin* [1988] suggests it would take >1000 years for woody

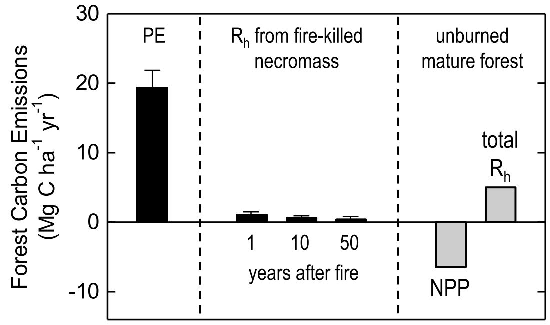

Figure 6. Forest carbon emissions from heterotrophic respiration (R_h) of necromass killed in the Biscuit fire, compared to the one-time pyrogenic emissions (PE) incurred during the fire and the biological fluxes typical of unburned mature forests of the Klamath region. Error bars on R_h are propagated 95% confidence intervals in decomposition rates. Pyrogenic emissions and uncertainty estimated by *Campbell et al.* [2007]. Net Primary Production (NPP) modeled by *Turner et al.* [2007] and consistent with empirical observations of *Hudiburg et al.* [2009]. Total R_h, which includes both the heterotrophic fraction of soil surface efflux and dead wood decay, modeled by *Turner et al.* [2007] and consistent with empirical observations of *Campbell et al.* [2004a, 2004b].

debris to reach a site-level steady state in Western Oregon, and as such, most forests in the region exist in a state of dead wood disequilibrium defined by site-specific disturbance history. In this study, the measured magnitude and modeled duration of carbon released to the atmosphere through the decomposition of fire-killed trees speaks to how disturbance-generated mortality shapes not only the amount woody debris present at any given time, but in the exchange of carbon occurring between forest and atmosphere at any given time.

As shown in Figure 6, 10 years after the Biscuit Fire the annual flux of carbon from fire-killed trees into the atmosphere is estimated to be 0.6 Mg ha^{-1} yr^{-1}, which is only 10% the total heterotrophic respiration rates to which these forests hypothetically equilibrate once mature [*Turner et al.,* 2007; *Campbell et al.,* 2004a] and only 3% the one-time pyrogenic emissions released during the fire [*Campbell et al.,* 2007]. Clearly, the capacity of this relatively modest carbon flux to shape carbon exchange between forest and atmosphere has not to do with its magnitude, but rather its duration and the fact that other ecosystem carbon fluxes such as net primary production, and potentially soil surface efflux, are greatly reduced in the initial period following wildfire.

Several studies suggest that high-severity wildfire, despite generating substantial additions to the dead wood pool, actually reduces total heterotrophic respiration by about one half [*Meigs et al.,* 2009; *Dore et al.,* 2012]. This is because wildfire typically consumes the forest floor (the substrate from which up to 30% of total heterotrophic respiration arises; *Campbell et al.* [2004b]) and temporally cuts off the supply of fine root turnover (a sizable contribution to belowground heterotrophic respiration). It was not the purpose of the paper to compute postfire NEP which would depend largely on uncertain patterns of forest regrowth and mineralization of soil carbon; however, NPP of regenerating and surviving vegetation need only reach 0.57 Mg ha^{-1} yr^{-1} by the 10 year following fire in order to compensate for the respiration from the remaining fire-killed necromass. Preliminary measurements (unpublished data) suggest that shrub production alone 10 years after the Biscuit Fire has already far exceeded this rate, consistent with other studies showing NPP over 1.5 Mg C ha^{-1} by 2 years postfire in dry forests [*Irvine et al.,* 2007].

4.5. Regional Carbon Disequilibrium

Single, large disturbances like the Biscuit Fire make for valuable examples because they provide a broad range of conditions over which to stratify measurements. The specificity with which we evaluated mortality, fall, and decay within the Biscuit Fire was limited only by resources, not by opportunity. But quantifying the impacts of single events such as the Biscuit Fire also sheds light on the unique importance of rare events in shaping regional carbon exchange and the need to accurately account for them when either upscaling terrestrial measurements or downscaling atmospheric measurements.

It is reasonable to postulate, as *Odum* [1969], that over a sufficiently large landscape, disturbance-induced disequilibrium in any one location will be balanced in other locations experiencing similar disturbances at different times, and as long as the region-wide frequency of such disturbances remains constant, this shifting mosaic will operate with mass neutrality (e.g., NEP). However, within many ecoregions forest fires may not occur at fine-enough grain and high-enough frequencies for such equilibriums to arise. In fact, the self-organizing behavior of fire across landscapes dictates that most of the area burned in any given fire regime is the result of relatively few, very large events [*Malamud et al.,* 1998; *Reed and McKelvey,* 2002]. This disproportional impact of large infrequent disturbances thwarts landscape equilibriums in two dimensions. First, it can extend the area required to balance disturbance effects at any given time beyond meaningful ecological boundaries. Second, it can extend the time horizon required for any bounded area to achieve equilibrium beyond the period we expect disturbance regimes to be reasonable stable. This second constraint on landscape equilibrium is especially relevant considering climate change may now be altering probabilistic fire regimes faster than the return interval of the most important events [*Zinck et al.,* 2011], rendering the realized impacts of fire on processes such as carbon emission wildly stochastic in space and time.

As illustrated in Figure 6, the carbon emissions attributed to the decomposition of trees killed in the Biscuit Fire documented in this study, as well as the pyrogenic emissions released by the Biscuit Fire documented in *Campbell et al.* [2007], attest to the importance single-disturbance events can have in regional carbon dynamics, especially in large biomass systems confined to relatively small ecological boundaries. Predicting the frequency of these rare events will be increasingly difficult in a changing environment, but our ability to accurately assess their impacts on regional carbon flux is slowly approaching sufficiency.

Acknowledgments
This work was supported, in part, by the Joint Fire Science Program, administered through the U.S. Bureau of Land Management in cooperation with the Department of Forest Ecosystems and Society at Oregon State University (project 11-1-1-4, cooperative agreement L11AC20276). The views and conclusions contained in this document are those of the authors and should not be interpreted as representing the opinions or policies of the U.S. Government or any other institution. We would like to extend special thanks to Melody Culp of the Rogue-Siskiyou National Forest for her invaluable assistance throughout the project. The data used in this study are available, without charge, by request of the corresponding author.

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Surprising Diversity of Bats in Senegal

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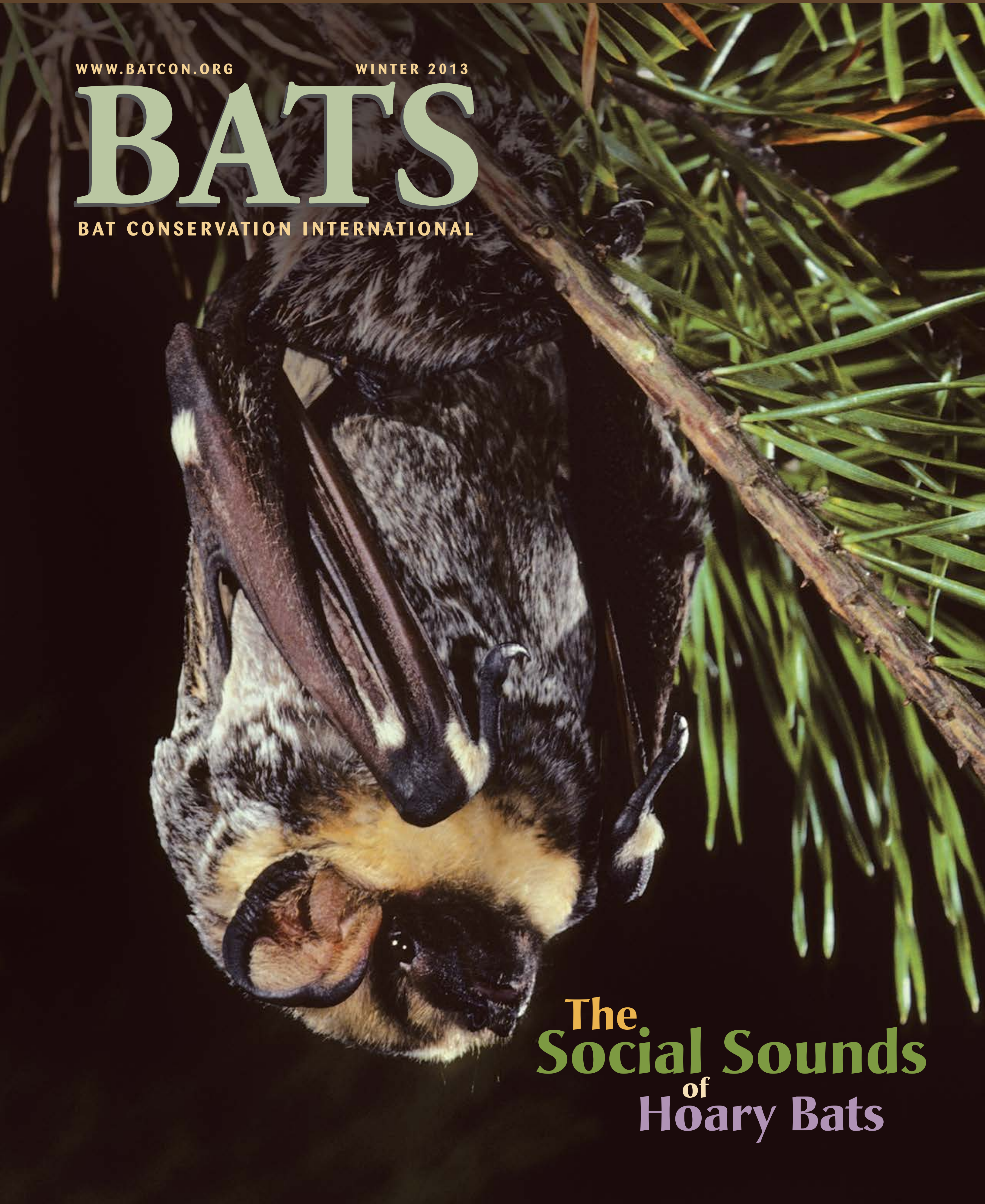

The Social Sounds of Hoary Bats

BATS IN THE BURNS
Studying the impact of wildfires and climate change

COURTESY OF ERIN SAUNDERS

by Carol Chambers and Erin Saunders

Researchers captured bats in mist nets over this pond in a severely burned area of the Apache-Sitgreaves National Forests in Arizona during their study of how bats respond to forest fires.

The Wallow Fire began with an abandoned campfire on the Apache-Sitgreaves National Forests in Arizona's White Mountains on May 29, 2011. By the time it was controlled 40 days later, it had become the largest wildfire in the state's history. Flames blazed across 538,000 acres that range from high-country grasslands to the giant pine forests favored by bats. And bats, like most other wildlife, will likely face more and more charred habitat in the years to come. Thanks to decades of fire suppression and livestock grazing, plus the stirrings of climate change, wildfires are becoming bigger and more frequent throughout the American West.

Our field crew, a half-dozen biologists – plus 50 volunteers from Virginia to California who stepped up to help for a week – spent an intense and arduous summer within the boundaries of that immense fire last summer as part of a study into how bats adapt to a burned-over landscape. We captured bats in mist nets over ponds, attached tiny radio transmitters to reproductive females and tracked them back to often-surprising maternity roosts. We call our research project, a collaboration of Northern Arizona University and the National Forests, "Bats in the Burns," and we hope to expand into other wildfire-burned forests in the Southwest.

Our preliminary evidence suggests that, not surprisingly, bats prefer unburned areas for travel, foraging and drinking. Roost selection was a different story: bats of some species chose roosts in completely charred tree trunks, including some surrounded by burned-over forests.

The forests of the White Mountains range from short-statured piñon pine and juniper woodlands around 5,000 feet (1,500 meters) elevation to subalpine meadows above 9,000 feet (2,750 meters). In between are forests of tall ponderosa pine, quaking aspen, and Douglas-fir trees. During summers, the White Mountains are green, cool and lush with scattered ponds, lakes and streams. At least 10 bat species spend their summers here, roosting in live trees and the dead trees known as snags. Many of them gather by species into maternity colonies to give birth and raise pups.

Previous research has found that bats typically use snags of more than two feet (60 centimeters) in diameter. They roost in vertical cracks in the snags, but will also wedge themselves under patches of loose bark that can house anywhere from one bat to hundreds, depending on the species of bat and the size of the sheltering bark. More than 900 Arizona myotis (*Myotis occultus*) were once counted as they emerged from a single snag.

Wildfires, meanwhile, have been part of forest ecosystems of the southwestern United States for centuries. Until the mid-1800s, lightning-caused fires burned through the ponderosa pine forests every 2 to 20 years. The low flames of those fires burned grasses and shrubs, but moved too fast to kill large pine trees with their thick, fire-resistant bark. That changed when Euro-Americans arrived. Livestock grazing eliminated much of the understory vegetation that had maintained low-intensity fires in the past. Plus, these new settlers considered such fires destructive and eventually began to extinguish them quickly.

Then, in the early twentieth century following a bumper seed crop and a wet year, millions of pine seedlings germinated and, without low-intensity fires to kill many of the tiny seedlings, tree densities increased from tens to thousands per acre. And these now-dense forests are facing yet another stressor in the form of changing climate. The unusually dry summers and winters that the Southwest is now experiencing have changed the way fires burn in forests. Tall flames now reach forest canopies and incinerate whole trees and snags. The decades of accumulated needles and forest litter smolder on the ground, killing old pine trees that would usually survive the fast-moving, pre-settlement fires. Today's forest fires can be so hot they create their own weather and wind patterns: a virtual firestorm. In addition, humans are now one of the leading causes of fires.

The Wallow Fire scorched or incinerated many existing bat-friendly snags. Although new snags were created from trees killed by fire, many were smaller than the size preferred by bats. So the question becomes: would bats accept or reject these blackened snags?

To find out, we captured bats at 20 livestock ponds. Not all the area burned, so we split our efforts among ponds in areas of high severity (at least 75 percent of surrounding landscape burned) or low (25 percent or less). Despite some rainy nights, between mid-June and the end of July, we captured more than 650 bats of 13 species, including the uncommon Allen's big-eared bat (*Idionycteris phyllotis*). The long-legged myotis (*Myotis volans*) was the most common capture, accounting for 25 percent of the total. Arizona myotis, long-eared myotis (*M. evotis*), silver-haired bats (*Lasionycteris noctivagans*) and big brown bats (*Eptesicus fuscus*) rounded out the top five, which represented 83 percent of our captures for the summer.

With long days of driving over rough, rocky and muddy roads, plus rugged hikes into forested ravines, we tracked our radiotagged females back to their roosts. We also occasionally resorted to telemetry flights to locate roosts from the air. In all, we found 19 roosts, including one snag that was shared by an Arizona myotis and a long-legged myotis colony, each of which used a different part of the snag.

More than half the roosts (58 percent) were large ponderosa pine snags, while 21 percent were Douglas-fir, 16 percent quaking aspen and 5 percent white fir. The pine snags averaged 24 inches (62 centimeters) in diameter and the Douglas-fir snags were 17 inches (43 centimeters). The average height of the roost snags was 80 feet (24 meters).

Most of the bats roosted in unburned snags, and bats were mostly captured while foraging and drinking at ponds in habitat relatively untouched by fire. The Arizona myotis and long-legged myotis roosted in unburned snags surrounded by unburned forest. However, four individuals of three species (long-eared myotis, fringed myotis [*M. thysanodes*] and Allen's big-eared bat) used snags that were completely charred – picture a huge, black toothpick. And big brown bats, long-eared myotis, fringed myotis and the single Allen's big-eared bat roosted in the midst of burned-out forest. What causes these species to choose burned or unburned areas for roosting? Perhaps thermal properties of roosts at these high elevations are important. We hope to find out more next summer, when we will be back in the White Mountains to hunt down still more roosts.

This project has been full of surprises, not the least of which is that so many people are willing to volunteer to work at night in remote and challenging terrain. And we were amazed at how bats choose and use roosts in this wildfire-burned area. We were astonished when 70 bats emerged from a completely charred pine snag. We found species segregating the use of snags based on the severity of fire damage in the surrounding landscape. That bats can bear and raise pups at elevations above 8,000 feet (2,400 meters) in such cold temperatures shows how unique and tough these little animals can be.

We will continue our investigation next summer to expand our initial results into how bats are using the Wallow Fire zone. And we hope in the future to explore the remnants of large fires in Arizona and New Mexico. Given the certainty of climate change, it is imperative that we learn how this complex assemblage of bats in the Southwest responds to this transformed habitat.

CAROL CHAMBERS is a Professor of Wildlife Ecology and ERIN SAUNDERS is a Master of Science Candidate in the School of Forestry at Northern Arizona University in Flagstaff.


COURTESY OF ERIN SAUNDERS

Field Assistant Steven Granroth removes a bat from a mist net in a burned forest in Arizona.

Preventing DISASTER

Home Ignitability in the Wildland–Urban Interface

Wildland-urban interface (W-UI) fires are a significant concern for federal, state, and local land management and fire agencies. Research using modeling, experiments, and W-UI case studies indicates that home ignitability during wildland fires depends on the characteristics of the home and its immediate surroundings. These findings have implications for hazard assessment and risk mapping, effective mitigations, and identification of appropriate responsibility for reducing the potential far home lass caused by W-UI fires,

By Jack D. Cohen

Once largely considered a California problem, residential fire losses associated with wildland fires gained national attention in 1985 when 1,400 homes were destroyed nationwide (Laughlin and Page 1987). The wildland fire threat to homes is increasing and is commonly referred to as the wildland—urban interface (W-UI) fire problem. Since 1990, W-UI fires have threatened and destroyed homes in Alaska, Arizona, California, Colorado, Florida, Michigan, New Mexico, New York, and Washington. Extensive or severe fires in Yellowstone in 1988, Oakland in 1991, and Florida in 1998 attracted much media coverage and focused national attention on wildland fire threats to people and property

Federal, state, and local land management and fire agencies must directly and indirectly protect homes from wildfire within and adjacent to wildlands. Davis (1990) indicated that since the mid-1940s, a major population increase has occurred in or adjacent to forests and woodland areas. Increasing residential presence near fire-prone wildlands has prompted agencies to take actions to reduce W-UI fire losses.

When an apparently all-encompassing, seemingly unstoppable W-UI fire occurs, the rapid involvement of many homes over a wide area produces a surreal impression; some homes survive amid the complete destruction of surrounding residences. After the 1993 Laguna Hills fire, some termed this seemingly inexplicable juxtaposition a “miracle.’ Miracles aside, the characteristics of the surviving home and its immediate surroundings greatly influenced its survival.

Wildland fire and home ignition research indicates that a home’s exterior and site characteristics significantly influence its ignitability and thus its chances for survival. Considering home and site characteristics when designing, building, siting, and maintaining a home can reduce W-UI fire losses.

W-UI Fire Loss Characteristics

W-UI residential fire losses differ from typical residential fire losses. Whereas residential fires usually involve one structure with a partial loss, W-UI fires can result in hundreds of totally destroyed homes. Particularly during severe W-UI fires, numerous

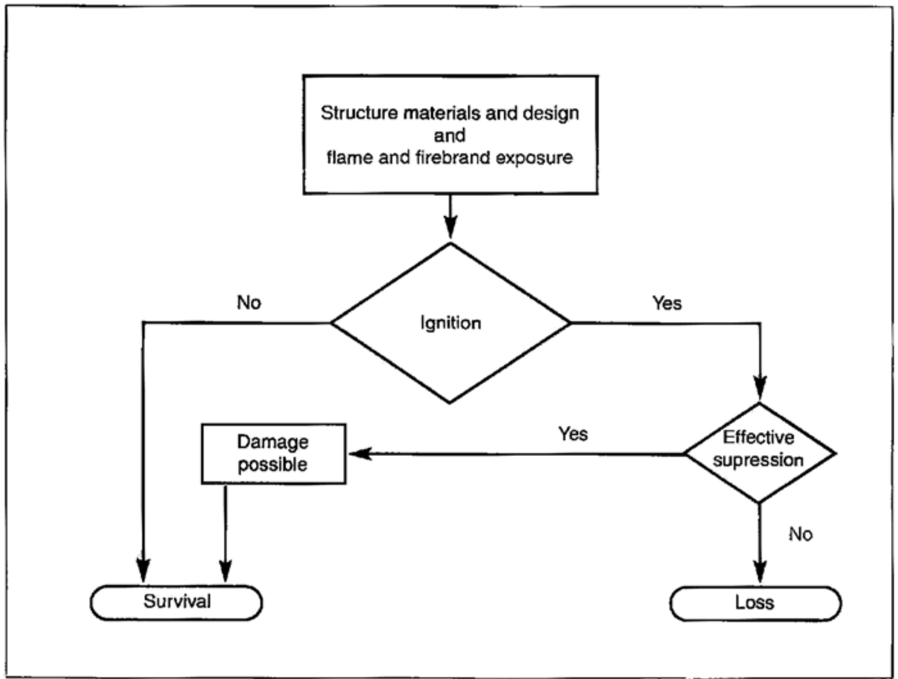

Figure 1. **The structure survival process**

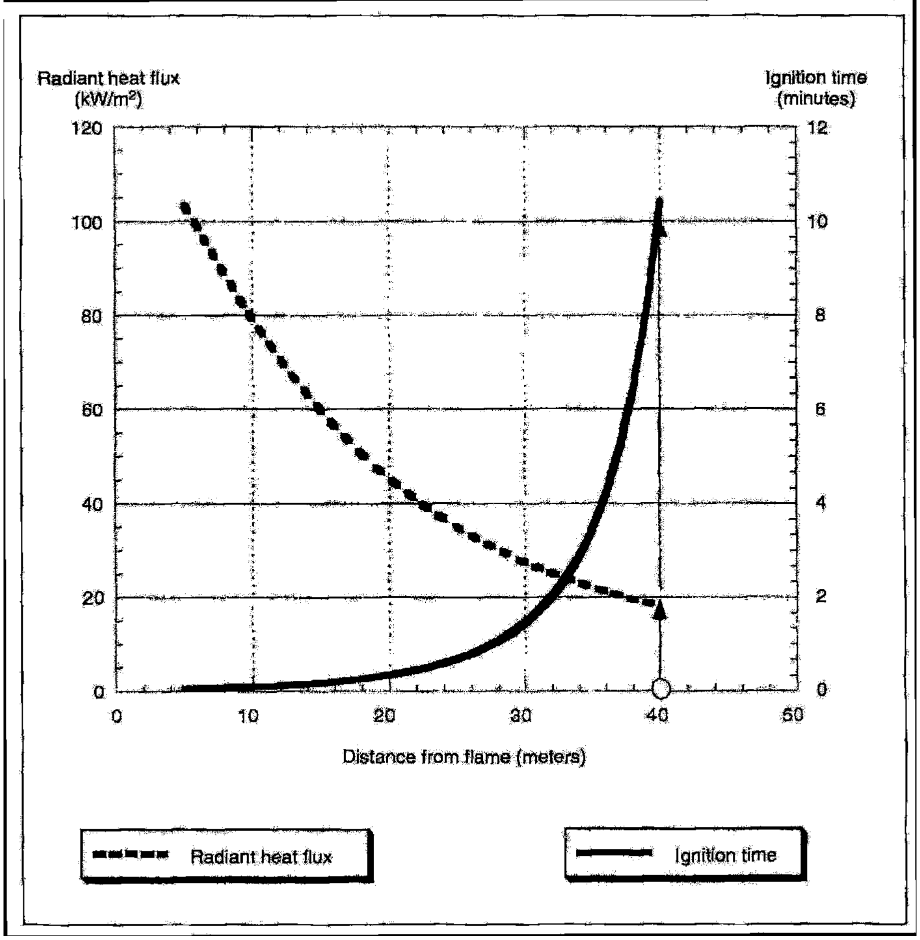

***Figure* 2.The incident radiant heat flux is shown as a function of a wall's distance from a flame 20 meters high by 50 meters wide, uniform, constant, 1,200 K, black-body. The minimum time required for a piloted wood ignition is shown given the corresponding heat flux at that distance.**

homes can ignite in a very short time. The usual result is that a home either survives or is totally destroyed; only a few structures incur partial damage (Foote 1994).

The W-UI Fire commonly originates in wildland fuels. During dry, windy conditions in areas with continuous fine fuels, a wildland fire can spread rapidly, outpacing the initial attack of firefighters. If residences arc nearby, a wildland fire can expose numerous homes to flames and lofted burning embers, or firebrands.

A rapidly spreading wildland fire coupled with highly ignitable homes can cause many homes to burn simul-taneously. This multistructure involvement can overwhelm fire protection Capabilities and, in effect, result in unprotected residences. Severe W-UI fires can destroy whole neighborhoods in a few hours—much faster than the response time and suppression capabilities of even the best—equipped and staffed firefighting agencies. For example, 479 homes were destroyed during the 1990 Painted Cave fire in Santa Barbara, most of them within two hours of the initial fire report. The 1993 Laguna Hills fire in southern California ignited and burned nearly all of the 366 homes destroyed in less than five hours.

Whether a home survives depends initially on whether it ignites; if ignitions with continued burning occur, survival then depends on effective fire suppression. *Figure 1* shows that home survival begins with attention to the factors that influence ignition. These factors determine home ignitability and include the structure's exterior materials and design combined with its exposure to flames and firebrands. The lower the home ignitability the lower the chance of incurring an effective ignition.

Ignition: A local Process

Ignition and spread of fire, whether on structures or in wildland vegetation, is a combustion process. Fire spreads as a continuing ignition process whether from the propagation of flames or from the spot ignitions of firebrands. Unlike a flash flood or an avalanche, in which a mass engulfs objects in its path, fire spreads because the requirements for combustion are satisfied at locations along the path. The basic requirements for combustion—the fire triangle—are fuel, heat, and oxygen. An insufficiency of any one of the three components, which can occur over a relatively short distance, will prevent a specific location from burning. “Green islands” that remain after the passage of a severe, stand-replacement fire demonstrate this phenomenon. Commonly one can find a green, living tree canopy very close to a completely consumed canopy.

The requirements for combustion equally apply to the W-UI fire situation. In the wildland fire context, fire managers commonly refer to vegetation as fuel. However, for the specific context of W-UI residential fire losses, a house becomes the fuel. Heat is supplied by the flames of adjacent burning materials that could include firewood piles, dead and live vegetation, and neighboring structures. Firebrands from upwind fires also supply heat when they collect on a house and adjacent flammable materials. The atmosphere amply supplies the third necessary component, oxygen.

A wildland fire cannot spread to homes unless the homes and their adjacent surroundings meet those combustion requirements. The home ignitability determines whether these requirements are met, regardless of how intensely or fast—spreading distant fires are burning. To use an extreme example, a concrete bunker would not ignite during any wildland fire situation. At the other extreme, some highly ignitable homes have ignited without flames having spread to them. These homes directly ignited from firebrands.

Firebrands are a significant ignition source during W-UI fires, particularly when flammable roofs are involved. Foote (1994) found a significant difference in home survival solely based on roof f1ammability. Homes with nonflammable roofs had a 70 percent survival rate compared with 19 percent for homes with flammable roofs. Davis (1990) reported similar results related to roof flammability.

Reducing W-UI fire losses in the context of home ignitability involves mitigating the fuel and heat components sufficiently to prevent ignitions. However, the question of sufficiency (or efficiency) remains: How much, or perhaps more appropriately, how little fuel and heat reduction must be done to effectively reduce home ignitions? To answer this question, we must first quantify the heat source in terms of the fuel’s ignition requirements; specifically, how close can flames be to a home’s wood exterior before an ignition occurs?

Research Insights

Diverse research approaches are providing clues for assessing the fuel and heat requirements for residential ignitions. Structure ignition modeling, fire experiments, and W-UI fire case studies indicate that the fuel and heat required for home ignitions only involve the structure and its immediate surroundings—the home ignitability context.

Modeling. The Structure Ignition Assessment Model (SIAM) (Cohen 1995) is currently being developed to asses the potential for structure ignitions from flame exposure and firebrands during W-UI fires. One function of SIAM is to calculate the total heat transferred, both radiation and convection, to a structure for varying flame sizes and from varying distances. From the calculated heat transfer, SIAM calculates the amount of heat over time that common exterior wood products can sustain before the occurrence of a piloted ignition (Tran et al. 1992).

Piloted ignition When wood is sufficiently heated, it decomposes to release combustible volatiles. At a sufficient volatile—air mixture, a small flame or hot spark can ignite it to produce flaming; thus, a piloted ignition.

Based on severe-case assumptions of flame radiation and exposure time, SIAM calculations indicate that wildland flame fronts comparable to crowning and torching trees (flames 20 meters high and 50 meters wide) will not ignite wood surfaces at distances greater than 40 meters (Cohen and Butler, in press). *Figure 2* shows the radiant heat a wall would receive from flames depending on its distance from the fire. The incident radiant heat flux, defined as the rate of radiant energy per unit area received at an exposed surface, decreases as the distance increases.

Figure 2 also shows that the time required for ignition depends on the distance to a flame of a given size. At 40 meters the radiant heat transfer is less than 20 kilowatts per square meter *(kW/m^2),* which translates to a minimum piloted ignition time of more than 10 minutes.

1 kW/m^2	Maximum for indefinite skin exposure.
10.4 kW/m^2	Pain after three seconds of skin exposure.
16.0 kW/m^2	Second-degree burn after five seconds of skin exposure.

Source: Drysdale 1985.

Ten minutes, however, is significantly longer than the burning time of wildland flame fronts at a location. Large flames of wildland fires typically depend on fine dead and live vegetation, which limits the intense burning duration at a specific location to less than a few minutes. Recent crown fire experiments have demonstrated a location-specific burning duration of 50 to 70 seconds.

Experiments. Field studies conducted during the International Crown Fire Modelling Experiment (Alexander et al. 1998) provide data for comparisons with SIAM model estimates. Total heat transfer (radiation and convection) and ignition data were obtained from heat flux sensors placed in wooden wall sections.

The instrumented walls were located on flat, cleared terrain at 10, 20, and 30 meters downwind from the edge of the forested plots. The wall section at 10 meters was 2.44 meters wide and 2.44 meters high with a 1.22-meter eave and roof section (*fig. 3a).* Exterior plywood (T-1-11) covered the wall with oriented-strand board covering the roof section and the eave soffit. Trim boards were solid wood with wood fiber composition board on the cave fascia. None of the materials were treated with fire retardant.

(a) 10-meter wood wall section before the crown fire. (b) Experimental crown fire.

Figure 3. International Crown Fire Modelling Experiment.

The forest was variably composed of an overstory of jack pine *(Pinus banksiana)* about 14 meters high with an understory of black spruce *(Picea mariana).* The spreading crown fire produced flames approximately 20 meters high. *Figures 3b and 3c* show examples of the experimental crown fire.

Five burns were conducted where wall sections were exposed to a spreading crown fire. As the crown fires reached the downwind edge of the plot, turbulent flames extended into the clearing beyond the forest edge. In two of the five burns, flames extended beyond 10 meters to make contact with the 10-meter wall section. When flame contact occurred, the 10-meter walls ignited; however, without flame contact, only scorch occurred, as shown in *figure 3d.* The wooden panels at 20 meters experienced light scorch when flames extended beyond 10 meters from the experimental plot, and no scorch from the other burns. The 30-meter wall section had no scorch from any of the crown fires.

Figure 4 displays the average total incident heat flux (radiation and convection combined) corresponding to the wall at 10 meters *(fig. 3d)* and the crown fire shown in figures *3b and 3c.* The average total incident heat flux is calculated from two sensors placed 1 meter apart in the wall. The amount of heat received by the wall increased as the flame front approached and decreased as the fine vegetation was consumed. The initial heat flux "spike" was caused by a nonuniform crowning flame front.

The flux-time integral shown in *figure 4* indicates whether sufficient heating has occurred to pilot-ignite wood (Tran et al. 992). SIAM uses the flux-time integral for calculating ignition potential, a correlation of the incident heat flux and the time required for piloted wood ignition.

The flux-time correlation identifies two principal ignition criteria: (1) A minimum heat flux of 13 kW/m^2 must occur before a piloted ignition can occur for any exposure time, and (2) piloted ignition depends on attaining a critical heating dosage level (heat transfer and its duration). These criteria are graphed in *figure 4.* The flux-time integral only increases for incident heat fluxes greater than the minimum of 13 kW/m^2, and the flux-time integral threshold value of 11,500 is shown as the ignition threshold. As seen in the figure, the flux-time integral does not reach the ignition threshold, indicating an exposure insufficient for ignition and corresponding to no actual occurrence of a wall ignition. Therefore, a home at some distance from a large flame front, such as a crown fire, may not receive sufficient energy to meet the minimum for ignition over any time period. In addition, a home closer to a large flame front can receive a high heat flux (for example, 46 kW/m^2 as shown in *figure 4*), but without the necessary duration to meet the threshold for ignition.

The flux-time integral plot indicates the duration of the heat transfer relevant to ignition. The heat transfer duration relevant to ignition combines the heat transfer from the approaching crown fire plus the burning time of the fire after it has reached the end of the plot. The observed time required for the flux-time integral to increase from zero to its maximum value corresponds to the heat transfer duration significant for ignition. *Figure 4* indicates a duration of 65 seconds (flux-time plot from 75 seconds to 140 seconds).

Case studies. Case studies of actual W-UI fires provide an independent comparison with SIAM and the crown

(c) Experimental crown fire.

(d) After crown fire exposure the wall scorched but did not ignite. Note the lack of wall scorch under the eave because of the radiation "shading" from the eave.

fire experiments. The actual fires incorporate a wide range of fire exposures. The case studies chosen examine significant factors related to home survival for two fires that destroyed hundreds of structures. The Bel Air fire resulted in 484 homes destroyed (Howard et al. 1973) and the Painted Cave fire destroyed 479 homes (Foote 1994).

Analyses of both fires indicate that home ignitions depend on the characteristics of a structure and its immediate surroundings. Howard et al. (1973) observed 86 percent survival for homes with nonflammable roofs and a clearance of 10 meters or more.

Dicussion

A comparison of the SIAM model calculations in *figure 2* with the observed heat flux from the experimental crown fire in *figure 4* indicated that the model overestimates the heat flux. The model calculation at 10 meters reveals a radiant heat flux of 70 kW/m^2, which exceeds the highest total heat flux of 46 kW/m^2 observed

At the 10-meter wall section in *figure 4*. SIAM calculations

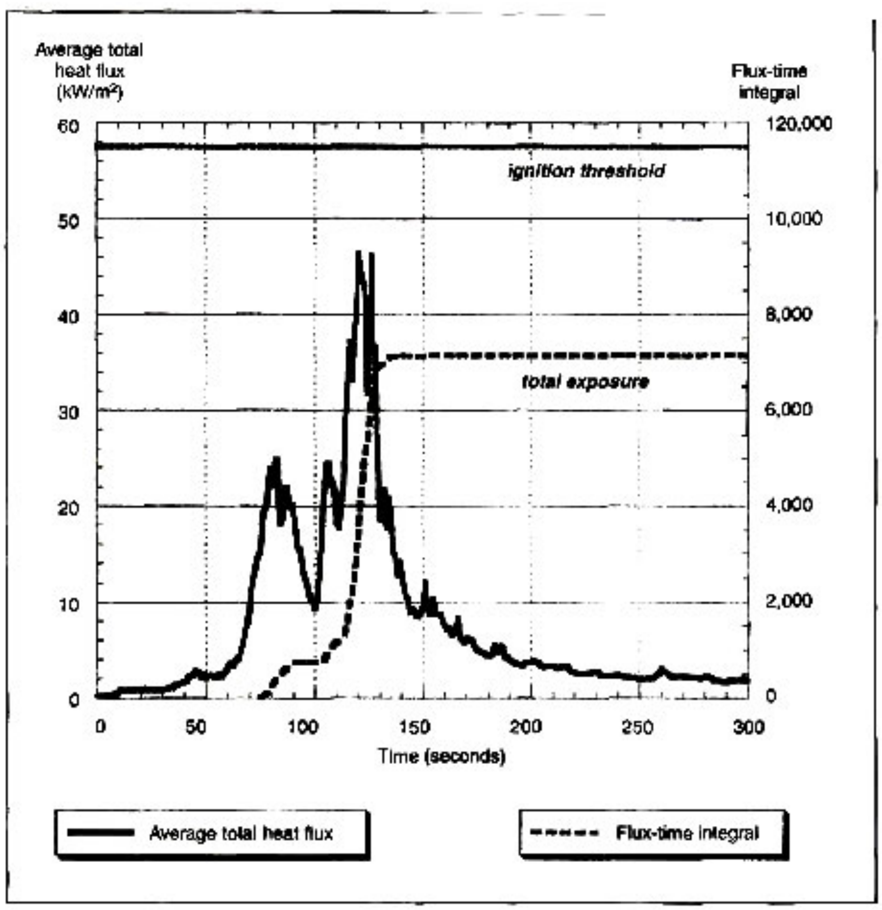

Figure 4. Actual average total incident heat flux and flux-time integral for the crown fire and 10-meter wall section shown in figure 3.

overestimate the heat transfer because the severe-case assumptions designate a homogeneous, black-body radiating flame front. Real flame fronts do not meet these assumptions and produce a significantly smaller radiant heat flux by comparison. For a given flame front, the SIAM calculations represent an extreme-case estimate of radiant heat transfer, and thus an extreme-case estimate of ignition potential.

Given the duration of the experimental heat flux (65 seconds), we can calculate the heat flux and corresponding distance required for ignition. At 65 seconds, the ignition time graph (*fig. 2)* indicates ignition at a flame distance of less than 30 meters. If the heat flux duration is extended by a factor of five to 325 seconds, the flame distance for ignition is less than 40 meters. By comparison, the 10-meter wall sections in the crown fire experiment did not ignite without flame contact and all burns produced little or no scorch to wall sections at 20 and 30 meters. The W-UI fire case studies indicated approximately 90 percent survival with a vegetation clearance on the order of 10 to 20 meters for homes with nonflammable roofs. Thus, the case studies support the general flame-to-structure distance range of 10 to 40 meters as found through modeling and experiments.

However, firebrands can also cause homes to ignite during wildland fires. Although firebrands capable of ignition can originate from a fire several kilometers away, homes can only be threatened if the firebrands ignite the home directly or ignite adjacent flammable materials that then ignite the home.

Analyses of potential home ignitions using modeling, experiments, and case studies did not explicitly address firebrand ignitions. However, firebrand ignitions were implicitly considered because of the firebrand exposures that occurred during the crown fire experiments and the case studies. The experimental crown fires provided a firebrand exposure that resulted in spot ignitions in the dead wood and duff around the wall sections hut not directly on the walls. In the case studies, firebrand ignitions occurred throughout the areas affected by the Bel Air and Painted Cave fires. The high survival rate for homes with nonflammable roofs and 10- to 20-meter vegetation clearances included fire-brands as an ignition factor, thus indicating that firebrand ignitions also depend on the ignition characteristics of the home and the adjacent flammable materials.

Conclusions

The key to reducing W-UI home fire losses is to reduce home ignitability. SIAM modeling, crown fire experiments, and case studies indicate that a home's structural characteristics and its immediate surroundings determine a home's ignition potential in a W-UI fire. Using the model results as guidance with the concurrence of experiments and case studies, we can conclude that home ignitions are nor likely unless flames and firebrand ignitions occur within 40 meters of the structure. This finding indicates that the spatial scale determining home ignitions corresponds more to specific home and community sites than to the landscape scales of wildland fire management. Thus, the W-UI fire loss problem primarily depends on the home and its immediate site.

Consequently if the community or borne site is not considered in reducing W-UI fire losses, extensive wildland fuel reduction will be required. For highly ignitable homes, effective wildland fire actions must riot only prevent fires from burning to home sites, but also eliminate firebrands that would ignite the home and adjacent flammable materials. To eliminate firebrands, wildland fuel reductions would have to prevent firebrand production from wildland fires for a distance of several kilometers away from homes.

Management Implications

Because home ignitability is limited to a home and its immediate surroundings, fire managers can separate the W-UI structure fire loss problem from other landscape-scale fire management issues. The home and its surrounding 40 meters determine home ignitability, home ignitions depend on home ignitability, and fire losses depend on home ignitions. Thus, the W-UI fire loss problem can be defined as a home ignitability issue largely independent of wildland fuel management issues. This conclusion has significant implications for the actions and responsibilities of homeowners and fire agencies, such as defining and locating potential W-UI fire problems (for example, hazard assessment and mapping), identifying appropriate mitigating actions, and determining who must take responsibility for home ignitability

W-UI fire loss potential. Because home ignitions depend on home ignitability, the behavior of wildland fires beyond the home or community site does not necessarily correspond to W-UI home fire loss potential. Homes with low ignitability can survive high-intensity wildland fires, whereas highly ignitable homes can be destroyed during lower-intensity fires.

This conclusion has implications for identifying and mapping W-UI fire problem areas. Applying the term wildland-urban interface to fire losses might suggest that residential fire threat occurs according to a geographic location. In fact, the wildland fire threat to homes is not a function of *where* it happens related to wildlands, but rather to *how* it happens in terms of home ignitability. Therefore, to reliably map the potential for home losses during wildland fires, home ignitability must be the principal mapping characteristic. The home threat information must correspond to the home ignitability spatial scale, that is, those characteristics of a home and its adjacent she within 40 meters.

Home fire loss mitigation. W-UI home losses can be reduced by focusing efforts on homes and their immediate surroundings. At higher densities where neighboring homes may occupy the immediate surroundings, loss reductions may necessarily involve a community. If homes have a sufficiently low home ignitability, a community exposed to a severe wildfire can survive without major fire destruction. Thus, there is a need to examine the reduction of wildland fuel hazard for the specific objective of home protection. There are various land management reasons for conducting wildland vegetation management. However, when considering the use of wildland fuel hazard reduction specifically for protecting homes, an analysis specific to home ignitability should determine the treatment effectiveness.

Responsibility for home ignitability. If no wildfires or prescribed fires occurred, the wildland fire threat to residential development would not exist. However, our understanding of the fire ecology for most of North America indicates that fire exclusion is neither possible nor desirable. Therefore, homeowners who live in and adjacent to the wildland fire environment most take primary responsibility for ensuring that their homes have sufficiently low home ignitability. Homes should not be considered simply as potential victims of wildland fire, but also as potential participants in the continuation of the fire at their location.

A change needs to take place in the relationship between homeowners and the fire services. Instead of home-related presuppression and fire protection responsibilities residing solely with fire agencies, homeowners must take the principal responsibility for ensuring adequately low home ignitability.

The fire services should become a community partner providing homeowners with technical assistance as well as fire response in a strategy of assisted and managed community self-sufficiency (Cohen and Saveland 1997). For this approach to succeed, it must be shared and implemented equally by homeowners and the fire services.

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Assessing the value of roadless areas in a conservation reserve strategy: biodiversity and landscape connectivity in the northern Rockies

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Summary

1. Roadless areas on United States Department of Agriculture (USDA) Forest Service lands hold significant potential for the conservation of native biodiversity and ecosystem processes, primarily because of their size and location. We examined the potential increase in land-cover types, elevation representation and landscape connectivity that inventoried roadless areas would provide in a northern Rockies (USA) conservation reserve strategy, if these roadless areas received full protection.
2. For the northern Rocky Mountain states of Montana, Wyoming and Idaho, USA, we obtained GIS data on land-cover types and a digital elevation model. We calculated the percentage of land-cover types and elevation ranges of current protected areas (wilderness, national parks and national wildlife refuges) and compared these with the percentages calculated for roadless and protected areas combined. Using five landscape metrics and corresponding statistics, we quantified how roadless areas, when assessed with current protected areas, affect three elements of landscape connectivity: area, isolation and aggregation.
3. Roadless areas, when added to existing federal-protected areas in the northern Rockies, increase the representation of virtually all land-cover types, some by more than 100%, and increase the protection of relatively undisturbed lower elevation lands, which are exceedingly rare in the northern Rockies. In fact, roadless areas protect more rare and declining land-cover types, such as aspen, whitebark pine, sagebrush and grassland communities, than existing protected areas.
4. *Synthesis and applications*. Landscape metric results for the three elements of landscape connectivity (area, isolation and aggregation) demonstrate how roadless areas adjacent to protected areas increase connectivity by creating larger and more cohesive protected area 'patches.' Roadless areas enhance overall landscape connectivity by reducing isolation among protected areas and creating a more dispersed conservation reserve network, important for maintaining wide-ranging species movements. We advocate that the USDA Forest Service should retain the Roadless Area Conservation Rule and manage roadless areas as an integral part of the conservation reserve network for the northern Rockies.

Key-words: conservation, elevation zones, land-cover types, landscape metrics, national forests, reserve design

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Introduction

A growing body of scientific evidence indicates that the current USA system of federal protected areas (designated wilderness areas, national parks and national wildlife refuges) may be too small and disconnected to protect against the decline and loss of native species diversity or to accommodate large natural ecosystem processes (Wright, Dixon & Thompson 1933; MacArthur & Wilson 1967; White 1987; Wilcove 1989; Baker 1992; Turner *et al*. 1993; Noss & Cooperrider 1994; Reice 1994; Newmark 1995; Sinclair *et al*. 1995; Soule & Terborgh 1999). Expanding road networks, human settlements, resource extraction and other encroachments on the landscape have increased the fragmentation and loss of natural areas. Such disturbances have isolated many protected areas, causing them to function as terrestrial 'islands' surrounded by a matrix of lower quality altered lands (Harris 1984; Pickett & White 1985; Wiens, Crawford & Gosz 1985; Turner 1989; Saunders, Hobbs & Margules 1991). The long-term persistence of many species within protected areas is dependent on the degree of human activities and land-use practices on lands adjacent to and near protected areas. There is a need to identify relatively undisturbed lands located outside protected areas that may increase the potential of protected areas in maintaining native biodiversity and certain ecological processes, and to include these lands within the conservation reserve system before they are lost or altered.

Inventoried roadless areas, large tracts of relatively undisturbed land on USA Forest Service lands, are often left out of landscape assessments for identifying functional conservation reserves. Only two studies (DeVelice & Martin 2001; Strittholt & DellaSala 2001) have analysed the contribution that roadless areas make to the current protected areas reserve network. However, more than one-third of inventoried roadless areas on national forests are adjacent to protected areas (DeVelice & Martin 2001). They hold the potential to increase the size and connectivity of designated wilderness areas, national parks and national wildlife refuges, thus increasing the ability of protected areas to maintain natural landscape dynamics and native species population viability over the long term. Smaller, isolated roadless areas are also important because they may contain rare species, capture more habitat variation, including underrepresented habitat types, and may function as 'stepping stones' that connect current protected areas across a landscape (Shafer 1995; Strittholt & DellaSala 2001).

There is a precedent for the protection of national forest roadless areas. The USA Congress has designated as wilderness more than half, 6 million ha, of roadless areas that the Forest Service inventoried in national forests in the 1970s. In 1998, the Forest Service began to devise regulations aimed at protection of roadless area characteristics in national forests. In May 2000, the agency released its proposed rule, familiarly known as the Roadless Rule, and draft environmental impact statement. Eight months later, the Forest Service adopted the rule. In July 2004, the Forest Service proposed to repeal the Roadless Rule and replace it with a state petition and rule-making process, which would offer less protection by presumably opening national roadless areas to all forest service activities and requiring state governors to 'opt in' Roadless Rule protections affirmatively for any roadless area.

Included in the Roadless Rule environmental impact statement was an evaluation of the potential contribution that protection of roadless areas could make to the conservation of biodiversity at a national scale (USDA Forest Service 2000b). In that evaluation, DeVelice & Martin (2001) found that the inclusion of roadless areas in the network of federal protected areas would expand representation of ecoregions in protected areas, increase the acreage of reserved areas at lower elevations, and increase the number of areas large enough to provide refuge for wide-ranging species.

Strittholt & DellaSala (2001) focused on similar questions at a regional scale for the Klamath-Sikiyou area in southern Oregon and northern California, USA. They found that roadless areas protect a wide range of ecological attributes, especially at mid- to lower elevations, important in this region. They also concluded that roadless areas increase the connectivity among ecoregions.

The northern Rocky Mountain states of Montana, Wyoming and Idaho comprise a region particularly rich in roadless areas, roughly 2·6 million ha, providing a unique opportunity to create a relatively intact reserve design that captures important elements of conservation for the northern Rockies. Using two key concepts in conservation biology, biodiversity representation and landscape connectivity, we investigated the potential contributions of national forest roadless areas to the protected areas reserve network across the northern Rocky Mountain region.

DIVERSITY REPRESENTATION

An important goal in the design and establishment of conservation reserves is to represent a full range of native biodiversity (Shelford 1926; Margules, Nicholls & Pressey 1988; Church, Stoms & Davis 1996; Possingham, Ball & Andelman 2000). Even though this goal has been articulated for some time, most protected areas are demarcated around areas with high scenic and recreational attributes (Davis *et al*. 1996). As a result, existing protected areas in the northern Rockies are, for the most part, concentrated at higher elevations, where other important elements of biodiversity are most likely to be poorly represented (Scott *et al*. 2001).

Representation of a full range of biodiversity in reserves requires an understanding of all species and ecosystem processes operating within a given landscape. However, many researchers have used ecological communities and elevation ranges as coarse-scale surrogates for native biodiversity in the design of conservation reserves (Scott *et al*. 1993; Host *et al*. 1996). This concept is based on the idea that if a full range of ecological communities and elevation ranges is protected, it is more likely that many ecological communities, wide-ranging species and ecosystem processes will be maintained in the reserves. In the northern Rockies, ecological communities are often associated with elevation gradients (Hansen & Rotella 1999). Hence, roadless areas situated at middle and lower elevations may make valuable contributions in protecting many elements of biodiversity that are currently not well represented in protected areas (DeVelice & Martin 2001).

LANDSCAPE CONNECTIVITY

Connectivity refers to the degree to which the structure of a landscape helps or hinders the movement of wildlife species or natural processes such as fire (Wiens, Crawford & Gosz 1985; Turner *et al*. 1993; Noss & Cooperrider 1994; Bascompte & Solé 1996; With 1999). A 'well-connected' area can sustain important elements of ecosystem integrity, namely the ability of species to move and natural processes to function, and is more likely to maintain its overall integrity compared with a highly fragmented area.

Roads are highlighted in the scientific literature as major causes of landscape fragmentation, and function as barriers to organism movements, resulting in a reduction of overall landscape connectivity for many native species. The effects of roads are broad and include mortality from collisions, modification of animal behaviour, disruption of the physical environment, alteration of chemical environments, spread of exotic and invasive species, habitat loss, increase in edge effects, interference with wildlife life-history functions and degradation of aquatic habitats through alteration of stream banks and increased sediment loads (Franklin & Forman 1987; Andrews 1990; Noss & Cooperrider 1994; Reice 1994; Reed, Johnson-Barnard & Baker 1996; Trombulak & Frissell 2000; McGarigal *et al*. 2001). Thus, the addition of roadless areas to existing protected areas reserve is likely to maintain or increase landscape connectivity, as well as increase the integrity of protected areas.

With the advent of landscape metrics, it is now possible to quantify connectivity for landscapes, land-cover types, species' habitats, species' movements and ecosystem processes across a given region (O'Neill *et al*. 1988; McGarigal & Marks 1995; Gustafson 1998; With 1999). Many different metrics that quantify spatial characteristics of patches or entire landscape mosaics have been described (Turner & Gardner 1991; McGarigal & Marks 1995; Ritters *et al*. 1995; Hargis, Bisonette & David 1998; Dale 2000; Jaeger 2000; McGarigal & Holmes 2002). We chose metrics that measure three elements of landscape connectivity: area, isolation and aggregation.

Area

It is known that larger areas (patches) generally contain more species, more individuals, more species with large home ranges and/or sensitive to human activity, and more intact ecosystem processes than smaller areas (Robbins, Dawson & Dowell 1989; Turner *et al*. 1993; Newmark 1995; Shafer 1995). Higher numbers of patches will usually contribute to greater resilience of populations and may also increase the utility of patches that act as 'stepping stones' or connectors across a landscape (Buechner 1989; Lamberson *et al*. 1992).

Isolation

The distance between patches plays an important role in many ecological processes. Studies have shown that patch isolation is the reason that fragmented habitats often contain fewer bird and mammal species than contiguous habitats (Murphy & Noon 1992; Reed, Johnson-Barnard & Baker 1996; Beauvais 2000; Hansen & Rotella 2000). As habitat is lost or fragmented, residual habitat patches become smaller and more isolated from each other, species movement is disrupted, and individual species and local populations become isolated (Shinneman & Baker 2000).

Aggregation

The spatial arrangement of patches may help to explain how certain species are found in patches located close together and are not found in patches that are more isolated, or vice versa (Ritters *et al*. 1995; He, DeZonia, & Mladenoff 2000). This concept generally follows the ideas developed in island biogeography theory (MacArthur & Wilson 1967) and metapopulation theory (Levins 1969, 1970).

For some species or natural processes, the isolation or aggregation of patches across the landscape may be more important, for others, area may be the key element. Together, these three elements offer a comprehensive assessment of the importance of roadless areas to the maintenance of overall landscape connectivity and ecosystem integrity of current protected areas in the northern Rockies.

In this study, we aimed to assess the extent to which roadless areas increase biodiversity representation and landscape connectivity when they are included in the protected areas reserve network for the northern Rockies.

Methods

STUDY AREA

Of the 84 million ha of land that stretch across Montana, Wyoming and Idaho in the USA, roadless areas cover 2·6 million ha and existing federal protected areas (wilderness areas, national parks, special management areas and national wildlife refuges) protect almost 8·7 million ha. Within this region, three large, relatively

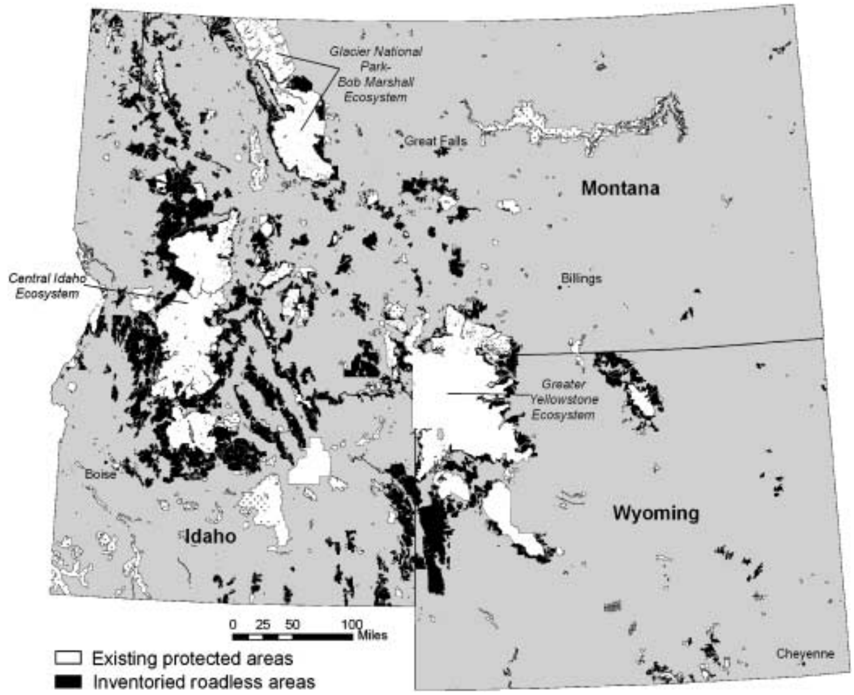

Fig. 1. Roadless areas and protected areas across the states of Idaho, Montana and Wyoming, USA.

undisturbed, mountain ecosystems are delineated around national parks and/or wilderness complexes. These are the Greater Yellowstone Ecosystem, Glacier National Park–Bob Marshall Ecosystem, and the Central Idaho Ecosystem (Fig. 1).

The topography of the northern Rocky Mountain states spans steep physical gradients in elevation, slope, aspect, temperature and precipitation that give rise to diverse vegetation types. Elevations range from 150 m to 4200 m. Average precipitation ranges from 28 cm to 51 cm (Franklin 1983). The northern Rockies comprise a variety of non-forested and coniferous forest types. Low-lying valleys are characterized by grasslands, sagebrush (*Artemisia* spp.) and desert shrublands, interspersed with juniper (*Juniperus* spp.) and riparian woodlands. Ponderosa pine *Pinus ponderosa* dominates lower elevation montane forests, while xeric coniferous forests of mainly Douglas fir *Psuedotsuga mensiezia*, ponderosa pine, grand fir *Abies grandis*, lodgepole pine *Pinus contorta* and aspen *Populus tremuloides* occur at mid-elevations. Mesic forests in the north and west largely contain western larch *Larix occidentalis*, grand fir, western red cedar *Thuja plicata* and mountain hemlock *Tsuga mertensiana*. Higher elevations are composed of Engelmann spruce *Picea engelmannii*, subalpine fir *Abies lasiocarpa*, alpine larch *Larix lyalli* and whitebark pine *Pinus albicaulis* intermixed with subalpine meadows. Herb lands, rock, alder *Alnus sinuata* shrubfields and snowfields/ice occur at the highest elevations.

DATA COLLECTION

We used a land management status GIS coverage and classification system developed by the USA Geological Survey's Biological Resources Division in its nationwide GAP Analysis Programme (Scott, Tear & Davis 1996) to delineate 'protected areas'. This programme devised a ranking scheme to represent various levels of protection, ranging from the least protected lands (category 4, e.g. private lands) to those with the highest level of protection (category 1, e.g. wilderness areas) for all public lands in the GIS spatial database. For this study, we assumed that categories 1 and 2 represent adequate protection as their primary management objective is conservation (Scott, Tear & Davis 1996), and selected these categories as our protected areas on all forest service lands located in the three states.

We used the federal inventoried roadless areas GIS database (USDA Forest Service 2000a). This includes areas that are greater than 2000 ha in size, where road building is prohibited under current National Forest Plan decisions and where road building is presently allowed. We recognize that our decision leaves out smaller roadless areas that were not considered during the inventory of federal roadless areas and that these areas serve important conservation goals (Strittholt & DellaSala 2001). For this study, the term 'roadless areas' refers to inventoried roadless areas.

We used three independently derived land cover maps for Montana, Wyoming and Idaho from the GAP Analysis Programme (Scott, Tear & Davis 1996). The Montana and Idaho GAP products were produced based on classification techniques by Redmond *et al.* (1998) for raw Landsat Thematic Mapper (TM) satellite imagery. Spatial resolution of the grid was 90 m for Montana and 30 m for Idaho. The Wyoming GAP Analysis Programme digitized land cover data in a vector format from Landsat TM satellite imagery at a scale of 1 : 100 000 (Gap Analysis Wyoming 1996). We converted Wyoming's vector map into a grid format and resampled the three data sets to 90-m resolution. Then we merged the three land cover maps into a single image and a common land cover classification scheme (Appendix 1).

Similar to most GIS databases, errors are associated with the land management status, inventoried roadless areas and land-cover grids. These grids represent a composite of data from many sources and include variations in mapping procedures and possible misclassifications that could potentially cause inconsistencies that are difficult to detect. However, we believe, based on professional judgement, that the error rate is not large enough to affect conclusions drawn from this large regional-scale analysis.

To investigate the representation of roadless areas at various elevation classes, we downloaded a digital elevation model from the 30-m National Elevation Dataset produced by the USA Geological Survey's EROS Data Center (Sioux Falls, SD). We reclassified the elevation range into 21 equal-interval classes ranging in 200-m increments from approximately 150 m to 4200 m.

DATA ANALYSIS

All data analyses were conducted in ARC/INFO and ArcView GIS software from Environmental Systems Research Institute (Redlands, CA).

Land cover representation

Using ARC/INFO, we combined the protected areas database with the land cover map. To calculate the percentage representation of each land-cover type, we divided the protected portion of each land-cover type by the total area of each land-cover type across the study area. Next, we appended the national forest inventoried roadless areas to the existing protected areas and repeated the same calculation described above to measure the additional representation of each land-cover type because of the inclusion of roadless areas. In addition, we calculated the percentage increase between each land cover percentage representation for protected areas alone and protected areas and roadless areas combined. This measure quantified the 'relative' ecological contribution from roadless areas for each land-cover type. We then ranked these land-cover types according to the level of representation within the existing protected areas.

Elevation representation

Using ARC/INFO, we combined the protected areas database with the 30-m digital elevation model. Similar to the procedure for land-cover types described above, we added the roadless areas to the existing protected areas, intersected this image with the elevation data, and calculated the change in representation for each elevation class provided by protection of roadless areas.

To examine the potential increase of landscape connectivity caused by roadless areas, we used ARC/INFO and FRAGSTATS (McGarigal & Marks 1995; McGarigal & Holmes 2002), a computer program developed to quantify heterogeneity of the landscape. We identified five landscape metrics available in FRAGSTATS to assess our three elements of landscape connectivity (McGarigal & Holmes 2002). To assess area, we used the metrics percentage land (PLAND), number of patches (NP) and patch size (AREA). We included the metrics NP and AREA to help explain the context of an increase in PLAND. For example, an increase in PLAND and AREA and a decrease in NP would indicate that the added roadless patches were located next to existing conservation patches, resulting in an increase in the size of patches and a decrease in the number of patches across the landscape. Conversely, a decrease in AREA and an increase in NP would indicate that the added patches were generally smaller and did not combine with existing patches.

To assess isolation we used nearest neighbour distance (ENN). A decrease or increase in ENN would indicate that patches are either located closer together or farther apart, respectively, across the landscape.

To assess aggregation, we used contagion (CONTAG). An increase in CONTAG would indicate that patches are, to a certain extent, aggregated together across the landscape.

Using FRAGSTATS, we selected and ran our five landscape metrics on the two grids described above (current protected areas only, and roadless areas and current protected areas combined). Each grid was a binary map where all grid cells that comprised the 'protected' and 'roadless' patches were classified as 1 and all other 'non-protected' grid cells were masked out as background (–99). For each landscape metric, we computed the mean, area-weighted mean and coefficient of variation where applicable. We then compared the differences in metrics between the two grids. In addition, differences in the mean, area-weighted mean and coefficient of variation helped to explain how the range of values for each metric were distributed when existing protected areas were compared with the conservation system including roadless areas.

Results

LAND COVER REPRESENTATION

In existing protected areas, burned forest and snowfields/ice had the highest land cover representation, 88% and 86%, respectively. Representation of other land-cover types, such as alpine meadows, whitebark pine, exposed rock/soil, subalpine meadows, wetlands, mixed subalpine forest and lodgepole pine, ranged from 31% to 71%.

The inclusion of roadless areas increased the representation of all land-cover types except for one, sand dunes (Table 1). Relative percentage increases ranged

Table 1. Additional representation and percentage increase in representation of each land-cover type across the northern Rockies when national forest roadless areas are added to existing protected areas

Land-cover type	Existing level of representation (%)	Potential level of representation including roadless areas (%)	Percentage increase including roadless areas
Burned forest	88·12	93·09	5·65
Snowfields/ice	86·12	97·48	13·19
Alpine meadow	71·51	94·18	31·70
Mixed whitebark pine	59·62	84·94	42·46
Exposed rock/soil	44·67	59·92	34·12
Subalpine meadow	40·49	68·85	70·05
Wetlands	37·34	38·68	3·61
Mixed subalpine forest	32·20	68·63	113·11
Lodgepole pine	31·35	59·42	89·54
Mixed barren lands	21·66	22·61	4·37
Sand dunes	18·44	18·44	0·00
Mixed conifer	16·97	37·24	119·44
Mesic upland shrub	10·74	26·14	143·44
Shrub-dominated riparian	7·98	12·77	59·91
Forest-dominated riparian	7·18	12·14	69·11
Sagebrush	6·33	9·91	56·55
Juniper	5·87	6·80	15·95
Xeric upland shrub	5·85	7·97	36·33
Vegetated sand dunes	5·69	6·03	5·89
Western red cedar	5·57	22·00	295·08
Mud flats	5·33	7·39	38·79
Ponderosa pine	4·94	9·88	99·97
Aspen	4·48	25·99	479·80
Shrub–grassland associations	4·25	5·89	38·46
Western hemlock	3·36	23·62	602·54
Grasslands	2·49	3·64	46·31
Grass-dominated riparian	2·15	3·07	43·01
Salt-desert shrub flats	1·58	1·71	8·63
Bur oak woodland	0·00	2·40	NA

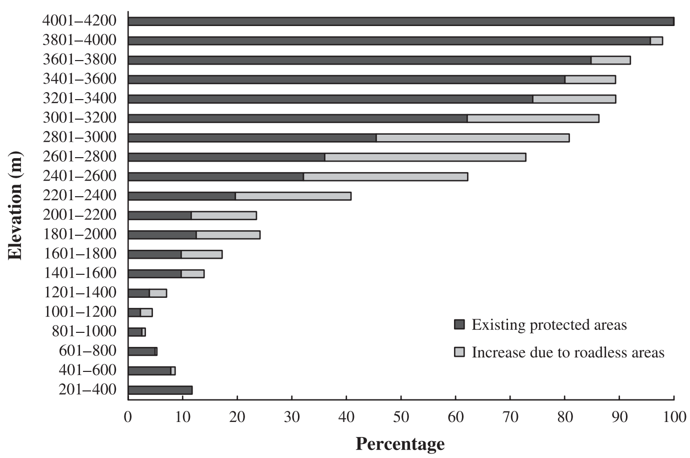

Fig. 2. Additional representation of elevation ranges resulting from the inclusion of roadless areas with protected areas for the northern Rockies. The *x*-axis represents elevation in 200-m increments and the *y*-axis shows absolute increase in percentage representation when roadless areas are added to protected areas. Black bars represent protected areas and grey bars represent roadless areas.

from 5% to 600%. Fifteen land-cover types increased by more than 40%, among them important ecological communities, western hemlock, aspen, ponderosa pine, western red cedar and sagebrush, each of which has less than 10% representation in current protected areas. Moreover, the addition of roadless areas represented one land-cover type, bur oak *Quercus macrocarpa* woodland, not present in protected areas.

ELEVATION REPRESENTATION

Our elevation analyses showed that elevations in the range of 2200–4200 m were well represented in protected areas (Fig. 2). The addition of roadless areas resulted in a large increase in representation of lands at elevations ranging from 1000 m to approximately 3400 m. For elevation ranges below 1000 m and above 3400 m, the

Table 2. Landscape metrics comparing the spatial pattern of protected areas alone with a scenario that includes protected areas and national forest roadless areas combined for the northern Rockies. + and – indicate an increase or decrease, respectively, in the metric value caused by the addition of roadless areas

Landscape Metrics	Protected areas	Protected and roadless areas	+/ –
Area			
Class area (ha)	8 814 900	15 673 600	+
Percentage land	9	16	+
Number of patches	770	722	–
Patch size (mean, ha)	11 447·92	21 708·59	+
Patch size (area-weighted mean)	1 105 055·78	2 505 909·11	+
Patch size (coefficient of variation)	977·39	1 069·74	+
Isolation			
Nearest neighbour (m)	7 013·72	5 353·11	–
Nearest neighbour (area-weighted mean)	3 153·73	2 518·75	–
Nearest neighbour (coefficient of variation)	122·47	134·16	+
Aggregation			
Contagion index	72·56	58·64	–

contribution of roadless areas was small. However, the proportion of area represented at lower elevations increased when we included roadless areas with protected areas.

CONNECTIVITY

Results from the landscape metrics showed that the addition of roadless areas increased regional connectivity for all three connectivity elements (Table 2). Area metrics demonstrated that the addition of roadless areas almost doubled the amount of area protected, rising from 9% to 16%, and the mean patch size in protected areas changed from 11448 ha to 21709 ha. The number of patches decreased from 770 to 722. Area-weighted mean patch size increases and the patch size coefficient of variation increased from 977 to 1070. Isolation metrics showed a decrease in the mean and area-weighted mean nearest-neighbour metrics when roadless areas were added. The mean distance between nearest protected patches decreased from 7014 m to 5353 m. The decrease in the area-weighted mean was less than the overall mean when patches of all sizes were considered. The coefficient of variation also increased for this metric. The aggregation metric (contagion) decreased from 72·56 to 58·64 when roadless areas were included, signifying more dispersion of patches across the landscape.

Discussion

BIODIVERSITY REPRESENTATION

A review of the literature suggests that a given vegetation community is adequately represented when 12–25% of it is included in a conservation area (World Commission on Environment & Development 1987; Noss & Cooperrider 1994), although it is not certain that these thresholds are truly adequate to protect vegetation communities. Based on this range, we define land-cover types above 25% as adequately protected, land-cover types within the range of 12–25% as minimally protected, and those below 12% as underrepresented, similar to DeVelice & Martin (2001).

Our results show that roadless areas make a substantial contribution in maintaining regional biodiversity. One of our most important findings is that roadless areas would protect a wider range of land-cover types and elevation ranges than protected areas alone, especially those characteristic of mid- to low elevations that are underrepresented in protected areas. These lands are among the last remnants of biologically productive lands that have not been significantly altered through human settlements, resource extraction and road construction (Scott *et al*. 2001; Strittholt & DellaSala 2001). We also found that protected areas adequately represent land-cover types that are characteristic of higher elevations. This finding supports the generally accepted notion that wilderness areas and national parks mainly protect higher elevation ecological communities (Davis *et al*. 1996; Possingham, Ball & Andelman 2000). Contrary to DeVelice & Martin (2001), whose study found that roadless areas mainly occurred at mid- to lower elevations, but similar to Strittholt & DellaSala (2001), we found that roadless areas considerably increase the protection of higher elevations and corresponding cover types as well. The different results are probably because of the scale at which the studies were implemented. DeVelice & Martin's (2001) study included all roadless areas across the nation, incorporating a wide range of elevations from sea level to the highest peaks. Our study, and that of Strittholt & DellaSala (2001), focused on smaller regions at higher elevations.

Across the northern Rockies region (Montana, Wyoming and Idaho), protected areas adequately represent nine land-cover types, whereas five biologically important land-cover types, western hemlock, aspen, ponderosa pine, western red cedar and mesic upland shrub, are underrepresented in protected areas. However, the addition of roadless areas increases representation of two cover types (western hemlock and western red cedar) to the minimally protected threshold and two cover types (aspen and mesic upland shrub) to the adequately represented threshold (greater than 25%). Ponderosa pine, even though it increases by nearly 100%, remains underrepresented. Overall, the magnitude of the increased representation, from 100% to 600%, indicates that roadless areas can make substantial contributions to the protection of land-cover types that are not well represented in protected areas.

Increased representation of certain rare ecological communities is particularly important in a northern Rockies conservation strategy. Aspen, for example, is thought to be declining in the northern Rockies (Gallent *et al*. 1998). When roadless areas are added to protected areas, aspen moves up two full categories: from underrepresented to adequately represented, a 480% increase in representation for this forest type, on which many avian species depend upon (Hansen & Rotella 2000). Representation of whitebark pine changes from 60% to 85% when roadless areas are added. Whitebark pine is declining throughout North America due to blister rust *Cronartium ribicola*, an introduced disease, and is a 'keystone species' important for many higher elevation species (Keane, Morgan & Menakis 1994).

Elevation representation results demonstrate that protected areas are mainly located at higher elevations. We also found that roadless areas are generally concentrated at mid- to high elevations and represent a wider range of elevations, especially low- to mid elevations, than protected areas. However, our results show that protected areas encompass more lower elevation lands than roadless areas. This situation is somewhat deceiving. Representation of lower elevations in protected areas is largely a result of two well-placed low-elevation conservation areas: Hell's Canyon National Recreation Area and Missouri Breaks National Monument. In fact, low-elevation lands below 1000 m are not well represented in either protected areas or roadless areas. As a majority of lower elevation lands in the northern Rockies have been converted to other uses, it is of utmost importance to increase representation of lower elevation sites in protected areas (Strittholt & DellaSala 2001). Protection of these lower elevation roadless areas would contribute greatly to the conservation of lower elevation species and ecological communities that are poorly represented in protected areas.

Landscape connectivity

Our analyses of three elements of connectivity show that roadless areas increase connectivity across the northern Rockies, and increase both the area and size of protected area patches. In addition, the number of protected area patches decreases with the addition of roadless areas because they combine with protected areas to form one larger patch. Larger patches will protect more species and more individuals, species with large home ranges, species sensitive to human activity, and more intact ecosystem processes than smaller areas (Askins, Philbrick & Sugeno 1987; Robbins, Dawson & Dowell 1989; Turner *et al*. 1993; Newmark 1995; Shafer 1995). Roadless areas also reduce the distance between protected areas and create a more evenly dispersed reserve system, critical for maintaining many species' movements and a large distribution of local populations (MacArthur & Wilson 1967; Murphy & Noon 1992; Reed, Johnson-Barnard & Baker 1996; Ritters *et al*. 1996; Beauvais 2000; Hansen & Rotella 2000; He, DeZonia, & Mladenoff 2000; Shinneman & Baker 2000).

Our results show an increase in the coefficient of variation for patch size and isolation metrics, which may be an important consideration in delineating conservation reserve systems capable of maintaining movements of various species and ecological processes (Wiens & Milne 1989; Wilcove & Murphy 1991; Noss 1992; Noss *et al*. 1996; O'Neill *et al*. 1996). Smaller patches may supplement larger reserves by protecting rare species that occur only in certain areas (Franklin & Forman 1987; Hansen *et al*. 1991; Shafer 1995). The dispersion of roadless areas may also contribute to greater resilience or survival of island populations by allowing a greater chance for species exchange, essentially maintaining a metapopulation or source–sink population structure (Wiens, Crawford & Gosz 1985; Pullium 1988; Gilpin & Hanski 1991; Murphy & Noon 1992). Many studies are investigating how species move through landscapes and their use of stepping-stone habitats, especially in fragmented landscapes (Freemark *et al*. 1993; With 1999; Beauvais 2000; Hansen & Rotella 2000; Holloway, Griffiths & Richardson 2003; Johnson, Seip & Boyce 2004). Being relatively undisturbed and well-distributed among protected areas, roadless areas are top candidates for the delineation of high-quality 'habitat connections' across the northern Rockies, particularly those that target rare or declining species. The loss or alteration of roadless areas may further reduce the movement of species among interdependent island populations located in protected areas and roadless areas, resulting in greater isolation.

Moreover, the addition of roadless areas increases the effective size of the three largest wilderness and national park complexes in the northern Rockies: the Greater Yellowstone Ecosystem, the Glacier National Park–Bob Marshall Ecosystem and the Central Idaho Ecosystem, where management challenges include maintaining large-scale ecological processes such as species' movements and natural fire across jurisdictional boundaries (Pickett & White 1985; Turner *et al*. 1993). Roadless areas not immediately adjacent to these complexes are dispersed in the surrounding landscape, which helps to decrease the degree of isolation between the complexes and possibly allows for species movement among these ecosystems.

Management implications

Using research to guide reserve design and develop land protection policies is the strongest approach in conservation. The importance of intact, functioning natural ecosystems to the maintenance of native biodiversity and ecological processes is unquestioned (Wright, Dixon & Thompson 1933; MacArthur & Wilson 1967; Usher 1987; White 1987; Shafer 1995; Noss, O'Connell & Murphy 1997). The negative impacts of roads in natural areas are well known (Andrews 1990; Foreman & Wolke 1992; Reed, Johnson-Barnard & Baker 1996; Spellerberg 1998; Trombulak & Frissell 2000; McGarigal *et al*. 2001). Our landscape assessment demonstrates how roadless areas, the remaining relatively undisturbed forested lands in the northern Rockies, are essential for maintaining biodiversity and landscape connectivity in a conservation reserve strategy for this area. This has direct bearing on management decisions regarding the protection of roadless areas in this region. Our results, along with the findings of DeVelice & Martin (2001) and Strittholt & DellaSala (2001), highlight the important role of roadless areas in USA conservation efforts and contribute to the larger policy dialogue surrounding roadless areas.

The methods used in this study can help land managers determine appropriate guidelines to identify and assess roadless areas that are critical in maintaining regional biodiversity, ecosystem processes, landscape connectivity and overall intact ecosystem integrity. Land managers should avoid activities such as road building, logging, spread of exotic species, off-road vehicle use and exurban development in roadless areas that would result in their degradation or loss. If roadless areas are not protected from these activities as a matter of priority, it is possible that their potential contribution to conservation effort in the future will be diminished and existing protected areas surrounded by or in close proximity to roadless areas will be negatively affected as well. We recommend that roadless areas receive full protection and are managed responsibly, so that they can function as an important part of the current conservation reserve system in the USA.

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Supplementary material

The following material is available from http://www.blackwellpublishing.com/products/journals/suppmat/JPE/JPE996/JPE996sm.htm.

Appendix 1. Land-cover types across the northern Rocky Mountain region reclassified from USA Geological Survey's Biological GAP Analysis Programme (Scott, Tear & Davis 1996).

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ERRATA

The preceding paper unfortunately was published with several minor errors that are corrected below:

Page 181: Gregory H. Aplet's address is The Wilderness Society, 1660 Wynkoop Street, Ste. 850, Denver, CO 80202, USA.

"Forest Service" should be capitalized throughout.

Page 183, col. 2: The first sentence of the Study Area section should read: Of the 84 million ha of land that stretch across Montana, Wyoming and Idaho in the USA, roadless areas cover 6.8 million ha, and existing federal protected areas (wilderness areas, national parks, special management areas and national wildlife refuges) protect 8.8 million ha.

Page 184, par. 1, line 13: The correct spelling is: Douglas-fir *Pseudotsuga menziesii*.

Page 185, col. 2: The top of the column should begin with the heading *Landscape connectivity.*

Page 187, col. 2, par. 1.: The last sentence should read: Our study, and that of Strittholt and DellaSala (2001), focused on smaller regions, where national forests are concentrated at higher elevations.

Page 188, col. 1, par. 1: The third sentence should read: When roadless areas are added to protected areas, aspen moves up two full categories: from underrepresented to adequately represented, a 480% increase in representation for the forest type, upon which many avian species depend (Hansen and Rotella 2000).

Page 188, col. 1, par. 2: The first two sentences should read: Elevation representation results demonstrate that higher elevations are well represented in existing protected areas. We also found that roadless areas would add substantially to protected areas at mid- to high elevations.

Page 189, col. 2: The reference to Beauvais (2000) should refer to F.W. Smith.

Page 191, col. 2, last line: The manuscript was received 30 December 2003.

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Assessing Crown Fire Potential in Coniferous Forests of Western North America: A Critique of Current Approaches and Recent Simulation Studies

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Assessing crown fire potential in coniferous forests of western North America: a critique of current approaches and recent simulation studies

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Abstract. To control and use wildland fires safely and effectively depends on creditable assessments of fire potential, including the propensity for crowning in conifer forests. Simulation studies that use certain fire modelling systems (i.e. NEXUS, FlamMap, FARSITE, FFE-FVS (Fire and Fuels Extension to the Forest Vegetation Simulator), Fuel Management Analyst (FMAPlus®), BehavePlus) based on separate implementations or direct integration of Rothermel's surface and crown rate of fire spread models with Van Wagner's crown fire transition and propagation models are shown to have a significant underprediction bias when used in assessing potential crown fire behaviour in conifer forests of western North America. The principal sources of this underprediction bias are shown to include: (i) incompatible model linkages; (ii) use of surface and crown fire rate of spread models that have an inherent underprediction bias; and (iii) reduction in crown fire rate of spread based on the use of unsubstantiated crown fraction burned functions. The use of uncalibrated custom fuel models to represent surface fuelbeds is a fourth potential source of bias. These sources are described and documented in detail based on comparisons with experimental fire and wildfire observations and on separate analyses of model components. The manner in which the two primary canopy fuel inputs influencing crown fire initiation (i.e. foliar moisture content and canopy base height) is handled in these simulation studies and the meaning of Scott and Reinhardt's two crown fire hazard indices are also critically examined.

Additional keywords: canopy base height, canopy bulk density, crown fire behaviour, crown fraction burned, crowning, Crowning Index, dead fuel moisture content, fire behaviour, fire behaviour modelling, fireline intensity, foliar moisture content, forest structure, rate of fire spread, Torching Index, wind speed.

Introduction

Crowning forest fires are exceedingly exciting to observe but like most natural phenomena, are dangerous as well. The safe and effective management of fire in most coniferous forest ecosystems is thus dependent to a very large extent on the ability to reliably assess or forecast crown fire potential based on predictive aids produced by research coupled with the skill and knowledge of the user.

Many advances have been made in crown fire behaviour research in recent years, including more intensively monitored experimental crown fires (Stocks *et al.* 2004) and physical-based modelling (Butler *et al.* 2004; Cruz *et al.* 2006*a*, 2006*b*). Nevertheless, crown fire behaviour is sometimes portrayed as a complex phenomenon for which we possess very limited knowledge and understanding of the exact physical processes involved (Cohen *et al.* 2006). Although this may very well be true, a substantial number of observations garnered from conducting outdoor experimental fires (Alexander and Quintilio 1990) and monitoring wildfires coupled with case study documentation (Cruz and Plucinski 2007) over the years have provided a solid foundation on several aspects of crown fire phenomenology as well as benchmark data on expected fire characteristics under certain environmental conditions, at least on an empirical basis.

Understanding the environmental conditions required for the onset or initiation and sustained propagation of crown fires is necessary to implement fuel management programs aimed at mitigating the likelihood of large, high-intensity crowning wildfires in the conifer-dominated forests found in western North America. Keyes and Varner (2006) have recently outlined just how complicated the processes involved are in using silvicultural methods to treat forest fuels in order to modify potential crown fire

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behaviour. The need for research into the effectiveness of fuel treatments in reducing crown fire potential has received considerable attention in recent years (Graham *et al.* 2004; Agee and Skinner 2005; Peterson *et al.* 2005). Roccaforte *et al.* (2008) classified research of this type into three categories: experimental, observational and simulation modelling.

Martinson and Omi (2008) have recently reported that more than half of the published studies aimed at quantifying fuel treatment effectiveness rely solely on modelling simulations. Commonly, these simulation studies characterise the fuel structure of distinct forest stands and through the use of fire modelling systems, coupled with specified fire weather, fuel moisture and slope conditions, attempt to integrate this information into a few fire behaviour descriptors in order to assess the relative 'flammability' of the fuel complex (McHugh 2006), and in turn, are able to gauge the effectiveness of fuel management strategies to mitigate the possibility of crown fires occurring (Graham *et al.* 1999; Keyes and O'Hara 2002).

Various fire modelling systems, such as NEXUS (Scott and Reinhardt 2001), Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS) (Reinhardt and Crookston 2003), FARSITE (Finney 2004), Fuel Management Analyst (FMAPlus®) (Carlton 2005), FlamMap (Finney 2006) and BehavePlus (Andrews *et al.* 2008), are extensively used in these simulation studies to assess potential crown fire behaviour in the western US (Keyes and Varner 2006; McHugh 2006; Varner and Keyes 2009) and to a lesser extent to date in western Canada (e.g. Bessie and Johnson 1995; Feller and Pollock 2006). The technical basis and intended uses of these modelling systems are contrasted elsewhere (McHugh 2006; Andrews 2007; Peterson *et al.* 2007).

All of the fire modelling systems referred to previously implement, link or integrate (or both) Rothermel's (1972, 1991) models for predicting surface and crown fire rates of spread with Van Wagner's (1977, 1993) crown fire transition and propagation models in various ways, and provide an output of several fire behaviour characteristics (e.g. rate of fire spread, fireline intensity, type of fire, crown fraction burned). Some of the systems also output two crown fire hazard indices – the Torching index (TI) and the Crowning Index (CI) as per Scott and Reinhardt (2001). The TI and CI represent the threshold wind speeds required for the onset of crowning and active crown fire propagation in coniferous forests respectively. Each TI and CI value is tied to a unique set of surface fuelbed characteristics (expressed in terms of a stylised or custom fuel model), dead and live moisture contents of surface fuels, crown fuel properties (canopy base height and bulk density, foliar moisture content), and slope steepness. This approach of using fire modelling systems to assess potential crown fire behaviour has gained widespread popularity within the US wildland fire research community, as evident by the number of published simulation studies over the past 10 years or so (e.g. Scott 1998*a*; Stephens 1998; Raymond and Peterson 2005; Harrington *et al.* 2006; Graetz *et al.* 2007; Mason *et al.* 2007; Battaglia *et al.* 2008). Scott and Reinhardt's (2001) two crown fire hazard indices are now being recommended for use in Canada (Gray and Blackwell 2008).

Our cursory critique of these simulation studies has revealed that many of them have produced unrealistic outcomes in terms of crowning potential, as evident by the resulting TI and CI values, given the specified environmental conditions and fuel characteristics. Quite often, critically dry fuel moisture levels are specified along with very low canopy base heights and relatively high canopy bulk densities and yet the simulations suggest that exceedingly strong winds are commonly required to initiate crowning and for fully developed or active crown fires to occur.

We have subsequently discovered that the fire modelling systems used in assessing crown fire potential in these simulation studies have an inherent underprediction bias associated with them as a result of the underlying models or the manner in which they have been implemented (Cruz *et al.* 2003*a*). The primary purpose of the present paper is to accordingly document the unrealistic nature of the outputs from these simulation studies and the level of underprediction bias involved in the models or modelling systems (or both), and then to explain the reasons for such results. Finally, comments are made on the manner in which two of the canopy fuel characteristics (i.e. foliar moisture content and canopy base height) involved in these simulation modelling studies are handled as well the interpretation of the two crown fire hazard indices.

Wind speeds quoted in this article are in terms of the international 10-m open standard (Lawson and Armitage 2008) unless otherwise stated. For the convenience of the reader, a summary list of the variables, including their symbols and units, referred to in the equations and text is given at the end of this article.

Evidence for underprediction of crowning potential in relation to environmental conditions

The notion of an underprediction trend associated with the modelling systems used in various simulation studies has also been hinted at by others. Hall and Burke (2006) found in applying the NEXUS modelling system to prefire fuel complex data collected in the area burned by the 2002 Hayman Fire in north-central Colorado (Graham 2003) that the system failed to simulate the crowning activity actually observed under the weather and fuel moisture conditions that prevailed. Similarly, Agee and Lolley (2006) noted that the low torching potential found in their simulations was 'contradictory to local and regional experience on recent wildfires'. Fulé *et al.* (2001*a*) also recognised that simulation outputs from the NEXUS modelling system appeared contradictory to actual wildfire experience, noting that 'simulated fires using our fuel and weather conditions proved nearly impossible to crown using realistic data, even though real fires had crowned under similar or even less severe conditions'. Here, we specifically discuss and provide evidence for the underprediction bias in terms of wind speed and dead fuel moisture content.

Wind speed and dead fuel moisture combinations

The simulations produced in several studies examining fuel treatment effectiveness reveal a rather low potential for crown fire behaviour relative to the specified environmental conditions (e.g. Scott 1998*a*; Graves and Neuenschwander 2001; Fulé *et al.* 2002; Perry *et al.* 2004; Raymond and Peterson 2005; Agee and Lolley 2006; Hall and Burke 2006; Harrington *et al.* 2006; Page and Jenkins 2007; Roccaforte *et al.* 2008). This is reflected in the threshold wind speeds required for the onset of crowning as represented by the TI and for active crown fire spread as represented by the CI. Both values are generally quite high considering that the simulations are generally based on extremely dry fuel moisture conditions. In many cases, these simulation studies have reported TI and CI values associated with gale-force winds (i.e. sustained winds greater than $\sim$100 km h^{-1}). Such winds seldom occur inland, but when they do, they generally result in trees and whole forest stands being blown down over large areas (List 1951). Scott (2006) has indicated that these very high wind velocities simply indicate 'a very low potential for initiating a crown fire' and that wind speeds at or in excess of 100 km h^{-1} 'occur so rarely that crown fire can be considered nearly impossible to initiate'. Stephens *et al.* (2009) suggest that such levels of wind strength should be 'interpreted as a characteristic of a forest structure that is extremely resistant to passive crown fire'. Although these are possible explanations, they aren't the only ones.

It can be argued that the outcomes of these simulation studies are realistic in that they simply reflect the fact that both strong winds and dry fuels are required to achieve any sort of torching or crowning activity. Although this may be intuitively true for areas that have undergone some form of fuel treatment, for control or untreated areas, the simulation results do not appear realistic based on general observation and experience (Fig. 1 and Table 1), thereby suggesting that the authors of these simulation studies have failed to compare their simulation outputs with empirical observation in order to gauge that their results are realistic (Alexander 2006). Empirical evidence from outdoor experimental crown fires (Stocks *et al.* 2004; Cruz *et al.* 2005) and from wildfire case study documentation (Alexander and Cruz 2006) provides a ready test of this assertion. Fig. 1*a* is a plot of the range in the fine dead fuel moisture (FDFM, %) as per Rothermel (1983) and 10-m open wind speed (U_{10}, km h^{-1}) associated with a dataset of 54 documented crowning wildfires from across North America as taken from a summary given in Alexander and Cruz (2006). FDFM is referred to as the 'estimated fine fuel moisture' in Cruz *et al.* (2004, 2005), Alexander and Cruz (2006), and Alexander *et al.* (2006).

Also plotted in Fig. 1*a* is the 1-h time-lag fuel moisture content (Fosberg and Deeming 1971; Deeming *et al.* 1977) – in lieu of the FDFM – and U_{10} pairs used in the control or no-treatment fuel complexes for a selected set of fuel treatment effectiveness simulation studies. It is apparent from Fig. 1*a* that the conditions used in these simulation studies are extremely severe and not representative of the conditions commonly encountered in large, high-intensity wildfire incidents that involve extensive crowning activity.

Fig. 1*b* illustrates the level of underprediction bias associated with crown fire rate of spread for nine simulation studies by comparing the resultant outputs with observed wildfire rates of spread in relation to U_{10}; some additional observations are given in Table 1. As a general trend, the simulation studies, even though they are relying on extremely dry fuel moisture conditions, require almost a doubling in the U_{10} to attain the level of fire spread rates contained within the wildfire dataset. It is evident from the plots of the TI and CI values (Fig. 1*c*) – the outputs sought by these studies in order to quantify stand or landscape 'flammability' – that the simulation results constitute a distinctly different population from the dataset compiled by Alexander and Cruz (2006) that is based largely, but not exclusively, on wildfires in the western and northern North American coniferous forests. The TI and CI values presented in Fig. 1*c* are applicable to stands with mostly low (i.e. $<$3 m) to moderately high (i.e. 3–8 m) canopy base heights. The various simulation studies generally indicate that exceptionally dry fuel conditions and very strong winds are required for passive and active crowning activity compared with the conditions associated with the documented wildfires.

Wind speed limits

Also noteworthy in Fig. 1*c* is the magnitude of simulated wind speeds, especially in respect to the TI, in several cases in excess of 100 km h^{-1}, given in some of these and other studies (e.g. Scott 1998*a*; Fiedler and Keegan 2003; Monleon *et al.* 2004; Perry *et al.* 2004; Fried *et al.* 2005; Ager *et al.* 2007; Moghaddas and Stephens 2007; Stephens *et al.* 2009). This is consentaneous with other studies aimed at quantifying the potential crown fire behaviour associated with specific fuel complex structures that have reported winds close to or in excess of 1000 km h^{-1} (e.g. Raymond and Peterson 2005; Hall and Burke 2006; Johnson 2008; Stephens *et al.* 2009; Vaillant *et al.* 2009*a*). Some authors have chosen to simply express their TI and CI (6.1-m open wind speeds) values as $\geq$40.2 km h^{-1} or the CI separately as $\geq$64.4 km h^{-1} (e.g. Skog *et al.* 2006; Huggett *et al.* 2008), thereby masking the possibility of very high speeds presumably required for crowning; $\geq$85 km h^{-1} has also recently appeared (Battaglia *et al.* 2008) and $>$145 km h^{-1} (Fiedler *et al.* 2010) have also recently appeared. More recently, some authors have elected to cite only the CI values (e.g. Ager *et al.* 2007; Brown *et al.* 2008; Finkral and Evans 2008).

In contrast to the winds reported in Fig. 1*c*, the 10-m open winds associated with the eight crown fire rate of spread observations used in the formulation of the Rothermel (1991) crown fire rate of spread model averaged 38 km h^{-1} and ranged from 20 to 83 km h^{-1}. The highest wind speed (i.e. 83 km h^{-1}) was associated with the later stages of the major run of the 1967 Sundance Fire in complex mountainous terrain in northern Idaho (Anderson 1968). If this one observation was removed, the winds would have averaged 32 km h^{-1}. Thus, based on all of the available evidence (i.e. Rothermel 1991; Alexander and Cruz 2006; Table 1), one can say with some degree of confidence that there has been no documented active crown fire of any size associated with sustained winds greater than $\sim$80 km h^{-1} reported to date.

Dead fuel moisture levels

In the development of his crown fire rate of spread model, Rothermel (1991) equated the FDFM of Rothermel (1983) to the 1-h time-lag fuel moisture content; this lack of distinction has undoubtedly led to some of the confusion now seen in several simulation studies. He then estimated the 10- and 100-h time-lag values by adding 1.0 and 2.0% to the FDFM value respectively. Some simulation studies (e.g. Cram *et al.* 2006), including many of those identified in Fig. 1*a* and 1*b*, have chosen to use the dead fuel moisture time-lags generated by the US National Fire Danger Rating System (NFDRS) (Deeming *et al.* 1977) rather than estimating the 1-h time-lag fuel moisture content from the FDFM or using the seasonal moisture condition scenarios (or both) presented in Rothermel (1991).

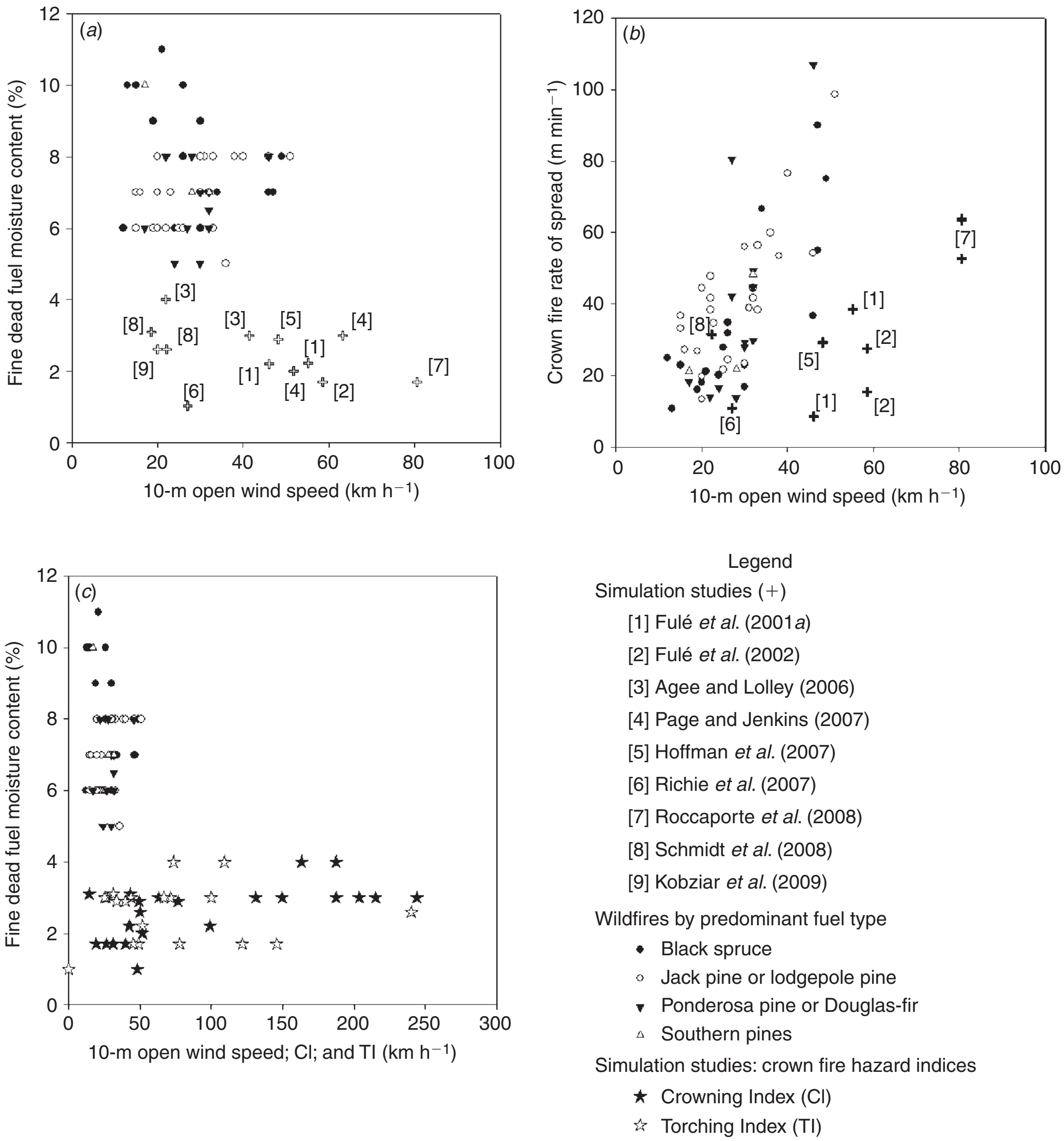

Fig. 1. Environmental conditions and associated crown fire rates of spread and indices of crown fire hazard for a dataset of actively crowning wildfires assembled by Alexander and Cruz (2006) and for a sample of selected simulation studies that have appeared in the scientific peer-reviewed literature: (*a*) fine dead fuel moisture *v.* 10-m open wind speed; (*b*) crown fire rate of spread *v.* 10-m open wind speed; and (*c*) fine dead fuel moisture *v.* 10-m open wind and Scott and Reinhardt's (2001) two crown fire hazard indices. Level terrain is assumed in all cases.

For the purpose of their simulations, Roccaforte *et al.* (2008) assumed 1-, 10- and 100-h time-lag fuel moisture contents of 1.7, 3.0 and 4.5% respectively, representing the 97th percentile level of fire weather severity based on 34 years of archived NFDRS calculations. DeRose and Long (2009) similarly applied values of 1.9, 2.1 and 3.2% respectively in their simulations. In calculating TI and CI values at the time that the 2002 Cone Fire in north-eastern California burned into their experimental fuel treatment plots, Ritchie *et al.* (2007) applied the NFDRS 1-h time-lag fuel moisture content of 1.0% as computed at a nearby fire weather station. The 10- and 100-h values both registered 2.0%. These three situations represent extremely low fuel moisture conditions for coniferous forests in all three categories.

Rothermel (1991) reported value ranges of 3–8, 4–9 and 5–9% respectively for the 1-h (i.e. FDFM was regarded as a surrogate), 10-h and 100-h time-lag fuel moisture contents associated with the wildfires used in the development of his crown fire rate of spread model. Even for his worst case 'late summer, severe drought' scenario, Rothermel (1991) only used 1-h (i.e. FDFM), 10-h and 100-h time-lag fuel moisture contents of 3.0, 4.0 and 6.0% respectively.

Table 1. Characteristics of some of the best-known or well documented (or both) crowning wildfires in conifer forests of North America and Australasia associated with exceptionally strong winds on level to gently undulating terrain not included in Fig. 1

Predominant fuel types are sand pine (*Pinus clausa*) and radiata pine (*P. radiata*). FDFM, fine dead fuel moisture; ROS, rate of spread; U_{10}, 10-m open wind speed. For U_{10}, the World Meteorological Organization standard to express wind speed at a height of 10 m in the open was followed here (Lawson and Armitage 2008). Winds measured at a height of 6.1 m in the open, as per the standard for fire danger rating and fire behaviour prediction used in the United States (Deeming *et al.* 1977; Rothermel 1983), were increased by 15% to approximate the U_{10} standard (Lawson and Armitage 2008)

Reference	Name of fire	Geographical location	Date (dd/mm/yy)	Predominant fuel type(s)	FDFM (%)	ROS ($m\,min^{-1}$)	U_{10} ($km\,h^{-1}$)	Type of crown fire
Stocks and Walker (1973)	Garden Lake	Ontario, CAN	02/06/30	Black spruce–jack pine–balsam fir	11	–[A]	48	Active
Folweiler (1937)	Big Henry	Florida, USA	12/03/35	Sand pine	–[B]	135–150	68	Active
Prior (1958)	Balmoral	New Zealand	26/11/55	Radiata pine	8	28	60	Passive and active
Schaefer (1957)	Dudley Lake	Arizona, USA	14/06/56	Ponderosa pine	6	–[C]	64	Active
Dieterich (1979)	Burnt	Arizona, USA	02/11/73	Ponderosa pine	9	30	74	Passive[D]
Geddes and Pfeiffer (1981)	Caroline	South Australia	02/02/79	Radiata pine	5	67	45	Active
Keeves and Douglas (1983)	Mount Muirhead	South Australia	16/02/83	Radiata pine	4	207	80	Active
NFPA (1992)	Spokane area	Washington, USA	16/10/91	Ponderosa pine	10[E]	30	66[E]	Passive and active[F]

[A]According to Stocks and Walker (1973), the extremely strong winds caused 'crowning and contributed greatly to the very fast spread of the fire' that saw two sustained runs of 24 and 64 km take place over a 26-h period on 1–2 June.
[B]According to Folweiler (1937), no measurements of relative humidity were available but it was 'probably low'.
[C]According to Schaefer (1957), the fire made a sustained run of 16 km. Dieterich (1976) estimated Byram's (1959) fireline intensity to have been 52 925 $kW\,m^{-1}$.
[D]According to Dieterich (1979), 'Damage from this fast-spreading fire was extremely variable ranging from complete destruction of crown material in patches of saplings and pole timber and an occasional mature tree, to large areas where the only evidence of fire was a blackened litter layer and slight scorch on the lowest portions of the crowns', and that much of the ponderosa pine was 'open grown, and tree crowns extended to within 4–5 feet [1.2–1.5 m] of the ground'. Alexander (1998) computed the fireline intensity at the head of the fire to be 5251 $kW\,m^{-1}$ using Eqn 4 and the critical surface fire intensity for initial crown combustion to be just 343 $kW\,m^{-1}$ using Eqn 1.
[E]Based on Alexander and Pearce (1992).
[F]According to the NFPA (1992) case study report, it was observed that: typically stands of ponderosa pine contain dead branches extending to the ground. In some cases, these 'ladder fuels' enabled the fire to reach the crowns of the 30- to 100-foot pine trees and would result in the fire spreading at extremely high rates. Unlike other severe wildland fires, however, this 'crowning' was fairly limited.

As illustrated in Fig. 1*a*, Alexander and Cruz (2006) found for a large database composed mainly of western and northern North American wildfires that the FDFM commonly varied between 6 and 10%. The moisture content of shaded needle litter in conifer forest stands very seldom is less than 2.5–3.0% (Countryman 1977; Harrington 1982; Rothermel *et al.* 1986; Hartford and Rothermel 1991; Wotton and Beverly 2007). The 1-h time-lag NFDRS fuel moisture content can easily be ~2.0% less than the shaded condition represented by the FDFM owing to the effects of solar radiation on fully exposed fuels. This is the reason for the very low fuel moisture conditions commonly associated with the simulation studies on fuel treatment effectiveness (Fig. 1*a*). Considering that the fine, dead fuels represented by the 1-h time-lag fuels are the principal carrier for surface fire spread, the use of the NFDRS computation in lieu of the FDFM represents a significant departure in the application of Rothermel's (1991) crown fire rate of spread model.

Reasons for underprediction of potential crown fire behaviour

The comparison of simulation results with actual observed data presented in Fig. 1 suggests there is a problem in the fundamental underlying models or the manner (or both) in which the models were implemented in the modelling systems. An in-depth analysis of the modelling system framework as dictated by the linkages between the Rothermel (1972, 1991) and Van Wagner (1977, 1993) models reveals that the underprediction bias in the assessment of potential crown fire behaviour arises from three principal sources: (1) incompatible model linkages; (2) use of surface and crown fire rate of spread models that have an inherent underprediction bias; and (3) the reduction in crown fire rate of spread based on the use of crown fraction burned functions. A further potential source of bias is the use of uncalibrated custom fuel models. All but one of these bias sources (i.e. the second one) arise from what we believe is unsubstantiated use of the cited models.

Rothermel (1972) surface fire–Van Wagner (1977) crown fire initiation model linkages

The implemented linkage between the outputs of the Rothermel (1972) surface fire model (i.e. rate of spread and intensity) and the Van Wagner (1977) crown fire initiation model overlooks an important assumption of the latter model. Through a combination of physical reasoning and empirical observation, Van Wagner (1977) defined quantitative criteria to predict the onset of crowning. He defined the critical surface fire intensity for initial crown combustion (I_o, kW m^{-1}) as a function of the canopy base height (CBH, m), and heat of ignition (h, kJ kg^{-1}):

$$I_o = (C \cdot \mathrm{CBH} \cdot h)^{1.5} \tag{1}$$

where h is in turn determined by the foliar moisture content (FMC, %) (Van Wagner 1989, 1993):

$$h = 460 + 25.9 \cdot \mathrm{FMC} \tag{2}$$

Van Wagner (1977) considered the quantity C in Eqn 1, the criterion for initial crown combustion, 'is best regarded as an empirical constant of complex dimensions whose value is to be found from field observations'. Van Wagner (1977) derived a value for the proportionality constant C using the following transformation of Eqn 1 on the basis of a blend of three experimental crown fires carried out in a red pine (*Pinus resinosa*) plantation:

$$C = \frac{I_o^{0.667}}{(\mathrm{CBH} \cdot h)} \tag{3}$$

The surface fire intensity at the onset of crowning was estimated to be ~2500 kW m^{-1} (Van Wagner 1968). Thus, for a CBH of 6.0 m and FMC of 100%, $C = 0.010$ (kW$^{2/3}$ kJ^{-1} kg m$^{-5/3}$).

Van Wagner (1977) equated I_o to Byram's (1959) fireline intensity (I_B, kW m^{-1}), which he calculated from measurements of fire spread rate and fuel consumption:

$$I_B = H \cdot w_a \cdot r \tag{4}$$

where H is the low heat of combustion (kJ kg^{-1}), w_a is the fuel consumed in the active flaming front (kg m^{-2}), and r is the rate of fire spread (m s^{-1}) (Alexander 1982). It is possible to express the requirements for the onset of crowning in terms of the surface fire spread rate by replacing I_o for I_B in Eqn 4 and working backwards (Van Wagner 1989, 1993; Forestry Canada Fire Danger Group 1992), giving the following result:

$$R_i = \frac{60 \cdot I_o}{H \cdot w_a} \tag{5}$$

where R_i is the critical surface fire rate of spread for crown fire initiation (m min^{-1}).

Modelling systems such as NEXUS, FlamMap, BehavePlus, FARSITE, FFE-FVS, and FMAPlus calculate fireline intensity from Rothermel's (1972) reaction intensity (I_R, kW m^{-2}) (Albini 1976):

$$I_B = I_R \cdot t_r \cdot r \tag{6}$$

where t_r is the flame-front residence time (s). Fireline intensities calculated in this manner are consistently lower than per the original Byram (1959) formulation (Cruz *et al.* 2003*a*, 2004). The extent of the differences is a function of the fuelbed characteristics. For the original 13 standard US fire behaviour fuel models as described by Anderson (1982), Byram's (1959) fireline intensity (Eqn 4) is larger than the Rothermel (1972) I_R-derived fireline intensity by a factor of 2 to 3 (Cruz *et al.* 2004).

The implication of these differences within a modelling system such as NEXUS is that higher simulated surface fire rates of spread, and consequently stronger wind speeds and hence larger TI values, are necessary to induce crowning than if the model linkages were to follow the original model assumptions. The end result is increasingly large TI values. Fig. 2 presents a graphical representation of the magnitude of this error for the Anderson (1982) Fuel Model 2 – Timber (grass and understorey) and Fuel Model 10 – Timber (litter and understorey) considering an I_o of 2935 kW m^{-1} per Eqns 1 and 2 based on a CBH of 5.0 m and an FMC of 140%; the output of Fuel Model 9 – Hardwood litter would be very similar to that of Fuel Model 10. The increase in mid-flame wind speed required for the onset of crowning is 72% (i.e. from 6.5 to 10.9 km h^{-1}) for Fuel Model 2 and 48% (i.e. from 8.2 to 12.1 km h^{-1}) for Fuel Model 10. The differences observed in this modelling exercise are considered as conservative in nature. The calculations of Byram's (1959) fireline intensity undertaken here assume that the fuels consumed in the flame front and thus contributing to the upward heat fluxes are the fine, dead and live fuels plus the 10-h time-lag fuels, whereas Van Wagner (1977) in his original formulation did not specifically differentiate between the fuels consumed during flaming as opposed to flaming and smouldering or glowing combustion. In other words, he assumed w_a was equivalent to the difference he obtained from pre- and post-burn fuel sampling – i.e. the fuel consumed in the active flaming front and by glowing or smouldering combustion following passage of the front (w, kg m^{-2}).

Conceptually, the two methods of computing Byram's (1959) fireline intensity should, in theory, yield nearly identical results. The main differences between these two arise from the use of the I_R and t_r models in the Rothermel (1972) model to calculate Byram's (1959) fireline intensity. I_R is estimated from an empirical model developed for homogeneous fuelbeds under no-wind/no-slope conditions in a laboratory setting. How well these assumptions hold for natural surface fuelbeds, with heterogeneous fuel particle and moisture content distributions is unknown, as the model has never been evaluated against field data to our knowledge other than the attempt by Brown (1972) involving simulated slash fuelbeds.

The use of Anderson's (1969) model to estimate t_r in Eqn 6 is the most likely source for the differences between the two methods of determining Byram's (1959) fireline intensity. Research on t_r in natural fuelbeds has identified fuel load, compactness, particle size and moisture as well as wind speed as the most influential variables (Cheney 1981; Nelson 2003). Anderson's (1969) model predicts t_r solely from the characteristic or average weighted size of individual fuel particles.

Nelson (2003) developed and evaluated a semi-physically based model to predict t_r that takes into account fuelbed structure and combustion zone properties. A comparison between the Anderson (1969) and Nelson (2003) t_r models reveals that the former model consistently yields lower t_r values when w_a exceeds ~0.5 kg m^{-2} (Fig. 3). Evaluation data for simulated fuelbeds of slash pine (*Pinus elliottii*) needle litter (Nelson and Adkins 1988) and ponderosa pine (*P. ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) slash (Brown 1972) reveal a marked underprediction of t_r by Anderson's (1969) model and general agreement with Nelson's (2003) model.

If Nelson's (2003) model is considered to provide an acceptable prediction of t_r, as supported by Fig. 3 and his own evaluation against an array of artificial fuelbeds, the Anderson (1969) model is underpredicting t_r in fuel beds with medium to high available fuel loads. This error is propagated within the modelling system and leads to low fireline intensities, and in turn, a low potential for crown fire initiation as illustrated in Fig. 2.

Underprediction bias in the Rothermel (1972) surface fire rate of spread model

In addition to the incompatibility between the various US fire modelling systems and Van Wagner's (1977) criteria for crown

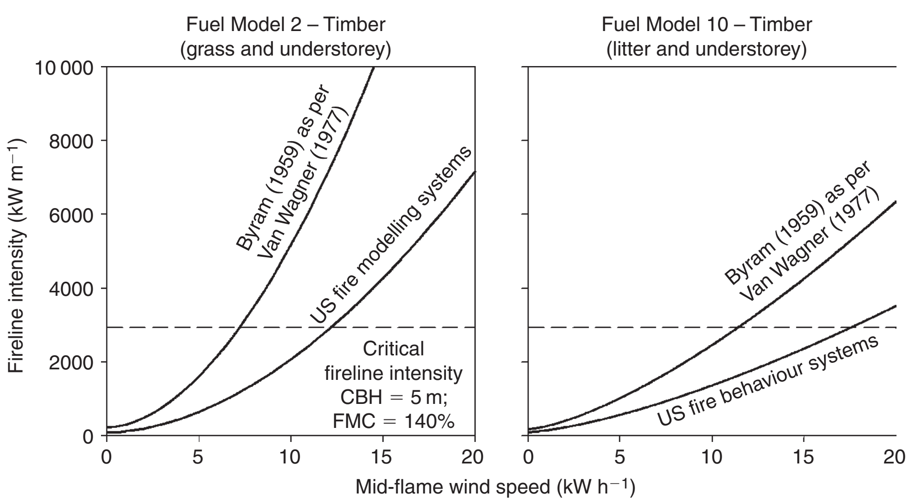

Fig. 2. Critical mid-flame wind speeds required for crown fire initiation as per Van Wagner's (1977) critical surface fire intensity criteria for two US fire behaviour fuel models (Anderson 1982) based on different methods of calculating fireline intensity (i.e. Byram (1959) as per Van Wagner (1977) *v.* US fire behaviour modelling systems) for a particular canopy base height (CBH) and foliar moisture content (FMC) equating to a critical surface fire intensity for initial crown combustion (I_o) of 2920 kW m^{-1}. The following environmental conditions were held constant: slope steepness, 0%; fine dead fuel moisture, 4%; 10- and 100-h time-lag dead fuel moisture contents, 5 and 6% respectively; live woody fuel moisture content, 75%; and live herbaceous fuel moisture content, 75%. The associated 10-m open winds would be a function of forest structure and can be approximated by multiplying the mid-flame wind speed by a factor varying between 2.5 (open stand) and 6.0 (dense stand with high crown ratio) (Albini and Baughman 1979).

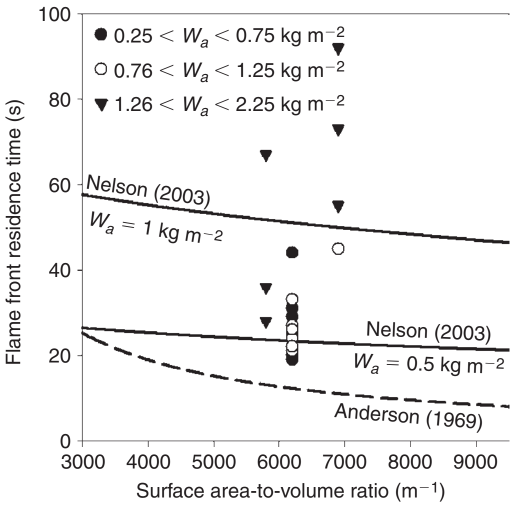

Fig. 3. Sensitivity of Anderson (1969) and Nelson (2003) flame front residence time models to surface area-to-volume ratio and fuel consumed in the active flaming front (w_a). For Nelson's (2003) model, the following environmental conditions were held constant: fuelbed depth, 0.1 m; fuel moisture content, 5%; and mid-flame wind speed, 5 km h^{-1}. Data points represent computed flame front residence times from experimental fires conducted in simulated fuelbeds of slash pine needle litter (Nelson and Adkins 1988) and ponderosa pine and Douglas-fir slash (Brown 1972), where it was implicitly assumed that w was $\sim w_a$.

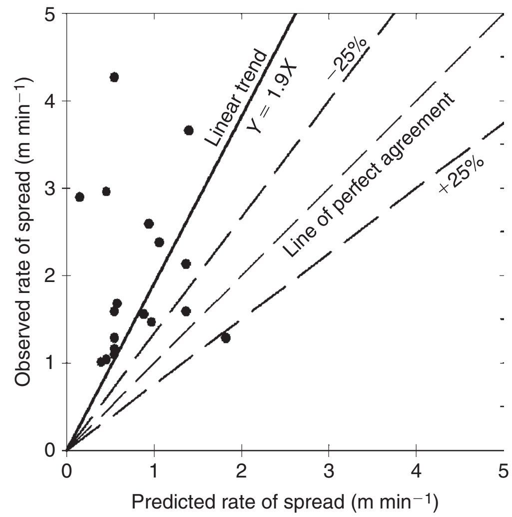

Fig. 4. Observed head fire rates of spread >1 m min^{-1} associated with prescribed burning experiments in ponderosa pine forests of Yosemite National Park, CA, *v.* predictions based on the Rothermel (1972) surface fire rate of spread model for Anderson (1982) Fuel Model 9 – Hardwood litter (adapted from van Wagtendonk and Botti 1984). The dashed lines around the line of perfect agreement indicate the ±25% error interval.

fire initiation with respect to determining w_a, a certain amount of uncertainty exists as to whether the Rothermel (1972) surface fire model can in fact reliably predict, in certain conifer forest stand types, the spread rate of moderate- and high-intensity surface fires that would lead to crowning. Studies that have evaluated Rothermel's (1972) fire spread model for any of the Anderson (1982) stylised 'timber' fuel models (numbers 2, 8, 9 and 10) have identified underprediction trends (Norum 1982; van Wagtendonk and Botti 1984; Grabner *et al.* 1997, 2001). This underprediction trend or bias arises from the sensitivity of the Rothermel (1972) fire spread model to the compactness of the horizontally oriented surface fuelbeds associated with these fuel models (Catchpole *et al.* 1993) and has been discussed in detail by Cruz and Fernandes (2008). Most investigators commonly develop an adjustment factor for rate of spread predictions on the basis of their performance testing (Rothermel and Reinhart 1983). Stephens (1998) for example used the adjustment factors derived by van Wagtendonk and Botti (1984) in his simulation study.

Modelling systems like NEXUS are widely applied to western US ponderosa pine forests (e.g. Johnson *et al.* 2007) and yet performance testing of Rothermel's (1972) model in such fuel complexes is limited to a single outdoor field study by van Wagtendonk and Botti (1984). The same underprediction bias seen in other studies is also evident in their study (Fig. 4 and Table 2). Considering that surface rate of fire spread is a factor in determining the onset of crowning in coniferous forests, the use of unadjusted predictions from stylised fuel models constitutes yet another source of underprediction bias in assessing crown fire potential.

Underprediction bias in the Rothermel (1991) crown fire rate of spread model

Until recently, the only comparison of observed crown fire spread *v.* predictions from Rothermel's (1991) model was that undertaken by Goens and Andrews (1998) on the 1990 Dude Fire that occurred in central Arizona. They found good agreement between predicted and observed spread distances. However, the Dude Fire was considered by Rothermel (1991) as a plume-dominated crown fire as opposed to a wind-driven crown fire, for which he considered his predictive methods were not applicable.

Several studies (Cruz *et al.* 2003*a*, 2005; Stocks *et al.* 2004; Alexander and Cruz 2006) have separately evaluated the Rothermel (1991) crown fire rate of spread model against outdoor experimental crown fire and wildfire datasets (Table 3). A composite summary of those evaluations is presented in Fig. 5. Rothermel's (1991) model underpredicted all 34 experimental observations, with a mean absolute error of 71% (Table 2).

A distinct underprediction bias was also evident in the wildfire observations (Fig. 5*b*). All 54 observations were underpredicted with a mean absolute error of 61%; 63 and 58% for the US and Canadian wildfires respectively (Table 2). The Rothermel (1991) model consistently underpredicted the four observed spread rates in ponderosa pine forests extracted from the 2002 Hayman Fire in north-central Colorado (Finney *et al.* 2003; Graham 2003) by a factor of 2.8 (Alexander and Cruz 2006).

Scott (2006) has acknowledged the underprediction trends evident in Fig. 5 and suggested the use of a correction or adjustment factor (1.7) to obtain what Rothermel (1991) defined as the near-maximum crown fire rate of spread derived on the basis of five 'chance' observations of temporary escalations in

Table 2. Model performance statistics for the Rothermel (1972), Rothermel (1991) and Schaaf *et al.* (2007) rate of fire spread models evaluated against different types of data sources

Statistic	Rothermel (1972)	Rothermel (1991)		Schaaf *et al.* (2007)
	Prescribed fires	Experimental fires	Wildfires	Wildfires
Number of observations	18	34	54	15
Root mean square error	1.54	27	30.7	22.2
Mean absolute error	1.23	22.2	26.0	15.2
Mean absolute percentage error	57	70.8	60.7	41.6
Mean bias error	−1.16	−22.2	−25.9	−15.7
Percentage within ±25% error	6	3	4	20
Over and under predictions	1, 17	0, 34	0, 54	1, 14

Table 3. Basic descriptive statistics associated with the experimental fire and wildfire datasets used in the evaluation of the Rothermel (1991) crown fire rate of spread model as shown in Fig. 5

For Experimental fires, refer to Table 1 in Cruz *et al.* (2005) and to Stocks *et al.* (2004) for the specific details on data sources. For Wildfires, refer to Alexander and Cruz (2006) for the specific details on data sources

Variable	Experimental fires ($n = 34$)				Wildfires ($n = 54$)			
	Mean	s.d.	Min.	Max.	Mean	s.d.	Min.	Max.
10-m open wind speed (km h^{-1})	15.6	5.9	5	35	28.2	9.92	12	51
Air temperature (°C)	25.7	3.9	18.5	31.4	26.6	4.2	20	36
Relative humidity (%)	36.1	7.5	23	52	28	10.6	5	56
Fine dead fuel moisture (%)	7.8	1.9	4	12	7.2	1.37	5	11
Rate of fire spread (m min^{-1})	29.2	16.9	10.7	69.8	39.8	22.1	10.7	107

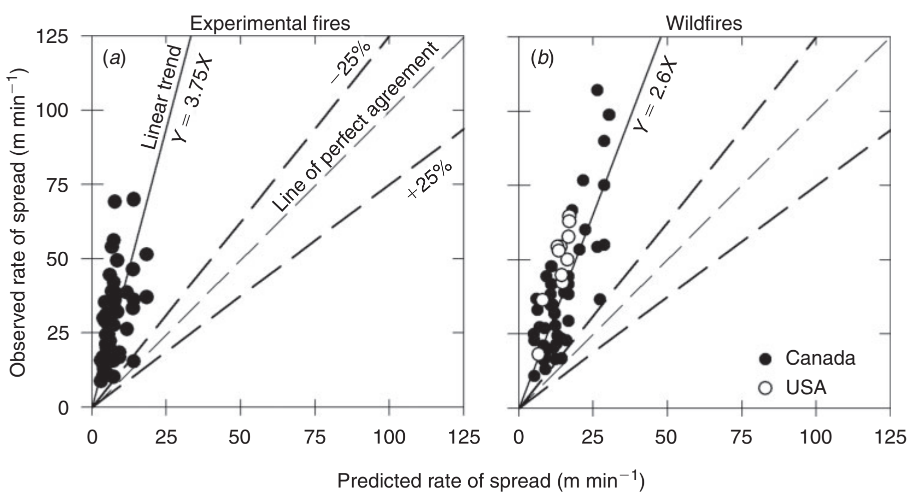

Fig. 5. Observed rates of spread of (*a*) experimental active crown fires; and (*b*) wildfires that exhibited extensive active crowning *v.* predictions based on the Rothermel (1991) crown fire rate of spread model. The dashed lines around the line of perfect agreement indicate the ±25% error interval.

crown fire spread but without any corresponding wind speed measurements. However, according to Rothermel (1991, p. 25), the near-maximum crown fire rate of spread adjustment was intended solely for predicting short bursts in crown fire spread that could be expected to occur during upslope runs and not as a general adjustment factor.

Why is the Rothermel (1991) model consistently under-predicting by a factor of ~2.5–3.0 and why does it also appear to be relatively insensitive to burning conditions? It is likely due to a multitude of interacting factors (Alexander 2006).

The Rothermel (1991) model is a simple relationship consisting of a correlation derived between the observed average crown fire rate of spread based on eight observations involving seven western US wildfires and the output of the Rothermel (1972) surface fire spread model using Fuel Model 10 and a wind-reduction factor of 0.4 (R_{10}, m min^{-1}) in order to adjust the 6.1-m open wind speed to a mid-flame height value (Albini and Baughman 1979). The Rothermel (1991) model for predicting active crown fire rate of spread (R_a, m min^{-1}) is as follows:

$$R_a = 3.34 \cdot R_{10} \quad (7)$$

Only four of the eight observations used in the model development involved level terrain, so the difficulty of obtaining representative winds in complex terrain relative to observed spread rate can be called into question. Furthermore, the overall average observed rate of spread for five of the eight observations used in the model development was 43 m min^{-1}, which seems reasonable for active or fully developed crown fires in light of the wildfire database compiled by Alexander and Cruz (2006). However, three of eight observations had spread rates of only 14 m min^{-1}. Without knowing what the associated canopy bulk density (CBD) values were for these three observations, such spread rates are low for active crown fires (Cruz *et al.* 2005; Alexander and Cruz 2006). This raises the issue as to the stage of development or degree of crown fire activity (i.e. passive crowning *v.* active crowning) associated with these three crown fire observations and their relative magnitude in the derivation of the Rothermel (1991) model.

From a conceptual perspective, it can be argued that the underlying relationships in the Rothermel (1972) model (i.e. developed from shallow surface fuelbeds in a laboratory setting) do not apply to crown fire phenomena, where the dimension of the fuelbed sustaining fire propagation and the heat flux generated are orders of magnitude higher. Rothermel (1972) readily acknowledged this point and clearly stated in the preface of his publication that the nature and mechanisms of heat transfer in a crown fire are considerably different than those for a surface fire and therefore stated that 'the model developed in this paper is not applicable to crown fires'. Thus, using R_{10} as a correlative or independent variable in what amounts to a statistical model is questionable. The underprediction tendency associated with Rothermel's (1991) model shown in Fig. 5 has also been found to occur with the crown fire rate of spread model developed recently by Schaaf *et al.* (2007) as part of the Fuel Characteristic Classification System (Ottmar *et al.* 2007). The Schaaf *et al.* (2007) model, based on a reformulation of the Rothermel (1972) model by Sandberg *et al.* (2007), is specifically designed to predict the rate of spread of crown fires in coniferous forests. Schaaf *et al.* (2007) undertook to test model performance on the basis of data extracted from Alexander and Cruz (2006) for 15 actively crowning wildfires in black spruce (*Picea mariana*) forests of Canada (Fig. 6 and Table 2). Cronan and Jandt (2008) observed the same underprediction bias evident in Fig. 6 with the experimental fires they conducted in Alaskan black spruce forests.

Another possible reason for the underprediction trend in the Rothermel (1991) model is its low sensitivity to changes in wind speed. As noted, the Rothermel (1991) crown fire spread model is a direct function of Fuel Model 10. Considering that heat transfer is optimised for vertically oriented, high-porosity fuelbeds (Rothermel 1972), the wind speed–rate of spread relationship of a litter and understorey fuelbed may not be representative of phenomena occurring in deep, low-packing-ratio fuel layers such as canopy fuels in a conifer forest stand. Cohen *et al.* (2006) have described in some detail the inadequacies of the Rothermel (1972) model framework to represent the processes determining crown fire propagation in conifer forests.

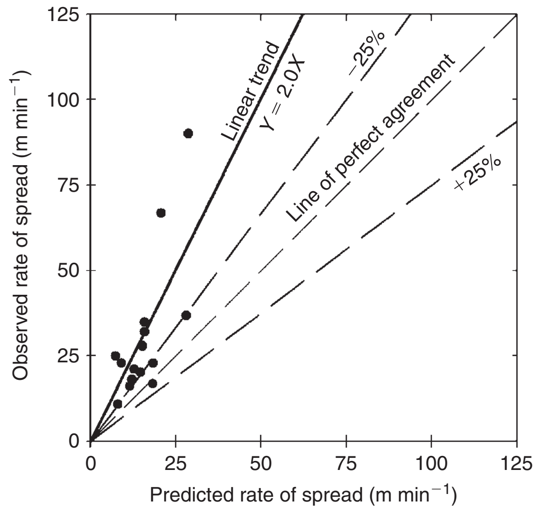

Fig. 6. Observed rates of spread of actively crowning wildfires in black spruce forests *v.* predictions based on the Schaaf *et al.* (2007) crown fire rate of spread model. The dashed lines around the line of perfect agreement indicate the ±25% error interval.

The seven wildfires used in the development of the Rothermel (1991) crown fire rate of spread model encompass a wide range in fuel complex structure and composition, although it is difficult to critically assess this factor because formal case study documentation is only available for two of the seven wildfires (Anderson 1968; NFPA 1990) that Rothermel (1991) used in his model development. The Rothermel (1991) crown fire rate of spread model does not explicitly take into account any stand or canopy fuel structure variables as inputs (e.g. CBH, CBD). Hence, crown fire behaviour in the Rothermel (1991) model is independent of the physical fuel characteristics associated with conifer forest stands (Finney 2004).

Rothermel (1991) indicated that the correlation he obtained between the observed crown fire rate of spread and the prediction of surface fire rate of spread from Fuel Model 10 did 'give reasonable results'. However, he was also quick to point out that 'It is readily apparent that more research is needed to strengthen this analysis', and emphasised that his guide represented 'first-order approximations of crown fire behavior' designed to aid operational decision-making.

All 34 experimental fires and 39 of the 54 wildfire observations presented in Fig. 5 involve boreal or boreal-like forest fuel complexes. Thus, it could be argued that the fires selected for evaluation are not 'applicable to the Northern Rocky Mountains or mountainous areas with similar fuels and climate' as per one of Rothermel's (1991) assumptions. Strictly speaking, this is a valid comment.

However, the Rothermel (1991) model has been directly and also indirectly applied through the application of fire modelling systems like NEXUS, FlamMap, FARSITE, FFE-FVS, FMAPlus and BehavePlus, to other distinctly different forest stand types and in other regions of the western US, including for example, the Sierra Nevada (Stephens and Moghaddas 2005*a*, 2005*b*; Dicus *et al.* 2009), north-central (Kobziar *et al.* 2009) and north-eastern (Ritchie *et al.* 2007) regions of California as well as the whole state (Vaillant *et al.* 2009*a*, 2009*b*), south-central (Hummel and Agee 2003), north-eastern (Graves and Neuenschwander 2001) and western Washington (Agee and Lolley 2006), north-eastern (Williamson 1999; Ager *et al.* 2007), central (Fitzgerald *et al.* 2005) and western Oregon (Raymond and Peterson 2005), south-western Utah (Stratton 2004), central Arizona (Goens and Andrews 1998), northern Arizona (Fulé *et al.* 2001*a*, 2001*b*, 2002, 2004), south-central New Mexico (Mason *et al.* 2007), northern Arizona–north-central New Mexico (Clifford *et al.* 2008), and even the north-eastern US (Duveneck and Patterson 2007). In defence of the datasets incorporated in Fig. 5, the fuel characteristics associated with montane and subalpine forests in the Northern Rocky Mountains – namely, ponderosa pine, lodgepole pine (*Pinus contorta*), Englemann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) are not that dissimilar structurally from forests composed of pure and mixed stands of red pine, jack pine (*Pinus banksiana*), black spruce, white spruce (*Picea glauca*) and balsam fir (*Abies balsamea*).

Reduction of crown fire rate of spread due to use of crown fraction burned functions

All of the fire modelling systems mentioned here (i.e. NEXUS, FlamMap FARSITE, FFE-FVS and FMAPlus), with the exception of BehavePlus, that integrate or link the Rothermel (1972, 1991) and Van Wagner (1977, 1993) models to predict the full range of fire behaviour apply a reduction factor to the predicted crown fire rate of spread based on a crown fraction burned (CFB) function (Table 4) as used for example in the Canadian Forest Fire Behaviour Prediction (FBP) System (Van Wagner 1989; Forestry Canada Fire Danger Group 1992).

The CFB, which indicates the proportion of tree crowns involved in the spread of the fire, varies from 0.0 (surface fire with no crown fuel involvement) to 1.0 (fully developed crown fire). In the FBP System, passive crown fire spread or intermittent crowning and continuous crowning or active crown fire spread is judged to occur at CFB values ranging from 0.1 to 0.89 and $\geq$0.9 respectively (Forestry Canada Fire Danger Group 1992).

The final rate of fire spread (R, m min^{-1}), whether surface or crown, is computed as follows:

$$R = R_s + \mathrm{CFB} \cdot (R_a - R_s) \quad (8)$$

where R_s is the predicted surface fire rate of spread (m min^{-1}) per Rothermel's (1972) model and R_a by Rothermel (1991) per Eqn 7.

The CFB adjustment scheme devised by Van Wagner (1993) provides for a gradual transition in a fire's spread rate from the initial onset of crowning (i.e. passive crown fire spread), as defined by Eqn 5, to the point of active crown fire development

Table 4. Description of computation procedures involved in predicting passive and active crown fire rate of spread in terms of crown fraction burned (CFB) within the various US fire modelling systems

FMAPlus performs the same crown fire computations as FARSITE and NEXUS

Spread regime	BehavePlus	NEXUS and FFE-FVS	FARSITE and FlamMap
Passive crown fire	Does not calculate a CFB for use in computations. Does not provide a spread rate output specifically associated with passive crown fires but does identify passive crown fires as a distinct type of fire.	Calculates CFB between 0.0 and 1.0 using a simple linear transition function between surface and active crown fire rates of spread (Scott and Reinhardt 2001). CFB values are higher than those produced by the CFB function used in FARSITE or FlamMap. Spread rate for passive crown fires is determined to be intermediate between the active crown fire and surface fire rates of spread based on the calculated CFB value.	Calculates CFB between 0.0 and 1.0 based on an exponential transition function between surface and active crown fire rates of spread developed by Van Wagner (1993). CFB values are lower than those produced by the CFB function used in NEXUS or FFE-FVS. Spread rate for passive crown fires assumed to be the same as the surface fire rate of spread.
Active crown fire	Uses Rothermel's (1991) model to predict the average active crown fire rate of spread.	Uses Rothermel's (1991) model to predict the average active crown fire rate of spread. There is also the option to apply the near-maximum crown fire rate of spread multiplier (i.e. 1.7).	Uses Rothermel's (1991) model to predict a reference active crown fire rate of spread that is then adjusted on the basis of the CFB. Because calculated CFB values are lower than those produced by the CFB function used in NEXUS and FFE-FVS, active crown fire spread rates remain less than that predicted by Rothermel's (1991) model even after an active crown fire is judged to have occurred.

based on Van Wagner's (1977) concept of a critical minimum spread rate for active crowning (R_o, m min^{-1}):

$$R_o = \frac{S_o}{\text{CBD}} \quad (9)$$

where S_o is the critical mass flow rate for solid crown flame (kg m^{-2} min^{-1}) and CBD is the canopy bulk density (kg m^{-3}). Van Wagner (1977) provided one estimate of S_o, namely 3.0 kg m^{-2} min^{-1} (Alexander 1988), based largely on a single experimental crown fire in a red pine plantation plot exhibiting a CBD of 0.23 kg m^{-3} (Van Wagner 1964). Cruz *et al.* (2005) have since confirmed the robustness of this estimate based on an examination of a relative large ($n = 37$) dataset of experimental crown fires carried out in several different conifer forest fuel complexes (Fig. 7*a*).

Dickinson *et al.* (2009) claim to have recalibrated Van Wagner's (1977) model represented by Eqn 9 on the basis of the foliar biomass per unit area or available canopy fuel load (CFL, kg m^{-2}) rather than the CBD:

$$R_o = \frac{23.4}{\text{CFL}} \quad (10)$$

This formulation implies that the propagation of active crown fire is not dependent in any way on the stand structure (i.e. height or crown depth) or, in other words, the vertical distribution of the available canopy fuel. It appears from the available experimental evidence that the Dickinson *et al.* (2009) modification of Van Wagner's (1977) R_o model is not as reliable at distinguishing active crown fires from passive crown fires as originally envisioned (Fig. 7*b*).

In deriving his estimate of S_o, Van Wagner (1977) computed the CBD as the available canopy fuel load divided by the canopy depth (Cruz *et al.* 2003*c*) and assumed that all the fuel was uniformly distributed. Admittedly, this is not always the case, for example, in multistoried stands (Reinhardt *et al.* 2006*b*) and even to a certain extent in red pine plantations (Sando and Wick 1972, pp. 6–7) such as Van Wagner (1964, 1968, 1977) worked in. Nevertheless, Alexander *et al.* (1991*b*) found that Van Wagner's (1977) simple model represented by Eqn 9 worked well at distinguishing between surface and crown fires in a black spruce–lichen woodland fuel complex that exhibited large gaps between clumps of trees and crowns that extended down to the ground surface. In their implementation of Eqn 9 in NEXUS, Scott and Reinhardt (2001) initially defined CBD as the maximum 4.5-m vertical running mean bulk density; this was later changed to a 3.0-m interval, although no reason was given (Peterson *et al.* 2005; Scott and Reinhardt 2005, 2007; Scott 2006). This represents a distinct departure from the manner in which Van Wagner (1977) calculated CBD and undoubtedly leads to higher CBD values and hence lower R_o values required for active crowning to occur. As such, it constitutes a violation of one of the fundamental assumptions of Van Wagner's (1977) active crown fire propagation model represented by Eqn 9.

The form of the CFB function varies among the fire modelling systems. FARSITE uses the original exponential form presented by Van Wagner (1993). NEXUS, however, assumes a linear adjustment when the rate of fire spread is between R_i and R_o (Scott and Reinhardt 2001). This gives distinctly different results even if the core models are the same (Fig. 8). Scott and Reinhardt (2001) explored the impact of Van Wagner's (1993) CFB function in FARSITE and found that even under extreme burning conditions, the crown fire rate of spread predicted by the Rothermel (1991) model was reduced by approximately one-third. Regardless of which CFB function is used, the result is a further increase in the underprediction bias (Stocks *et al.* 2004).

The BehavePlus modelling system (Andrews *et al.* 2008) has separately implemented the Rothermel (1972, 1991) surface and crown fire rate of spread and Van Wagner (1977) crown fire initiation and propagation models rather than attempt to directly link them using a CFB function. Thus, BehavePlus doesn't

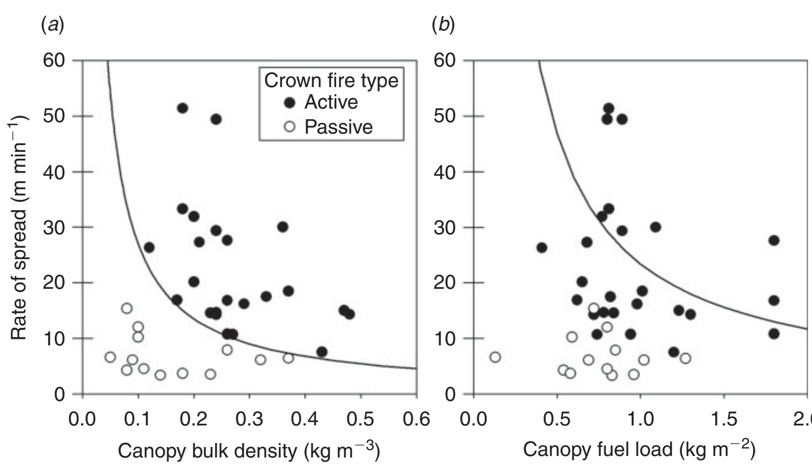

Fig. 7. Scatterplots of experimental crown fire rates of spread by type of spread regime *v.* two canopy fuel properties (adapted from Cruz *et al.* 2005). The curve in (*a*) represents Van Wagner's (1977) criterion for active crowning represented by Eqn 9, assuming an S_o value of 3.0 kg m^{-2} min^{-1}. The curve in (*b*) represents the Dickinson *et al.* (2009) recalibration of the Van Wagner (1977) criterion using canopy fuel load rather than canopy bulk density as represented by Eqn 10.

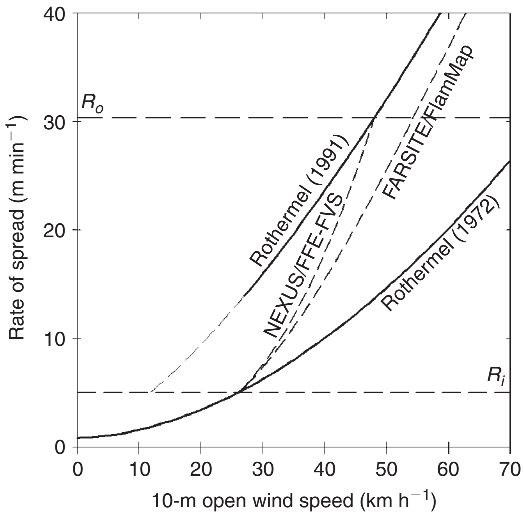

Fig. 8. Comparison of the effect of crown fraction burned functions on rate of fire spread used in the NEXUS and FFE-FVS (Scott and Reinhardt 2001; Reinhardt and Crookston 2003) *v.* FARSITE and FlamMap (Finney 2004, 2006) modelling systems in relation to the Rothermel (1972, 1991) surface and crown fire rate of spread models and Van Wagner's (1977) criteria for the critical minimum spread rates for crown fire initiation (R_i) and active crowning (R_o) for the Anderson (1982) Fuel Model 2 – Timber 2 (grass and understorey) with a canopy bulk density of 0.1 kg m^{-3}, canopy base height of 1.5 m, and a wind reduction factor of 0.2 (Albini and Baughman 1979). The following environmental conditions were held constant: slope steepness, 0%; fine dead fuel moisture, 6%; 10- and 100-h time-lag dead fuel moisture contents, 7 and 8% respectively; live woody fuel moisture content, 75%; live herbaceous fuel moisture content, 75%; and foliar moisture content, 140%. The dashed portion of the Rothermel (1991) curve represents output below the original dataset bounds for rate of spread.

provide a spread rate for passive or intermittent crowning but rather provides a transition to crowning ratio and an active crown fire spread ratio based on the values generated by Eqns 4 *v.* 1 and Eqns 7 *v.* 9 respectively in a manner analogous to Anderson's (1974) index of crowning potential as dictated by the ratio of predicted flame height *v.* an observed or measured CBH.

There is no experimental or sound theoretical evidence for a CFB effect on crown fire rate of spread. Furthermore, general observations of wildfires (e.g. Alexander *et al.* 1991*a*; Cohen *et al.* 2006) and documentation of experimental crown fires (e.g. Van Wagner 1964; Bruner and Klebenow 1979; Burrows *et al.* 1988; Fernandes *et al.* 2004; Stocks *et al.* 2004) indicate that a rather abrupt transition between surface and crown fire regimes is far more commonplace than a gradual transition as implied by a CFB function (Alexander 1998) and as illustrated in Fig. 8.

Use of uncalibrated custom fuel models

Understandably, the use of standard, stylised fuel models (Anderson 1982) in simulation studies examining fuel treatment effectiveness on potential crown fire behaviour limits the extent to which one can gauge the influence of surface fuelbed characteristics on the start and spread of crown fires. Furthermore, there is no empirical proof produced to date to substantiate that by simply increasing the number of fuel models (Scott and Burgan 2005) or reformulating Rothermel's (1972) surface fire rate of spread model (Sandberg *et al.* 2007) would greatly improve matters.

The use of calibrated custom fuel models to represent surface fuelbeds is thus seen by some as a more realistic alternative. However, the use of uncalibrated custom models (e.g. Bessie and Johnson 1995; Battaglia *et al.* 2008; Cheyette *et al.* 2008) can constitute another potential source of underprediction bias. Custom fuel models (Burgan and Rothermel 1984; Burgan 1987) are likely to be unsuccessful when developed without calibrating the predictions or tuning the parameters against field observations of fire behaviour (e.g. Hough and Albini 1978; Cruz and Fernandes 2008).

Studies that have evaluated custom fuel models in horizontally oriented fuels, such as found in conifer litter surface fuelbeds, have identified strong underprediction trends (e.g. Lawson 1972; McAlpine and Xanthopoulos 1989; Hély *et al.* 2001) and in other forest fuel complexes as well (e.g. Burrows 1994; Grabner *et al.* 1997). The effect of this underprediction trend or bias is noticeable in the studies of potential crown fire behaviour that rely on uncalibrated custom fuel models based on field sampling using methods such as those of Brown *et al.* (1982).

Agee and Lolley (2006), for example, predicted a flame height of 1.4 m for their control or untreated ponderosa pine–Douglas-fir fuel complex for simulations based on a 1-h time-lag fuel moisture content of 3% and 6.1-m open wind speeds of 36 km h^{-1}. Comparatively, the Hayman Fire in north-central Colorado (Finney *et al.* 2003; Graham 2003) went from ~5000 to 25 000 ha over a period of 12 h on 9 June 2002 under more moist fuel conditions (FDFM 6–7%) than that of the Agee and Lolley (2006) simulated situation and with a maximum U_{10} of 30–40 km h^{-1} at its peak (Alexander and Cruz 2006).

Similar unrealistic predictions of potential fire behaviour have been reported by others, for example by Page and Jenkins (2007) for lodgepole pine stands infested with mountain pine beetle (*Dendroctonus ponderosae*) in northern and north-eastern Utah and central Idaho (e.g. rates of spread of ~2.0 m min^{-1} for FDFM of 6% and 6.1-m open winds of 50 km h^{-1}) and by Stephens and Moghaddas (2005*a*, 2005*b*) for California mixed-conifer forests (rate of spread of 1.9 m min^{-1} for a 1-h time-lag fuel moisture content of 3.9% and 6.1-m open winds of 22 km h^{-1}). The low spread potential of these custom fuel model predictions explains the need for very dry fuels and high wind speeds in order to induce crown fire activity, as illustrated in Fig. 1*c*.

Other simulation modelling and interpretation issues

Selection of foliar moisture content levels

Van Wagner's (1977) crown fire initiation model is sensitive to FMC (Fuglem and Murphy 1980; Alexander 1988). Changing the FMC from 80 to 140% will almost double the surface fire intensity required for the onset of crowning (Alexander 1988). Within the simulation framework of the fire behaviour modelling systems like NEXUS, this will lead to a large increase in the critical surface fire rate of spread required for crown fire initiation and hence wind speed or fuel dryness (or both) necessary to initiate crown fire activity. Varner and Keyes (2009) recently pointed out that some modellers have assigned FMC 'values without justification or use values that lie on the extremes of published data'.

Scott and Reinhardt (2001) suggested using a constant or default FMC value of 100% as 'a reasonable approach' until better data exist. They also suggested that future research should be directed at compiling existing FMC data and then conducting field research to fill in data gaps. Keyes (2006) concluded on the basis of a review of FMC studies that a single FMC default value 'ignores established differences amongst tree species'. However, he also stated that 'For species lacking published FMC data, a low default value of 90 or 100% remains a prudently conservative assignment'. As a general rule of thumb, an FMC of 90% seems unduly low based on existing information. Chandler *et al.* (1983) regarded crown fire potential as 'high' when the FMC fell below 100%. Some authors have used an FMC of 100% in their simulation studies (e.g. Brown *et al.* 2008; Vaillant *et al.* 2009*b*), whereas others have elected to use much lower values.

Roccaforte *et al.* (2008) used an FMC of 80% in their simulations for ponderosa pine fuel complexes in north-western Arizona without any justification. Although this value might be appropriate for ponderosa pine forests in the south-western US, which typically experience their fire season much earlier in the year, it would be unduly low for other areas in the western US given the seasonal dynamics in FMC found to date in ponderosa pine. Several studies conducted in the western US indicate that the FMC typically ranges from 100 to 120% for 1-year-old ponderosa pine needles between July and September (Philpot and Mutch 1971; Agee *et al.* 2002; Finney *et al.* 2003; Faiella and Bailey 2007), the traditional peak burning period in the western US. Agee *et al.* (2002) and Faiella and Bailey (2007) in turn report FMC in the range of 250–335% and 180–220% respectively for new needle growth. Simulations should consider an aggregate or composite FMC taking into account the differences in moisture contents between new and old needles and the relative proportions of each as well as seasonal changes (cf. Van Wagner 1974). The proportion of new and 1-year and older needle growth is dependent on species, canopy position and site characteristics (Reich *et al.* 1995). Needle longevity for ponderosa pine has been reported to vary between 2 and 4 years in low to moderate elevation sites, but reaching 6 to 9 years in high-stress environments such as arid and alpine habitats (Ewers and Schmid 1981; Richardson and Rundel 1998). Assuming that new needle foliage makes up approximately one-third of the foliage biomass (Van Wagner 1967, 1974) and taking into account the midpoint of Faiella and Bailey's (2007) foliar moisture content ranges for 1-year and older needle foliage (i.e. 110%) and for new growth (i.e. 200%), a nominal FMC value for summertime conditions in ponderosa pine would be ~140%.

It appears the use of low FMC values is becoming commonplace in simulation studies examining potential crown fire behaviour. Stephens and Moghaddas (2005*a*, 2005*b*) used 75% for mixed conifers and Page and Jenkins (2007) used 70% for lodgepole pine. Neither study sampled FMC directly, referenced any previous studies of FMC or otherwise rationalised their FMC selection. Similarly, Stephens *et al.* (2009) used an FMC of 75% without any justification. In their study in ponderosa pine, Ritchie *et al.* (2007) indicated the FMC 'was estimated to be 75% since the Cone Fire burned under dry, north wind conditions following the long, dry summer'. Certainly FMC values this low have occasionally been observed (Keyes 2006). Van Wagner (1993) in fact computed FMC values that average 67% based on a weighting of the moisture contents of old needle foliage and fine, dead woody crown material relative to their separate fuel loadings (Van Wagner 1977). However, such low FMC levels have typically been reported in boreal coniferous tree species just before needle flushing in the spring (Van Wagner 1967, 1974; Fuglem and Murphy 1980).

The National Wildfire Coordinating Group (2008) recently recommended that in the absence of specific information on FMC, one should assume that the FMC is equal to the live woody fuel moisture content input given in BehavePlus, which presently allows for the FMC to vary from 30 to 300%. The moisture content of understorey shrub vegetation can reach 30% (Rothermel 1983) or less and thereby be treated as dead fuel. Existing information on the moisture contents of conifer trees and shrubs sampled at the same time and at the same location does not support this recommendation (e.g. Philpot 1963; Agee *et al.* 2002).

Some authors have selected FMC values below 30% in their application of fire behaviour modelling systems like NEXUS to insect-killed conifer forest stands (e.g. Cheyette *et al.* 2008). Given the empirical nature of Van Wagner's (1977) crown fire initiation model with respect to FMC, applying FMC values any lower than ~70% is not recommended, even if the computer software associated with modelling systems such as NEXUS or BehavePlus allow for it. What is needed is the derivation of a *C* value for use in Eqn 1 based on a carefully documented outdoor experimental fire(s) carried out at very low FMC levels in order to determine crown fire potential in canopy fuel layers comprised largely of fine, dead fuels (e.g. Kuljian and Varner 2010).

Canopy base height criteria

Another input in Van Wagner's (1977) crown fire initiation model, and one that readily favours the occurrence of crowning activity is the CBH. In fact, the natural variation in CBH would allow for a much greater effect on crowning potential than would the observed variation in FMC (Fuglem and Murphy 1980; Alexander 1988).

Van Wagner's (1977) crown fire initiation model has an empirical basis and was parameterised using the mean crown base height of the trees within a red pine plantation experimental plot (Van Wagner 1968). In their simulation studies, Ritchie *et al.* (2007) and Roccaforte *et al.* (2008) used the lowest quartile CBH value. We do not dispute the fact that the lowest quartile could possibly be a better descriptor of a fuel complex's vertical continuity than the average value when applying a physical-based model. Nonetheless, the use of the lowest quartile in the context of Van Wagner's (1977) crown fire initiation model, as represented by Eqn 1, violates one of the fundamental assumptions of this semi-empirical-based model.

Defining what constitutes an effective CBH can admittedly be difficult at times (Williamson 1999; Scott and Reinhardt 2001; Cruz *et al.* 2004; Menning and Stephens 2007; Mitsopoulos and Dimitrakopoulos 2007), especially in forest stands with highly complex vertical fuel distributions. Muraro (1971) was the first to suggest a threshold CBD value (i.e. $0.320\,kg\,m^{-3}$) as a means of quantitatively defining the CBH. Sando and Wick (1972) indicated that 'little is known about the amount of fuel required to support combustion vertically'; they ended up selecting an arbitrary threshold value as well (i.e. $0.037\,kg\,m^{-3}$), which

Williams (1977) simply doubled for his application (i.e. 0.074 $kg\,m^{-3}$). Roussopoulos (1978) arbitrarily defined CBH as the height separating the lower 5.0% of the total needle foliage load from the upper 95%.

In determining CBH, the majority of simulation studies examining potential crown fire behaviour have followed Scott and Reinhardt's (2001) definition – i.e. 'the lowest height above ground at which there is a sufficient amount of canopy fuel to propagate fire vertically into the canopy'. Scott and Reinhardt (2001) also selected an arbitrary CBD value (0.011 $kg\,m^{-3}$) as the basis for determining CBH. In the intervening years, this approach has come to be an accepted standard with little or no questioning of its origin. Reinhardt *et al.* (2006*a*) readily admit that this threshold value is 'not based on any kind of combustion physics, but it seems to perform well', although they offer no details regarding their performance testing. Thus, the lack of an objectively defined threshold CBD value for determining CBH remains a continuing research need (Alexander 2006).

Meaning of the two crown fire hazard indices

TI and CI values are outputs of NEXUS, FFE-FVS and FMA-Plus but not of the BehavePlus, FARSITE or FlamMap modelling systems. The TI and CI concept were initially introduced by Scott (1998*b*) and later elaborated on by Scott and Reinhardt (2001) for the purpose of assessing crown fire hazard in coniferous forests. Scott (2008) has also extended the methodology to shrubland and open forest woodland fuel complexes. The TI might have been more appropriately termed the 'passive or intermittent crowning index' as torching is more commonly associated with calm to light winds (e.g. Lawson 1972; Dyrness and Norum 1983) and a single tree torching does not make for even a passive crown fire (Forestry Canada Fire Danger Group 1992). Similarly, the CI could have been labelled the 'active or continuous crowning index'.

Although the TI and CI are to be regarded as relative numerical values (Fulé *et al.* 2004; Roccaforte *et al.* 2008; Stephens *et al.* 2009), Scott and Reinhardt (2001) chose to express both indices in terms of the wind speed (in either $km\,h^{-1}$ or miles h^{-1}) as taken at a height of 6.1 m (20 feet) above open ground per the standard for fire danger rating and fire behaviour prediction used in the US (Deeming *et al.* 1977; Rothermel 1983). Later on, Scott (2006) expressed TI and CI in terms of the 10-m open wind standard used for fire danger rating and fire behaviour prediction in Canada (Lawson and Armitage 2008) and elsewhere (e.g. Australia and New Zealand).

The present practice of calculating TI and CI values by various authors does not readily allow for direct comparison between different studies or assessments. For example, the fuel moisture contents selected are based on one of the various scenarios presented by Rothermel (1991) or on percentile values derived from a fire weather database, each of which has value. Added to this is the fact that both the FDFM (Rothermel 1983) and the NFDRS 1-h time-lag fuel moisture content (Fosberg and Deeming 1971; Deeming *et al.* 1977) are used in computing the two crown-fire hazard indices and they do not result in the same numerical value for a given set of weather conditions. Some authors have failed to specify the associated environmental conditions (e.g. Graves and Neuenschwander 2001; Fiedler *et al.* 2004; Monleon *et al.* 2004; Mason *et al.* 2007) or the description remains vague (e.g. Moghaddas and Craggs 2007). Furthermore, some authors have failed to explicitly specify the FMC applied in their simulations (e.g. Stephens 1998; Monleon *et al.* 2004; Johnson *et al.* 2007; DeRose and Long 2009). The situation is further complicated by the lack of standardisation of the index scale as dictated by the use of two different units of measure (i.e. $km\,h^{-1}$ and miles h^{-1}) and to a much lesser extent, two different open wind-speed exposure heights (i.e. 6.1 and 10 m). To make matters worse, some authors have now chosen to express TI and CI outputs in $m\,s^{-1}$ (e.g. Ritchie *et al.* 2007; Finkral and Evans 2008). The basic premise of any index is that it has a consistent scale.

Summary and concluding remarks

The ready availability of a multitude of fire modelling systems in the US in recent years has led to their widespread use in numerous simulation studies aimed at assessing various fire behaviour characteristics associated with specific fuel complex structures, including the propensity for crown fire initiation and spread (McHugh 2006). The results of these simulations, often aimed at evaluating fuel treatment effectiveness, are in turn utilised in a whole host of applications (e.g. Scott 2003; Fiedler *et al.* 2004; Skog *et al.* 2006; Johnson *et al.* 2007; Finkral and Evans 2008; Huggett *et al.* 2008; Johnson 2008; Reinhardt *et al.* 2010) and thus have significant implications for public and wildland firefighter safety, community fire protection, fire management policy-making, and forest management practices. As Cheney (1981) has noted, 'The reality of fire behaviour predictions is that overestimates can be easily readjusted without serious consequences; underestimates of behaviour can be disastrous both to the operations of the fire controller and the credibility of the person making the predictions'.

A critical review of several of these simulation studies, as documented here, has found that the results are often unrealistic for a variety of reasons. It's recognised that the authors of these studies commonly point out the limitations of the models and modelling systems being used through a customary disclaimer concerning the unknowns regarding crown fire behaviour (e.g. Stephens *et al.* 2009). Nevertheless, the fact that the fuel treatment evaluation studies referenced here are based on modelling systems that utilised model linkages for gauging potential crown fire behaviour that have not previously undergone any form of performance evaluation against independent datasets or any empirical observations should be of concern. There appears, however, to be an aversion within an element of the fire research community to do so (e.g. Scott and Reinhardt 2001; Scott 2006; Stephens *et al.* 2009). Nevertheless, such testing is now generally regarded as a basic tenet of modern-day model development and evaluation (Jakeman *et al.* 2006).

Fire modelling systems like NEXUS (Scott and Reinhardt 2001), FFE-FVS (Reinhardt and Crookston 2003), FARSITE (Finney 2004), FMAPlus (Carlton 2005), FlamMap (Finney 2006), and BehavePlus (Andrews *et al.* 2008) that are based on separate implementations or linkages between Rothermel's (1972, 1991) rate of fire spread models and Van Wagner's (1977, 1993) crown fire transition and propagation models have been shown to have a marked underprediction bias when used to assess potential crown fire behaviour. What has been allowed to evolve is a family of modelling systems composed of independently developed, linked models that were never intended to work together, are sometimes based on very limited data, and may propagate errors beyond acceptable limits.

We have documented here the sources of the bias based on empirical evidence in the form of published experimental fire and wildfire datasets. By analysing model linkages and components, we have described the primary sources of such bias, namely: (1) incompatible model linkages; (2) use of surface and crown fire rate of spread models that have an inherent underprediction bias; and (3) reduction in crown fire rate of spread based on use of unsubstantiated CFB functions. The use of uncalibrated, custom fuel models to represent surface fuelbeds is considered another potential source of bias.

Our analysis has also shown that the crown fire initiation underprediction bias inherent in all of these fire modelling systems could possibly be rectified by modifying the method used to calculate the surface fireline intensity for the purposes of assessing crown fire initiation potential, namely using Nelson's (2003) model to estimate t_r in place of Anderson's model (1969). Other modelling systems exist for predicting the likelihood of crown fire initiation and other aspects of crown fire behaviour (Alexander *et al.* 2006; Cruz *et al.* 2006*b*, 2008). Mitsopoulos and Dimitrakopoulos (2007) have, for example, made extensive use of this suite of models in their assessment of crown fire potential in Aleppo pine (*Pinus halepensis*) forests in Greece. These systems are based on models that have undergone performance evaluations against independent datasets and been shown to be reasonably reliable (Cruz *et al.* 2003*b*, 2004, 2006*b*; Cronan and Jandt 2008). Resolving the underprediction bias associated with predicting active crown fire rate of spread inherent in the Rothermel (1991) model would require substantial changes, including a reassessment of the use of a CFB function, if not complete replacement with a more robust empirically developed model (Cruz *et al.* 2005) that has been extensively tested (Alexander and Cruz 2006) or a physically based one that has undergone limited testing (Butler *et al.* 2004).

Alexander (2007) has emphasised that assessments of wildland fire potential involving simulation modelling must be complemented with fire behaviour case study knowledge and by experienced judgment. This review has revealed an overwhelming need for the research users of fire modelling systems to be grounded in the theory and proper application of such tools, including a solid understanding of the assumptions, limitations and accuracy of the underlying models as well as practical knowledge of the subject phenomena (Brown and Davis 1973; Albini 1976; Alexander 2009*a*, 2009*b*).

List of symbols, quantities and units used in equations and text

C, criterion for initial crown combustion (kW$^{2/3}$ kJ^{-1} kg m$^{-5/3}$)
CBD, canopy bulk density (kg m^{-3})
CBH, canopy base height (m)
CFL, canopy fuel load (kg m^{-2})
CFB, crown fraction burned
CI, crowning index (km h^{-1})
FDFM, fine dead fuel moisture (%)
FMC, foliar moisture content (%)
h, heat of ignition (kJ kg^{-1})
H, low heat of combustion (kJ kg^{-1})
I_B, fireline intensity (kW m^{-1})
I_o, critical surface fire intensity for initial crown combustion (kW m^{-1})
I_R, reaction intensity (kW m^{-2})
r, rate of fire spread (m s^{-1})
R, final rate of fire spread, surface or crown (m min^{-1})
R_a, active crown fire rate of spread (m min^{-1})
R_i, critical surface fire rate of spread for crown fire initiation (m min^{-1})
R_s, surface fire rate of spread (m min^{-1})
R_o, critical minimum spread rate for active crowning (m min^{-1})
R_{10}, predicted surface fire rate of spread for Fuel Model 10 using a 0.4 wind reduction factor (m min^{-1})
S_o, critical mass flow rate for solid crown flame (kg m^{-2} min^{-1})
t_r, flame front residence time (s)
TI, torching index (km h^{-1})
U_{10}, 10-m open wind speed (km h^{-1})
w, fuel consumed in the active flaming front and by glowing or smouldering combustion following passage of the front (kg m^{-2})
w_a, fuel consumed in the active flaming front (kg m^{-2})

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Does increased forest protection correspond to higher fire severity in frequent-fire forests of the western United States?

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Abstract. There is a widespread view among land managers and others that the protected status of many forestlands in the western United States corresponds with higher fire severity levels due to historical restrictions on logging that contribute to greater amounts of biomass and fuel loading in less intensively managed areas, particularly after decades of fire suppression. This view has led to recent proposals—both administrative and legislative—to reduce or eliminate forest protections and increase some forms of logging based on the belief that restrictions on active management have increased fire severity. We investigated the relationship between protected status and fire severity using the Random Forests algorithm applied to 1500 fires affecting 9.5 million hectares between 1984 and 2014 in pine (*Pinus ponderosa*, *Pinus jeffreyi*) and mixed-conifer forests of western United States, accounting for key topographic and climate variables. We found forests with higher levels of protection had lower severity values even though they are generally identified as having the highest overall levels of biomass and fuel loading. Our results suggest a need to reconsider current overly simplistic assumptions about the relationship between forest protection and fire severity in fire management and policy.

Key words: biodiversity; climate; fire frequency; fire severity; fire suppression; Gap Analysis Program levels; logging; protected areas.

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Introduction

It is a widely held assumption among federal land management agencies and others that a lack of active forest management of some federal forestlands—especially within relatively frequent-fire forest types such as ponderosa pine (*Pinus ponderosa*) and mixed conifers—is associated with higher levels of fire severity when wildland fires occur (USDA Forest Service 2004, 2014, 2015, 2016). This prevailing forest/fire management hypothesis assumes that forests with higher levels of protection, and therefore less logging, will burn more intensely due to higher fuel loads and forest density. Recommendations have been made to increase logging as fuel reduction and decrease forest protections before wildland fire can be more extensively reintroduced on the landscape after decades of fire suppression (USDA Forest Service 2004, 2014, 2015, 2016). The concern follows that, in the absence of such a shift in forest management, fires are burning too severely and may adversely affect forest resilience (North et al. 2009, 2015, Stephens et al. 2013, 2015, Hessburg 2016). Nearly every fire season, the United States Congress introduces forest management legislation based on this view and aimed at increasing mechanical fuel treatments via intensive logging and weakened forest protections.

However, the fundamental premise for this fire management strategy has not been rigorously tested across broad regions. We broadly assessed the influence of forest protection levels on fire severity in pine and mixed-conifer forests of the western United States with relatively frequent-fire regimes to test this assumption. We used vegetation burn severity data from all fires >405 ha over a three-decade period, 1984–2014, in forests with varying levels of protection.

Study area

Pine and mixed-conifer forests at low/mid-elevations, where historical fires were relatively frequent, are broadly distributed across several ecoregions in the western United States (Fig. 1; Appendix S1: Table S1). Although ponderosa pine often dominates these forests, they can also include Jeffrey pine (*Pinus jeffreyi*), which in places intermix with, and are similar to, ponderosa pine forests, and Madrean pine–oak (*Quercus* spp.) forests with a diversity of pines. Mixed-conifer forests at low/mid-elevations are also broadly distributed across multiple ecoregions (Fig. 1). They can include additional pines (e.g., lodgepole pine, *Pinus contorta*; sugar pine, *Pinus lambertiana*), true firs (*Abies* spp.), Douglas-fir (*Pseudotsuga menzeisii*), and incense-cedar (*Calocedrus decurrens*).

Methods

We used Gap Analysis Program (GAP) protection classes (USGS 2012), as described below, to determine whether areas with the most protection (i.e., GAP1 and GAP2) had a tendency to burn more severely than areas where intensive management is allowed (i.e., GAP3 and GAP4). We compared satellite-derived burn severity data for 1500 fires affecting 9.5 million hectares from years for which there were available data (1984–2014) among four different forest protection levels (Fig. 1), accounting for variation in topography and climate. We analyzed fires within relatively frequent-fire forest types comprised of pine and mixed-conifer forests mainly because these are the predominant forest types at low to mid-elevations in the western United States, there is a large data set on fire occurrence, and they have been a major concern of land managers for some time due to decades of fire suppression. We defined geographic extent of forest types from the Biophysical Settings data set (BpS) (Rollins 2009; *public communication*, http://www.landfire.gov) that derived forest maps from satellite imagery and represents plant communities based on NatureServe's Ecological Systems classification. Baker (2015) noted that some previous work found ~65% classification accuracy of this system with regard to specific forest types and, accordingly, he analyzed groups of related forest types in order to improve accuracy. We followed his approach (see Appendix S1: Table S1). The categories selected from the Biophysical Settings map were ponderosa/Jeffrey pine and mixed-conifer forest types with relatively frequent-fire regimes (e.g., Swetnam and Baisan 1996, Taylor and Skinner 1998, Schoennagel et al. 2004, Stephens and Collins 2004, Sherriff et al. 2014), compared to other forest types with different fire regimes such as high-elevation forests and many coastal forests not studied herein. Forest types in our study totaled 29.2 million hectares (Fig. 1; Appendix S1: Table S1). We used the BpS data to capture areas that were classified as forests before fire, because postfire vegetation maps can potentially show these same areas as temporarily changed to other vegetation types. We sampled our response and predictor variables on an evenly spaced 90 × 90 m grid within these forest types using ArcMap 10.3 (ESRI 2014). This created a data set of 5,580,435 independent observations from which we drew our random samples to create our models. The 90-m spacing was chosen because it was the smallest spacing of points that was computationally practical with which to operate.

Fires

The Monitoring Trends in Burn Severity project (MTBS, *public communication*, http://www.mtbs.gov) is a U.S. Department of Interior and Department of Agriculture-sponsored program that has compiled burn severity data from satellite imagery, which became available in 1984, for fires >405 ha, and was current up to 2014 (Eidenshink et al. 2007). The MTBS Web site allows bulk download of spatial products that include two closely related indices of burn severity: differenced normalized burn ratio (dNBR) (Key and Benson 2006) and relative differenced normalized burn ratio (RdNBR) (Miller and Thode 2007). Both indices are calculated from Landsat TM and ETM satellite imagery of reflected light from the earth's surface at infrared wavelengths from before and after fire to

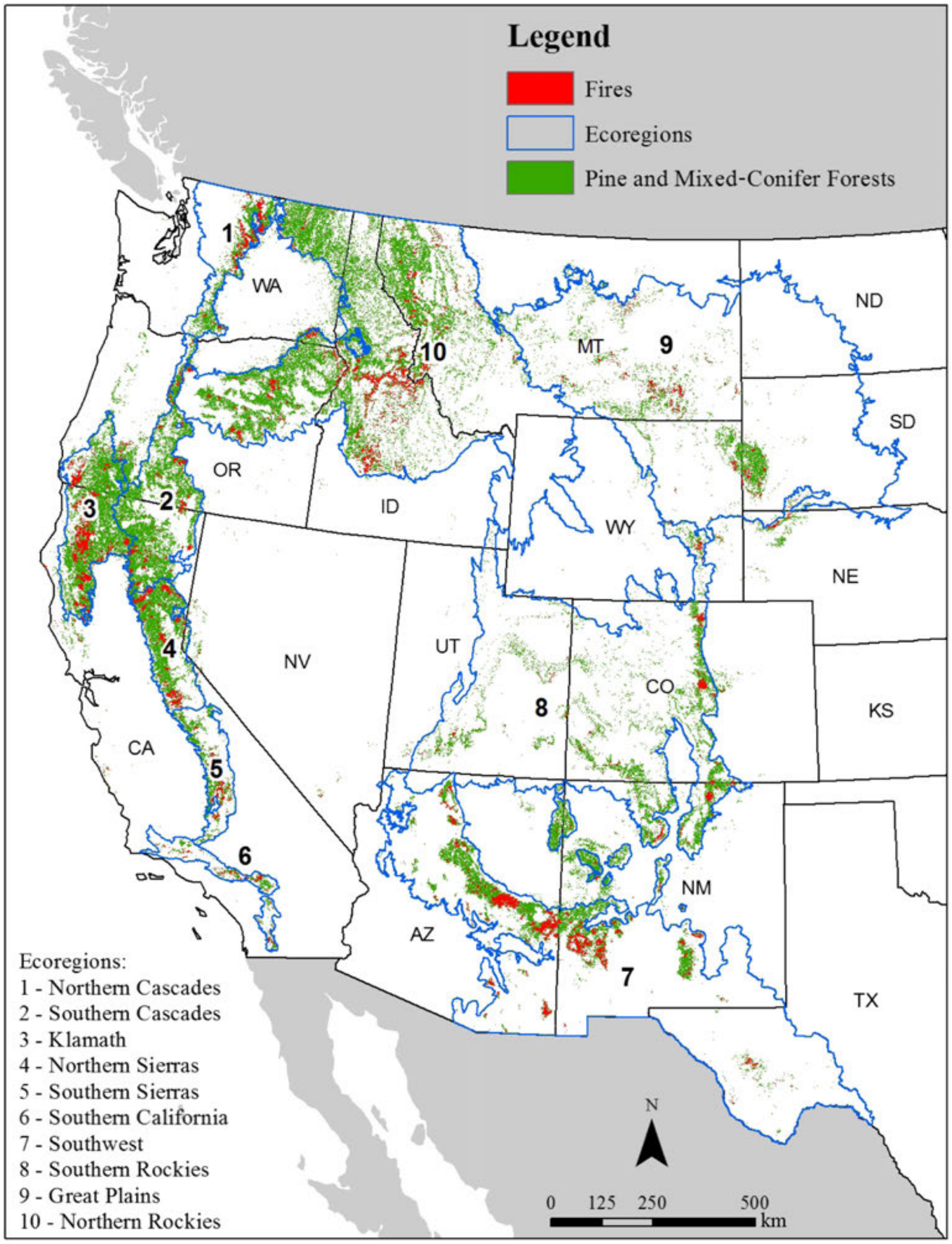

Fig. 1. Pine and mixed-conifer forests, fires, and ecoregions analyzed in this study.

measure associated changes in vegetation cover and soil characteristics. We defined burn severity with the RdNBR index because it adjusts for prefire conditions at each pixel and provides a more consistent measure of burn severity than dNBR when studying broad geographic regions with many different vegetation types (Miller et al. 2009*a*, Norton et al. 2009). RdNBR values typically range from negative 500 to 1500 with values further away from zero representing greater change from prefire conditions. Negative values represent vegetation growth and positive values increasing levels of overstory vegetation mortality. The RdNBR values could be used to classify fires into discrete burn severity classes of low, medium, and high but this was not performed in our study, as we desired to have a continuous response variable in our models.

We intersected forest sampling points with fire perimeters downloaded from MTBS to determine fires that occurred in our analysis area, and censored fires with <100 sampling points (81 ha). The remaining points represented sampling locations from 2069 fires (Fig. 1). We extracted RdNBR values at each sampling point as our response variable as well as predictor variables that included topography, geography, climate, and GAP status. These sampling points were used to investigate the relationship between forest protection levels and burn severity (Appendix S1: Tables S2 and S3). We chose topographic and climatic variables based on previous studies that quantified the relationship between burn severity, topography, and climate (Dillon et al. 2011, Kane et al. 2015).

Topographic and climatic data

To account for the effects of topographic and climatic variability, we derived several topographic indices (Appendix S1: Table S2) from seamless elevation data (*public communication*, http://www.landfire.gov/topographic.php) downscaled to 90-m^2 spatial resolution due to computational limits when intersecting sampling points. These indices capture categories of topography, including percentage slope, surface complexity, slope position, and several temperature and moisture metrics derived from aspect and slope position. We used the Geomorphometry and Gradient Metrics Toolbox version 2.0 (*public communication*, http://evansmurphy.wix.com/evansspatial) to compute these metrics. We also computed several temperature and precipitation variables (Appendix S1: Table S3) by downloading climatic conditions for each month from 1984 to 2014 from the PRISM climate group (*public communication*, http://prism.oregonstate.edu). Climate grids record precipitation and minimum, mean, and maximum temperature at a 4-km grid scale created by interpolating data from over 10,000 weather stations. To determine the departure from average conditions, we subtracted each climate grid by its 30-yr mean monthly value. These "30-yr Normals" data sets were also downloaded from the PRISM Web site and reflected the mean values from the most recent full decades (1981–2010). We determined mean seasonal values with summer defined as the mean of July, August, and September of the year before a given fire; fall being the mean of October, November, and December of the previous year; winter the mean of January, February, and March of the current year of a given fire; and spring the mean of April, May, and June of the current year.

Protected area status and ecoregion classification

We used the Protected Areas Database of the United States (PAD-US; USGS 2012) to determine forest protection status, which is the U.S. official inventory of protected open space. The PAD-US includes all federal and most State conservation lands and classifies these areas with a GAP ranking code (see map at: http://gis1.usgs.gov/csas/gap/viewer/padus/Map.aspx). The GAP status code (herein referred to interchangeably as GAP class or protection status) is a metric of management to conserve biodiversity with four relative categories. GAP1 is protected lands managed for biodiversity where disturbance events (e.g., fires) are generally allowed to proceed naturally. These lands include national parks, wilderness areas, and national wildlife refuges. GAP2 is protected lands managed for biodiversity where disturbance events are often suppressed. They include state parks and national monuments, as well as a small number of wilderness areas and national parks with different management from GAP1. GAP3 is lands managed for multiple uses and are subjected to logging. Most of these areas consist of non-wilderness USDA Forest Service and U.S. Department of Interior Bureau of Land Management lands as well as state trust lands. GAP4 is lands with no mandate for protection such as tribal, military, and private lands. GAP status is relevant to the intensity of both current and past managements.

We made one modification to GAP levels by converting Inventoried Roadless Areas (IRAs) from the 2001 Roadless Area Conservation Rule (S_USA.RoadlessArea_2001, *public communication*, http://data.fs.usda.gov/geodata/edw/datasets.php) to GAP2 unless these areas already were defined as GAP1. We considered most IRAs as GAP2 given they are prone to policy changes and because they allow for certain limited types of logging (e.g., removal of predominately small trees for fuel reduction in some circumstances).

However, we note that very little logging has occurred within IRAs since the Roadless Rule, although there occasionally have been proposals to log portions of some IRAs pre- and postfire, and fire suppression often occurs.

We modified level III ecoregions (U.S. Environmental Protection Agency (EPA) 2013) to create areas of similar climate and geography (Fig. 1). We did this by extracting ecoregions and combining adjacent provinces in our study region.

Random Forests analysis

We investigated the relationship between protection status and burn severity using the data-mining algorithm Random Forests (RF) (Breiman 2001) with the "randomForestSRC" add-in package (Ishwaran and Kogalur 2016) in R (R Core Team 2013). This algorithm is an extension of classification and regression trees (CART) (Breiman et al. 1984) that recursively partitions observations into groups based on binary rule splits of the predictor variables. The main advantage of using RF in our study is that it can work with spatially autocorrelated data (Cutler et al. 2007). It can also model complex, nonlinear relationships among variables, makes no assumption of variable distributions (Kane et al. 2015), and produces accurate predictions without overfitting the available data (Breiman 2001).

Our independent observations were a random subset of our 5.5 million points, from which we drew three random samples of 25,000 points each. Each sample consisted of 500 fires randomly selected without replacement from the pool of 2069 fires. Fifty points were then randomly selected within each of the 500 fires. Our dependent variables were all continuous (Appendix S1: Tables S2 and S3) except for the main variable of interest, protected area status, which included the four GAP levels. The three observation samples were used to create three RF model runs, each consisting of 1000 regression trees. We conducted three RF model runs to assess whether our random samples of 25,000 points produced fairly consistent results.

The RF algorithm samples approximately 66% of the data to build the regression trees, and the remaining data are used for validation and to assess variable importance. We used this validation sample to determine the amount of variance explained and variable importance. The algorithm also produces individual variable importance measures by calculating differences in prediction mean-square-error before and after randomly permuting each dependent variable's values. Variable importance is a measure of how much each variable contributes to the model's overall predicative accuracy.

Unlike linear models, RF does not produce regression coefficients to examine how a change in a predictor variable affects the response variable. The analogy to this in RF is the partial dependence plot which is a graphical depiction of how the response will change with a single predictor while averaging out the effects of the other predictors, such as the climatic and topographic variables (Cutler et al. 2007). We used this approach, in addition to using RF to determine overall variable importance as described above, in order to determine the effect of GAP status, in particular, on fire severity, while averaging out effects of climate and topography.

Mixed-effects analysis

We performed a linear mixed-effects analysis using the "nlme" add-on package in R (Pinheiro et al. 2015). We used a random intercept model and identified year of fire (n = 31) and ecoregion (n = 10) as random effects. Similar to our RF models, our independent observations were a random subset of our 5.5 million points but for these models we drew three random samples of 50,000 points each. Each sample consisted of 500 fires randomly selected without replacement, and within each of those fires, 100 points were randomly selected. Our dependent variables were the same used in our RF models, and we log-transformed the non-normal variables of slope, surface roughness, and topographic radiation aspect index. We removed dependent variables that were correlated with each other (Pearson's $r > 0.5$), retaining 21 of 45 candidate dependent variables, and centered these on their means. Model reduction was performed in a stepwise process using bidirectional elimination with Bayesian information criterion selection criterion.

Spatial autocorrelation analysis

Spatial autocorrelation (SA) is the measure of similarity between pairs of observations in relationship to the distance between them. Ecological variables are inherently autocorrelated because

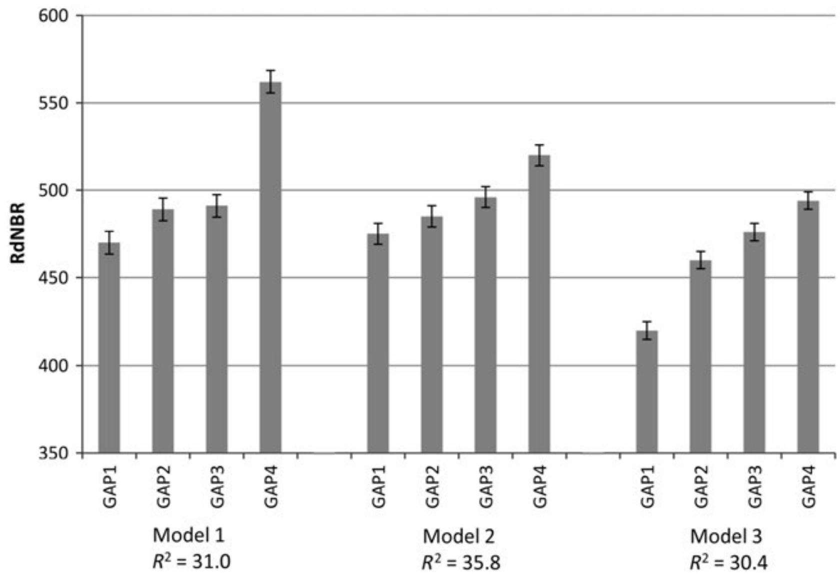

Fig. 2. Random Forests partial dependence of protection status vs. RdNBR burn severity for each model (n = 25,000). The variance explained is shown as pseudo R^2.

landscape attributes that are closer together are often more similar than those that are far apart.

We assessed the SA in the Pearson residuals with inspection of Moran's I autocorrelation index using the "APE" package add-in in R (Paradis et al. 2004) after removing points that shared the same x and y coordinates. Moran's I is an index that ranges from −1 to 1 with the sign of the values indicating strength and direction of SA. Values close to zero are considered to have a random spatial pattern. Our mixed-effects models all had a Moran's I values statistically different from 0 at the 95% confidence level ($P < 0.001$) so we included a spatial correlation structure in our model using the "nlme" package in R. Of Gaussian, exponential, linear, and spherical spatial correlation structures, we determined that the exponential structure produced the lowest Akaike's information criterion (AIC). Despite these additions, our second measurements still found relatively small, but significant, autocorrelation (Moran's I for model runs 1, 2, 3 = 0.10, 0.08, 0.10, all $P < 0.001$).

Results

With regard to ranking of variables in the model runs, variable importance plots from the three RF model runs show that protection status was consistently ranked as one of the 10 most important of the 45 variables in explaining burn severity (Appendix S1: Table S4). The most important variable explaining burn severity was ecoregion for models 1 and 2 and maximum temperature from the previous fall for model 3.

With regard to the GAP status variable in particular, after averaging out the effects of climatic and topographic variables, the RF partial dependence plots show an increasing trend of fire severity with decreasing protection status (Fig. 2). Fires in GAP4 had mean RdNBR values greater than two standard errors higher than all other GAP levels. Fires in GAP3 had mean RdNBR values two standard errors higher than GAP1 in all model runs. GAP3 differences with GAP2 were less pronounced with only one model showing differences greater than two standard errors. Fires in GAP1 were consistently the least severe, being two standard errors less than GAP3 in all model runs and two standard errors less than GAP2 in two of three model runs.

Our mixed-effects models validated these findings with similar results (Fig. 3, Appendix S1: Table S5). Like our RF models, our linear mixed-effects models showed GAP4 fires to have significantly higher RdNBR values and GAP1 fires to have significantly lower RdNBR values when compared to all other GAP classes. Fires in GAP

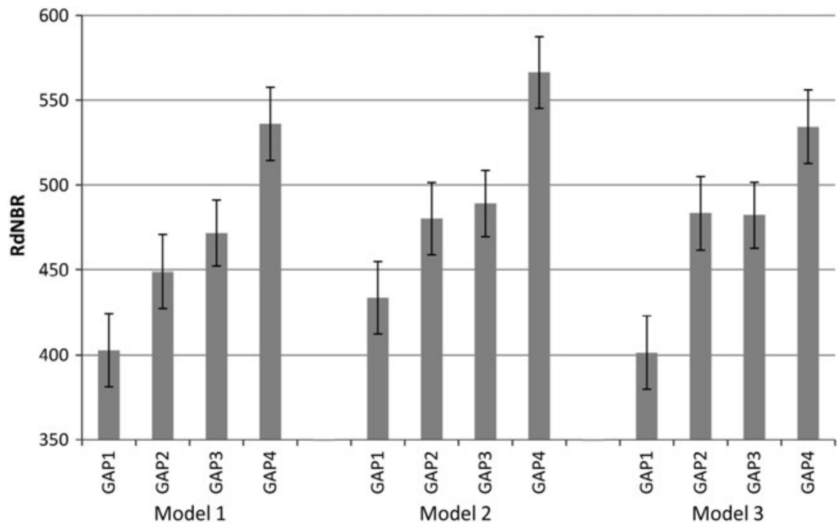

Fig. 3. Linear mixed effects models of protection status vs. RdNBR burn severity (n = 50,000).

status levels 2 and 3 were not significantly different in the mixed-effects models. Although the level of autocorrelation was significant, it was small in our model (Moran's I ~0.1) and not enough to account for such a substantial difference in burn severity among protection classes.

Discussion

Protected forests burn at lower severities

We found no evidence to support the prevailing forest/fire management hypothesis that higher levels of forest protections are associated with more severe fires based on the RF and linear mixed-effects modeling approaches. On the contrary, using over three decades of fire severity data from relatively frequent-fire pine and mixed-conifer forests throughout the western United States, we found support for the opposite conclusion—burn severity tended to be higher in areas with lower levels of protection status (more intense management), after accounting for topographic and climatic conditions in all three model runs. Thus, we rejected the prevailing forest management view that areas with higher protection levels burn most severely during wildfires.

Protection classes are relevant not only to recent or current forest management practices but also to past management. Millions of hectares of land have been protected from logging since the 1964 Wilderness Act and the 2001 Roadless Rule, but these areas are typically categorized as such due to a lack of historical road building and associated logging across patches >2000 ha, while GAP3 lands, for instance, such as National Forests lands under "multiple use management," have generally experienced some form of logging activity over the last 80 yr.

We expect that the effects of historic logging from nearly a century ago to gradually lessen over time, as succession and natural disturbance processes reestablish structural and compositional complexity, but it was beyond the scope of this study to attempt to assess the relative role of recent vs. historical logging. Similarly, industrial fire suppression programs that intensified in the 1940s influenced fire extent across forest protection classes. While more recent let-burn policies have been applied in GAP1 and GAP2 forests in some circumstances, evidence indicates that protected forests nevertheless remain in a substantial fire deficit, relative to the prefire suppression era (Odion et al. 2014, 2016, Parks et al. 2015). Thus, we believe it is unlikely that recent decisions to allow some backcountry fires to burn, largely unimpeded, account for much of the differences in fire severity among protection classes that we found, simply because such let-burn policies have not been extensive enough to remedy the ongoing fire deficit.

While forests in different protection classes can vary in elevation, with protected forests often occupying higher elevations, our results indicate that protection class itself produced notable differences in fire severity after averaging out the effects of elevation and climate (see Fig. 2 and *Results* above). In our study, GAP1 forests were 284 m on average higher in elevation than GAP4 forests, while GAP1 forests experienced lower fire severity. This is the opposite of expectations if elevation was a key influence because higher elevation forests are associated with higher fire severity (see, e.g., Schoennagel et al. 2004, Sherriff et al. 2014). We note that we are not the first to determine that increased fire severity often occurs in forests with an active logging history (Countryman 1956, Odion et al. 2004).

Prevailing forest–fire management perspectives vs. alternative views

An extension of the prevailing forest/fire management hypothesis is that biomass and fuels increase with increasing time after fire (due to suppression), leading to such intense fires that the most long-unburned forests will experience predominantly severe fire behavior (e.g., see USDA Forest Service 2004, Agee and Skinner 2005, Spies et al. 2006, Miller et al. 2009*b*, Miller and Safford 2012, Stephens et al. 2013, Lydersen et al. 2014, Dennison et al. 2014, Hessburg 2016). However, this was not the case for the most long-unburned forests in two ecoregions in which this question has been previously investigated—the Sierra Nevada of California and the Klamath-Siskiyou of northern California and southwest Oregon. In these ecoregions, the most long-unburned forests experienced mostly low/moderate-severity fire (Odion et al. 2004, Odion and Hanson 2006, Miller et al. 2012, van Wagtendonk et al. 2012). Some of these researchers have hypothesized that as forests mature, the overstory canopy results in cooling shade that allows surface fuels to stay moister longer into fire season (Odion and Hanson 2006, 2008). This effect may also lead to a reduction in pyrogenic native shrubs and other understory vegetation that can carry fire, due to insufficient sunlight reaching the understory (Odion et al. 2004, 2010).

Another fundamental assumption is that current fires are becoming too large and severe compared to recent historical time lines (Agee and Skinner 2005, Spies et al. 2006, Miller et al. 2009*b*, Miller and Safford 2012, Stephens et al. 2013, Lydersen et al. 2014, Dennison et al. 2014, Hessburg 2016). However, others have shown that this is not the case for most western forest types. For instance, using the MTBS (www.mtbs.gov) data set, Picotte et al. (2016) found that most vegetation groups in the conterminous United States exhibited no detectable change in area burned or fire severity from 1984 to 2010. Similarly, Hanson et al. (2009) found no increase in rates of high-severity fire from 1984 to 2005 in dry forests within the range of the northern spotted owl (*Strix occidentalis caurina*) based on the MTBS data set. Using reference data and records of high-severity fire, Baker (2015) found no significant upward trends in fire severity from 1984 to 2012 across all dry western forest regions (25.5 million ha), nearly all of which instead were too low or were within the range of historical rates. Parks et al. (2015) modeled area burned as a function of climatic variables in western forests and non-forest types, documenting most forested areas had experienced a fire deficit (observed vs. expected) during 1984 to 2012 that was likely due to fire suppression.

Whether fires are increasing or not depends to a large extent on the baseline chosen for comparisons (i.e., shifting baseline perspective, Whitlock et al. 2015). For instance, using time lines predating the fire suppression era, researchers have documented no significant increases in high-severity fire for dry forests across the West (Williams and Baker 2012*a*, Odion et al. 2014) or for specific regions (Williams and Baker 2012*b*, Sherriff et al. 2014, Tepley and Veblen 2015). Future trends, with climate change and increasing temperatures, may be less simple than previously believed, due to shifts in pyrogenic understory vegetation (Parks et al. 2016).

This is more than just a matter of academic debate, as most forest management policies assume that fire, particularly high-severity fire, is increasing, is in excess of recent historical baselines, and needs to be reduced in size, intensity, and occurrence over large landscapes to prevent widespread ecosystem damages (policy examples include USDA Forest Service 2002, Healthy Forests Restoration Act 2003, USDA Forest Service 2009, HR 167: Wildfire Disaster Funding Act 2015). However, large fires (landscape scale or the so-called megafires) produce myriad ecosystem benefits underappreciated by most land managers and decision-makers (DellaSala and Hanson 2015*a*, DellaSala et al. 2015). High-severity fire patches, in particular, provide a pulse of "biological legacies" (e.g., snags, down logs, and native shrub patches) essential for complex early seral associates (e.g., many bird species) that link seral stages from new forest to old growth (Swanson et al. 2011, Donato et al. 2012, DellaSala et al. 2014, Hanson 2014, 2015, DellaSala and Hanson 2015*a*). Complex early seral forests are most often logged after fire, which, along with aggressive fire suppression, exacerbates their rarity and heightens their conservation importance (Swanson et al. 2011, DellaSala et al. 2014, 2015, Hanson 2014).

Limitations

One limitation of our study is that, due to the coarseness of the management intensity variables that we used (i.e., GAP status), we cannot rule out whether low intensities of management decreased the occurrence of high-severity fire in some circumstances. However, the relationship between forest density/fuel, mechanical fuel treatment, and fire severity is complex. For instance, thinning without subsequent prescribed fire has little effect on fire severity (see Kalies and Yocum Kent 2016) and, in some cases, can increase fire severity (Raymond and Peterson 2005, Ager et al. 2007, Wimberly et al. 2009) and tree mortality (see, e.g., Stephens and Moghaddas 2005, Stephens 2009: Figure 6)—the effects depend on the improbable co-occurrence of reduced fuels (generally a short time line, within a decade or so) and wildfire activity (Rhodes and Baker 2008) and can be over-ridden by extreme fire weather (Bessie and Johnson 1995, Hély et al. 2001, Schoennagel et al. 2004, Lydersen et al. 2014). Empirical data from actual fires also indicate that postfire logging can increase fire severity in reburns (Thompson et al. 2007), despite removal of woody biomass (tree trunks) described by land managers as forest fuels (Peterson et al. 2015). While our study did not specifically test for these effects, such active forest management practices are common on GAP3 and GAP4 lands. Recognizing these limitations, researchers have stressed the need for managers to strive for coexistence with fire by prioritizing fuel reduction nearest homes and allowing more fires to occur unimpeded in the backcountry (Moritz 2014, DellaSala et al. 2015, Dunn and Bailey 2016, Moritz and Knowles 2016).

Follow-up research at finer scales is needed to determine management emphasis and history in relation to fire severity. However, we believe our findings are robust at the subcontinental and ecoregional scales.

Conclusions

In general, our findings—that forests with the highest levels of protection from logging tend to burn least severely—suggest a need for managers and policymakers to rethink current forest and fire management direction, particularly proposals that seek to weaken forest protections or suspend environmental laws ostensibly to facilitate a more extensive and industrial forest–fire management regime. Such approaches would likely achieve the opposite of their intended consequences and would degrade complex early seral forests (DellaSala et al. 2015). We suggest that the results of our study counsel in favor of increased protection for federal forestlands without the concern that this may lead to more severe fires.

Allowing wildfires to burn under safe conditions is an effective restoration tool for achieving landscape heterogeneity and biodiversity conservation objectives in regions where high levels of biodiversity are associated with mixed-intensity fires (i.e., "pyrodiversity begets biodiversity," see DellaSala and Hanson 2015*b*). Managers concerned about fires can close and decommission roads that contribute to human-caused fire ignitions and treat fire-prone tree plantations where fires have been shown to burn uncharacteristically severe (Odion et al. 2004). Prioritizing fuel treatments to flammable vegetation adjacent to homes along with specific measures that reduce fire risks to home structures are precautionary steps for allowing more fires to proceed safely in the backcountry (Moritz 2014, DellaSala et al. 2015, Moritz and Knowles 2016).

Managing for wildfire benefits as we suggest is also consistent with recent national forest policies such as 2012 National Forest Management Act planning rule that emphasizes maintaining and restoring ecological integrity across the national forest system and because complex early forests can only be produced by natural disturbance events not mimicked by mechanical fuel reduction or clear-cut logging (Swanson et al. 2011, DellaSala et al. 2014). Thus, managers wishing to maintain biodiversity in fire-adapted forests should appropriately weigh the benefits of wildfires against the ecological costs of mechanical fuel reduction and fire suppression (Ingalsbee and Raja 2015) and should consider expansion of protected forest areas as a means of maintaining natural ecosystem processes like wildland fire.

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Supporting Information

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Are Wildland Fires Increasing Large Patches of Complex Early Seral Forest Habitat?

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Abstract: High-severity fire creates patches of complex early seral forest (CESF) in mixed-severity fire complexes of the western USA. Some managers and researchers have expressed concerns that large high-severity patches are increasing and could adversely impact old forest extent or lead to type conversions. We used GIS databases for vegetation and fire severity to investigate trends in large (>400 ha) CESF patches in frequent-fire forests of the western USA, analyzing four equal time periods from 1984 to 2015. We detected a significant increase in the total area of large patches relative to the first time period only (1984–1991), but no significant upward trend since the early 1990s. There was no significant trend in the size of large CESF patches between 1984 and 2015. Fire rotation intervals for large CESF patches ranged from ~12 centuries to over 4000 years, depending on the region. Large CESF patches were highly heterogeneous, internally creating ample opportunities for fire-mediated biodiversity. Interior patch areas far removed from the nearest low/moderate-severity edges comprised a minor portion of high-severity patches but may be ecologically important in creating pockets of open forest. There was ample historical evidence of large CESF patches but no evidence of increases that might indicate a current risk of ecosystem-type shifts.

Keywords: complex early seral forest; conifers; biodiversity; high-severity fire; western USA

1. Introduction

High-severity fire patches represent the component in fires that kill all or nearly all of the overstory trees within mixed-severity fire areas in conifer forests of the western USA [1,2], creating a unique forest habitat type known as the complex early seral forest (CESF) [3]. CESFs are distributed as small (<1 ha) to large patches (>400 ha) in mixed-severity burns in the lower/middle-montane conifer forests of the Sierra Nevada [2] and within other frequent-fire forest types of the western USA [4–6]. Unlike early seral produced by a clear-cut or otherwise intensively logged area, a CESF is more complex in its structure, and is characterized by a heterogeneous mix of abundant standing dead trees (snags) and downed logs, naturally regenerating conifers, other trees, shrub patches, and abundant wildflowers [3].

Whether high-severity fire is increasing and the ultimate causes of presumed increases (e.g., climate change, increase in tree densities) is the subject of much recent debate. For instance, the areal extent and proportion of high-severity fire within large fire complexes have not changed markedly in recent decades in most forested regions of the West [4,7–11], but results are equivocal in the Rocky Mountains and Southwestern US, e.g., see [9,11,12]. In the Sierra Nevada, some studies have reported increasing trends for high-severity fire, e.g., [13,14], whereas subsequent research [15,16] indicated no increases. Moreover, the size of CESF patches within large fire complexes has been used as a key metric to hypothesize whether fire regimes are operating within historical bounds [6,17–21]. Some have expressed concerns that large high-severity patches are increasing as a component of a recent increase in so-called megafires and that this may signal ecosystem-type shifts and the loss of old-growth forests [6,18,20,22], while others have predicted potential overall decreases in the future occurrence of high-severity fire in general [23]. Concern over high-severity fires and the resulting large patches of CESF has been a catalyst for fundamental changes to federal forest management policies (e.g., Healthy Forest Restoration Act of 2003, 2012 National Forest Management Act Planning Rule) and has been recently used to promote proposed congressional legislation that would substantially curtail environmental protections and dramatically increase logging in federal forests (e.g., The Resilient Federal Forests Act of 2019). Concerns over high-severity fires overall are missing a biodiversity perspective that is necessary to fully evaluate fire management proposals in the context of ecosystem benefits from such fires and not just their potential impacts on people [24,25].

Notably, patches of CESF support unique fire-adapted communities, including many plants [26], avifauna [27,28], mammals [29], bats [30], terrestrial [31] and aquatic invertebrates [32]. The Black-backed Woodpecker (*Picoides arcticus*) is associated with large CESF patches (typically ~100–800 ha for a single pair, depending on habitat quality) for nesting and foraging [33–36]. The California Spotted Owl (*Strix occidentalis occidentalis*), which is being petitioned for federal listing under the Endangered Species Act, actively forages in CESF patches [37,38]. Thus, policies aimed at suppressing large fires that otherwise would maintain and replenish CESF patches may have unintended consequences for fire-mediated biodiversity [24,25].

Our objectives were to determine whether there has been a recent trend (increase or decrease) in large CESF patches in fire areas within frequent-fire conifer forests of the western USA [4,39], to evaluate the spatiotemporal extent of such patches in these forests, assess their internal heterogeneity, and investigate historical evidence for the occurrence of such patches. Our study is the first to analyze the occurrence of large high-severity fire patches by distinct time periods. Additionally, our findings may have relevance to policy makers and forest-fire managers seeking to integrate biodiversity benefits of large CESF patches with wildfire risk reduction to people and natural resource management [24,25].

2. Methods

We analyzed the same western USA frequent-fire forest types, and used the same vegetation databases as in our related study [40] (Figure 1). These areas are dominated by mixed-conifer forests, as well as ponderosa pine (*Pinus ponderosa*) and Jeffrey pine (*Pinus jeffreyi*) forests.

We downloaded burn severity maps derived from satellite imagery from the Monitoring Trends in Burn Severity project (MTBS; http://www.mtbs.gov). Within the conifer forests of our study area, we defined CESF patches as areas experiencing high-severity fire, using a threshold of Relative Delta Normalized Burn Ratio (RdNBR) values ≥641 [41]. The same or similar thresholds have been used to define high-severity fire in multiple forest regions of western USA [21,42–44] and thus, our findings are directly applicable with consistent use of MTBS across studies. Although there is no accepted or standard definition of large CESF patches, we chose to analyze patches >400 ha in order to address concerns expressed by researchers that CESF patches hundreds of hectares or larger may not have occurred historically [6,18,21], may create homogeneity and inhibit post-fire forest regeneration due to lack of seed sources [20,22] and/or may reduce forest resilience to climate change [45–47]. We used an inclusive approach such that any high-severity fire pixels of conifer forest (30 × 30-m each) with sides touching were considered to be part of the same patch.

We used a Mann–Kendall test to determine whether there is any trend in (a) the combined total annual area of CESF patches >400 ha, and (b) the size of individual CESF patches >400 ha, for the years 1984–2015 (the period for which consistently mapped MTBS datasets were available for the US), analyzing both the annual area of large CESF patches, and the size of individual large CESF patches, as continuous variables. Mann–Kendall is a non-parametric test for monotonic upward or downward trends over time and has been used in similar studies [9,15,48]. Compared to other tests, including parametric tests, the Mann–Kendall has been found to have an equal or greater statistical power to detect trends in environmental time series data when the data are non-parametric, such as wildland fire trend data [15].

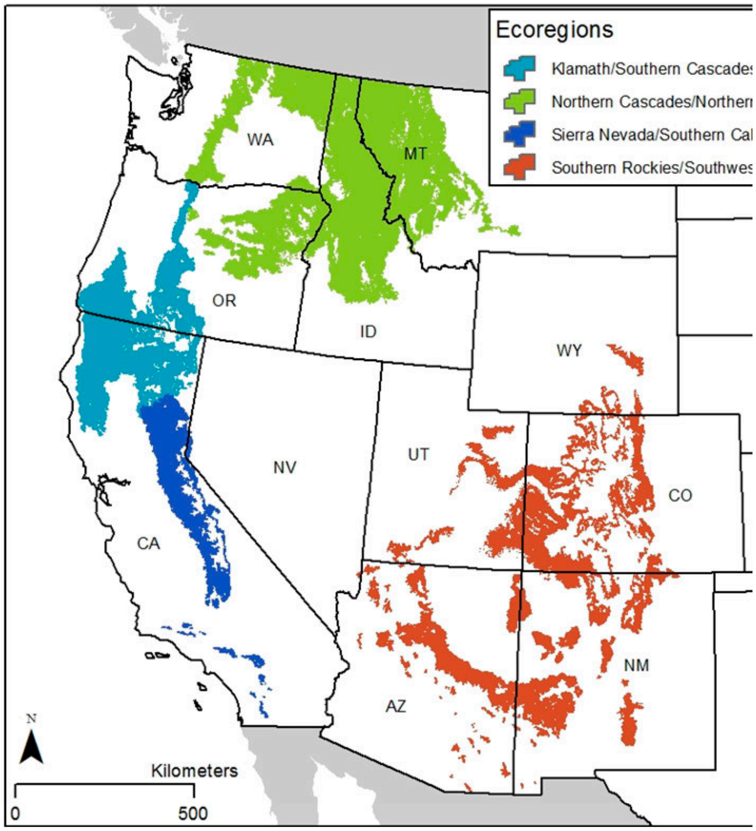

Figure 1. Ecoregions with pine and mixed conifer forests analyzed for large high-severity fire patches in our study modified from [40]. Two-letter acronyms shown on the map represent different U.S. states.

Since we were interested in determining the specific timing of any differences in occurrence in large CESF patches, we used a Nemenyi non-parametric test for multiple comparisons among groups with an equal sample size [49] to analyze whether there have been increases or decreases in large (>400 ha) patches of CESF, created by high-severity fire, for total annual area across four equal time periods (1984–1991, 1992–1999, 2000–2007, 2008–2015). To determine which specific time groups were significantly different with regard to individual patch sizes, we used a Dunn non-parametric test for multiple comparisons with unequal sample sizes [49]. In all analyses, significance was assessed at α = 0.05. We conducted this analysis because we wanted to determine whether any trend in the occurrence of large CESF patches is current and ongoing or happened at some point in the past, during the 1984–2015 time series, but may not be ongoing. This is not possible when large CESF patch occurrence is analyzed as continuous variables across the entire time series. For these two multiple comparison analyses, we chose to assess four groups of eight years each, rather than, for example, eight groups of four years each because the latter reduces sample size within each group to levels considered to be statistically inadvisable, and because using eight groups of four years increases the critical threshold to determine differences among groups, thus making it more difficult to reveal such differences when they exist [49].

In order to understand the spatiotemporal extent and context of large CESF patches across the forested landscape, we calculated fire rotation intervals [9] for high-severity fire patches >400 ha in each of four regions in the western USA: Sierra-Nevada/Southern-California, Klamath/Southern-Cascades, Northern-Cascades/Northern-Rockies, and Southern-Rockies/Southwest. The rotation interval for the occurrence of large CESF patches is equal to the average interval between occurrences of large patches across the study landscape [9].

We also analyzed the internal heterogeneity of CESF patches >400 ha in the four western USA regions by determining the percentage of the total area of such patches that was 1–100 m, 101–200 m, 201–300 m, and >300 m from the nearest unburned, low, or moderate-severity pixel (from either outside or inside the patch) within the frequent-fire conifer forest types analyzed in this study [40]. We included a specific analysis of internal heterogeneity of large high-severity patches because some authors have hypothesized that such patches would be internally homogeneous and have expressed concern about the potential for natural succession in this regard [6,20]. The distance intervals selected for this analysis were based on biologically meaningful relationships in levels of natural post-fire conifer regeneration at increasing distances from seed sources. We assumed lower levels of conifer recruitment at greater distances from live trees, consistent with natural succession to more open forest conditions [45,50–53].

Finally, although it was beyond the scope of this study to attempt to compare current versus historical rates of occurrence of large CESF patches, we included a table summarizing evidence for historical occurrence of patches >400 ha, focusing on low/middle-montane, frequent-fire forest types, given questions expressed about whether large CESF patches occurred historically in these forests [6,18,21].

3. Results

Over the entire time series, 1984–2015, there was a significant increasing trend in the combined total area of CESF patches >400 ha in each year (τ = 0.407, p = 0.001), but no trend in patch size (τ = 0.009, p = 0.802). However, when the data were analyzed by time periods, there was only one significant difference in the annual area of CESF habitat created by high-severity fire relative to the earliest time period (1984–1991), but no significant differences were detected among time periods since the early 1990s (Table 1, Figure 2). With regard to the size of individual large CESF patches, there were no significant differences detected among time periods (Table 2). Figure 3 shows the distribution of individual large CESF patches over the entire time series.

Table 1. Critical values ($q_{0.05,\,4}$), absolute difference between mean of ranks ($|R_A$-$R_B|$), standard errors (SE), and test statistics (q) to assess statistical significance, at α = 0.05 of any differences among the four time groups (1 = 1984–1991, 2 = 1992–1999, 3 = 2000–2007, and 4 = 2008–2015) for total annual area of CESF patches >400 ha using the Nemenyi non-parametric test for multiple comparisons between groups with an equal sample size (n = 8 years for each time group). The statistical significance of the levels of q are shown as "Y" (significant) or "N" (not significant).

Time Group Comparison	$q_{0.05,4}$	$\lvert R_A$-$R_B \rvert$	SE	q	Significant? (Is q > $q_{0.05,4}$?)
1–2	3.63	45.0	26.53	1.70	N
1–3	3.63	108.0	26.53	4.07	Y
1–4	3.63	107.0	26.53	4.03	Y
2–3	3.63	63.0	26.53	2.37	N
2–4	3.63	62.0	26.53	2.34	N
3–4	3.63	1.00	26.53	0.04	N

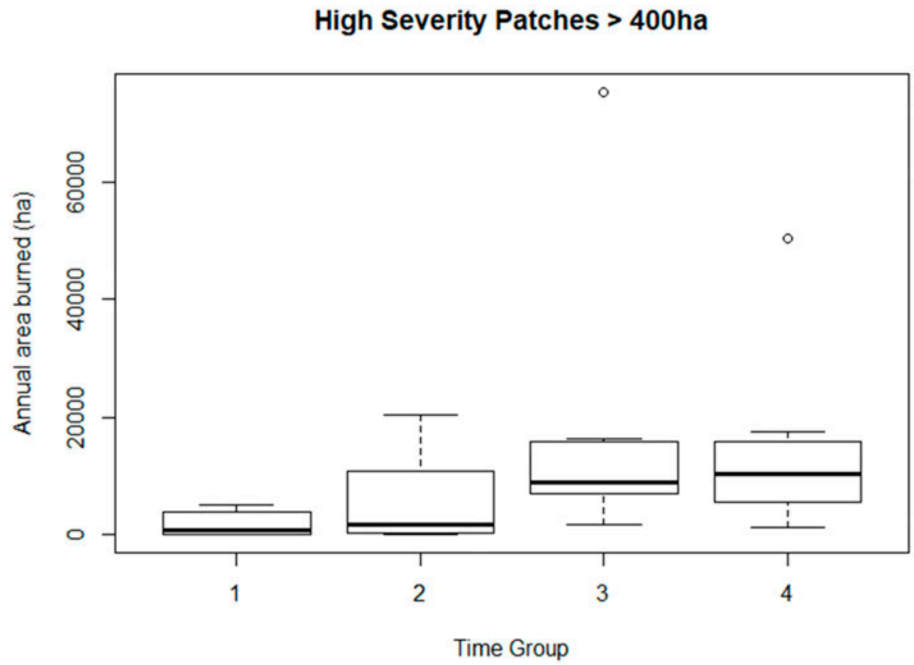

Figure 2. Annual area of large (>400 ha) CESF patches in the four time periods (see Tables 1 and 2 for time periods).

Table 2. Critical values ($q_{0.05,4}$), absolute difference between mean of ranks. ($|_{A}-_{B}|$), standard errors (SE), and test statistics (Q) to assess statistical significance at α = 0.05 of any differences among the four-time groups (1 = 1984–1991, 2 = 1992–1999, 3 = 2000–2007, and 4 = 2008–2015) for the size of individual CESF patches >400 ha using the Dunn non-parametric test for multiple comparisons. The statistical significance of levels of Q are shown as "Y" (significant) or "N" (not significant). For time groups 1, 2, 3, and 4, n = 17, 46, 134, and 130 CESF patches >400 ha, respectively.

Time Group Comparison	$Q_{0.05,4}$	$|_{A}-_{B}|$	SE	Q	Significant? (Is Q > $Q_{0.05,4}$?)
1–2	2.64	2.73	26.91	0.10	N
1–3	2.64	26.50	24.37	1.09	N
1–4	2.64	15.08	24.42	0.62	N
2–3	2.64	23.77	16.23	1.46	N
2–4	2.64	12.35	16.29	0.76	N
3–4	2.64	11.42	11.60	0.98	N

Over the 32-year study period, high-severity fire patches >400 ha occurred on ~0.7% to ~2.7% of the total area of frequent-fire conifer forest, depending on the region, such that the rotation intervals for occurrence of large (>400 ha) CESF patches, created by high-severity fire, ranged from 1181 years to 4354 years (Table 3).

Table 3. Total area and fire rotation interval for occurrence of CESF patches >400 ha in the four regions of the study area from 1984 to 2015.

Region	Area of Forest (ha)	Area (ha) of Patches >400 ha (% of Ecoregion)	Rotation Interval [1] (Years)
Sierra Nevada/Southern California	2,395,288	64,895 (2.709)	1181
Klamath/Southern Cascades	5,741,930	100,112 (1.744)	1835
Northern Cascades/Northern Rockies	10,057,451	73,936 (0.735)	4354
Southern Rockies/Southwest	6,956,201	72,851 (1.047)	3056

[1] Rotation intervals for high-severity patches were calculated by dividing the total area of the conifer forest by the average area of large high-severity patches per year.

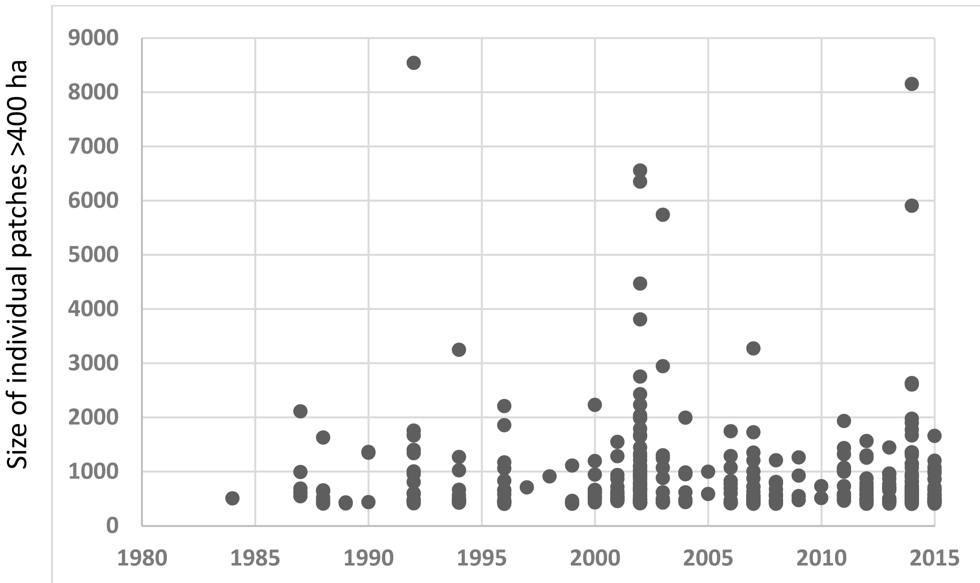

Figure 3. Scatter plot of size of individual large (>400 ha) CESF patches, 1984–2015.

Overall, 52% of the area within the boundaries of CESF patches >400 ha was within 100 m of unburned, low, or moderate-severity edges/inclusions, and 78% of the total area was within 200 m of such edges and inclusions. The results were similar in all four western USA regions (Table 4). Figure S1 is an example illustration of various distances from potential seed sources in very large (>1000 ha) high-severity patches in two areas: Rim fire 2013 (Stanislaus National Forest, Sierra Nevada, CA) and Hayman Fire 2002 (northwest Colorado Springs area).

Table 4. Percentages of the total area within the boundaries of CESF patches >400 ha, created by high-severity fire, that were at increasing distances from unburned or low/moderate-severity edges and inclusions.

Distance (m)	Sierra-Nevada/ Southern-California	Klamath/ Southern-Cascades	Northern-Cascades/ Northern-Rockies	Southern-Cascades/ Southwest
<100	49.3	55.6	46.8	54.7
101–200	27.6	25.5	25.2	26.0
201–300	13.5	11.2	12.8	10.6
>300	9.6	7.7	15.3	8.7

There is historical evidence of numerous large CESF patches created by high-severity fire prior to widespread fire suppression in every region of the western USA in low/middle-montane forests (Table 5). Historical patches >400 ha ranged from ~400 ha to >20,000 ha for our study area.

Table 5. Examples of historical occurrence of CESF patches >400 ha, created by high-severity fire, in low/middle-montane forests of the western USA [1].

Source	Region	Forest Type	Evidence Type	Patch Size/s (ha)	Time Period
[54,55]	Northern Sierra Nevada	Mixed-conifer and ponderosa pine	Historical USGS mapping, and current GIS analysis	400–~9000	19th century
[8]	Sierra Nevada	Mixed-conifer and ponderosa pine	Reconstruction, using 19th-century General Land Office data	Largest = 8050 (northern) and 9400 (southern)	19th century
[56]	Eastern Washington Cascades	Mixed-conifer	Reconstructions of past high-severity from historical aerial photos	400–10,500	19th century, and early 20th
[57]	Eastern Oregon Cascades	Mixed-conifer and ponderosa pine	Reconstruction from 19th-century General Land Office data	400–~5000	19th century
[58]	Oregon Klamath	Mostly ponderosa pine	Historical account, early 20th century U.S. Geological Survey report	~14,000	19th century
[59]	Colorado Front Range	Mostly ponderosa pine	Reconstruction from 19th-century General Land Office data	400–~22,000	19th century
[59]	Blue Mountains, Oregon	Ponderosa pine	Reconstruction from 19th-century General Land Office data	400–~12,000	19th century
[59]	Central/eastern Arizona	Ponderosa pine	Reconstruction from 19th-century General Land Office data	400–~40,000	19th century
[60]	Black Hills, South Dakota	Ponderosa pine, some lodgepole pine	Historical account	~19,000	mid-19th century
[61,62]	Northern Rockies	Ponderosa pine, some Douglas-fir	Reconstruction from historical aerial photos	~35,000	1910

[1] Some patches may have resulted from more than one fire. This represents all available data on historical occurrence of high-severity fire patches >400 ha known to currently exist within western US frequent-fire conifer forest types. For context, the largest individual high-severity fire patches in each of the four current time periods analyzed in this study are (in chronological order, by time period) 2109, 8539, 6554, and 8153 ha.

4. Discussion

Despite concerns about there being too many large CESF patches produced by big fires, we found that while an increase in the total area of such patches did occur initially in the time series, this happened over two decades ago and there has been no subsequent increase since the 1990s. We did not find an increase in the size of individual CESF patches >400 ha at any point during the time series (1984–2015)—i.e., patches >400 ha did not get significantly larger in more recent time periods. The rotation intervals for large patches ranged from about twelve centuries to over four millennia, depending on the region. A posteriori, we conducted the same analyses regarding whether there had been an increase in the area of large high-severity fire patches, but with a smaller patch size threshold (>100 ha), and we found the same result—i.e., significant differences between the first time period and the third and fourth time periods, but no other significant differences (Table S1, Figure S2). We did not conduct a posteriori analysis for patches >100 ha regarding the question of whether individual high-severity patches had been getting larger, since there were no significant or marginally significant differences with the >400 ha threshold.

Importantly, in large CESF patches, within-patch heterogeneity was high, with the great majority of patch area occurring within 200 m of the potential seed sources of unburned, low, or moderately burned conifer forest. In this regard, our findings are similar to those in the Northern US Rockies [63]. Depending on site factors, natural post-fire conifer regeneration generally occurs most quickly and abundantly within 100 m of low/moderate-severity and unburned recruitment areas, and secondarily at 100–200 m from unburned or low/moderate-severity areas [45,50–53,64]. It also occurs—typically more slowly and less densely—in the portion of large CESF patches that are >200 m from unburned or low/moderate-severity areas [51,53,64]. However, in these more distant areas, we can expect pockets of more open conifer forest or dense vegetation dominated primarily by oaks (*Quercus* spp.) and aspen (*Populus* spp.) and secondarily by conifers [51,64]. This internal patch heterogeneity indicates that large CESF patches play an important role in creating and maintaining pockets of open forest stands and increasing the heterogeneity (beta diversity) of forest structure across the landscape [64].

We also found considerable evidence of historical occurrence of large CESF patches in all regions, indicating that such patches are a component of natural fire regimes in low/middle-elevation, frequent-fire conifer forests of the western USA. More research is needed to compare current versus historical extents of such patches.

Modeling studies regarding wildland fire in western forests project overall increases [65], or more complex mixes of increases and decreases within and among regions, mediated by interactions between climate and vegetation shifts [65–67]. Thus, it will be important to continue to monitor high-severity fire occurrence and patch sizes periodically to understand any patterns that emerge in patch dynamics and conifer recruitment rates. Our findings also differ from some previous work regarding high-severity fire trends in western U.S. conifer forests. Some researchers [13,14], for instance, noted increasing trends in overall high-severity fire occurrence in mixed-conifer forests of the Sierra Nevada. Subsequent analyses [15,16] found that the use of a vegetation database by these researchers post-dated the time series being analyzed and led to an unintended omission of much of the high-severity fire in the earlier years of the time series, causing the appearance of an upward trend where no such trend existed. In other words, it was later found that the vegetation database used by these studies often did not reflect the vegetation that existed at the time of the fires analyzed, since much of the conifer forest that experienced high-severity fire in the earlier years of the time series was later reclassified as chaparral or other non-conifer vegetation—a phenomenon that occurred less for more recent fires in the time series.

Others [46,68,69] reported an increasing trend in the interior area of high-severity fire patches in the Sierra Nevada, but also used a vegetation database that post-dated the time series and omitted more of the high-severity fire in the earlier years of the time series [15]. They did not account for small low/moderate-severity inclusions within large high-severity fire patches, while inclusions of this size were common in our analyses.

Our results indicate that large CESF patches have high levels of heterogeneity (beta diversity), even within the most interior portions, which may facilitate heterogeneous natural forest regeneration in ecologically beneficial ways [25,53,55,70]. Some delayed tree mortality can, of course, occur in the years following a fire in low/moderate-severity inclusions, and this could potentially influence the internal patch complexity along with conifer seedling establishment. Yet, even in such cases, individual trees experiencing delayed mortality would provide seed source in the interim years, and research into delayed post-fire mortality indicates fairly modest levels of such occurrences in low/moderate-severity pixels [71].

Some researchers have expressed concern about type conversion to non-forest following fires, especially high-severity fires, e.g., [47,72]. Although a detailed discussion of this issue is beyond the scope of our study, we note that areas described as examples of possible post-fire type conversion nevertheless had substantial post-fire conifer regeneration, generally within the described natural range of variability for the specific forest type [72], and the areas with no regeneration occurred at the spatial scale of very small plots [47,72]. Thus, we suggest that there may be a scale-of-observation issue at work here, and much larger plots indicate more consistent post-fire regeneration [64]. Moreover, while recent research has suggested somewhat lower regeneration in more recent fires, time-since-fire was not accounted for, and far fewer years of post-fire succession had occurred at the time of field sampling in the more recent fires, which might account for the difference [47]. Nevertheless, some researchers have predicted that in a hotter and drier climate in certain areas, such as the Klamath region of northwestern California, recurrent high-severity fire could limit the recruitment of some conifer species in future decades [73]. Thus, more research is needed to address this question after taking spatial and temporal scale and time-since-fire into account.

5. Conclusions

Our findings have specific management and policy relevance. In particular, we counter claims made by some researchers, and often used by decision-makers, to justify large-scale forest “thinning” and post-fire logging projects—specifically, the assumption that such logging projects are needed to prevent type conversion in response to a perceived increase in CESF patch sizes and conifer regeneration failures in “megafires” (see [6,18,20,22]). Lack of a biodiversity perspective has created underlying tensions among researchers over the role of high-severity fires in maintaining CESF, and we hope that our findings will now inform this ongoing discussion. Additionally, contrary to assumptions made by land managers in the course of proposing extensive post-fire logging and creation of artificial tree plantations following large fires, we found ample evidence of patch heterogeneity–and presumably natural conifer establishment–in large severely burned patches, in addition to the occurrence of large high-severity patches in the historical record. This finding has key relevance to current forest management policy, since the assertion that current large CESF patches are unprecedented is not substantiated by our data but is being used to justify legislative and regulatory proposals to severely weaken environmental laws on U.S. federal lands.

Notably, numerous studies have found high levels of native plant and animal richness and abundance in large fires of mixed severity that produce CESF patches in severely burned areas, see [3,24–31,70,74,75]. Such fires facilitate high levels of beta diversity at landscape scales, providing a broad suite of habitat for both fire-seeking and fire-avoiding species [25], including many early seral birds that have been declining due to a lack of “diverse early seral habitat” [76]. Thus, far from being indicative of “catastrophic” (or “megafire”) ecosystem shifts, large CESF patches have consistently been found to support a unique ecological community that is otherwise most often post-fire logged because of perceptions that this forest type has limited wildlife value, see [25,75]. Instead, we found that large CESF patches are extremely infrequent at landscape scales in ponderosa/Jeffrey-pine and mixed-conifer forests of the western U.S., and whether high-severity fire that produces this important seral stage is increasing in western USA forests remains debatable, e.g., [4,9–11,13–16,19,21,23].

Regarding the human implications of our findings, we recommend that land managers focus limited resources on community fire safety and defensible space of homes as a means of getting to coexistence with wildfire [77–79] and for managing wildfire under safe conditions for a myriad of ecosystem benefits.

Supplementary Materials: The following are available online at http://www.mdpi.com/1424-2818/11/9/157/s1: Figure S1: Example of CESF patches >1,000 ha, showing distances from areas of unburned, low, and moderate severity fire within the patch boundaries in the Rim (Stanislaus National Forest, CA) and Hayman fires (Colorado Front Range). Figure S2: Annual area of large patches (>100 ha) of CESF in the four time periods; Table S1: Critical values ($q_{0.05,\infty,4}$), absolute difference between mean of ranks ($|R_A\text{-}R_B|$), standard errors (SE), and test statistics (q) to assess statistical significance, at $\alpha = 0.05$ of any differences between the four time groups (1 = 1984–1991, 2 = 1992–1999, 3 = 2000–2007, and 4 = 2008–2015) for total annual area of CESF patches >100 ha using the Nemenyi non-parametric test for multiple comparisons between groups with an equal sample size (n = 8 years for each time group). The statistical significance of levels of q is shown as Y (significant) or N (not significant).

Author Contributions: D.A.D. and C.T.H. were equally involved in all aspects of this study.

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An Ecologically Based Strategy for Fire and Fuels Management in National Forest Roadless Areas

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During the challenging 2000 fire season, the local and national headlines trumpeted daily news about the "worst fires in recent memory." The media showered us with the latest statistics on wildland fires in the West: "More than 6 million acres charred in 13 Western States…more than 25,000 firefighters deployed…over 80 blazes raging out of control…hundreds of homes consumed."

Amid the media frenzy, one Presidential candidate—George W. Bush—sought to improve his position in the public opinion polls by stating that greatly reduced logging levels on national forests during the previous decade had "made the forests more dangerous to fire." The implication was that the USDA Forest Service's proposed policy for protecting roadless areas was akin to putting a lit match into a tinderbox.

Others called for massive logging, roadbuilding, and a rash of prescribed fires as a quick fix for the previous 50-100 years of fire suppression. While conservationists advocated for roadless area protection on the grounds that roadless areas are the last remnants of formerly large and intact forests, critics asserted that fiery conflagrations would inevitably occur if the same forest remnants were not intensively managed. The rest of us pondered: Where is the science in all this? Is every acre doomed to "catastrophic" fire if not intensively managed? Is it appropriate to treat all forests the same, regardless of whether or not they contain existing road systems?

After all the hyperbole – a combination of media hype, electoral politics, and misinformation spread to promote special interests – it's time to take a sober look at the questions raised by the 2000 fire season. Specifically, what evidence exists on the relationship between wildland fire and timber management in roaded vs. roadless areas? What effects might silvicultural treatments and prescribed fire have on ecosystems in roadless areas? Is there an ecologically based strategy for identifying, on a case-by-case basis, where active management might be appropriate for maintaining fire-dependent forest ecosystems?

Fire and Roadless Areas

Level of Fire Hazard. Scientists widely agree that protecting roadless areas on the national forests from roadbuilding, logging, and other forms of development will greatly enhance biodiversity and ecosystem conservation (Ercelawn 1999; Henjum and others 1994; Noss and Cooperider 1994; Strittholt and DellaSala [in press]). However, some critics of roadless area protection (Bernton 1999; Hansen 1999; Schlarbaum 1999) have repeatedly made two assertions:

- Road building prohibitions in roadless areas will restrict access and timber management, which in turn will increase the frequency of large, intense fires.
- Widespread silvicultural treatments (such as low thinning and crown thinning) in roadless areas will be necessary to reduce the fire hazard.

Does the relevant scientific literature support these claims?

Broad scientific assessments were completed in 1996 and 1997, respectively, for Federal lands in the Sierra Nevada in California and the Interior Columbia River Basin in portions of Idaho, Montana, Nevada, Oregon, Washington, and Wyoming. These studies provide the most comprehensive analysis to date for comparing fire, fuel, and vegetation conditions in intensively managed areas to conditions in roadless areas. Both assessments found the fire hazard to be significantly higher in intensively managed areas.

According to the Sierra Nevada assessment, “Timber harvest, through its effects on forest structure, local microclimate and fuel accumulation, has increased fire severity more than any other recent human activity” (SNEP 1996). The Interior Columbia Basin assessment similarly concluded that “fires in unroaded areas are not as severe as in roaded areas because of less surface fuel….Many of the fires in the unroaded areas produce a forest structure that is consistent with the fire regime, while the fires in the roaded areas commonly produce a forest structure that is not in sync with the fire regime. Fires in the roaded areas are more intense, due to drier conditions, wind zones on the foothill/valley interface, high surface-fuel loading, and dense stands” (Hann and others 1997).

Even within the forest types most altered as a result of fire suppression (such as dry forests with a regime of frequent low-intensity fires), intensively managed forests on federal lands in the Interior Columbia Basin are denser and carry higher fuel loads than do roadless areas. Accordingly, intensively managed lands were found to be at higher risk of tree mortality from fire, insects, disease, and other disturbance agents (Hann and others 1997).

Others have reported similar findings for portions of the interior West. In the Sierra Nevada, McKelvey and others (1996) and Weatherspoon (1996) identified timber harvest as the single most important factor responsible for an increase in potential fire severity. In the Klamath Mountains of northwestern California, Weatherspoon and Skinner (1995) found that partial-cut stands with fuels treatment (lop and scatter or broadcast burning) burned more intensely and suffered higher levels of tree mortality than adjacent areas left uncut and untreated. Fire and fuel models also suggest that mechanical treatments alone, including silvicultural thinning and biomass removal, are not likely to be effective at reducing fire severity in dense stands (van Wagtendonk 1996).

In eastern Oregon and Washington, Lehmkuhl and others (1995) and Huff and others (1995) reported a positive correlation between logging, on the one hand, and fuel loadings and predicted flame lengths, on the other hand. They attributed the increased fire hazard in intensively managed areas to leftover slash fuels from tree removal activities (including thinning) and to the creation of dense, early-successional stands through overstory removal. A postfire study of the effectiveness of fuels treatments (including thinning) on previously nonharvested lands on the Wenatchee National Forest in Washington found that harvest treatments likely exacerbated fire damage (USDA Forest Service 1995).

Overall, the scientific literature shows that forests in areas without roads are less altered from historical conditions and present a lower fire hazard than forests in intensively managed areas, for three reasons:

1. Timber management activities often increase fuel loads and reduce a forest's resilience to fire.
2. Areas without roads have been less influenced by fire suppression than intensively managed lands.
3. Widespread road access associated with intensively managed lands raises the risk of human-caused ignitions.

As summarized in a recent review of national forest management organized by the Ecological Society of America, "There is no evidence to suggest that natural forests or reserves are more vulnerable to disturbances such as wildfire than intensively managed forest stands. Indeed, there is considerable evidence to the contrary, evidence that natural forests are actually more resistant to many types of both small- and large-scale disturbances" (Aber and others 2000). Assertions about increased wildfire made by critics of roadless area protection are not based in fact, as the evidence is clear that the forests most in need of fuels treatment are not roadless areas but areas that have already been roaded and logged, "where significant investments have already been made" (USDA/USDI 1997).

Effectiveness of Fire Suppression. Some evidence exists that fire suppression activities have had a lower impact on roadless areas than on roaded portions of the national forests (Hann and others 1997; SNEP 1996). The lower impact may be attributable to limited access and steep terrain, which prevent the application of large, ground-based suppression strategies in roadless areas (Agee 1993; Fuller 1991; Pyne 1996; Schroeder and Buck 1970).

Fires in roadless areas tend to be more remote from human habitations than are fires on roaded lands. Accordingly, they are often the lowest priority for suppression during years when fire-fighting resources are in short supply. Although data are limited, findings from the Interior Columbia Basin assessment on this topic might apply to other regions as well. The assessment concluded that a “combination of past harvest practices and more effective fire suppression moved the roaded landscapes much further from their unaltered biophysical templates, as measured by dominant species, structures, and patterns, relative to unroaded areas….In general, all forests which show the most change from their historical condition are those that have been roaded and harvested” (Hann and others 1997). Furthermore, the forests that are most susceptible to moisture stress, insects, disease, and unnaturally intense fire tend to be at the lowest elevations, which typically border private, state, tribal, or other landownerships (Everett and others 1994).

Another reason why fire suppression has had less impact on forests in roadless areas is associated with differences in vegetation and fire regimes. Most roadless areas on the national forests, particularly in the interior West, are at mid- to high-elevations (Beschta and others 1995; Henjum and others 1994; Merrill and others 1995). The exceptions are in the Eastern United States, where elevational gradients are limited, and the Klamath–Siskiyou ecoregion in northwest California and southwest Oregon, where very steep slopes at lower elevations have limited road construction (Strittholt and DellaSala [in press]).

Higher elevations are cooler, receive more moisture, and have a shorter summer dry season than lower elevations. They are typically characterized by a regime of low frequency, high-intensity fires (Agee 1993; Baker 1989; van Wagner 1983). Roadless areas are therefore less likely to have current fire regimes that are significantly different from historical conditions (Agee 1997; Beschta and others 1995).

For fires in high-elevation forests, weather rather than fuels is often the primary variable determining fire severity and extent (Agee 1997; Bessie and Johnson 1995; Flannigan and Harrington

1988; Johnson and Wowchuck 1993; Turner and others 1994). Under severe fire weather, the efficacy of fire suppression decreases dramatically in forest types characterized by high-intensity fires (Agee 1998, SNEP 1996). Even substantial investments of financial and human firefighting resources often fail to control large fires; they are extinguished only when the weather changes (Romme and Despain 1989).

Risk of Human-Caused Ignitions. Roadless areas have a lower potential for high-intensity fires than roaded areas partly because they are less prone to human-caused ignitions (DellaSala and others 1995; USDA Forest Service 2000; Weatherspoon and Skinner 1996). Roads constructed for timber management and other activities provide unregulated motorized access to most national forestlands and are heavily used by the general public.

In the Western United States, many of the more than 378,000 miles of national forest roads traverse heavily managed forests with the greatest potential for high-severity fire. According to the Forest Service, more than 90 percent of wildland fires are the result of human activity, and ignitions are almost twice as likely to occur in roaded areas as they are in roadless areas (USDA Forest Service 1998, 2000). While it can be argued that roads provide improved access for fire suppression, this benefit is more than offset by much lower probabilities of fire starts in roadless areas.

The Case Against Mechanical Fuels Treatments in Roadless Areas

Some land managers and policy makers advocate the widespread use of silvicultural treatments (often mechanical thinning of merchantable trees) in western roadless areas to reduce fuel loads and tree stocking levels and thereby decrease the probability of large, intense fires. Although thinning has long been a part of intensive forest management, its efficacy as a tool for fire hazard reduction at the landscape scale is controversial, largely unsubstantiated, and fundamentally experimental in nature (DellaSala and others 1995; FEMAT 1993; Henjum and others 1994; SNEP 1996; USDA Forest Service 2000).

Few empirical studies have tested the relationship, even on a limited basis, between thinning or other fuels treatments and fire behavior. These studies, supported by anecdotal information and the analysis of recent fires, suggest that thinning treatments have highly variable results. In some instances, thinning intended to reduce the fire hazard appeared to have the opposite effect (Huff and others 1995; van Wagtendonk 1996; Weatherspoon 1996). Thinning might reduce fuel loads, but it also allows more solar radiation and wind to reach the forest floor. The net effect is usually reduced fuel moisture and increased flammability (Agee 1997; Countryman 1955).

Moreover, mechanical treatments fail to mimic the ecological effects of fire, such as soil heating, nutrient cycling, and altering forest community structure (Chang 1996; DellaSala and others 1995; Weatherspoon and Skinner 1999). In fact, according to the SNEP (1996), "although silvicultural treatments can mimic the effects of fire on structural patterns of woody vegetation, virtually no data exist on their ability to mimic the ecological functions of natural fire. Silvicultural treatments can create patterns of woody vegetation that appear similar to those that fire would create, but the consequences for nutrient cycling, hydrology, seed scarification, non-woody vegetation response, plant diversity, disease and insect infestation, and genetic diversity are almost unknown."

Although our current understanding of the ecological effects of thinning is incomplete, evidence indicates that mechanical treatments, even when carefully conducted, can have additional environmental impacts:

- Damage to soil integrity through increased erosion, compaction, and loss of litter layer (Harvey and others 1994; Meurisse and Geist 1994);
- Increased mortality of residual trees due to pathogens and mechanical damage to boles and roots (Filip 1994; Hagle and Schmitz 1993);
- Creation of sediment that might degrade streams (Beschta 1978; Grant and Wolff 1991);
- Increasing levels of fine fuels and near-term fire hazard (Fahnestock 1968; Huff and others 1995; Weatherspoon 1996; Wilson and Dell 1971);
- Disruption of mycorrhizal fungi – plant relationships that are important to ecosystem function – and shrubs and perennial native bunchgrasses involved in fungal linkages (Amaranthus and Perry 1994, Massicotte and others 1999, pers. comm. D. Southwort and L. Valentine, Southern Oregon University);

- Dependence on roads, which have numerous adverse effects of their own (Henjum and others 1994; Megahan and others 1994); and
- Reduced habitat quality for sensitive species associated with cool, moist microsites or closed-canopy forests (FEMAT 1993; Thomas and others 1993).

These adverse impacts of mechanical treatments should be of particular concern in managing roadless areas, where ecological values are especially high. Moreover, roadless areas are often in steep, unstable terrain that is highly sensitive to human disturbance (Henjum and others 1994; Wilderness Society 1993). According to the Forest Ecosystem Management Assessment Team, most existing roadless areas "are considered inoperable because timber harvest and road construction would result in irretrievable loss of soil productivity and other watershed values. These lands consist of erosion- and landslide-prone landforms such as inner gorges, unstable portions of slump earthflow deposits, deeply weathered and dissected weak rocks, and headwalls" (FEMAT 1993).

Similarly, the Interior Columbia Basin assessment found "a high risk to watershed capabilities from further road development in these [roadless] areas. In general, the effects of wildfires in these areas are much lower and do not result in the chronic sediment delivery hazards exhibited in areas that have been roaded. In contrast, the already roaded areas have high potential for restoration action" (USDA/USDI 1997). Given the potential for adverse impacts from silvicultural treatments in roadless areas, many scientists recommend limiting experimental treatments to previously managed lands already degraded by fire suppression and logging (Aber and others 2000, Beschta and others 1995; DellaSala and others 1995; Franklin and others 1997; Hann and others 1997; Henjum and others 1994; McKelvey and others 1996; Perry 1995).

In summary, scientific assessments of federal lands in several western regions generally conclude that previously roaded and logged areas should be the highest priority for fuels reduction and forest restoration treatments (FEMAT 1993; Hann and others 1997; SNEP 1996). Silviculture has a role to play in a scientifically based approach to fire and fuel management on federal lands, but current evidence indicates that widespread mechanical treatments in roadless areas would most likely increase rather than decrease ecosystem degradation. Therefore, experimentation with mechanical treatments for fire hazard reduction should proceed primarily in areas with road access and adjacent to private lands where the ecological risks are lower and the threat of fire to human lives and property is far greater.

Roadless areas should only be considered for mechanical treatment after all other, higher priority areas are addressed and only if it can be demonstrated that such treatments will not degrade ecological values. Any experimental treatments in roadless areas should occur in small roadless areas (less than 5,000 acres (2,000 ha)) that have relatively good access, are near the wildland-rural interface, and exhibit high fire hazard due to past suppression. Only small trees (generally less than 12" diameter) should be considered for removal and under no circumstances should new or temporary roads be built to conduct mechanical treatments.

The Case for Prescribed Fire in Roadless Areas

The Forest Service should treat roadless areas primarily by reintroducing fire, both natural and prescribed. Restoration of ecological processes is key to ecosystem integrity and biological diversity (Samson and Knopf 1993), particularly in unroaded areas. Use of prescribed fire has been successful in restoring wildland fire regimes to many fire-adapted ecosystems (Wright and Bailey 1982), and a widespread consensus exists that additional burning is necessary (Arno 1996; Mutch 1994, 1997; USDA/USDI 1995; Walstad and others 1990).

Prescribed fire has important advantages over mechanical treatments in areas where ecological integrity and biodiversity conservation are important management objectives (Hann and others 1997; SNEP 1996; Weatherspoon and others 1992). Prescribed fire also appears to be the most effective treatment for reducing fire severity and rate of spread (Stephens 1998; van Wagtendonk 1996). In addition to reducing fuel loading and continuity, prescribed fire may decrease pest outbreaks, provide germination sites for shade-intolerant species, release nutrients, and create wildlife habitat (Agee 1993; Biswell 1999; Chang 1996; Walstad and others 1990).

Positive outcomes associated with prescribed fire are, of course, contingent on detailed site-specific planning, adequate budgetary support, and careful execution by trained personnel. In roadless areas with forests characterized by low-intensity, high-frequency fire regimes, repeated prescribed burns within a relatively short timeframe might be required to sufficiently reduce fuels and ensure that fire intensities remain within an acceptable range (Biswell 1999). After initial treatment, the frequency of prescribed burns can be designed to reflect the inherent disturbance regime and range of variability associated with particular forests. Data from the Sierra Nevada suggest that prescribed burning is likely to be considerably cheaper for treating fuels than either mechanical treatments or fire suppression (Husari and McKelvey 1996; see Deeming (1990) for a summary of the literature on the cost-effectiveness of prescribed burning versus other fuel treatments).

In addition to prescribed fire, ecological benefits could flow from allowing some naturally ignited fires to burn in roadless areas under specific environmental conditions. Traditionally, the Forest Service has suppressed most wildland fires without adequately considering the potential resource benefits of a “confine-and-contain” strategy. However, Federal policies introduced in 1995 encourage careful management of naturally ignited wildland fires if they meet resource objectives and are consistent with historical fire regimes (USDA/USDI 1995). Less than full control strategies for fire suppression could be employed, provided the strategy chosen is projected to incur the least cost of suppression and the least loss of resource values (McKelvey and others 1996).

Carefully planned wildland fire use should be fully considered for roadless areas, based on fire regime, expected fire behavior, and other variables, as an alternative to costly firefighting in remote areas where there is little or no danger to lives and property. In 2000, the Forest Service spent more than $91 million fighting two large fires in Idaho, the Burgdorf Junction Fire and the Clear Creek Complex Fire. Together, the fires burned more than 280,000 acres, mostly in remote roadless and wilderness areas (Morrison and others 2000; NIFC 2000a). On such fires, wildland fire is likely to be the most sensible as well as ecologically appropriate strategy.

Roadless areas could instead benefit from proactive fuels management using fire. Fire management in roadless areas should be based on (1) a standard set of guidelines for identifying and prioritizing roadless areas based on their fire hazard and risk at the national or regional level (see sidebar); and (2) a subsequent step-down process for planning fire treatments at the local level, designed to allow fire to play a more important role while minimizing risks to ecological values.

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Integrated Management Strategies are Needed

Roadless areas do not exist in isolation from other land designations. It follows that an effective fire and fuel management strategy should be developed at the landscape scale. This means first identifying areas of highest priority for fire/fuels treatments and then planning treatments that are consistent with management standards to ensure protection of soil, water, wildlife and other ecological values. For roadless areas, high-priority treatment areas should first be identified at the national and regional scale. Then site-specific burn plans can be developed for individual roadless areas, or for complexes of areas, by integrating spatial information on fire hazard (fuel load, fuel continuity, and topography); fire risk (ignition history and weather); and ecosystem values (old-growth forests, wildlife habitat, and sensitive watersheds) (Agee 1995; Bunting 1996; Crutzen and Goldhammer 1993; Johnson and others 1997; Weatherspoon and Skinner 1996). By employing this kind of tiered prioritization, limited resources can be directed to areas that are most in need of fire and fuels reduction.

Over time, as fire is reintroduced into roadless areas – coupled with fire and other fuels treatments on adjacent, intensively managed lands – the occurrence of large, high-intensity wildland fires might become of less concern. In rare cases, limited low thinning (removal of small understory trees) may be appropriate in some roadless areas as a prerequisite for prescribed fire. However, more experimentation and research on the efficacy of mechanical treatments should first be conducted in intensively managed forests before broadly applying them to roadless areas. Such a cautious approach is warranted, given that a mere 4 percent of roadless lands present a high fire hazard; the vast majority of areas at risk of uncharacteristically intense fire are in the intensively managed, roaded landscape (USDA Forest Service 2000).

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Although much can be done to reduce fire hazards, there is no “magic bullet” to reverse many decades of fire suppression activities. Despite our best intentions, the fire situation may yet worsen as more homeowners build cabins deeper into fire-prone forests and climate change potentially produces hotter and drier conditions in some areas. Moreover, it is important to note that despite all the media hype, the 2000 fire season was relatively light by historical standards: In the 1930’s, more than 39 million acres (15.6 million ha) burned on average each year (NIFC 2000b).

The strategy outlined here is consistent with the Clinton Administration’s recent policy recommendations that emphasize treatment of the highest priority areas first in non-controversial areas – the wildland-rural interface and designated municipal watersheds (Council on Environmental Quality 2000). To ensure that current fire management policy avoids ecological risks associated with the logging of large trees and other ecosystem values, we recommend that thinning in priority areas target only the removal of small, non-commercial material that has most likely increased as a result of fire exclusion and is of greatest concern for hazardous fuel reduction. This is consistent with Chief Dombeck’s letter (5/23/00 file code 1500) to Senator Bingaman emphasizing that emergency appropriations be used to remove small trees <12 inch dbh (30 cm) from priority areas.

In contrast, timber industry representatives such as Butch Bernhardt of the Western Wood Products Association insist that “cutting some larger trees” is “the incentive” needed to “markedly improve forest health” by allowing “more sunlight and nutrients to reach the remaining growth” (Associated Press 2000). Commercial harvest is designed for profit, not to address ecological need; the timber industry’s claims to the contrary are inconsistent with the available science on fire and fuels management. Only through an integrated approach that emphasizes protection of roadless values and focuses treatment where it is most needed – in the roaded landscape – are we likely to make significant progress in restoring the resiliency of western forest ecosystems.

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[DESIGNER: Please set up the following as the first sidebar.]

Prioritizing Roadless Areas for Prescribed Fire

Land managers need a comprehensive set of criteria for prioritizing roadless areas for prescribed fire treatments. The following list provides a preliminary guidepost for determining high-priority areas for treatment. Prescribed fire should be considered for roadless areas where:

- Most of the area is covered by dry forest types that are characterized by low-intensity, high-frequency fire regimes;
- A long interval has passed since the last major fire (for example, more than three natural fire cycles have been missed);
- The topographic and elevational gradients are relatively gentle, permitting relatively low-risk prescribed fire treatments and raising the likelihood that past firefighting efforts have increased the fire hazard;
- Areas of high fire risk are nearby, such as the wildland–rural interface, major population centers, transportation routes, or residential developments and other infrastructure; and
- Ecological risk factors are absent or low, such as—
    - Populations of threatened and endangered species or rare communities that are known to be adversely affected by fire;
    - Vegetation changes that would predictably result from fire treatments; or
    - Fish refugia where burning could impair hydrological processes or degrade critical fish habitat through sedimentation.

[DESIGNER: Please set up the following as the second sidebar.]

Principles for Fire and Fuels Management

Land managers need a comprehensive, landscape-level strategy for fire/fuels management that takes into account the important values associated with roadless areas and directs treatments where they are needed the most. The strategy should be based on the following principles:

- Limit mechanical treatments to high-priority areas, primarily roaded areas of dense, dry forest within the wildland–rural interface.
- Define the wildland–rural interface by treating areas immediately adjacent to rural settlements as a first line of defense. Provide homeowners with assistance grants to reduce the fire hazard on private land by creating a defensible space around homes.
- Conduct watershed or landscape-scale assessments that identify restoration priorities before fire/fuel treatments are initiated.
- Eliminate commercial incentives for mechanical removal of merchantable trees by decoupling goods from services (that is, pay a fixed fee for tree removal services that is not tied to timber volume).
- Restrict thinning to small-diameter trees (e.g., less than 12 inches (30 cm) in diameter at breast height or less than the average stand diameter) where it can be demonstrated that current forest stand densities are outside the historical range of variability.
- Minimize impacts to soils, below-ground processes and related species, accumulation of surface fuels from thinning, and exposure to solar radiation and reduction of soil moisture retention.
- Conduct mechanical treatments in priority areas in compliance with all relevant environmental statutes (e.g., National Environmental Policy Act, National Forest Management Act, Endangered Species Act, etc).

FEATURE

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Roadless areas and clean water

Dominick A. DellaSala, James R. Karr, and David M. Olson

Clean water, like biodiversity, is most closely linked to undisturbed natural ecosystems. When undisturbed watersheds in roadless and protected areas (e.g., national parks, state parks, wilderness areas, national monuments) are fragmented by roads, logging, and intensive recreation development, both water quality and biodiversity decline as hydrological integrity is lost (USFS 1972, 1979, 2001; Alexander and Gorte 2008; Anderson 2008). In the United States, inventoried roadless areas (IRAs) are lands without roads exceeding 2,000 ha (5,000 ac) that have been inventoried by the USDA Forest Service. IRAs collectively amount to approximately one third of the 77 million ha (193 million ac) of the 155 national forests but are disproportionately concentrated in western states (figure 1) (Trout Unlimited 2004; Anderson 2008). The roaded, intensively managed landscapes of the other national forest lands have been closely correlated with heavily sediment-laden streams and dramatic changes in flow regimes (Espinosa et al. 1997; Trombulak and Frissell 2000; CBD et al. 2001; Coffin 2007; Frissell and Carnefix 2007). While the biodiversity benefits of IRAs are well documented (DeVelice and Martin 2001; Strittholt and DellaSala 2001; Loucks et al. 2003; Strittholt et al. 2004; Gelbardi and Harrison 2005), little has been made of the importance of IRA water for downstream users and wildlife.

In this paper, we assess the importance of IRAs from a water quality perspective, including the likely water quality effects of developing IRAs. We provide conservative estimates of the economic impact of intact unroaded watersheds on national forests for clean water and associated water resource benefits. In particular, rising demand and shrinking water supply associated with changing climate will likely make intact areas in drought-prone regions of the West even more valuable and crucial to protect. Thus, our findings are especially relevant to drought-prone states considering development of IRAs. The state of Colorado, for example, with approximately 1.7 million ha (4.2 million ac) of IRAs, has been seeking federal permission to develop its IRAs for logging, expanding ski areas, coal-bed mining, and producing oil and gas (figure 2) (Anderson 2008; Colorado Division of Wildlife 2010; Colorado, State of 2010; Straub 2010, USFS 2011). Although we focus on IRAs throughout the western United States, we also emphasize the importance of uninventoried roadless areas (unroaded) <2,000 ha (Henjum et al. 1994; Greenwald 1998; Beschta et al. 2004) that collectively cover an area roughly 1.5 times that of the total IRA network (USFS 2000; Strittholt et al. 2004). Those smaller unroaded areas also play a strategic role in maintaining reliable supplies of high-quality water and protecting aquatic ecosystems.

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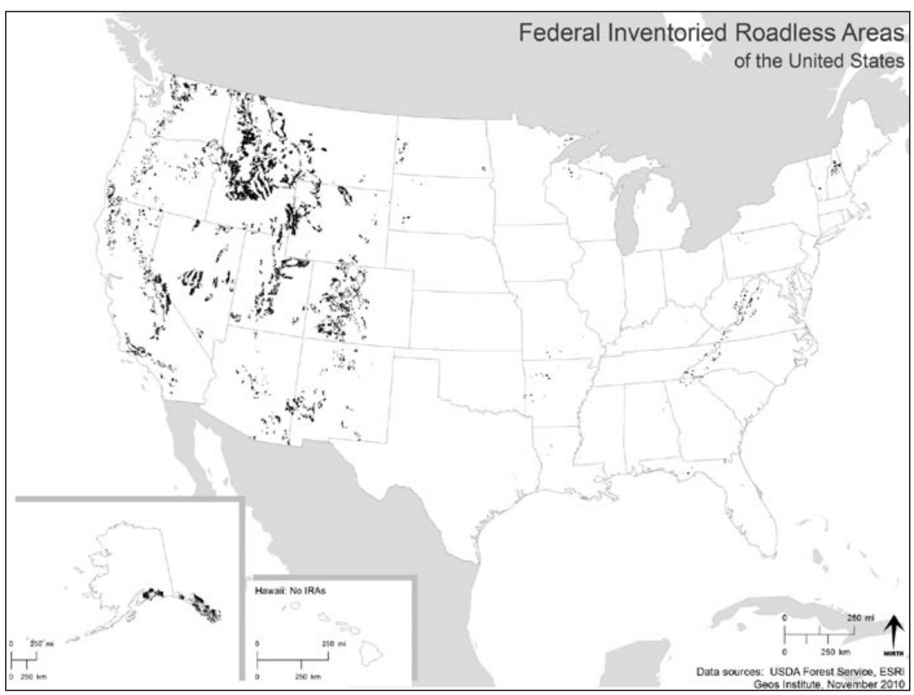
ROADLESS AREAS PROVIDE SUBSTANTIAL WATER RESOURCE BENEFITS

IRAs benefit society in many ways, including providing a valuable and increasingly rare natural supply of abundant, clean, and naturally reliable water (Sedell et al. 2000); affordable drinking water for municipal and rural communities; water for agricultural and industrial uses; flood control; in-stream aquatic recreation; aquifer recharge; flood protection; reliable water supply; diverse and productive fisheries; healthy aquatic ecosystems; resident and migratory waterfowl habitat; recovery of endangered species; and, increasingly, the vitality and sustainability of local economies (table 1). These benefits accrue nationally and at the local and regional levels.

National Benefits of Clean Roadless-Area Water. At least 124 million Americans directly benefit from water

Figure 1

Federal inventoried roadless areas (IRAs) of the United States (Source: USDA Forest Service).



originating from national forests (Sedell et al. 2000). In fact, national forests provide about 15% of the nation's runoff with an estimated net value of $3.7 (Sedell et al. 2000) to $27 billion (Krieger 2001). The water treatment value alone of National Forests ranges from $490 million (Loomis 2005) to $18 billion (Krieger 2001).

Because IRAs represent roughly a third of national forestland, by inference they contribute significantly to the overall runoff volume and value (Anderson 1997, 2008) estimated in billions of dollars annually (Loomis and Richardson 2001; Sechhi et al. 2005). For instance, using Forest Service data (USFS 2000), IRAs make up 661 of the 914 national forest watersheds, with 55% of the 914 watersheds acting as source areas for facilities that treat and distribute drinking water to the public. The cost-savings to water treatment plants and highway departments from avoiding sedimentation caused by logging in IRA watersheds is estimated at up to $18 billion annually (Loomis 1988). IRAs provide $490 million annually in waste treat-

Figure 2

Colorado's 2001 inventoried roadless areas (IRAs) are shown in light gray, the 2011 proposed Colorado roadless areas (CRAs) are shown in gray, and overlap between CRAs and IRAs is shown in black. Water quality will be most impacted by changes of allowable activities within existing IRAs relative to changes in designated areas (USFS 2011).

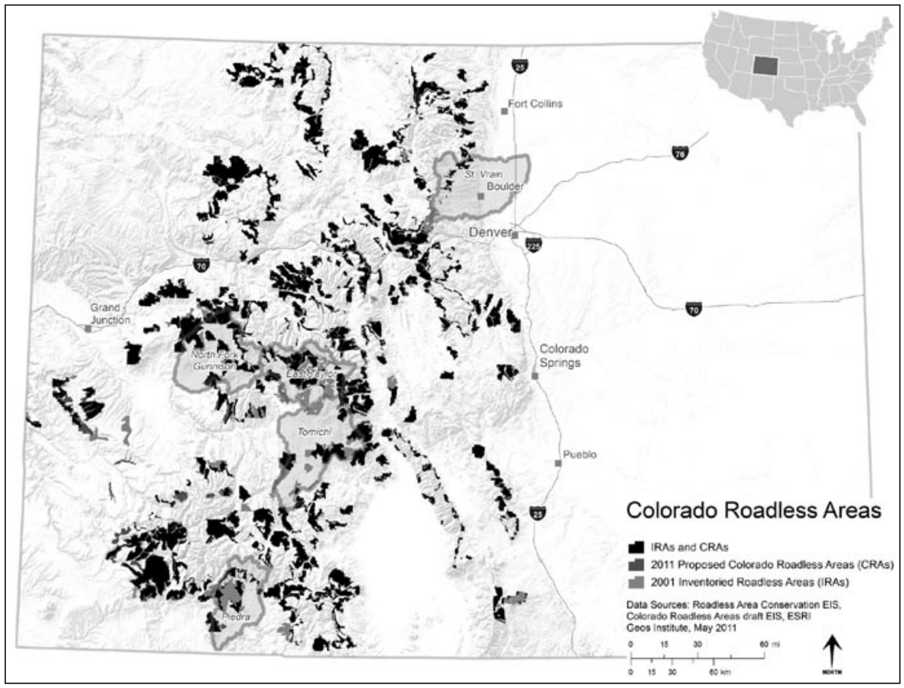


Table 1

General ecosystem services and benefits related to water that are provided by undisturbed IRAs and watersheds (derived from Greenway 1996; Costanza et al. 1997; Talberth and Moskowitz 1999; GAO 2000;Heal 2000, Loomis and Richardson 2001; Sedell et al. 2000; Krieger 2001; Dombeck 2003; Berrens et al. 2006).

	Benefits
Off-stream benefits	Low treatment costs for water for all beneficiaries
	Low price per unit volume costs for water for all beneficiaries
	High-quality and abundant drinking water for rural communities and municipal water supplies
	High-quality water for agricultural and industrial purposes
	High-quality water for downstream livestock production
	High-quality water for reduced health care and epidemic control
	Reduced costs of flood damage and flood control; enhanced local economies and property values
	Community benefits, including jobs, income, favorable trends for key economic indicators, and economic sustainability and stability
	Recharging of groundwater aquifers
	Healthy terrestrial and riparian ecosystems and their component species, sustained ecological and evolutionary processes, and resilient ecosystems
In-stream benefits	Healthy aquatic ecosystems
	Recovery of endangered species and protection of refugia
	Diverse and productive fisheries
	High-quality habitat for wildlife, including migratory waterfowl and game and nongame species
	Aquatic recreation such as swimming, rafting, and boating; enhancement of hiking and camping
	The inherent value of wild rivers and wilderness (including passive use benefits such as option, bequest, and existence values)
	Moderation of runoff and streamflows (e.g., lower peak flows, higher low flows, year-round water)
	Soil stabilization and erosion control
	Scientific value (intact watersheds are very rare today)
	Maintaining sediment production to streams at normal background rates
	Reducing potential for damage to downstream properties and water users during periods of high flow
	Breakdown and containment of waste and toxins (e.g., atmospheric, prior use)

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ment services through recovering mobile nutrients and cleansing the environment, both processes that involve water flow through intact watersheds (Loomis and Richardson 2001).

Regional Benefits of Clean Roadless-Area Water. In the US Rocky Mountains, roughly one third of utilized streamflow is derived directly from IRAs (which cover a quarter of Colorado's headwaters), with cities like Denver receiving about 30% of their water supply from IRA watersheds. Annually, IRAs in Colorado are estimated to provide an equivalent of nearly 2.5 times Denver's annual water use (Doyle and Gardner 2010; Denver Water 2010). Similarly, IRAs in New Mexico provide an estimated water quality benefit up to $42 million annually (Berrens et al. 2006).

Flood Control Protection and Inventoried Roadless Areas. The intact watersheds of IRAs are especially important for ameliorating the frequency and intensity of flooding, saving millions of dollars annually from averted floods and associated sedimentation, a service that will only increase in value as climate change drives more floods (Seeds 2010). Dredging reservoirs to increase capacity and channels to enable navigation costs cities, states, and ultimately taxpayers millions annually. Salem, Oregon, spent approximately $100 million on new treatment facilities after logging in upper watersheds created conditions leading to mass sedimentation in its watershed following storms in 1996 (Schwickert and Mauldin 1997; Talberth and Moskowitz 1999). In addition, Seattle, Washington, deferred a $150 million filtration plant expenditure through an intensive watershed rehabilitation program that will decommission 480 km (300 mi) of roads over a 10-year period, fix road erosion problems, and limit access and high-risk activities for fire and sedimentation within their watersheds (Seeds 2010).

Recreation Benefits and Strong Local Economies. IRA water benefits outdoor recreation and the people that either engage in or earn their living from outdoor recreation. The nation's IRAs generate $600 million annually from recreation (Loomis and Richardson 2001). Passive-use values (i.e., the intrinsic value of wilderness, wildlands, and benefits for the future) are estimated at an additional $280 million annually. At the regional scale, New Mexico IRA water provides an estimated $27 million active outdoor recreation benefit and a $14 million passive-use benefit annually (Berrens et al. 2006). For many visitors, much of the attraction to wildlands is associated with the presence of clean and abundant water—a dwindling resource as logging, grazing, and road-building continues across mountain landscapes and droughts from a changing climate intensify in much of the West (Saunders et al. 2008).

Freshwater Biodiversity and Healthy Fisheries. Clean water from IRAs also maintains healthy fisheries, such as salmon and trout fisheries, sustains viable aquatic ecosystems, and helps protect threatened species and ecosystems (Abell et al. 2000; Trout Unlimited 2004). Indeed, IRAs may act as important refugia for many salmon and trout populations, as well as for a diversity of endangered freshwater species (Henjum et al. 1994; Huntington 1998; NRC 1996; Trombulak and Frissell 2000; CBD et al. 2001; Strittholt and DellaSala 2001; Oechsli and Frissell 2002; Strittholt et al. 2004; Petersen 2005). Restoration of salmon and trout fisheries in places with high road densities will likely fail without the pivotal role provided by IRAs as fishery strongholds.

ROADLESS AREAS ARE IMPORTANT SOURCES FOR DRINKING WATER

The distribution of IRAs across prime hydrologic real estate—headwaters and upper watersheds—makes them particularly valuable for providing reliable supplies of clean water. In Colorado, IRAs occur in the headwaters of all major drainages, covering roughly a third of upper watersheds in the state. Indeed, most IRAs are located in mountainous terrain in western states, including Oregon, Idaho, New Mexico, Utah, Montana, California, and Washington. This extensive coverage of IRAs in headwaters, and because they are often the last minimally disturbed watersheds within larger landscapes of degraded lands, makes them hydrologic hotspots—areas with relatively small spatial extent that have a disproportionately important role in producing abundant and reliable clean water (Frissell and Carnefix 2007).

For many major drainages (entire watersheds of major rivers, such as the Columbia River Basin), IRAs and other wilderness areas represent the last few percentages (typically 1% to 5%) of the landscape with a minimally disturbed, or near natural, hydrology. As in many other ecological contexts, losing the last relatively natural systems typically results in major losses in water resource benefits, losses that can only be compensated by very expensive actions. The known relationship between watershed degradation and water quality decline deserves to be more rigorously incorporated as a central foundation for decisions on watershed management and protection.

Developing Roadless Areas Degrades Water Quality. In addition to their keystone location within watersheds, roadless areas typically encompass the most fragile of natural landscapes—montane forests and meadows. Road building and other intensive management in these otherwise intact areas damage their ability to provide clean water for downstream communities and biodiversity over both short and long terms (Beschta 1978; Forman and Alexander 1998; Lugo and Gucinski 2000; Trombulak and Frissell 2000; Gucinski et al. 2001; Coffin 2007). Logging, including post-disturbance, fire-risk reduction, forest health, and insect control; livestock grazing; mining; and road building are responsible for chronic and acute sedimentation of aquatic ecosystems, alter overland flow and stream structure, and change a range of physical and biological features by causing more frequent and intense floods, decreasing available water throughout the year, increasing stream and ambient temperatures, and elevating turbidity and nutrient levels (Beschta 1978; Fleischner 1994; Trombulak and Frissell 2000; DellaSala et al. 2006; Coffin 2007). Logging roads have been linked to great increases in erosion rates and sediment delivery to streams—up to 850% over rates in undisturbed habitat—with long-term and often catastrophic impacts on stream biota, aquatic ecosystems, and water quality (Fredricksen 1970; Megahan and Kidd 1972; Amaranthus et al. 1985; Bilby et al. 1989; King 1989, 1993; Haynes and Horne 1997; Jones et al. 2000; Wemple and Jones 2003).

Depending on severity and duration of impacts, disturbance can elevate average turbidity levels well above background levels (Seeds [2010] provides examples from Oregon), along with triggering more frequent and intense turbidity spikes that are a major source of excess costs to municipal water supply departments. Relative to roadless watersheds with intact natural vegetation, intensively managed watersheds also produce less available water (i.e., average monthly usable raw water) due to intensified high flows with very high turbidity and exacerbated low flow conditions (Seeds 2010). The monthly reliability of water is also diminished.

Even small disturbances in upper watersheds can result in significant, cumulative, and long-term impacts to downstream water and aquatic ecosystems (Platts and Nelson 1985; Boise National Forest 1993; McIntosh et al. 1994, 1995). In unstable terrain, for instance, small areas (e.g., less than 10% of a watershed's area) of low-intensity disturbance, including roads, may greatly increase the frequency and size of mass erosion events, with subsequent acute and chronic reduction in downstream water quality. Management activities that damage natural vegetation typically result in loads of suspended solids that exceed background levels and more frequent and intense spikes in suspended solids stemming from an increase in mass erosion events like landslides, debris flows, and bank failures. These impacts are strongly correlated with roads, as well as with logging and grazing (Amaranthus et al. 1985; Fleischner 1994; Trombulak and Frissell 2000; Coffin 2007).

Rising Demand and Climate Change Diminish Water Supply. Population in the West is projected to increase by 300% within just 30 years, with similar increases in demand for water (Sedell et al. 2000). Urban and exurban areas are growing exponentially, including communities adjacent to wilderness areas and IRAs (Theobald 2005). The demand for water in Colorado is expected to triple by 2050. Similarly, the number of people relying on national forest water has doubled in Oregon in the last 30 years, and 86% of the population of Washington rely on national forest water to some degree (Sedell et al. 2000).

The dramatic population growth in the West is concurrent with a warming and drying climate in many places. Temperatures are increasing, snow pack is declining and melting sooner, and drought and summer water deficits are more frequent and longer (Barnett et al. 2008; Mohammed and Tarboton 2008; Saunders et al. 2008; Miller et al. 2010). Streamflow reductions ranging from 10% to 35% are likely for the western states over the next half century as a consequence of climate change (Barnett and Pierce 2009). A 10% drop in streamflow is considered calamitous by municipal water districts. More frequent and intense flood events are also likely in places (Raff et al. 2009), despite drying conditions. Costs for flood control, repair and reconstruction, and insurance rates will also increase (GAO 2007). These events will worsen the severe and unprecedented droughts already afflicting much of the West (Drechsler et al. 2006; Saunders et al. 2008).

SOLUTION: A LIGHT HYDROLOGICAL FOOTPRINT IN ROADLESS AREAS

IRAs should be managed in the same way many municipalities manage their watersheds—sustaining a light ecological and hydrological footprint and hydrologic restoration through decommissioning or, even better, obliteration of roads (Barten et al. 1998; NRC 2000; Payne et al. 2004; Gallo et al. 2005; Postel and Thompson 2005; Seeds 2010). The most cost-effective and prudent approach to maintain water supplies and high-quality fresh water in the face of population growth and climate change is to manage upper watersheds in a roadless condition with undisturbed natural vegetation. The high, long-term economic cost of degrading clean water for millions of people, by itself, is argument strong enough to continue protection of the current roadless areas network either at national or state levels. Development of IRAs, as proposed in Colorado, would primarily provide opportunities for short-term gains, but the substantial and long-term impacts on water quality and availability will come at a time of increasing demand and shrinking supply. Managers should, therefore, treat IRAs as natural reservoirs of high quality water for downstream users before approving development projects. Cost-benefit analyses should include regionally and locally specific estimates of water quality to better inform project management decisions that may reduce the value of high-quality water in the short and long run.

CONCLUSIONS

Roadless areas and the relatively intact ecosystems they maintain provide many important biodiversity benefits, including acting as strongholds for threatened freshwater species. Beyond these important values, their role in producing clean and reliable water for people and economies is more likely to compel decision-makers to leave roadless areas undeveloped. We reviewed the importance of inventoried roadless areas on national forests in the United States to determine their importance in providing clean water for downstream users. We concluded that (1) many intact watersheds are in headwaters, (2) they supply downstream users with high-quality drinking water, and (3) developing these watersheds comes at significant costs associated with declining water quality and availability. Several case studies from the western United States, particularly Colorado, demonstrated the importance of assessing the diverse consequences of developing roadless areas. Managers should perform comprehensive cost-benefit analyses when weighing development options. A light-touch hydrological footprint is recommended to sustain the many values that derive from roadless areas, especially clean and abundant water.

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Complex Early Seral Forests of the Sierra Nevada: What are They and How Can They Be Managed for Ecological Integrity?

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R E S E A R C H   N O T E

Complex Early Seral Forests of the Sierra Nevada: What are They and How Can They Be Managed for Ecological Integrity?

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ABSTRACT: Complex early seral forests (CESFs) occupy potentially forested sites after a stand-replacement disturbance and before re-establishment of a closed-forest canopy. Such young forests contain numbers and kinds of biological legacies missing from those produced by commercial forestry operations. In the Sierra Nevada of California, CESFs are most often produced by mixed-severity fires, which include landscape patches burned at high severity. These forests support diverse plant and wildlife communities rarely found elsewhere in the Sierra Nevada. Severe fires are, therefore, essential to the region's ecological integrity. Ecologically detrimental management of CESFs, or unburned forests that may become CESF's following fire, is degrading the region's globally outstanding qualities. Unlike old-growth forests, CESFs have received little attention in conservation and reserve management. Thus, we describe important ecological attributes of CESFs and distinguish them from early seral conditions created by logging. We recommend eight best management practices in CESFs for achieving ecological integrity on federal lands in the mixed-conifer region of the Sierra Nevada.

Index terms: complex early seral forests, ecological integrity, mixed-severity fire, Sierra Nevada

INTRODUCTION

Early seral forests are ecosystems that occupy potentially forested sites after a stand-replacement disturbance and before re-establishment of a closed forest canopy (Swanson et al. 2011). Such forests are generated by disturbances that reset successional processes and follow a pathway that is influenced by biological legacies (e.g., large live and dead trees, downed logs, seed banks, resprout tissue, fungi, and other live and dead biomass) that were not removed during the initial disturbance (Franklin et al. 2000; Donato et al. 2012). Where these legacies are intact, complex early successional forests (CESFs) develop with rich biodiversity due to the function of the remaining biomass in providing resources to many life forms and because of habitat heterogeneity provided by mixed-severity fires that generated them (Odion and Sarr 2007; Swanson et al. 2011). In general, mixed-severity fires, which include patches of high-severity fire, create coarse-grained, high-contrast heterogeneity that results in CESFs, and, over time, a complex mosaic of seral stages at the landscape and local scales. Low to moderate fire severities create fine-grained, lower contrast heterogeneity that generate very little if any CESFs, although they create other conditions favorable to biodiversity. Many effects of fire cannot be mimicked by land-use disturbances (Odion and Sarr 2007). Suppression of fire and removal of biomass after a fire are thus causes of reduced biodiversity and ecological integrity.

While the unique "floral phoenix" that follows stand-replacing fire in many vegetation types such as the California chaparral has long inspired botanists in the United States (Brandegee 1891; Howell 1946) and elsewhere (Bond and van Wilgen 1996), similar attention has not been given to stand-replacing fire in Sierran forests. Instead, fire has been suppressed in these forests for many decades. Traditionally, stand-replacement processes have also been considered historically unimportant in these forests, simply because they occur less frequently than surface fires, which are largely non-lethal (Skinner and Chang 1996). Stand-replacing fire also has a negative connotation in resource management disciplines because of their narrow focus on impacts to timber values, and such fires frequently receive negative coverage from the mass media.

While much of the conservation attention in the Sierra Nevada has rightfully focused on iconic conifers like the giant sequoia (*Sequoiadendron giganteum*) and other old-growth forest types, even in the context of multiple-use management and conservation, there is still little appreciation for CESFs, which do not have the charismatic old-growth species and living structures (Swanson et al. 2011). Thus, for a variety of reasons, there is a paucity of literature on, or appreciation of, CESFs. Indeed, CESFs are not even recognized as a distinct habitat type in any current vegetation mapping used by the U.S. Forest Service in the Sierra Nevada (e.g., California Wildlife Habitat Relations). However, in terms of their contribution to biodiversity and vital life-history stages of many species, CESFs have disproportionately important ecological roles in the overall ecological integrity of forested landscapes. Thus, we call attention to this successional stage (Swanson et al. 2011) and the need for its inclusion in conservation strategies in the Sierra Nevada ecoregion.

It is timely to consider CESFs in Sierra conservation strategies because the Sequoia, Sierra, and Inyo National Forests (Figure 1) are undergoing forest plan revisions as part of the "early adopters" of the forest-planning rule (36 Code of Federal Regulations Part 219). The forest-planning rule directs the U.S. Forest Service to maintain or improve ecological integrity, defined as "the quality or condition of an ecosystem when its dominant ecological characteristics (for example, composition, structure, function, connectivity, and species composition and diversity) occur within the natural range of variation and can withstand and recover from most perturbations imposed by natural environmental dynamics or human influence" (Forest Planning Rule 36 CFR 219.19). Given the global importance of the Sierra Nevada ecoregion (Ricketts et al. 1999), many scientists and the public expect a high level of protection and stewardship in forest-planning decisions and they support managing for ecological integrity. But, as an often-overlooked seral stage, the role of CESFs in ecological integrity and conserving biodiversity has not been addressed.

We address three questions of management relevance to CESFs in the Sierra Nevada: (1) what are CESFs and why are they important to ecological integrity; (2) are there tradeoffs for managing species of conservation concern that occur at opposite ends of the successional continuum such as Black-backed Woodpeckers (*Picoides villosus*; avian taxonomy follows American Ornithologists' Union checklist of North and Middle American birds; http://checklist.aou.org/; active May 20, 2013) and California Spotted Owls (*Strix occidentalis occidentalis*); and (3) what are the principal threats to these forests? We also provide general recommendations for conserving, restoring, and researching the ecological integrity and biodiversity of Sierran CESFs.

STUDY AREA

The Sierra Nevada ecoregion spans some 63,111 km^2 along a north-south axis in California, and the USDA Forest Service manages the majority of montane forests in this region (Davis and Stoms 1996; Figure 1). The ecoregion is among the most diverse temperate conifer forests in the world and its conservation status is considered critically endangered due to extensive forest fragmentation and other land-use stressors (Ricketts et al. 1999). An extraordinary assortment of vegetation types and diverse forest successional stages occur across the region. For instance, based on potential vegetation mapping, 25 conifer, 23 hardwood forest/woodland types, 34 shrub and chaparral, and 5 herbaceous alliances are distributed across elevations, slopes, aspects, and soil types (USDA Forest Service 2008). Plant alliances mix together at zones of overlap resulting in high levels of beta diversity (change in numbers of species across environmental gradients). There are exceptional levels of endemic plants (e.g., approximately 405 vascular plants are endemic and 218 taxa are rare; Shevock 1996), especially in the southern Sierra, and some of the highest levels of mammal endemism in North America (Ricketts et al. 1999). Notably, areas with high concentrations of endemic species are a conservation priority because the restricted distribution of endemics predisposes them to extinction from habitat losses.

Mixed-conifer forests are the predominant forests in the Sierra that are typically found at middle elevations (760–1400 m) in the northern Sierra, higher elevations south (915–3050 m), and, to a lesser extent, on upper elevations (2130 m to 3040 m) along the east slopes (Chang 1996). They are replaced at higher elevations by pure red fir (*Abies magnifica,* Andr. Murray) and red and white fir (*A. concolor,* Gordon & Glend.) (Barbour et al. 2007). There are three forest types that comprise mixed-conifer forests in this region: (1) white fir/Jeffrey pine (*Pinus jeffreyi,* Grev. & Balf.) /lodgepole pine (*P. contorta,* Loudon); (2) Pacific Douglas-fir (*Pseudotsuga menziesii menziesii,* Franco), and ponderosa pine (*P. ponderosa*; at lower elevations); and (3) mid-elevation Douglas-fir (does not occur south of Yosemite National Park). These more typical conifers are associated with sugar pine (*P. lambertiana,* Douglas), incense cedar (*Calocedrus decurrens,* Torrey), black oak (*Quercus kelloggii,* Newb.), and patches of giant sequoia. Mixed-conifer forest types also support shrubs such as greenleaf manzanita (*Arctostaphylos patula,* E. Greene), huckleberry oak (*Q. vaccinifolia*, Kellogg), curleaf mountain mahogany (*Cercocarpus ledifolius,* Nutt.), snowbrush (*Ceanothus velutinus,* Dougl.), mountain alder (*Alnus incana* ssp. *tenuifolia,* Nutt.), mountain sagebrush (*Artemisia tridentate* ssp. *vaseyana,* Rydb.), and bitterbrush (*Purshia tridentata,* Pursh) (USDA Forest Service 2013a). Most of these forests consist of mid-sized trees that average 30–60 cm dbh and include areas with larger trees (>60 cm dbh; North 2013); nearly half of the mixed-conifer forest in the giant sequoia type is late seral (USDA Forest Service 2013a).

Very-long-interval, stand-replacement fire occurs in a patchwise fashion within low- and mixed-severity fires in moist mixed-conifer and white fir forests in this region, and variable (both short- and long-interval) stand-replacement fires occur in Douglas-fir and lodgepole pine (*Pinus contorta,* Loudon) forests (Leiberg 1902; Chang 1996). Prior to fire suppression, drier low-elevation forests burned relatively frequently and often at a low severity; but they also had significant mixed-severity effects, including occasional large high-severity fire patches (USDA Forest Service 1911).

What are Early Seral Forests and Why Are They Important?

In general, CESFs are rich in post-disturbance legacies (Photo Plates 1a, 1b, 1c) and post-fire vegetation (e.g., native fire-following shrubs/herbs, resprouting broad-leaved trees, and natural conifer regeneration) (Photo Plates 2a, 2b, 2c). We identify 12 ecological attributes that contribute to the prolific biological response common in CESFs and which are, therefore, key to the ecological integrity present in CESFs

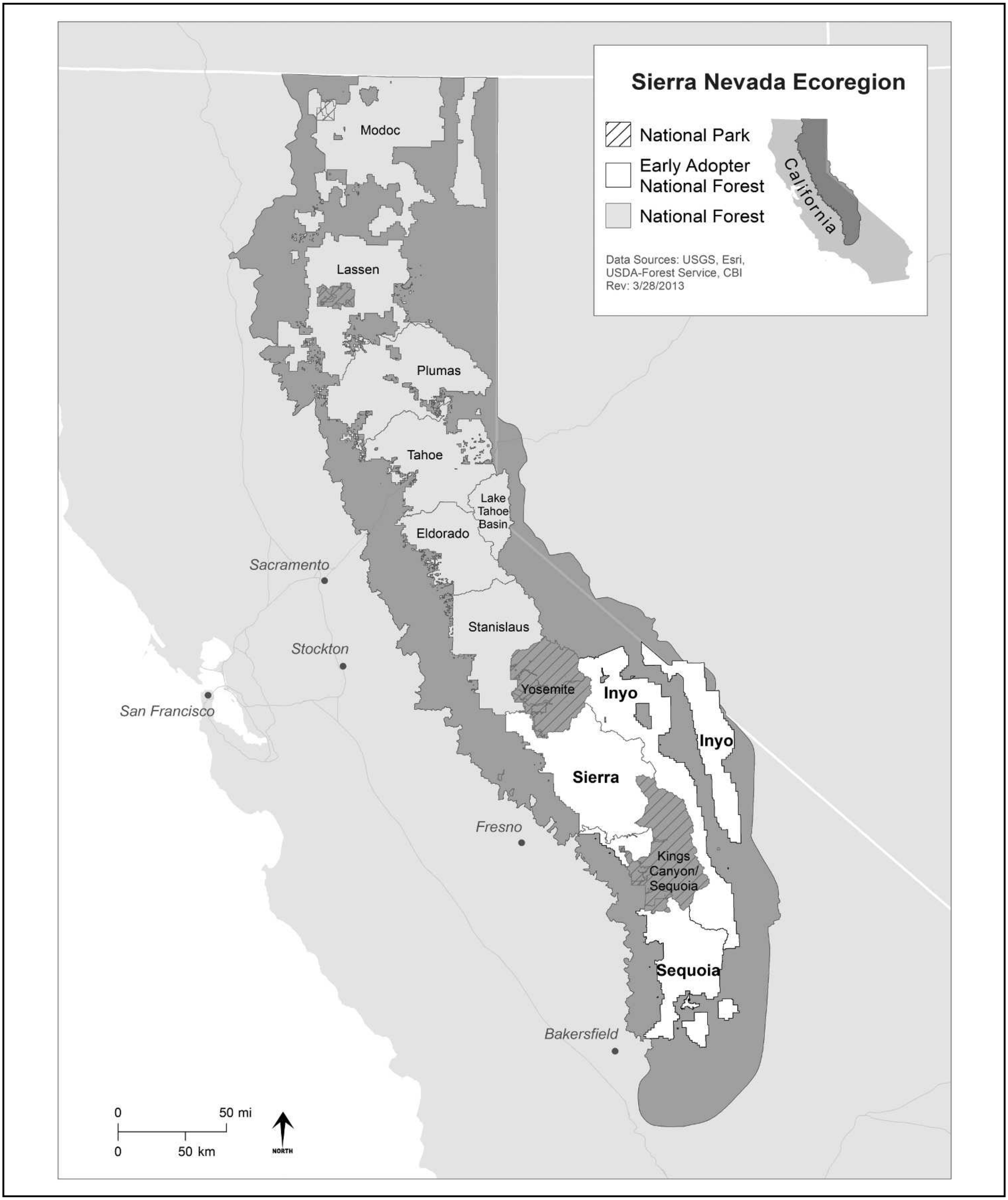

Figure 1. Location of Sierra Nevada ecoregion, northern California, and "early adopters" of the forest-planning rule involved in forest plan revisions.

Photo Plates. Extensive biological legacies, abundant forb cover, and abundant conifer regeneration present in complex early seral forests vs. early seral that has been post-fire logged. Post-fire logging in the Sierra Nevada and elsewhere sets back ecosystem processes creating a successional debt.

Plate 1a Plate 1b Plate 1c

Photo Plate 1. Star Fire of 2001, Northern Sierra, CA. (a) unmanaged with forbs (Doug Bevington, 2008); (b) natural conifer re-establishment (Chad Hanson, 2012); Storrie fire of 2000, Southern Cascades, CA. (c) unmanaged with snags and forbs (Chad Hanson, 2007).

Plate 2a Plate 2b Plate 2c

Photo Plate 2. Postfire logged portions of Fred's fire in the Eldorado National Forest, CA, showing lack of nitrogen-fixing shrubs (a) and presence of Klamath weed (*Hypericum perfoliatum*) and many readily ignitable, invasive grasses (b) (Dennis Odion, August 2011); (c) simplified system from Dinkey post-fire thin on west slopes of Southern Sierra (Chad Hanson, 2012).

(Table 1). When logging compounds the natural disturbance that created a CESF (Photo Plates 3a, 3b, 3c), each of these attributes is reduced or eliminated (Table 1). Such multiple disturbances often lead to alternative successional pathways, or loss of resilience (Paine et al. 1998; Odion and Sarr 2007), as has been documented in the Sierra Nevada following post-fire logging, which leads to dominance by the non-native ecosystem transformer, cheatgrass (*Bromus tectorum,* Linnaeus) (McGinnis et al. 2010).

Overall, compared to logged areas, CESFs are structurally more complex, contain more large trees and snags that originated from the pre-disturbed forest, have more diverse understories, functional ecosystem processes, and more diverse gene pools that, theoretically, should provide greater resilience in the face of climate change than that provided by the simplified early seral forests produced by logging. CESF attributes promote a high level of species richness, particularly bird communities that utilize these forests extensively (Hutto 1995; Kotliar et al. 2002; Fontaine et al. 2009; Appendix). The residual biomass of CESFs reduces disturbance stressors and provides for the rapid proliferation of new life (Odion and Sarr 2007). For example, seed banks and vegetation tissues give rise to dense, often rampant, forb cover, abundant grasses, and shrubs – especially nitrogen fixers (e.g., *Ceanothus* spp.) (Conard 1985; Busse et al. 1996; Busse 2001) and ectomycorrhizal associates (e.g., *Manzanita* spp.) that facilitate conifer growth (Zavitovsky and Newton 1968; Horton et al. 1999). Serotinous (closed cone) conifers like giant sequoia (Stephenson et al. 1991) also do well in these forests. Other plants that can abundantly colonize burns, such as conifers and fireweed (*Epilobium angustifolium,* Linnaeus), arrive by wind or animal dispersed seed. Thus, plant species richness of CESFs can be much higher than in unburned forests (Donato et al. 2009).

Other bird and small mammal communities that utilize CESFs forage extensively on the abundant insects and increased abundance of seeds from the post-fire flora (Lawrence 1966; Fontaine et al. 2009). These species, in turn, support an increase in raptors (Lawrence 1966). Bird species such as the Black-backed Woodpecker, Olive-sided Flycatcher (*Contopus cooperi*), Mountain Bluebird (*Sialia currucoides*), Chipping Sparrow (*Spizella passerina*), and Mountain Quail (*Oreortyx pictus*) (Appendix)

Table 1. Differences between early seral systems produced by natural disturbance processes vs. logging. For natural disturbances, assume that a disturbance originates from within a late-successional forest as legacies are maintained throughout succession. For logged sites, assume site preparation includes conifer plantings but no herbicides, which, if also applied, would magnify noted differences.

Attribute	Regeneration Harvest or Postfire Logged	Natural Disturbance
Large trees	rare	abundant and widely distributed
Large snags/downed logs	rare	abundant and widely distributed
Understory	dense conifer plantings followed by sparse vegetation as conifer crowns close (usually within 15-20 years depending on site productivity)	varied and rich flora
Species composition	few species mostly commercially stocked, deer initially abundant then excluded as conifer crowns close	varied and rich flora, rich invertebrates and birds, abundant deer
Structural complexity	simplified	highly complex; many biological legacies
Soils and below-ground processes	compacted and reduced mycorrhizae	complex and functional below ground mats
Genetic diversity	low due to emphasis on commercial species and nursery genomes	complex and varied
Ecosystem processes (predation, pollination)	moderate initially then sparse as conifer crowns close; limited food web dynamics	rich pollinators and complex food web dynamics
Susceptibility to invasives	moderate to high depending on site preparation, soil disturbances, livestock, road densities (see McGinnis et al. 2010)	low due to resistance by diverse and abundant native species and low soil disturbances
Disturbance frequency	commercial rotations (40-100 years or so)	varied and complex
Landscape heterogeneity	low	high; shifting mosaics and disturbance dynamics
Resilience/resistance to climate change	low due to nursery stock genomes but conifer plantings can be adjusted for locally anticipated climate envelopes	varied and complex genomes allow for resilience and resistance to climate change

achieve highest abundances in CESFs. In fact, in the Sierra Nevada, CESF habitat is comparable or higher in bird species richness and total bird abundance relative to unburned mature forest (Burnett et al. 2010). Bats (*Myotis, Idionycteris, Lasionycteris*, and *Eptesicus)*, which are an increasing conservation concern, are also favored by CESFs, likely because of greater insect prey as well as suitable roosts (Buchalski et al. 2013). Stand-replacing fires stimulate a flux of aquatic prey to terrestrial habitats, driving increases in riparian consumers (Malison and Baxter 2010). The trees killed by fire are highly beneficial to the ecological integrity of stream communities because they are a main source of large woody debris inputs (Minshall et al. 1997). There is also reproduction by some forest fungi species that are restricted to burns (e.g., morels, *Morchella* spp.) and the dead wood provides substrate for fungal growth that supports many arthropod species, including unique fire-following native beetles (Lindsey 1943; Bradley and Tueller 2001). Beetles, in general, colonize fire-killed trees in CESFs and their abundant larvae support species like Black-backed Woodpeckers (Hutto 2008).

Indicator Species for CESF Biodiversity (Figure 3)

Indicator species are valuable tools for conservation management because it is not practicable to monitor all biodiversity. When burned forests are logged after fire, one species that serves well as an ecological indicator for post-fire biodiversity, the Black-backed Woodpecker, declines substantially (Hutto 2008). Given that this woodpecker already is an indicator of the biodiversity supported by CESFs in the Sierra Nevada (USDA Forest Service 2013b), and is a fire specialist, we propose it as a Species of Conservation Concern. Designated Species of Conservation Concern are those whose population viability, or continued representation within a particular plan area, is of management concern. The forest-planning rule provides guidance to forest managers to use Species of Conservation Concern as a means for maintaining species diversity and wildlife population viability.

CESF habitat represented by Black-backed Woodpeckers is biologically unique (Hutto 1995; Bond et al. 2012). The Black-backed Woodpecker is an important primary excavator of nesting holes for many other cavity-nesting birds and mammals because it discards cavities after excavating them, and it uses a given cavity for one year (Tarbill 2010). Under a scenario with stand-replacing fire operating in a patchwise fashion in a landscape containing healthy populations of Black-backed Woodpeckers, the availability of nesting cavities across the landscape over time may be greatly enhanced compared to where fire is suppressed and/or fire-killed trees are removed. Black-backed Woodpeckers use CESFs for only several years (typically seven or eight) after fire and they depend upon the regular creation of CESFs to replenish their habitat (Hanson and North 2008; Tarbill 2010; Dudley et al. 2012; Siegel et al. 2013). When this does not occur, many other species that rely on nesting cavities are likely to be negatively affected. Thus, many species probably depend directly, or indirectly, on the continued occurrence of high-intensity natural disturbance across large landscapes to maintain their populations (Hanson and North 2008; Tarbill 2010; Dudley et al. 2012; Siegel et al. 2013).

Black-backed Woodpeckers have become increasingly rare because their optimal habitat has shrunk to a fraction of its historical extent (Figure 2 a – d); populations are estimated at <700 nesting pairs in burned forests (Bond et al. 2012). Importantly, the CESF habitat that the remaining pairs depend on has little or no protection on public lands managed by the U.S. Forest Service. Much of this CESF habitat is under mounting pressure from fire suppression and both pre- and post-fire logging (Hutto and Gallo 2006; Hanson and North 2008; Hutto 2008; Siegel et al. 2013), which prevent high-quality woodpecker habitat. That, in turn, may affect the biodiversity for which this woodpecker serves as an indicator.

Figure 3. Black-backed woodpecker – a fire dependent species in the Sierra (Photo – Monica Bond).

Are There Management Tradeoffs for Species of Conservation Concern at Opposite Ends of the Successional Continuum?

Wildlife management often involves tradeoffs when habitat for a particular species is emphasized. That is a problem with single-species management (managing for what one species needs), but is not a problem when managing for the maintenance of natural systems that a species may indicate. In the latter case, we would not enhance but would maintain natural levels of habitat for CESF indicators like the Black-backed Woodpecker, and for the biodiversity associated with its presence.

However, the California Spotted Owl is also a management indicator species but for late-seral forests in this region. Notably, all three subspecies (Mexican, California, Northern; Bond et al. 2002; Jenness et al. 2004; Roberts 2008; Bond et al. 2009; Clark et al. 2011; Roberts et al. 2011; Lee et al. 2012; Clark et al. 2013) appear to tolerate, or even benefit, from some degree of moderate- to high-severity fire within territories.

Managing CESFs for high levels of ecological integrity may provide important prey habitat (e.g., dusky-footed woodrat *Neotoma fuscipes*; Munton et al. 2002) for the spotted owl. In fact, the owl is known to reproduce in territories burned at all fire severities in this region, and preferentially selects high-severity fire areas for foraging (Bond et al. 2009). Owl reproduction has been found to be 60% higher in unmanaged mixed-severity fire areas than in unburned forests (Roberts 2008), and mixed-severity fire (with an average of 32% high severity) (Lee et al. 2012) does not reduce owl occupancy, though post-fire logging may precipitate territory abandonment (Clark et al. 2011, 2013; Lee et al. 2012). Moreover, because high-severity fire has been reduced by fire suppression, and current high-severity fire rotations are very long in the Sierra Nevada, if high-severity fire rates increased by even two- or three-fold, it would benefit CESF-associated species like the Black-backed Woodpecker, but would only reduce current old forest by a very small amount given old forest recruitment from ingrowth (Odion and Hanson 2013). Thus, protecting CESFs from post-fire logging and maintaining the spatial heterogeneity created by mixed-severity fires should provide habitat for all seral associates – there really are no management trade-offs when we manage for the maintenance of natural processes and systems.

What are Principal Threats to CESFS?

Management of CESFs has most often included post-fire (salvage) logging followed by tree planting, including burning of slash piles and associated soil disturbances, reseeding with grasses (often introducing invasive species inadvertently), use of straw-bales and other erosion prevention methods, herbicides to reduce shrub competition with conifers,

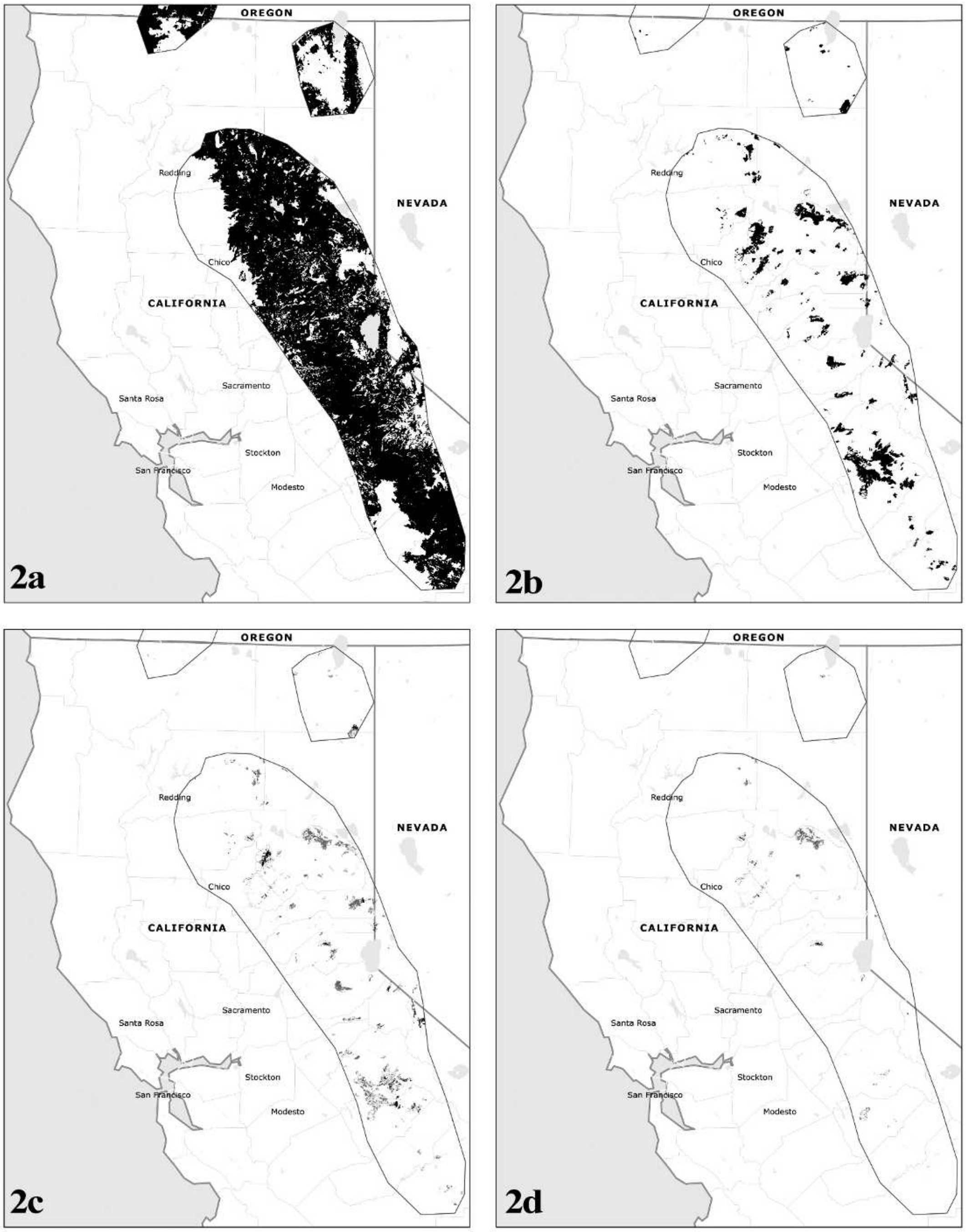

Figure 2. (a) Forest types used by Black-backed Woodpeckers in the Sierra Nevada management region; (b) fires since 1984 within the relevant forest types (private lands not included since they are rapidly logged); (c) moderate/high-severity fires resulting in >50% mortality (RdNBR >574 – see Hanson et al. 2010) of forests on public lands within the relevant forest types in the most recent decade for which there are fire severity data (2001–2010) (i.e., both high quality Black-backed Woodpecker habitat and moderate/low quality (older) habitat combined); and (d) moderate/high-severity fire on public lands within the relevant forest types in the most recent 5-year period for which fire severity data are available.

planting with conifer nursery stock, and livestock grazing (Swanson et al. 2011; Long et al. 2013; Table 1). These activities remove, or severely degrade, CESFs or, at a minimum, can narrow the window of duration for CESFs (Swanson et al. 2011), contributing to "landscape traps," whereby entire landscapes are shifted into, and then maintained in, a highly altered state as the result of cumulative impacts (Lindenmayer et al. 2011).

Climate change and forest fragmentation also have been identified as threats to biodiversity in the Sierra Nevada (USDA Forest Service 2013b). Since the 1980s, the region has experienced a decrease in annual number of days with below-freezing temperatures at higher elevations with more rain and less snowfall mainly in northern latitudes, more extreme heat days at lower elevations, earlier (5 to 10 days) snowmelt than decades ago, earlier (5–15 days) peak stream flows (Safford et al. 2012; Harpold et al. 2012), as well as an increase of approximately 1 °C since the early twentieth century, though some areas of the northern Sierra Nevada have seen a decrease in temperature (North 2012). Some regional climate models project further decreases in mountain snowpack, earlier snowmelt and peak stream flows, and greater drought severity (Overpeck et al. 2012). Such climatic changes are likely to affect the low-elevation ponderosa pine, which is projected to extend upward, while red fir and subalpine communities are projected to lose much of their climate envelope in the coming century (USDA Forest Service 2013b). It is unclear how such changes will affect CESFs. If fire increases in severity or frequency (Miller et al. 2009; Miller and Safford 2012), this could provide more opportunities for development of CESFs. This assumes there is not a concomitant increase in post-fire logging, and that fire suppression activities either cannot keep pace with climate-related fire events or prove ineffective due to the increasing influence of climate as a top-down driver of fire behavior. On the other hand, a number of climate models predict decreasing fire activity in these forests – even as temperatures rise – due to increasing precipitation, including summer precipitation and changes in vegetation (McKenzie et al. 2004; Krawchuk et al. 2009), and recent research using the largest fire severity data set to date has found no increase in fire severity in the Sierra Nevada since 1984 (Hanson and Odion, 2014; also see Odion et al. 2014 for related discussion).

Land-use stressors also magnify climate change effects on forest communities. For instance, Thorne et al. (2008) documented significant regional changes due to climate and land-use practices resulting in greater levels of disturbance compared to historical. Millar (1996) identified three paramount influences on Sierra Nevada ecosystems: (1) climate change and shifting hydrological patterns; (2) dense forests; and (3) rapidly expanding human populations. It is not known, however, whether these changes will act in concert to make CESFs more vulnerable to invading species, particularly those more suited to the changing climate and land-use disturbances.

Suggested Best Management Practices for CESF

For all the reasons outlined above, CESFs represent a neglected seral stage subject to multiple stressors that compromise ecological integrity. We, therefore, propose eight "best management practices" for stimulating conservation, restoration, and research interests in these unique forests. These principles can serve as appropriate guidelines where management goals include the maintenance of ecological integrity.

Conservation Focus

Principle 1 – "Rehabilitation" Is Not Needed After Fire Creates a Complex Early Seral Forest (Beschta et al. 2004; Swanson et al. 2011).

Fire acts as a natural restorative agent by resetting the successional clock and providing habitat for disturbance-dependent species. Although CESFs lack live trees initially and are populated by dead ones, this does not mean they require site rehabilitation or are "unhealthy" forests. In the context of ecological integrity, a functional forest system is one where the natural fire regime is of mixed-severity and has all stages of succession following stand-replacing fire. CESFs should be mapped and managed as a distinct forest habitat type.

Principle 2 – Protect Large, Old Forest Structures Across Seral Stages, and Retain Dense, Old Forests to Improve Ecological Integrity at Landscape Scales.

Large old-forest structures take decades to centuries to develop, and forest management has created a deficit through extraction. Dense, old forests provide high-quality habitat not only when they are green, but also when they experience mixed-severity fire (Hutto 2006, 2008), or snag pulses from beetles (Bond et al. 2012), as biological legacies remaining also serve to connect seral stages along the successional gradient.

Principle 3 – Mixed-severity Fire Should Be a Management Goal for Reserves.

Robust, reserve-based conservation strategies are needed to maintain the suite of seral stages and allow for climate-forced wildlife dispersals into suitable habitat. Thus, managers should allow fires to run their course in the backcountry and in reserves when not a risk to people or dwellings. This includes maintaining a landscape that includes diverse seral stages across environmental gradients (elevation, latitudinal).

Restoration and Management Focus

Principle 4 – Adopt Comprehensive Approaches to Restore Ecological Integrity in CESFs.

This starts with a restoration needs assessment (DellaSala et al. 2003) to evaluate and prioritize drivers of ecosystem degradation and best practices aimed at reducing specific stressors (see Principle 6). Most importantly, forests restored through fire usually do not need "restoration" otherwise.

Principle 5 – Limit Post-fire Management to Early Seral Forests Previously Degraded by Logging, Grazing, and Other Stressors.

Restoration approaches should identify comparable areas of high ecological integrity (e.g., unmanaged CESFs, DellaSala et al. 2003) to serve as a baseline or reference condition from which to restore degraded areas (e.g., burned plantations), and then surveillance, implementation, effectiveness, and ecological effects monitoring (Hutto and Belote 2013) should always be an integral part of the restoration activity.

Principle 6 – Reduce Land-use Stressors That Compromise the Ecological Integrity of CESFs.

Restorative measures can be active or passive depending on site-specific needs and should always be followed with well-funded monitoring (DellaSala et al. 2003). Examples include removal of livestock, invasive species abatement, road closures and obliteration, and reintroduction of fire.

Research Focus

Principle 7 – Determine Historical, Current, and Projected Future Distributions and Spatio-temporal Extent of CESFs as Well as Other Seral Stages Across the Planning Area.

This can be informed through "back-casting" approaches that reconstruct an historical baseline from combining age-structure reconstructions (e.g., from either FIA plot data or General Land Surveys from the 1800s; see techniques in Baker 2012; Williams and Baker 2012) with studies that link stand structure, disturbances and fire scar data (e.g., Sherriff and Veblen 2006), or other sources of information (e.g., USDA Forest Service 1911). Historical baselines can then be compared to current and future projected conditions under a changing climate in order to determine appropriate representation levels of CESFs and other seral stages in a planning area.

Principle 8 – Designate the Black-backed Woodpecker a "Species of Conservation Concern."

Continue, and expand upon, current monitoring efforts and, in partnership with the U.S. Fish and Wildlife Service and other experts, determine how best to meet population viability and habitat needs of this important CESF species. Although Black-backed Woodpecker populations decline as this seral stage advances (within seven years following fire), this species still functions as an indicator of early successional species because stable woodpecker populations would mean a steady supply of CESFs over time. Olive-sided Flycatcher, Chipping Sparrow, Mountain Bluebird, and other early seral species that have population peaks after declines in woodpeckers, may need to be monitored to ensure CESF conservation.

CONCLUSIONS

The forest-planning rule and its emphasis on ecological integrity, plant and animal community diversity, and Species of Conservation Concern provides the Forest Service with a unique opportunity to revise forest plans in the Sierra Nevada to meet the primary and cumulative threats that these forests now face – climate change and land-use stressors. Where the region's forests are to be managed for ecological integrity, managers will need to determine spatio-temporal occurrence of CESFs (historical and current) to allow for adequate representation of all seral stages across planning areas, particularly the rare ones that occupy opposite ends of the successional continuum (CESFs and late seral). This also means conducting field inventories in CESFs to better describe their unique attributes and ecological importance, treating CESFs as a distinctive wildlife habitat type in habitat classifications, and incorporating mixed- to high-severity fire into management goals at middle to upper elevations.

Clearly, climate change introduces uncertainties regarding how fire and other disturbance agents will operate on these forests in the future. Whether this will increase or further reduce CESFs remains to be seen. While managing for resilient ecosystems is a desired ecological objective of climate adaptation planning on the national forest system (36 Code of Federal Regulations § 219.5), it is important for managers to go beyond mechanical fuel reduction as a means for maintaining resilient ecosystem properties, and this includes acceptance of mixed- and high-severity fires as important ecosystem processes. However, resilient to natural disturbance does not necessarily mean resistant to disturbance. Sierran forests are disturbance dependent; they require severe fire for the production of CESFs.

The eight principles recommended for best management practices in CESFs in the Sierra Nevada would promote ecological resilience and allow the National Forests in this globally outstanding ecoregion to better adapt to climate change and increasing human development in the surroundings. We encourage conservationists and park managers to emphasize CESFs in reserve design and related conservation strategies as these forests are at least as important as their late-successional counterparts.

AUTHORS ENDNOTE

At the time of this publication, the Stanislaus National Forest was proposing extensive (~18,000 ha) post-fire logging of live (injured) and dead trees (including "roadside-hazard trees"), conifer re-planting, and shrub-eradication in the wake of the 2013 Rim Fire along the border of Yosemite National Park. The agency also proposes to plant conifers in high severity patches, thereby leap frogging important non-conifer dominant stages. Post-fire logging is incompatible with the needs of legions of species that depend on the presence of standing dead trees and montane chaparral.

Because of the significance of the Rim Fire as a pulse disturbance for generating CESFs, its proximity to an iconic national park, and the opportunity to educate the public about the importance of burned forest habitat, we believe the area warrants consideration for a national monument designation as did Mount St. Helens after the historic 1980 eruption. We urge managers and conservationists to give more attention to the ecological importance of CESFs in new protected areas proposals. This is especially important as we see the threat to these unique forests escalating due to increasing emphasis by federal agencies on extensive and intensive post-fire management projects.

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Appendix. Bird species present in complex early seral forests in the Sierra Nevada based on comparisons of burned and unburned plots (Raphael et al. 1987: east slopes of Sierra, University of California Sagehen Creek Field Station, pine-fir forests, ridgetop at 2100-m elevation, Burnett et al. 2012: Plumas National Forest, northeastern CA, mixed conifers, elevations 1094–2190 m: Storrie, Moonlight, and Cub mixed-severity fires). Only the Burnett et al. (2012) performed statistical analyses on bird abundances between burned and unburned plots. Taxonomy follows American Ornithologists' Union checklist of North and Middle American birds (http://checklist.aou.org/; active May 20, 2013).

Species	Present in both studies	Difference in abundance burned vs. unburned[2]
Mountain quail[3,4] *Oreortyx pictus*	+	+
American kestrel[3] *Accipiter cooperii*		
Mourning dove[3] *Zenaida macroura*		
Common nighthawk[3] *Chordeiles minor*		
Calliope hummingbird *Selasphorus calliope*	+	NS
Williamson's sapsucker *Sphyrapicus thyroideus*		
Red-breasted sapsucker *Sphyrapicus ruber*		
Hairy woodpecker *Picoides villosus*	+	+
White-headed woodpecker *Picoides albolarvatus*	+	+
Black-backed woodpecker *Picoides arcticus*	+	+
Northern flicker *Colaptes auratus*		
Olive-sided flycatcher[3] *Contopus cooperi*	+	+
Western wood-pewee *Contopus sordidulus*	+	+
Hammond's flycatcher *Empidonax hammondii*		
Dusky flycatcher *Empidonax oberholseri*	+	-
Cassin's vireo *Vireo cassinii*		-
Warbling vireo *Vireo gilvus*		NS
Stellar's jay *Cyanocitta stelleri*	+	NS
Mountain chickadee *Poecile gambeli*	+	-
Red-breasted nuthatch *Sitta canadensis*		-
White-breasted nuthatch *Sitta carolinensis*		

Cont'd.

Appendix. (Continued)

Species	Present in both studies	Difference in abundance burned vs. unburned[2]
Pygmy nuthatch[3] *Sitta pygmaea*		
Brown creeper *Certhia Americana*	+	+
Golden-crowned kinglet *Regulus satrapa*		-
Mountain bluebird[3] *Sialia currucoides*	+	+
Townsend's solitaire *Myadestes townsendii*		
Hermit thrush *Catharus guttatus*		-
American robin *Turdus migratorius*	+	NS
House wren[3] *Troglodytes aedon*	+	+
Nashville warbler *Oreothylypis ruficapilla*	+	-
MacGillivray's warbler *Geothlypis tolmiei*		NS
Yellow warbler[3] *Setophaga petechia*		NS
Yellow-rumped warbler *Setophaga coronata*	+	-
Hermit warbler *Setophaga occidentalis*		-
Green-tailed towhee[3] *Pipilo chlorurus*	+	+
Spotted towhee *Pipilo maculatus*		+
Chipping sparrow *Spizella passerina*	+	+
Brewer's sparrow[3] *Spizella breweri*		
Fox sparrow *Passerella iliaca*	+	+
Dark-eyed junco *Junco hyemalis*	+	+
Western tanager *Piranga ludoviciana*	+	-
Black-headed grosbeak *Pheucticus melanocephalus*		-
Lazuli bunting[3] *Passerina amoena*	+	+

Cont'd.

Appendix. (Continued)

Species	Present in both studies	Difference in abundance burned vs. unburned[2]
Brown-headed cowbird[3] *Molothrus ater*		
Cassin's finch *Carpodacus cassinii*	+	+
Red crossbill *Loxia curvirostra*		
Pine siskin *Carduelis pinus*	+	-
Evening grosbeak *Coccothraustes vespertinus*	+	-

[1] No fire severity estimate was given by Raphael et al. (1987) other than the Donner Ridge fire "consumed about 18,000 ha of pine-fir forests," a "few scattered mature pines and firs survived," and the fire area was "dominated by brush (primarily Ceanothus velutinus) and by regenerating conifers" (thus probably a high severity burn).

[2] (+) indicates species was significantly more abundant in burned, (-) means it was significantly less abundant, and NS means non-significant (P>0.05; Burnett et al. 2012).

AUTHOR QUERY FORM

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Cohen, 2004;

AU:1, page 380	should the brackets really be between "n" and "o"? Kindly advise.	☐
AU:2, page 382	In the provided author correction, the word “Logging” has been highlighted without any instruction in the list (u0140). Kindly advice.	☐

Chapter 13

c0065
Flight of the Phoenix: Coexisting with Mixed-Severity Fires

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s0010
ECOLOGICAL PERSPECTIVES ON MIXED-SEVERITY FIRE

p0090 We have presented compelling evidence of fire's beneficial ecological role mainly in western North America but with relevant case studies in other regions. Even though most people recognize the importance of maintaining fire on the landscape, few realize the myriad ecosystem benefits associated with large fires of mixed severity. Habitat heterogeneity, which may be maximized by mixed-severity fire that includes large patches of high severity, and the successional mosaic such fire creates, is one of the most dependable predictors of species diversity (Odion and Sarr 2007, Sitters et al., 2014). This ecological tenet has yet to be fully realized in management circles. If such fires are operating within historical bounds, then ecosystems will remain resilient to them; indeed, deficits of these fires relative to the natural range of variability, in places such as montane forests of western North America, are degrading to fire-dependent biodiversity (Odion et al., 2014a; Sherriff et al., 2014). This is particularly the case when reductions in fire extent and/or severity occur in combination with forest management practices, such as postfire logging, that undermine development of complex early seral forests (Chapter 11).

The Ecological Importance of Mixed-Severity Fire: Nature's Phoenix.

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p0095 Natural heterogeneity in vegetation types, stand structures, and successional age classes at all spatial scales and environmental settings is emerging as a strategy for enhancing forest ecosystem resilience to climate change, at least in North America (Moritz et al., 2014). This will help ensure that there will be enough habitat for species with varying postfire habitat requirements. The fire dynamic is changing in places, however, with climate change now poised in some systems to recalibrate fire behavior (Chapter 9). With the addition of ongoing pre- and postfire logging in forests and other development pressures, particularly in shrublands, this is having a combined negative impact on native biodiversity associated with both complex early seral and old-growth forest and chaparral ecosystems (e.g., Chapters 2–5).

s0015 Beneficial Fire Effects Often Take Time to Become Fully Realized

p0100 In general, for ecological acceptance of postfire landscapes to translate into improved management practices, as a prerequisite fire ecologists, land managers, and the general public all must recognize both pre- and postfire landscapes as irreplaceable habitat for fire-associated biodiversity. To a large extent, this depends on how one views the postfire landscape.

p0105 When considering the effects of fire, patience is clearly a virtue; postfire processes may take years, decades, or longer to unfold. However, land managers often rely on quick indices to assess fire effects, and this can have negative consequences. For instance, in the western United States, the US Forest Service's "burn area emergency response" (http://www.fs.fed.us/eng/rsac/baer/; accessed February 22, 2015) uses satellite images and other geospatial data in real time to classify soil "damages" immediately after fire. Similarly, the US Forest Services' Rapid Assessment of Vegetation Condition (RAVG) after Wildfire (http://www.fs.fed.us/postfirevegcondition/whatis.shtml; accessed February 22, 2015) provides estimates of "basal area losses" in forests 30-45 days following fires >400 ha. We saw in Chapter 11 that these types of rapid assessments can overestimate tree mortality given their immediate timeline compared with the delayed response of fire-affected trees. In forests, particularly pine and mixed conifer, this can lead to premature conclusions about fire "damages" and fire "catastrophes," as well as erroneous notions about high-severity fire patch size, along with a rush to "take action" at any cost and to advance "restoration" or "recovery" approaches that do far more harm than good (Box 13.1; see also DellaSala et al., 2014; Hanson, 2014).

p0110 Notably, differences in whether postfire vegetation is viewed as fuel or habitat (Haslem et al., 2011) most often are at the heart of heated conflicts between natural resource managers and conservationists. Witness these polar opposites: fire suppression (including both mechanical thinning and actions to halt active fires) versus [illegible]rn approaches for wildlife habitat (Chapter 12); postfire logging versus the pulse of biological legacies (Chapter 11); thinning versus habitat for closed-canopy species; and reseeding/replanting and shrub removal versus

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b0010 **BOX 13.1 Rapid Assessment of Vegetation Condition after Wildfire "Treatments" as Defined by the US Forest Service**

p0010 According to the US Forest Service RAVG assessments, the term *treatment* "describes any of a set of management activities that can assist the prompt recovery of forestlands. Management actions include any combination of live, dead, and dying wood removal, or disposal (with or without commercial value) by any feasible method, including but not limited to logging, piling, masticating, and burning, for site preparation. In addition, planting, seeding, and monitoring for natural regeneration without site preparation are appropriate management activities designed to foster the prompt recovery following wildfire. Treatments also include follow up activities to control vegetation that is believed to compete with desired trees during the early establishment period, usually 1 to 5 years after establishment, using any viable method that meets Land and Resource Management Plan direction."

the montane chaparral component of complex early seral forest. Where one stands on this debate can be a matter of principle and perspective, but can also stem from a lack of a comprehensive understanding of the effects of mixed-severity fire and successional processes after fire (see, e.g., Chapters 2–5). Further, while the public may consider fire to be a necessary change agent (see "Understanding the Public's Reaction to Fire," below), this seems to be tempered by whether fire is operating within "safe limits," constrained by prescribed (or "controlled") fire or reduced in intensity by tree thinning and shrub mastication. While prescribed fire is most appropriate for low-severity, high-frequency fire systems, it is not a replacement for the ecosystem benefits produced by large and higher-severity fire because prescribed fire does not mimic the patch mosaics or pulses of biological activity that higher-severity fires provide (Moritz and Odion 2004, DellaSala et al., 2014). Thus, understanding one's perspective is a starting point for potentially settling differences and developing ways to coexist safely and beneficially with fire. Being willing to respond competently to the cognitive dissonance created when perspectives do not align with new scientific information is also vital to the development of successful and ecologically sound fire management strategies (e.g., Chapter 7).

s0020 13.2 UNDERSTANDING THE PUBLIC'S REACTION TO FIRE

p0115 If ecologists and conservationists want a new discourse on fire that improves ecological understanding and fire management practices, then informed and sustained communications with the public, land managers, the media, and decision makers are vital. A common understanding is needed to move the public and land management agencies from a view of fire as the harbinger of death (Kauffman, 2004) to fire as nature's phoenix. Here we provide some insights from a public poll on fire attitudes in the United States that reaffirms our personal experiences about the prevailing attitudes of the public and of land managers when it comes to fire.

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s0025 Attitudes Toward Fire

p0120 In 2008 The Wilderness Society and The Nature Conservancy got together to construct a 10-year fire communications framework that was informed by a large national sample of public attitudes ($n = 2000$ respondents), focus groups in six regions of the United States where fire was a concern, and communications experts (Metz and Weigel, 2008). The task was to develop ecological messaging on fires that would "complement Smokey Bear's message" about being careful with fire.

p0125 Based on a summary of the survey findings, important messages on fire can be gleaned from survey data, some of which are remarkably aligned with fire ecology, whereas others are at odds with basic ecological principles. Most notably, the poll demonstrated the public's sophistication regarding the role of fire in ecosystems, but it was clearly tempered by safety concerns (Smokey Bear), notions regarding the importance of "controlled" burns, and a desire to let "some" fires burn in "natural areas." Education (higher levels) was associated with positive attitudes toward fires, and gender was a factor, with men being more risk tolerant and women more risk averse. Some of the poll's most relevant findings are displayed in Box 13.2. We

b0015 BOX 13.2 Key Findings on Public Fire Attitudes from the Study by Metz and Weigel (2008)

- u0010 Some fires can be beneficial, and a history of fire suppression has led to more large and destructive fires. (*Note that dramatic changes in fire behavior actually are associated with very few forest types in western North America* (Odion et al., 2014a)).
- u0015 Strong negative emotional reactions to fire persist based on safety issues (most view fire as "scary").
- u0020 Public understanding of fire's ecological role has increased over time.
- u0025 Public concerns about wildfire rank very low compared with other conservation issues.
- u0030 The most significant fire concerns pertain to effects on people and firefighters rather than ecosystem benefits.
- u0035 Allow fire teams to use "controlled burns" when and where doing so will safely reduce the amount of fuel for fires (*controlled burns are most relevant in low-severity rather than mixed-severity systems*).
- u0040 Cut and remove overgrown brush and trees in natural areas that act as fuel for fires (*this is largely true for low-severity systems, not higher-severity fires that are largely controlled by extreme weather*).
- u0045 Allow naturally started fires that do not threaten homes, people, or the health of natural areas to take their natural course, rather than putting them out.
- u0050 Shift some government funds from putting out practically all fires to proactively cutting and removing overgrown brush and trees and using controlled burns to reduce the amount of fuel for fires (*removing brush/trees and controlled burns are mostly ways to reduce fire severity in low-severity systems*).

also highlight in parentheses those beliefs that seem to be at odds with the ecological literature on mixed-severity fires.

p0130 Communication experts then advised the conservation groups that successful fire messaging should have the following five fundamental communication themes:

o0010 **1.** Protect people, property, and communities
o0015 **2.** Safeguard the health and regeneration of natural areas
o0020 **3.** Safely manage controlled burns to clear fuels (*this management is appropriate in low-severity systems only during the natural fire season*)
o0025 **4.** Save taxpayer money through controlled burns
o0030 **5.** Protect air and water by protecting the health of forests and natural areas and giving plants and wildlife the exposure to fire they need to survive

p0160 From focus groups and polling results, according to communication experts the following cogent messages are likely to reach the public:

u0075 • Safety is always the number one priority when it comes to fire. By putting out every single fire, however, we are actually creating more dangerous conditions (*in western North America, higher-severity fires are operating at an historical deficit*). Using controlled burns to thin out overgrowth and carefully managing natural fires help ensure the safety of neighborhoods in outlying areas.

u0080 • Forests and natural areas are important to our health; they act as natural filters to give us clean air and are the source of clean drinking water. We must ensure the health of forests and natural areas by allowing some fires to take their natural course.

u0085 • Taxpayer money is being wasted putting out fires that are far from people and their property. A far more cost-effective approach is to use controlled burns to prevent large, severe fires from spreading into areas where people live and to allow some fires to take their natural course (*and they are ecologically inappropriate when applied outside the natural fire season*).

p0180 For higher-severity fires, a good portion of this messaging may work to bridge the divide between science and public attitudes, whereas some of the recommendations of the communications experts in 2008 (refer to the italicized text in the parentheses above) do not incorporate the ecological importance of maintaining, and managing for, complex early seral forest created by mixed-severity f[illegible] particular, the poll's findings that fire safety matters most is still very muc[illegible]evant; thus putting out fires that are dangerous to human communities is still of primary importance. From a safety standpoint, Smokey Bear's cautionary fire safety tale needs to be updated so that the focus of fire management is ~~clearly~~ on creating "defensible space" aro[illegible]omes, the home ignition zone (HIZ), and introducing land use zoning to al[illegible]w fire to run its course unimpeded in natural areas under safe conditions (Making Homes Fire Safe, see below). And, while the poll found the public generally agreed that fire is necessary in natural

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areas, how far this tolerance would go in relation to large or higher-severity fires is unclear given that the poll's questions were ~~clearly~~ geared toward low-severity fires that can be either "controlled" or suppressed (through thinning or the use of fire retardants). Notably, in Chapter 12 we discussed how runaway expenditures in fire suppression have been ecologically damaging and fiscally irresponsible, and the public seems to agree with these fiscal concerns. In combination with economics, whether public attitudes will change, or are changing, regarding large or higher-severity fires is still unknown; this will require polling that is more specific to these kinds of fires along with enhanced public education (e.g., the videos referenced in the preface) regarding ways to coexist with large fires.

p0185 A core message—and one that will most certainly be difficult for much of the public to accept despite being fact based—is that large fires in any given location each year, at least in western North America, cannot be stopped no matter what we do. We at least need to be honest about that and clearly state the damages that can ensue from large-scale pre- and postfire management that attempts to control large, mainly climate-driven fires that are uncontrollable. We also need to clearly communicate to the public the current state of scientific knowledge regarding the ecological benefits and values of the habitats created by mixed-severity fire. This is especially so given the still all-too-common notions that such areas have been categorically damaged by fire, which in turn leads to misguided assumptions that ~~such areas~~ are in need of "restoration" or "recovery" management actions.

s0030 13.3 SAFE LIVING IN FIRESHEDS

p0190 Based on public attitudes toward fire there ~~clearly~~ are important challenges to coexistence with fire. These can be overcome, however, if we not only increase public education about current fire ecology but also act responsibly in reducing risks where they matter most. We note that by far the biggest challenge to coexistence with fire is the explosion of exurban sprawl in many rural communities triggered by those moving out of congested cities.

p0195 A case in point is Kalispell, Montana, the gateway to Glacier National Park. A November 17, 2014, article in *Greenwire*, the online source of information on the environment ("Where property rights are king, development continues despite growing wildfire threat"), reported that during the 1990s the county's population grew at twice the state's average as more and more people seeking a rural quality of life purchased 16-ha "ranchettes" scattered across Big Sky fire country. They were able to do so as a result of lax and often resisted land use zoning standards. Based on data provided by Headwaters Economics (2014), 11,000 houses in this Montana county lie within the wildland-urban interface (where towns, homes, and other built structures abut fire-prone wildlands)—more than any other county in Montana—and this number is growing at a phenomenal rate. As reported in the online article, public attitudes included the notion that fire will not directly affect them and strong views about private

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property rights (i.e., "don't tell me what to do on my land"). Some of the same people vocally oppose government actions in general then demand that public money be spent to remove "fuels" from wildlands. In essence, the lack of homeowner fire risk reductions and inappropriate fuel treatments is setting in motion the perfect storm of land use and fire conflicts.

p0200 To minimize these kinds of conflicts, landowners need to practice fire-safe (also known as "fire-wise" in the United States) planning to protect home structures. We suggest that landowners first declare a common "fireshed" boundary, as they do for watersheds. Firesheds are multidimensional spaces. They begin at the scale of a watershed and encompass the residential community with similar fire risks (Figure 13.1a). Within a fireshed, homeowners can take fire risk reduction measures together (preferably) or on their own (Figure 13.1b).

s0035 Making Homes Fire-Safe

p0205 Probably no research results are as relevant to fire safety science than those of Dr. Jack Cohen, whose seminal fire safe research recommendations are now standard risk reduction measures taken by many homeowners[1] and have caught on with risk-averse insurance companies[2]. The work of Syphard et al. (2012, 2014) on home loss in chaparral systems of southern California is strikingly similar.

p0210 According to Dr. Cohen, fire planning within an HIZ begins with defensible space nearest the home. Notably, research on HIZ risks shows that homes whose owners reduced vegetation and flammables within 10-18 m of the structure and built with nonflammable roof materials had an 86% (Foote, 1996) to 95% (Howard et al., 1973) "survival" rate when fires swept through an area (cf. Syphard et al. (2014) for more recent and similar home structure protection distances). Combined with home fire simulations by the insurance industry (http://www.extension.org/pages/63495/vulnerabilities-of-buildings-to-wildfire-exposures#.VHUr00snRNs; accessed February 15, 2015), Box **13.3** provides measures that are most critical for living safely in firesheds.

p0215 An example from a town in Idaho during an intense 2007 fire is instructive regarding the importance of the HIZ and fireshed management. As the *Idaho Statesman* newspaper reported (Druzin and Barker, 2008):

dq0020 *We spend billions attacking almost every wildfire, but scientists say that's bad for the forest, can put firefighters in unnecessary danger and doesn't protect communities as well—or as cheaply—as we now know how to do. A wall of fire barreled through the forest with a jet-engine roar near Secesh Meadows last August, and local fire chief Chris Bent knew his work was about to be tested.*

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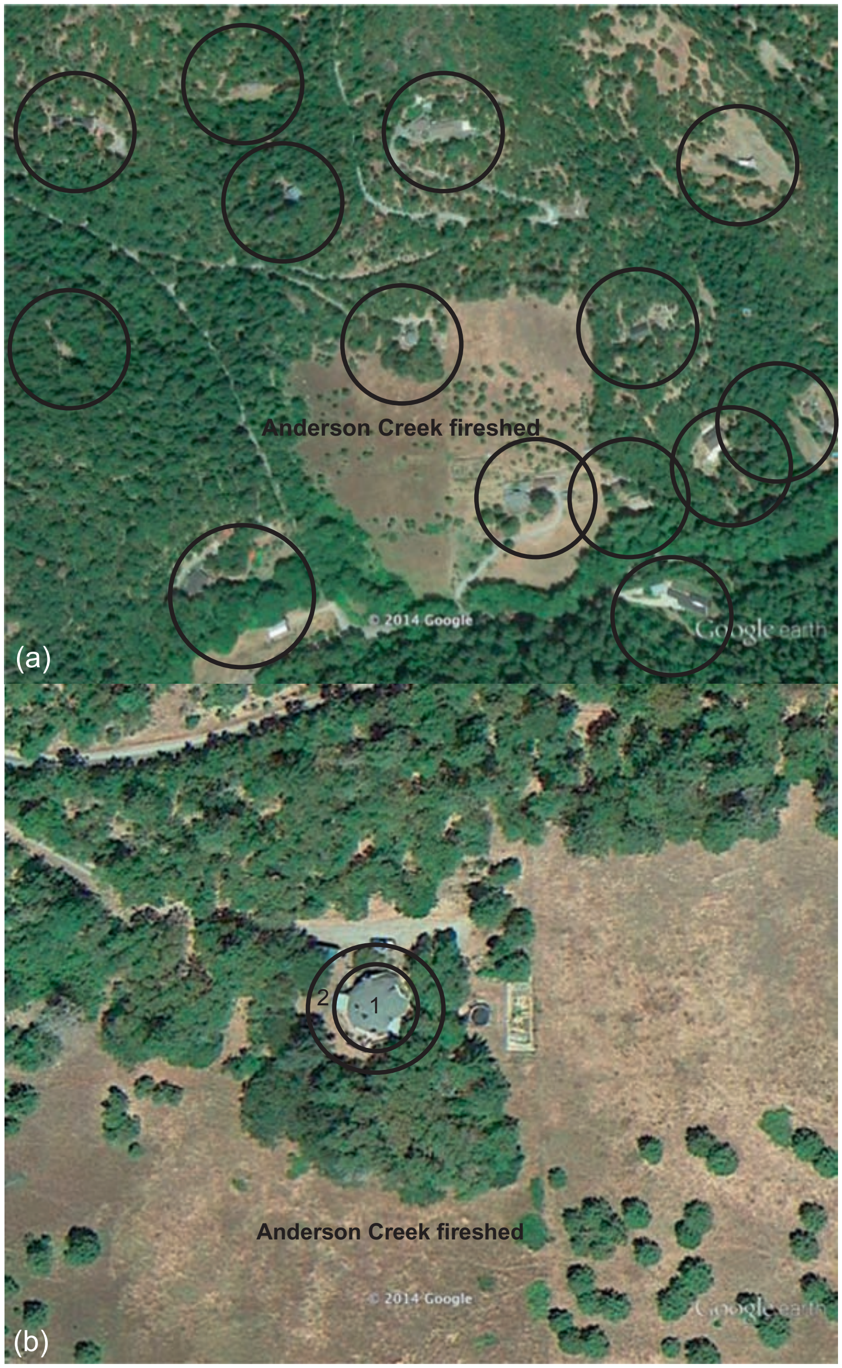
np0010 1. http://www.firewise.org/wildfire-preparedness/firewise-toolkit.aspx?sso=0; accessed November 25, 2014.

np0015 2. http://www.extension.org/pages/63495/vulnerabilities-of-buildings-to-wildfire-exposures#.VHUr00snRNs; active November 26, 2014.



f0010 **FIGURE 13.1** (a) Google Earth image of the Anderson Creek watershed and community fireshed in Talent, Oregon, showing a housing development (circled; the center house is depicted in b). Most members of this community reduced lower-strata fuels via thinning small trees in the surroundings, although tree densities are beginning to fill in and require repeat treatments. (b) Two fire-safety zones where the landowner built with fire-resistant material in the inner most zone (home ignition zone 1) and cleared most vegetation within a 10 m radius around the structure (zone 2). Tree crowns are touching in zone 2; however, lower branches were pruned to 3 m, and there are few ladder fuels to carry fire from the ground into tree crowns. Downslope grasses may pose a fire hazard but may not crown out given the precautions taken in zones 1 and 2.

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b0020 **BOX 13.3 Prudent Fire Risk Reduction Measures for Homeowners**

u0055
- Build homes with noncombustible roof covering and siding; keep roof and gutters clear of leaves/needles; keep firewood away; keep vegetation adjacent to homes to a minimum; cut overhanging limbs of trees closest to the home; and install ember-resistant attic vents.

u0060
- Clearing vegetation within 5-20 m of a home is the most effective treatment: Carefully space plants, reduce wood plant cover to <40% around the structure, and use varieties that grow low and are free of resins, oils, and waxes that burn easily; mow the lawn regularly and prune trees up to 3 m from ground; space conifer crowns ~3 m apart and remove lower limbs; trim back trees overhanging the house; create a "fire-free" area within 1.5 m of the house using noncombustible landscaping; remove dead vegetation; use fire-resistant furniture; remove firewood and propane tanks; and water plants or use xeriscaping.

u0065
- Additional measures include low-growing, well-irrigated, and relatively noncombustible vegetation in low planting densities; a mix of deciduous and conifer trees; fuel breaks like driveways and gravel walkways and lawns.

u0070
- Treatments >30 m from the home structures offer no additional protection (Syphard et al., 2014).

Flames danced atop lodgepole pines, smoke darkened the sky, and residents of the tiny mountain hamlet north of McCall prepared for the worst. Just a month earlier, a forest fire had burned 254 homes near Lake Tahoe and the 2007 fire season appeared ready to claim its next community. But as the raging East Zone Complex fire reached the cluster of loosely-spaced homes, the flames dropped to the ground, crackling and smoldering. The fire crept right up to doorsteps. But without the intense flames that spurred the fire just moments before, no homes burned—a feat fire managers attributed largely to Bent's push to clear flammable brush from around houses in the community. "It just blew through the area," Bent said. "We were well prepared." The town's ability to withstand a frontal assault by a major wildfire demonstrates what fire behavior experts have been saying for more than a decade. Clearing brush and other flammables and requiring fireproof roofs will protect houses even in an intense wildfire—without risking firefighters' lives. More provocatively, the research suggests that fighting fires on public lands to protect homes is ineffective and, in the long run, counter-productive. It is also far more expensive.

p0225 Importantly, clearing vegetation nearest a home is not enough, as fire risk reduction also needs to include the home structure itself (Figure 13.2). This is often missed in discussions about homeowner fire safety, and it is a crucial step in responsible fire risk reduction, as we illustrate in the following examples.

p0230 In a recent research paper concerning why homes burn in wildfires, Syphard et al. (2014) concluded that geography is key: where the house is located and where houses are placed on the landscape. Syphard and her coauthors gathered data on 700,000 addresses in the Santa Monica Mountains and part of San Diego

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f0015 **FIGURE 13.2** Homes burn because they are flammable. Many homes with adequate defensible space still burn in wildland fires because embers land on flammable materials around the home or enter through openings such as attic vents. These two homes burned during the 2014 Poinsettia Fire in Carlsbad, California, despite fire-safe landscaping, a firewall, and thinned wildland vegetation. Focusing exclusively on wildland vegetation clearing ignores the main reasons homes burn: they are flammable. *(Photo credit: Richard W. Halsey.)*

County. They then mapped the structures that had burned in those areas from 2001 to 2010, a time of significant wildfire activity in the region. Buildings on steep slopes, in Santa Ana wind corridors, and in low-density developments intermingled with wildlands were the most likely to have burned. Nearby vegetation was not a major factor in home destruction.

p0235 Looking at vegetation growing within roughly 800 m of structures, Syphard et al. (2014) concluded that the exotic grasses that often sprout in areas cleared of native habitat like chaparral could be more of a fire hazard than shrubs. Interestingly, they found that homes that were surrounded mostly by grass actually ended up burning more than homes with higher fuel volumes such as shrubs.

p0240 Similarly, during the 2007 Witch Creek Fire (San Diego County, CA), houses in Rancho Bernardo started burning by ember contact when the fire front was nearly 6 km away. Two-thirds of the burning homes were set on fire by embers (Maranghides and Mell, 2009).

p0245 During the 2007 Grass Valley Fire near Lake Arrowhead in California's San Bernardino Mountains, approximately 199 homes were destroyed or damaged. This was despite the fact that the US Forest Service had thinned the surrounding forest. The main cause of the losses was that individual homeowners failed to understand that vegetation management is only one part of the fire risk reduction equation. Fire will exploit the weakest link—and it did so in Grass Valley. In the detailed report of the fire, Forest Service researchers (Rogers et al., 2008) concluded: "Post-fire visual examination indicated a lack of substantial fire effects on the vegetation and surface fuels between burned homes. Lack of surface fire evidence in surrounding vegetation provides strong evidence that

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house-to-house ignitions by airborne firebrands were responsible for many of the destroyed homes."

p0250 Investments in making homes and communities fire safe are clearly fiscally prudent and ~~represent~~ responsible homeownership that can save lives and homes by reducing risks to all, especially firefighters. Moreover, proper land use zoning that reduces housing densities in firesheds is key to the survival of home structures over the larger area (Syphard et al., 2014).

p0255 In sum, these recent studies show that overcoming misperceptions about homeowner losses is urgently needed because those misconceptions are a driving factor in many inappropriate fuel reduction projects in wild areas. We hypothesize that with stepped-up planning directed at proper homeowner safety (as demonstrated in the above studies), public attitudes about large and intense fires may begin to shift from fear-based primal responses to more of a neocortex-like awareness of fire as nature's phoenix. This could be tested using before-and-after polling about large, higher-severity fires with and without proper public safety measures in places.

s0040 13.4 TO THIN OR NOT TO THIN?

p0260 One of the most significant challenges involved in changing the way land managers think about fire in the forests is how the US Forest Service views forest fires. The agency is deeply invested in continuing the fire management trajectory of the past—a situation compounded by the budgetary issues associated with the agency's direction of much, and often most, of their tax-based support to selling timber from public lands, and the agency's retention of most of the revenue from such timber sales to fund staff salaries and operations. Though in recent years we have learned much about the ecological benefits of higher-severity fire and the risks to fire-dependent wildlife species from further suppressing these fires, which are deficient in most western US conifer forests (Chapters 1–5), the Forest Service continues to aggressively promote landscape-level mechanical thinning (North, 2012; Stine et al., 2014) and postfire logging (Collins and Roller, 2013) ostensibly to reduce fuels and prevent and mitigate future fire. These forest management policies are promoted based on the assumption that decades of fire suppression have created forests "overloaded with fuel, priming them for unusually severe and extensive wildfires" (Stine et al., 2014; see also North, 2012). The basic concept being articulated by the Forest Service is that, because of decades of fire suppression and "fuel accumulations," we cannot simply allow wildland fires to burn because long-unburned forests will "uncharacteristically" burn almost exclusively at higher severities (North, 2012; Stine et al., 2014). Under this premise, recommendations focus on how to manage forests through logging and fire suppression to further reduce and prevent the significant occurrence of mixed-severity fire (North et al., 2009; North, 2012; Stine et al., 2014). Yet these sources do not include a discussion of the current deficit of these fires in most forests of western North America (Odion et al. 2014a; see also Chapters 1, 2, and 9) or meaningful content on the ecological importance of mixed-severity fire for many rare and imperiled wildlife species (Chapters 2–5). Nor do they explore the validity of the basic premise that long-unburned forests will burn much more severely.

p0265 Studies that empirically investigated the "time-since-fire" issue in the Sierra region of northern California and the Klamath Mountains of Oregon and California tended to find that, contrary to popular assumptions, the most long-unburned forests experience mostly low- and moderate-severity fire and do not have significantly higher levels of higher-severity fire than more recently burned forests (Odion et al., 2004, 2010; Odion and Hanson, 2006, 2008; Miller et al., 2012; van Wagtendonk et al., 2012). One modeling study predicted a modest increase in fire severity with increasing time since fire, but the strength of inference was limited by a lack of data for all but long-unburned stands, especially in the largest forest types, such as mixed-conifer forest. Even the most long-unburned forests were predicted to have ~70-80% low/moderate-severity effects (Steel et al., 2015), well within the range of natural variability (see Chapter 1). In fact, long-unburned forests sometimes have the lowest levels of higher-severity fire; understory vegetation and the lower limbs of conifers self-thin as canopy cover increases and available sunlight in the understory decreases with increasing time since fire (Odion et al., 2010). Therefore the argument that we cannot allow more wildland fires to burn without suppression in natural areas is not valid for many dry montane forests in western North America (Odion et al., 2010).

s0045 Problems with Fuel Models and Fire Liabilities

p0270 Government programs that aim to make forests safe places for people to live are based on theory rather than actual evidence about historical forests. As discussed above, the common argument has been that fuels have unnaturally accumulated from fire exclusion and land uses, and if fuels are restored to low levels, fires will burn primarily at low intensity rather than as high-intensity crown fires (e.g., Agee and Skinner, 2005). Thus forests can be restored while also making them safe places to live—a win-win solution that is appealing to the public. Little evidence about actual historical fuel amounts in forests to support this argument was available, however; instead, evidence is mostly based on the idea that frequent fires would have kept fuels at low levels. When records from land surveys before fire exclusion were examined (Baker, 2012, 2014; Baker and Williams, 2015; Hanson and Odion, in press), understory fuels (shrubs, small trees) that would naturally have promoted intense fires were found to have been common and often abundant in many areas, and small trees were dominant, not rare. This direct evidence suggests that fuel treatments would typically have to artificially remove natural shrubs and small trees and adversely alter habitat for native species in a quest to make forests safer places for people to live.

p0275 Fuel reduction also has been overpromised to be effective, using questionable logic and unvalidated models. First, fire intensity in most forest types is much more strongly affected by wind than by fuel. High fire-line intensity, the primary fire characteristic that promotes crown fires, is the product of the energy released by burning fuel and the rate of spread of fire (Alexander, 1982). Energy release by fuel varies over perhaps a 10-fold range, however, whereas rate of spread can vary over more than a 100-fold range; thus a high rate of spread caused by strong winds can easily overcome the limited reductions in fuel that are feasible (Baker, 2009). This was confirmed by a recent analysis of the 2013 Rim Fire in California, which concludes: "Our results suggest that even in forests with a restored fire regime, wildfires can produce large-scale, high-severity fire effects under the type of weather conditions that often prevail when wildfire escapes initial suppression efforts. . . . During the period when the Rim fire had heightened plume activity. . . [n]o low severity was observed [in thinned areas], regardless of fuel load, forest type, or topographic position" (Lydersen et al., 2014, p. 333). Second, common fire models used to show that forests would be fire-safe after fuel reductions have an underprediction bias and are not validated. These flawed models include NEXUS, FlamMap, FARSITE, FFE-FVS, FMAPlus, and BehavePlus (Cruz and Alexander, 2010; Alexander and Cruz, 2013; Cruz et al., 2014). The underprediction bias means that these models often predict that fuel reductions would reduce or eliminate the potential for crown fires in forests, when in fact fuel reductions do not achieve this effect. Fixing these models would be difficult and has not yet occurred (Alexander and Cruz, 2013). Also, these models have not been sufficiently tested and validated using a suite of actual fires, in which case they would likely be shown to fail (Cruz and Alexander, 2010). Alternative validated models are available and could be further developed, but they are not being used (Cruz and Alexander, 2010). Further, studies of tree mortality in thinned areas following fire do not typically take into account the mortality caused by the logging itself before the fire, leading to further biased results. Au1

p0280 These concerns should raise red flags about the effectiveness of fuel treatments, as well as issues regarding liability and responsibility. Imagine if a company sold airplanes with identified flawed designs and without adequate test flights, which then crashed. There are thus sound scientific reasons to closely scrutinize government wildland fuel-reduction programs. Meanwhile, we need to be honest and warn the public that living within or adjacent to natural forests prone to burn is inherently hazardous. Only treating fuels in the immediate vicinity of the homes themselves can reduce risk to homes, not backcountry fuel reduction projects that divert scarce resources away from true home protection (Cohen, 2000; Gibbons et al., 2012; Calkin et al., 2013; Syphard et al., 2014).

p0285 Finally, another land management liability that is frequently overlooked when assessing fire-related economic losses is the role of silviculture. For instance, before the 2013 Rim Fire, a significant portion of the Stanislaus National Forest in central California's Sierra Nevada Mountains consisted of even-aged monoculture tree plantations (following past clearcuts) distributed across large landscapes (Figure 13.3). Land managers often claim that clearcutting over large landscapes

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f0020 **FIGURE 13.3** Google image of the Stanislaus National Forest, central Sierra Nevada, taken on July 8, 2012, before the August 25, 2013, Rim Fire. The red boundary is where the Rim fire burned. Note numerous clearcuts within the burn area, where the fire later burned intensely. Figure provided by J. Keeley.

like this reduces fire spread, yet based on preliminary findings from the Rim Fire, clearcutting did nothing to stop the fire. In fact, the area with the most clearcutting had the largest contiguous area of high-severity fire of any portion of the Rim Fire (see Figure 13.3 and compare with Figure 11.11). In other areas with large portions of the landscape in tree plantations [illegible] past clearcutting, fires have a tendency to burn uncharacteristically severely, presumably because of homogenized fuel loads (e.g., Odion et al., 2004). Despite these observations, in postfire assessments land managers rarely discuss this effect or the liabilities it creates for economic losses related to intense burns.

s0050 13.5 FIRE SAFETY AND ECOLOGICAL USE OF WILDLAND FIRE RECOMMENDATIONS

p0290 Based on the ecological importance of higher-severity forest fires (e.g., Reinhardt et al., 2008; DellaSala et al., 2014; Hanson, 2014; Moritz et al., 2014) and home safety concerns (e.g., Cohen, 2000; Headwaters Economics, 2014), there are ways for people to live safely in firesheds and still allow fire to perform its vital ecosystem service. Below we provide some summary recommendations that, if widely implemented, would allow fire to take its natural course (i.e., ecological use of wildland fire) while reducing risks to people.

s0055 Fire Safety Recommendations

u0090
- Prepare to live safely with fire so that it can perform its ecologically beneficial functions. (The bulk of fire risk reduction should occur immediately adjacent to homes.)

- u0095 Develop negative financial consequences for landowners who increase fire risk within firesheds by not taking precautionary measures versus providing financial incentives for those who reduce risks (e.g., cost sharing for fire safety). As an example, mortgage and/or insurance rates could be increased for high risks from lack of fire safety and discounted for those who practice fire risk management principles. In this manner, planning for home fire safety would become as routine as taking out a mortgage to buy a home.
- u0100 Include HIZ and fire-safe principles in rural land use planning, including zoning restrictions that limit housing densities in firesheds deemed too risky for development.
- u0105 Require mandatory disclosure of fire risks to homebuyers.
- u0110 Have local and state governments contribute to firefighting costs to create a powerful incentive for improved land use planning, including zoning restrictions, which reduce fire suppression needs.
- u0115 Offer technology transfer to local governments and financial assistance to plan communities that are fire safe.
- u0120 Map high-risk areas where fire-safe standards are most prudent within a local county or other land use unit.
- u0125 Discourage rebuilding in the same high-risk place or require that building occurs with risk management conditions.
- u0130 Redirect funding away from backcountry fire suppression and fuel reduction programs and toward aiding willing homeowners in creating defensible space and reducing the ability of homes to ignite.
- u0135 Initiate strategies to reduce human-caused fire ignitions, especially along roadsides. Many wildland fires start along highways and streets.

s0060 Wildland Fire Recommendations

- u0140 Postfire "salvage" logging is especially damaging to complex early seral Au2 forests. If such forests were ecologically valuable or protected before fire, then they should also be recognized as uniquely valuable and protected after fire.
- u0145 Wildlands cannot be fireproofed by suppression (mechanical thinning or aerial retardants) or clearcutting; fuel treatments (thinning) are more likely to work in low-severity frequent fire systems and much less so in mixed- and higher-severity fire systems that tend to burn under extreme conditions, when suppression is least effective.
- u0150 Large fires, including high-severity patches, are the most efficient means of restoring fire-dependent ecosystems and natural heterogeneity where fire has been excluded for decades. When a fire burns under these conditions, fire-dependent communities are therefore restored. This should be encouraged, with public safety assured.
- u0155 The best way to buffer fire-dependent ecosystems from climate change is to increase ecological resilience, particularly in areas where a fire deficit

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exists, by allowing fires to burn naturally under safe conditions. This will require relatively large protected landscapes with proper land use zoning and logging restrictions.

u0160 • Implement strategies to reduce human-caused fires in ecosystems with excessive fire frequencies, such as the chaparral in southern California.

s0065 13.6 LESSONS FROM AROUND THE GLOBE

s0070 Africa

p0380 Of the five communication themes that arose from the polling in North America, the one most applicable to attitudes in sub-Saharan Africa is number 5, a broad statement to protect natural resources for the ecosystems services they provide (see Chapter 8). The public in South Africa, for example, assumes number 3, safety in controlled burns, because the public is already attuned to the widespread use of fire for habitat management, and when accessible, fuel wood is collected for heat and cooking. Of course, the South African public is not deluged by media reports of catastrophic losses caused by wildfire, so items 1, 2 and 4 are not part of a daily discourse in countries where wildfires in large forests are rare and most of the managed habitat is the much thinner type of woodland associated with savanna (see Chapter 8).

p0385 In terms of such issues as woodland thinning (directed silviculture or ad hoc management), in African savanna the public and policy makers are more concerned with maintaining herbivore populations as part of ecotourism and for the love of Africa's "big five" megafauna wildlife species. South Africa practices extensive silviculture, and it often is blended into wilderness areas (Tsitsikama National Forest lies adjacent to extensive tracts of forest plantation, where fire suppression is practiced because of economics of the wood industry). It seems the "fear" of fire so prevalent in North America is absent from rural areas of Africa for multiple reasons, but this results in a more sane approach to fire ecology. In Kruger Park managers learned over time that allowing wildfire is acceptable, and it is now a tool (although not frequent) integrated with controlled burns. They even seek to achieve as hot a fire as they can in certain habitat conditions to clear the invasive vegetation or just to suppress woody growth. The lesson learned in South Africa over 50 years of "experimenting," and from many decades of following the Serengeti system, is that monitoring is critical, and adapting to those results (adaptive management) is imperative.

s0075 Australia

p0390 In Australia prescribed burning is considered a staple part of the land management tool kit and is routinely applied with the aim of reducing the risk of large, unplanned wildfires to property and infrastructure (Clarke, 2008). In some cases fire is applied to the landscape in efforts to "restore" ecosystems or to create fine-scaled fire mosaics of mixed successional stages to encourage greater faunal and floral diversity (Bradstock et al., 2005). In response to the perceived need to apply fuel-reduction burns, the Victorian state government implemented a policy that mandated that 5% of the total land area under state jurisdiction be burned each year. This policy did not discriminate fire prescriptions between ecosystems and has been subject to widespread criticism from fire ecologists in Australia; it is currently under review (DELWP, 2015a). Although appropriate fire regimes have positive ecological outcomes in many systems, application of prescribed burning can lead to species declines and in some cases can cause irreversible changes in ecosystem state (Pardon et al., 2003, Pennman et al. 2011, Pastro et al., 2011).

p0395 Recent large wildfires in Australia have spurred new policies to address the growing public concern over the dangers presented by these fires (McLennan and Handmer, 2012; Whittaker et al., 2013). The royal commission that followed the 2009 "Black Saturday" fires suggested the implementation of new policies to encourage clearing around homes and to shift public perceptions toward recognition of bushfires as defensible events (i.e., homes can be effectively protected) that require early planning and avoidance actions (Teague et al., 2010). Residents in areas of high fire risk are now able to clear all vegetation within 50 m of their homes. These new measures, coupled with the 5% burn target, aim to reduce the potential of a repeat of the 2009 fires. This home protection approach is partially supported by science. Gibbons et al. (2012) highlighted that houses with vegetation cleared within 50 m were 70% more likely to survive a fire than those with no clearing. They revealed, however, that there was no effect of fuel reduction burning in nearby state forest or ecological reserves on house preservation following the 2009 fires in Victoria, Australia. Furthermore, in some of the most potentially pyrogenic systems, such as mountain ash forests, fuel reduction burns are rarely applied because moisture levels are normally high, and risk of fire spread is considered unacceptable when conditions are dry (DELWP, 2015b). A growing body of literature indicates that inappropriate fire regimes are contributing to species declines globally (Driscoll et al., 2010). In response to the increased fire risk caused by climate change, policy makers should seek to implement strategies with a proven ability to protect homes, while avoiding ineffective actions that detrimentally impact biodiversity.

s0080 Central Europe

p0400 In central Europe forest fires are relatively infrequent and mainly limited to regions with pine forest plantations growing on sands, gravel-sands and sandstone rocks. Any burned areas are mandatorily reclaimed within just 2 years of their formation; exceptions are possible in forests protected as national parks or nature reserves. The option to request avoidance of logging and replanting is used only rarely, however, and nearly all forests affected by fires are quickly logged and replanted.

p0405 Available evidence suggests that fire-induced bare soil patches, charred trunks, and dead wood resulting from the postfire dieback represent unique nesting resources for numerous species. The areas subject to mixed- and high-severity fires are associated with dynamic assemblages of plant and animal species, many of which are rare or even absent in the surrounding landscape. The burned forests serve as key habitats, particularly for aculeate Hymenoptera associated with cavities in dead wood (such as *Dipogon vechti*). Such cavities are considered limiting nesting resources, and their absence (and targeted removal of any newly emerging snags, which is mandatory by law) causes numerous specialized cavity adopters to be red-listed or extinct. Mounting evidence suggests that specific groups of organisms are strictly dependent on the occurrence of repeated fires. As long as sites of natural disturbances become extremely rare in the intensively cultivated landscape of central Europe, bare soil specialists and species that specialize in cavities of decaying wood will be completely absent where forests are subject to intense cultivation and rigorous dead wood removal. Dead wood thus should be considered an important habitat resource deserving conservation measures. Mosaic management of burned forest sites and retaining charred trunks are suggested as management measures supporting biodiversity at the sites of recent forest fires (Bogusch et al., 2015).

s0085 Canadian Boreal

p0410 There is emerging a new paradigm about the role of fire in the Canadian boreal forest. Historically, it was perceived as a simple system where "catastrophic" fire created landscapes of young, even-aged stands and where species diversity was poor. The reality is much more complex. There is an impressive range of fire cycle estimates—some as long as several centuries—suggesting that for at least part of the boreal forest region the abundance of old-growth forests in preindustrial times was much greater than expected (see Chapter 8). Associated with these old-growth forests is high understory diversity in black spruce (*Picea mariana*) stands and a number of rare species of nonvascular plants associated with balsam fir (*Abies balsamea*) stands. Similar findings have been made in boreal forests of Europe and Asia.

p0415 At the other end of the disturbance spectrum, there is now compelling evidence showing the importance of early seral burned habitats for the pyrocommunity, led by saproxylic insects (dependent on dead or decaying wood) and followed by primary cavity nesting birds (see Chapter 8). The retention of a wide range of burn conditions enhances saproxylic insect diversity. A link between this saproxylic community and nutrient cycling has been found, indicating a connection between biodiversity and ecosystem function in Canadian boreal forests. Large fires produce significant pulses of dead wood, which drive biodiversity and ecosystem processes through natural succession over time. Fire skips, or remnants left after large burns, also are critically important for biodiversity, species persistence, and recolonization and ecosystem recovery.

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p0420 For a long time, forest management was driven with a strong focus on timber extraction and developed a jargon that infiltrated the dialect of forestry, with words like "decadent" for old-growth forests, "waste wood" for trees that had been killed by natural disturbances, and "salvage" as the practice used to recover that "wasted" timber. Today, management in the boreal forest is increasingly driven by themes like ecosystem-based management and sustainable development. The new era will require conservation of boreal forests at different ends of the disturbance spectrum from newly created, postfire habitat to multicentury, old-growth forests.

s0090 13.7 ADDRESSING UNCERTAINTIES

p0425 Even though most people recognize the importance of maintaining fire on the landscape, there remain important questions about what might be the optimal postfire conditions for the broad suite of species with varying fire tolerances. For instance, we do not know whether there is a certain amount of burned forest or spatial distribution of burned forest patches, patch sizes, and fire frequencies necessary to maintain species at polar ends of the successional gradient. However, we hypothesize that in large, intact forested landscapes where fire is allowed to burn and logging is restricted (e.g., wilderness areas, large national parks, and other protected ecosystems) there should be ample habitat for all seral species over the long term and the best opportunities for coexistence with fire as a process (see Chapters 3–5). By contrast, in highly degraded landscapes, particularly those close to towns and homes, an optimal condition of recently burned and long-unburned patches is more difficult to ascertain because it may involve tradeoffs for public safety reasons (DellaSala et al. 2004).

p0430 Currently, megafires in western North American forested landscapes burn in mixed-severity patterns and seem to provide the necessary patch mosaics for a broad array of species (Chapters 2–6). Fire-related change of late seral habitat to complex early seral forest (Swanson et al., 2011; DellaSala et al., 2014; Hanson, 2014) has not been a threat to species dependent on such mature forest habitat, particularly given that there is generally much less high-severity fire in mixed-conifer and pine forests of western North America than there was historically (Odion et al., 2014a). Rates of old forest recruitment, as a result of growth, also outpace rates of high-severity fire in old forest by several times (Hanson et al., 2009; Odion and Hanson, 2013; Odion et al., 2014b). The situation is less clear in portions of Australia, however, where fewer vertebrate species have thus far been found to be fire dependent (see Chapters 3 and 4) and there are more species associated with late seral conditions that are especially at risk (Kelly et al., 2015). By contrast, other Australian research found bird species richness to be highest where there is the most successional diversity from higher-severity fire (Sitters et al., 2014) (see Chapter 8). Human-caused fires in North American chaparral, the Great Basin, and many desert ecosystems, which mostly replace stands, have exceeded historical bounds, adversely affecting this diverse shrubland

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community (Chapter 7). Thus, whether or not fire mosaics are correlated with high levels of biodiversity (cf. Martin and Sapsis, 1991 versus Parr and Andersen, 2006; Taylor et al., 2012; Kelly et al., 2015) depends on differences in biogeography, fire histories, land use histories, and life history requirements (including fire tolerances and dependencies) of species over long time lines and large landscapes (e.g., Scott et al., 2014; see Chapters 3–5).

p0435 In addition, climate change introduces uncertainty in how forests will respond to changes in fire extent, longer fire seasons, higher severities in places, how soon the current fire deficit in places will remain that way before exceeding historical bounds, and whether existing deficits will be exacerbated in some forests with increasing precipitation driven by climate change (see Chapter 9). Nonetheless, at least for mixed-severity fire systems there is no magic thinning or suppression bullet to forestall climate-mediated fire changes. Changes in fire behavior are a consequence of human-caused climate change. It is best to treat the cause—climate change—rather than the symptom (fire behavior) if we are truly concerned about climate effects on ecosystems and people.

s0095 13.8 CLOSING REMARKS

p0440 When viewing the natural world, as a matter of perspective, we are reminded of discussions we have often had with foresters regarding how we each see the value of postfire landscapes. Clearly, we see the world differently depending on our professional judgment and value system.

p0445 A professional forester views the fruits of his or her labor, imagining what the future "production" forest will look like after decades of growing wood fiber, and then being frustrated by nature run amuck when the forest goes up in flames.

p0450 For the fire-trained ecologist, the initiating fire is but a glimpse into a vibrant community that begins with a pulse of biological activity and ensures successional events, just one of the many important links to follow in a long chain of ecosystem changes. Even the most charred forest is transformed by fire on one of nature's grandest stages. Among the first actors to arrive on the postfire stage are the biological legacies that provide the supporting foundation for other postfire actors to enter with the passage of time. If we imagine what the stage will look like years after a severe burn (often only 1 year), we see a floral phoenix arising from the ashes, we hear a cacophony of songbirds and drumming woodpeckers, and the rhythmic buzzing of bees and other insects as they go about their business of pollinating the next explosion of flowering plants. Up close and personal, we see tiny native beetle larvae tucked neatly into galleries beneath the outer charred tree bark, wood-boring scorpion wasps recoiling long abdomens after depositing eggs into open crevices in tree bark, centipedes and millipedes working charred humus, and ravenous insect-loving bats and fly-catching birds feasting on all the buzz.

p0455 The postfire landscape is indeed a transformative place if we humans are willing to have the patience to look beyond the brief snapshot in time right after

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the initiating event. Only then will the postfire esthetic become apparent. Our human world of instant gratification pales in comparison to nature's seemingly infinite horizon. Meticulous observations by trained ecologists too often are drowned out by the noise of a fast-paced society preoccupied with one-size-fits-all solutions, do something at any cost, myopic economic benefits, and a fear-based media blitz of fire catastrophe reporting. But if we wait for the ecosystem actors to emerge in synchronicity, the postfire habitat unveiled is remarkably resilient, brilliant like the mythical phoenix, and even musical if we know how to listen. We hope that we have sufficiently portrayed an ecological awareness for this postfire symphony in the chapters of this book.

p0460 In this closing chapter we also have discussed the importance of education and outreach for a communications framework and improved ecological understanding of fire that follows fundamental ecological and safety principles.

p0465 From a communications standpoint, fire operates very much like an apex predator, thinning out and culling its prey, sometimes in large numbers, sometimes not. Apex predators are indeed vital to fully functioning ecosystems, yet they are either loved or hated based on one's perspective, which simply boils down to either an appreciation for wild things or a fear of being attacked or of losing a commodity. People view fire in much the same way. Decades of public outreach and campaigns in many places (most notably Europe and North America) have shifted public opinion to be more accepting of predators, and even to relish them in national parks and other protected landscapes where predators roam free and tourists flock to witness nature primeval. Clearly, fires, like apex predators, cannot be restricted to inside national parks, as the parks are not big enough to sustain them.

p0470 There is a lesson to be learned regarding the message of fear in both instances: As with predators, the risks of losses to people and property can be successfully mitigated by taking precautionary measures (e.g., just don't feed the bears, and remember to make loud noises while hiking in grizzly bear country!). In the case of fire, public safety of those living in firesheds is based on prudent fire risk reduction that with stepped-up outreach one day may become common knowledge. With a shift in this direction, we envision a move toward fire tolerance, and eventually coexistence, so that fire, in all its severities and forms, can continue to shape ecosystems into the next millennium. This will take a concerted effort of sophisticated and sustained message framing, an infusion of funds for stepped-up education that at least rivals predator-friendly campaigns, a commitment from land management agencies and the media to become more ecologically literate (including replacing Smokey Bear with nature's phoenix), conservation groups to see the value in mixed-severity and not just low-severity fire, and politicians to see the big picture that the postfire landscape has irreplaceable ecological value and is not just a money tree to be ravaged for short-term profit. Then nature's phoenix will truly take flight, reborn out of the ashes of a postfire landscape mosaic that is alive and well!

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Non-Print Items

Abstract

Throughout this book we present a compelling case for the ecological importance of mixed-severity wildfires in forests (though some chaparral systems currently experience too much fire), including, in many cases, megafires from western North America. Stand-replacing fire disturbances are underappreciated natural events that have been shaping fire-dependent ecosystems for millennia, and their ecosystem benefits are being compromised by management actions that carry unintended consequences. Mimicking the spatial, temporal, and structural heterogeneity of these fire effects through management is not possible. Moreover, fire management actions such as forest thinning, mastication, and postfire logging are creating novel fire regimes at the expense of historical ones. Dramatic improvements in fire management and public perceptions of wildfire are needed to accommodate wildfires where they are beneficial. We provide several closing recommendations for addressing public safety concerns and ecological use of fire in natural areas.

Keywords: Fire-dependent forests; Fire safety; Forest thinning; Habitat conservation; Mixed-severity fires; Public attitudes.

Forum: Issues, Management, Policy, and Opinions

Accommodating Mixed-Severity Fire to Restore and Maintain Ecosystem Integrity with a Focus on the Sierra Nevada of California, USA

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ABSTRACT

Existing fire policy encourages the maintenance of ecosystem integrity in fire management, yet this is difficult to implement on lands managed for competing economic, human safety, and air quality concerns. We discuss a fire management approach in the mid-elevations of the Sierra Nevada, California, USA, that may exemplify similar challenges in other fire-adapted regions of the western USA. We also discuss how managing for pyro-

RESUMEN

La política de fuego actual fomenta la permanencia de la integridad del ecosistema en el manejo del fuego. Sin embargo esto es difícil de implementar en tierras manejadas con multiplicidad de objetivos (económicos, de seguridad humana, o relacionados con la calidad del aire). Nosotros debatimos un enfoque sobre el manejo del fuego en las elevaciones medias de la Sierra Nevada en California, EEUU, que podría extenderse a casos similares que ocurren en otras regiones adaptadas al fuego en el oeste de los EEUU. También discutimos como el manediversity through mixed-severity fires can promote ecosystem integrity in Sierran mixed conifer and ponderosa pine (*Pinus ponderosa* Laws) forests. To illustrate, we show how coarse-filter (landscape-level) and complementary fine-filter (species-level) approaches can enhance forest management and conservation biology objectives as related to wildfire management. At the coarse-filter level, pyrodiverse mixed-severity fires provide landscape heterogeneity. Species and ecosystem characteristics associated with pyrodiversity can be maintained or enhanced by accommodating moderately severe fires, which hasten restoration by recreating a complex vegetation mosaic otherwise at risk from suppression. At the fine-filter level, managers can select focal species and species of conservation concern based on the degree to which those species depend on fire and accommodate their specific conservation needs. The black-backed woodpecker (*Picoides arcticus* [Swainson, 1832]) is an ideal focal species for monitoring the ecological integrity of forests restored through mixed-severity fire, and the California spotted owl (*Strix occidentalis occidentalis* [Xantus de Vesey, 1860]) is a species of conservation concern that uses post-fire habitat mosaics and is particularly vulnerable to logging. We suggest a comprehensive approach that integrates wildland fire for ecosystem integrity and species viability with strategic deployment of fire suppression and ecologically based restoration of pyrodiverse landscapes. Our approach would accomplish fire management goals while simultaneously maintaining biodiversity.

jo para lograr la pirodiversidad a través de fuegos de severidad mixta podrían promover la integridad del ecosistema boscoso de coníferas mixtas de estas Sierras y de bosques de pino ponderosa (*Pinus ponderosa* Laws). Para ilustrarlo, mostramos cómo los enfoques a gran escala (a nivel de paisaje) y complementariamente a pequeña escala (a nivel de especie), pueden favorecer los objetivos del manejo forestal y de la conservación biológica en relación al manejo del fuego. A nivel de gran escala, la pirodiversidad de los fuegos de severidad mixta resultó en la heterogeneidad del paisaje. Las características de las especies y del ecosistema asociadas a la pirodiversidad pueden ser mantenidas o favorecidas cuando se admite la ocurrencia de algunos fuegos moderadamente severos, los cuales aceleran la restauración recreando un mosaico complejo de la vegetación, lo que no ocurriría en caso de ser suprimidos. A nivel de pequeña escala, los gestores pueden seleccionar especies focales y especies relacionadas con la conservación, basados en el grado sobre el cual esas especies dependen del fuego y se adaptan a sus necesidades de conservación específicas. El pájaro carpintero negro (*Picoides arcticus* [Swainson, 1832]) es una especie focal ideal para monitorear la integridad ecológica de los bosques restaurados a través de fuegos de severidad mixta, y la lechuza moteada de California (*Strix occidentalis occidentalis* [Xantus de Vesey, 1860]) es una especie de interés para la conservación que utiliza mosaicos de hábitat post fuego y es particularmente vulnerable al aprovechamiento forestal. Nosotros sugerimos un enfoque comprensivo que integre los fuegos naturales para la integridad del ecosistema y la viabilidad de las especies, con la implementación estratégica de la supresión del fuego y la restauración de paisajes pirodiversos basada en principios ecológicos. Nuestro enfoque podría cumplir con los objetivos de manejo del fuego, manteniendo simultáneamente la biodiversidad.

Keywords: coarse filter, ecosystem integrity, fine filter, focal species, mixed-severity fire, pyrodiversity, Sierra Nevada, species of conservation concern

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INTRODUCTION

Pyrodiversity, the mean spatial variability in wildfire effects, results in complex post-fire vegetation mosaics that are associated with high levels of biodiversity. Large fires that produce a variety of severities (i.e., mixed-severity fires) in ponderosa pine (*Pinus ponderosa* Laws) and mixed-conifer forests of the western USA are increasingly recognized for their importance in generating pyrodiverse landscapes (e.g., Perry *et al.* 2011, Williams and Baker 2012, Odion *et al.* 2014, Marcoux *et al.* 2015). Top-down processes such as extreme fire weather, regional climate (which influences fuel moisture and ignitions), and bottom-up processes such as topographic relief, vegetation, and disturbance history govern the distribution and size of fire patches in mixed-severity fires (Perry *et al.* 2011, Dunn and Bailey 2016). Regional drought, high winds and temperatures, and other factors (e.g., surface fuel loading, crown base height, and crown bulk density; Cruz and Alexander 2010) drive crown fire behavior in these systems, producing small and large patches of high tree mortality within a predominantly surface-fire matrix of mostly surviving trees. Mixed-severity fires therefore generate complex stand structures and landscape heterogeneity—characteristics not typically produced by low-severity fire (Table 1). Low-severity fire, while also important ecologically, is preferred by many managers due to lower risks to economic values. Here, we focus on mixed-severity fires because they have received less attention by managers, but they result in pyrodiverse landscapes (DellaSala and Hanson

Table 1. Pyrodiversity attributes produced by mixed-severity fires associated with high levels of biodiversity and ecosystem functions.

Mixed-severity fire attribute	Ecological importance
Landscape heterogeneity	Habitat for wide array of species—early to late seral associates Mixture of foraging and nesting habitat for spotted owls
Complex stand structures	Biological legacies: large snags, down wood, shrubs, flowering plants Habitat for black-backed woodpeckers
Food web dynamics	Complex trophic structure connected across seral stages with abundant food for certain taxa (e.g., beetle larvae for woodpeckers) Pulsed nutrient inputs (aquatic and terrestrial)
Ecosystem processes	Nutrient cycling and soil nutrient exchange, energy transfer from live to dead material, pollination, predator-prey (owls-mice)
Species composition	Rich and varied, compared to old growth

2015). We demonstrate how ecosystem integrity can be met by managing for pyrodiverse landscapes mediated by mixed-severity fires in the biodiverse region of the Sierra Nevada, California, USA.

Although it is the subject of ongoing research and debate (Odion *et al.* 2016), it has been suggested that mixed conifer, ponderosa pine, and Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.) forests in this region historically experienced a mix of fire severities, including areas of high overstory tree mortality (DellaSala *et al*. 2014, Stevens *et al.* 2016). There is considerable variability in reported proportions and sizes of high-severity fire patches, with the greatest differences found in relatively smaller study areas or studies in which shorter time periods were analyzed (Table 2). High-severity patches commonly ranged from 0.4 ha to >50 ha, but the historical frequency of patches >1000 ha is still debated (e.g., Baker 2014, Stevens *et al.* 2016). While uncertainty remains on some issues, there is general agreement that most forests of the Sierra Nevada currently have less high-severity fire, in terms of annual or decadal area burned, than they did historically, prior to fire suppression (Mallek *et al.* 2013, Odion *et al.* 2014, Baker 2015). Additionally, drier low-elevation pine forests burned most frequently at low to moderate severity (Stephens *et al.* 2015), but those fires also contained variably sized high-severity patches (Leiberg 1902, Baker 2014, Hanson and Odion 2016*a, b*). Douglas-fir (*Pseudotsuga menziesii* Mirbel) (Odion *et al.* 2014) and Sierra lodgepole pine (*Pinus contorta* var. *murrayana* Grev. & Balf.) forests experienced mixed-severity fires as well (Caprio 2008).

Tree mortality is also an important component of mixed-severity fire effects characterized mostly by low-mortality levels (0% to 20% tree basal area), highly variable moderate-mortality levels (20% to 70%), and high-mortality levels (>70% tree mortality) (Perry *et al.* 2011; Figure 1). Agee (2005) noted that mixed-severity fires are not merely an intermediate state between low and high severity but, rather, are a unique type of disturbance that warrants careful study by ecologists.

While there are winners and losers in the immediate aftermath of any disturbance event, the net effect of mixed-severity fire is that it

Table 2. Historical fire severity proportions and maximum high-severity fire patch sizes in mixed-conifer and ponderosa pine forests, Sierra Nevada management region.

Study	Study area size (ha)	Fire severity (%)			Time period (yr)	Maximum high-severity patch size (ha)
		Low	Moderate	High		
Beaty and Taylor (2001)[1]	1587	1 to 60	14 to 47	6 to 86	43	no data
Bekker and Taylor (2001)	2042	2 to 4	35 to 44	52 to 63	75	no data
Baker (2014)	330000	13 to 26	42 to 48	31 to 39	110	9400
Hanson and Odion (2016*a,b*)	65296	no data	no data	22	60	697
Leiberg (1902)[2]	1193166	no data	no data	20	100	~16000
Stephens *et al.* (2015)	11500	no data	no data	1 to 6	~20 to 30	no data

[1] Fire severity percentages vary by slope position and aspect.
[2] Does not include high-severity fire patches <32.4 ha, so actual percent high-severity fire would be higher, if patches <32.4 ha had been mapped. Historical high-severity fire mapped polygons are from Leiberg (1902), and analysis of high-severity fire percent by forest type is from Hanson (2007), based on Leiberg (1902).

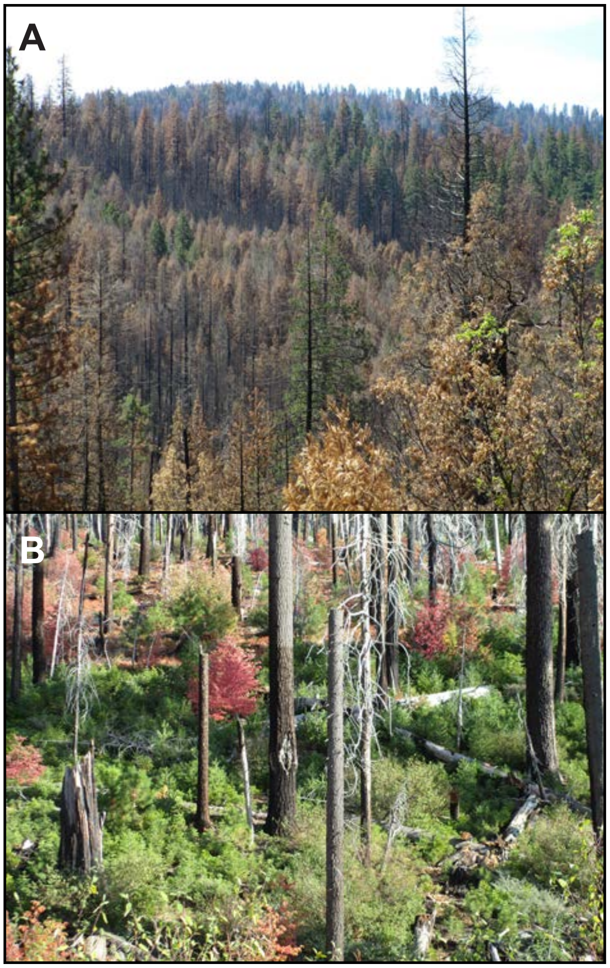

Figure 1. (A) Landscape view of mixed-severity fire effects in the Rim Fire 1 year post fire. The spatial pattern of fire severity patches and patch sizes results in a pyrodiverse landscape that provides habitat for wildlife across a post-fire vegetation gradient of low or unburned vegetation patches to severely burned vegetation patches. (B) Close-up of large patch of complex early seral forest created by high-severity fire in juxtaposition with abundant and varied "biological legacy" trees (complex structures, such as snags, logs, and shrubs that survive fire). Photos by C. Hanson.

provides a mosaic of habitat for a broad suite of species. For instance, songbirds have high levels of species richness and abundance in post-fire vegetation at mid elevations (Fontaine *et al.* 2009, Tingley *et al.* 2016). Black-backed woodpeckers (*Picoides arcticus* [Swainson, 1832]), mountain bluebirds (*Sialia currucoides* [Bechstein, 1798]), tree swallows (*Tachycineta bicolor* [Vieillot, 1808]), and numerous shrub-nesting birds preferentially use recently burned forests in the Sierra Nevada and other regions, presumably due to increased shrub cover and presence of snags (Fontaine *et al.* 2009, DellaSala *et al.* 2014, Hutto *et al.* 2015, Tingley *et al.* 2016). California spotted owls (*Strix occidentalis occidentalis* [Xantus de Vesey, 1860]) and olive-sided flycatchers (*Contopus cooperi* [Nuttall, 1831]) forage in severely burned patches where prey are abundant, and nest in unburned to moderately burned portions of the same fire mosaic (Bond *et al.* 2009, 2016; Hutto *et al.* 2015; Comfort *et al.* 2016). Bats make use of high snag densities (Buchalski *et al.* 2013) and fire-recruiting plants are associated with severely burned patches (Donato *et al.* 2009). Even mature-forest carnivores such as the Pacific fisher (*Pekania pennanti* [Erxleben, 1777]) actively forage in severely burned patches (Hanson 2015).

The high-severity patches within the mixed-severity mosaic provide a unique pulse of biological legacies—complex structures such as snags, downed logs, and native shrub patches from seed that survive fire and that are important in connecting seral stages through time (Franklin *et al.* 2000, Fontaine *et al.* 2009, Donato *et al.* 2012, DellaSala *et al.* 2014). The economic value of large dead and live trees within these patches means that commercial trees are most often targeted for harvest soon after fire. In addition, nursery-grown young trees are planted soon after fire and, to promote the crop of young trees, herbicides are often sprayed to kill competing vegetation (Lindenmayer *et al.* 2008, 2017). Logging slash from post-fire logging may contribute to subsequent fire behavior (Donato *et al.* 2006, Thompson *et al.* 2007), as can the fuel array of densely planted even-aged trees (Odion *et al.* 2004).

On public lands, current fire policy promotes thinning over large landscapes (e.g., USDA Forest Service 2002, US Congress 2003, USDA Forest Service 2009, US Congress 2015), which is costly (Schoennagel and Nelson 2011), infeasible over large areas (Calkin *et al.* 2013, North *et al.* 2015*a*, Parks *et al.* 2015), and largely ineffective under extreme fire weather conditions (Lydersen *et al.* 2014, Cary *et al.* 2016). For instance, from 2001 to 2008, over 11 million hectares were thinned on national forests (mostly in the western USA) at a cost of more than \$6 billion (Schoennagel and Nelson 2011). Mechanical vegetation treatments can cost over \$3700 per hectare for each round of thinning (Kline 2004), which would need to be repeated at least every 15 to 20 years to keep flammable vegetation at low levels. Additionally, from 1985 to 2015, suppression costs were more than \$25 billion to fight approximately 2 million fires on over 83 million hectares, mostly spent by the Forest Service (Ingalsbee and Raja 2015).

Thus, we concur with others that active management approaches could include more natural fire ignitions (Calkin 2013, Meyer 2015, North *et al.* 2015*b*) or resource objective wildfires (Meyer 2015) in which fire is put back on the landscape to hasten the process of forest restoration (Moritz *et al.* 2014, Moritz and Knowles 2016). This would also help to meet fire and fuels objectives and allow managers to better accommodate mixed-severity fire effects for ecosystem integrity (Meyer 2015, Dunn and Bailey 2016). We suggest that an ecosystem integrity approach is not inconsistent with current active fuel management on federal lands and may be a cost-effective way to achieve biodiversity goals (North *et al.* 2015*b*), while reducing some of the conflicts associated with extensive fuels-focused approaches—particularly impacts to imperiled species and at-risk ecosystems. We use the definition of ecosystem integrity common in the literature (e.g., Pimentel *et al.* 2000), also adopted by the USDA Forest Service (2012), as the ability of an ecological system to support and maintain a community of organisms that has a species composition, diversity, and functional organization comparable to those of natural habitats within a region.

Our focus is the Sierra Nevada region because of national attention given to so many recent fires therein. We include an example of a fire-adapted species (black-backed woodpecker) that uses high-severity patches, and an imperiled species (California spotted owl) known to decline within intensively managed post-fire landscapes. The Sierra Nevada is one of the most diverse temperate conifer forest regions on Earth and has exceptional levels of plant endemism (Ricketts *et al.* 1999). Approximately half of California's 7000 vascular plant species occur in this region, with 400 considered endemic and 200 rare. High levels of vertebrate richness and endemism also occur. Species composition varies across north-south, east-west, and elevational gradients, resulting in high levels of beta diversity.

Importantly, the 2012 forest planning rule (USDA Forest Service 2012) includes specific provisions for managing public resources to maintain or restore: (1) structure, function, composition, and landscape connectivity; (2) ecological conditions for recovery of imperiled and focal species; and (3) rare and unique habitat types (USDA Forest Service 2012). The National Cohesive Wildland Fire Management Strategy (USDI and USDA 2014) and Sierra national parks (e.g., Yosemite, Sequoia and Kings Canyon) also include multi-faceted approaches that promote greater wildfire ignitions. Though national forest lands compose most of the forested area in California, and are thus our focus herein, significant areas of federal forest in California are managed by the National Park Service (NPS), and a state agency, California Department of Forestry and Fire Protection (CAL FIRE), responsible for decisions and operations pertaining to fire suppression on private and state lands. NPS, like the Forest Service, is required to protect species listed under the Endangered Species Act (ESA), and CAL FIRE is subject to the California state ESA. Thus, our approach to wildfire management can be applied to these agencies and land ownerships regarding decisions about fire suppression and forest management that might impact imperiled or ESA-listed species associated with post-fire landscapes.

STUDY AREA

The Sierra Nevada management region is a 750 km long, north-south oriented mountain range in California composed of granitic rock, and distributed across three ecoregions: Sierra Nevada proper; portions of the Modoc Plateau; and the eastern portion of the southern Cascades (Bailey 1995; Figure 2). The regional climate is mediterranean with cool, wet winters, and warm, dry summers; precipitation generally decreases west to east and north to south (Millar1996).

There are 11 national forests totaling about 4.6 million hectares: Modoc, Lassen, Plumas, Tahoe, Eldorado, Stanislaus, Sequoia, Sierra, Inyo, Humboldt-Toiyabe (western portion), and Tahoe Lake Basin Management Unit. Forest planning is governed by the Sierra Nevada Framework (USDA Forest Service 2004), but the Forest Service is currently revising its forest plans for the Inyo, Sequoia, and Sierra national forests as "early adopters" (i.e., first national forests to test the planning rule) of the 2012 forest-planning rule (USDA Forest Service 2012). Three national parks—Lassen, Sequoia and Kings Canyon, and Yosemite—and several large wilderness and inventoried roadless areas >2000 ha also occur in the region.

Coarse-Filter and Fine-Filter Approaches to Ecosystem Integrity in Mixed-Severity Systems

Managers wishing to maintain ecosystem integrity via naturally ignited fires can do so using a combination of coarse- and fine-filter conservation approaches (Noon *et al.* 2003, USDA Forest Service 2012). Coarse filters invariably include relatively few indicators associated with the larger ecosystem of interest (e.g., major vegetation types or, in this case, different categories of burn severity). Their presence is meant to indicate that essential components of the whole system are intact, and they operate at broad spatial scales such as those associated with large fires (hundreds of square kilometers). Coarse filters are typically used to guide reserve design based on fundamental principles of conservation biology, including spatially redundant reserve complexes representative of the major forest types and fire severities interconnected across large landscapes. To achieve a pyrodiverse landscape, perhaps the best coarse filter would include high-severity fire patches interspersed with fire refugia (unburned areas) and low- to moderate-severity patches.

Fine-filter considerations complement coarse filters by adding site-specific or habitat elements associated with focal species, guilds, or other species groupings (USDA Forest Service 2012). Application of this kind of filter allows managers to evaluate whether habitat and special conservation needs are met through a given management plan, and ground-truth the utility of burn severity maps by linking mapped fire severities to habitat needs of target species. In addition, the approach allows managers to meet national forest planning requirements to monitor and evaluate a small suite of focal species selected to assess the degree to which ecological conditions are supporting the diversity of plant and animal communities within a given planning area (USDA Forest Service 2012). Focal species can, therefore, be used to monitor the integrity of the larger system to which they belong, and researchers (e.g., Seavy and Alexander 2014, Stephens *et al.* 2015, Siegel *et al.* 2016) have suggested using patterns of plant and animal distributions as a passive management strategy to accommodate mixed-severity systems. The Forest Service also now considers species of conservation concern as "a species, other than federally recognized threat-

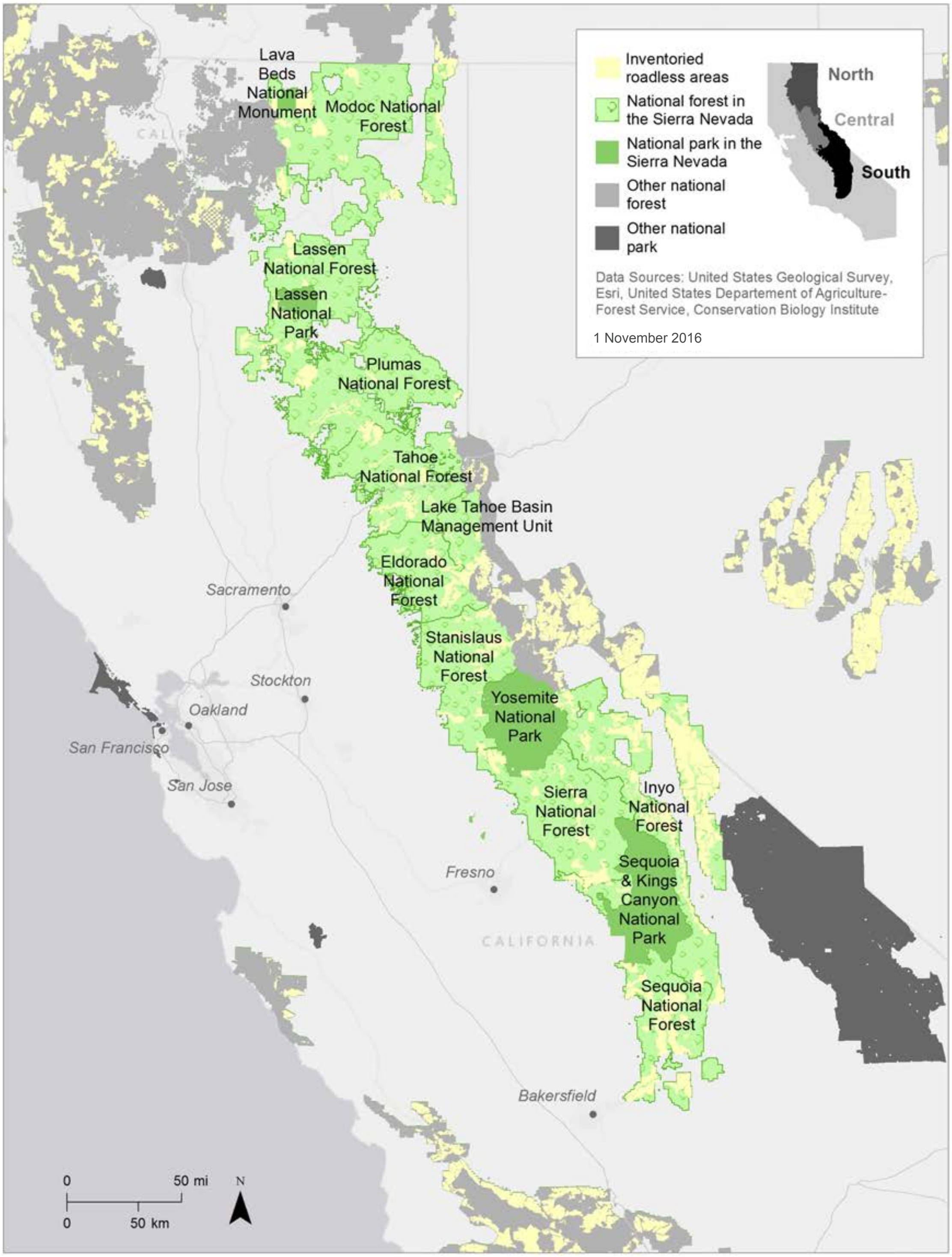

Figure 2. Sierra Nevada study region showing national forests, national parks, and inventoried roadless areas.

ened, endangered, proposed, or candidate species, that is known to occur in the plan area and for which the regional forester has determined that the best available scientific information indicates substantial concern about the species' capability to persist over the long-term in the plan area" (36 CFR 219.9(c); https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5359595.pdf, accessed 12 May 2017). The agency is required to maintain suitable habitat for these species to ensure viable populations are present in the planning area (USDA Forest Service 2012).

Comprehensive Wildland Fire Management

We recognize that land managers face many constraints (legal and social) and often competing regulatory and management objectives that limit wildfire management options. However, the Planning Rule and the National Cohesive Wildland Fire Management Strategy (USDI and USDA 2014) offer opportunities to put more fire back on the landscape whether through prescribed burning or managed wildfires. We provide some general concepts that managers might apply with pyrodiversity outcomes realized through mixed-severity fires that meet ecosystem integrity objectives.

Integrating Wildland Fire and Targeted Fire Suppression (Coarse Filter)

Mixed-severity fire effects for ecosystem benefits can be integrated with targeted suppression and fire-risk reduction efforts near towns using this coarse-filter approach. While we acknowledge that there was concern about the size and severity of the 2013 Rim Fire (Lydersen *et al.* 2014), the largest fire in recent Sierra Nevada history, we note that even this fire produced mostly low- to moderate-severity effects (i.e., ~20% of the burn was high severity based on Monitoring Trends in Burn Severity [MTBS]; http://mtbs.gov/MTBS_Uploads/data/2013/maps/ca3785712008620130817_map.pdf, accessed 23 April 2017), and a wide range of high-severity patch sizes, which contributed to significant heterogeneity at landscape scales. Thus, we concur with others (e.g., Moritz *et al.* 2014, Ingalsbee and Raja 2015, Dunn and Bailey 2016, Moritz and Knowles 2016, Schoennagel *et al.* 2017) that suppression could be focused narrowly to lands surrounding towns and used in combination with defensible space management nearest homes (Cohen 2000, 2004) so that more wildland fires can burn safely in the backcountry.

Notably, one way to safely modify fire suppression activity would be to restrict large fire crews and heavy equipment to protect homes and communities within the Wildland Urban Interface (WUI). The WUI is usually considered to extend to ~2 km from an at-risk community (US Congress 2003, USDA Forest Service 2004), even though most vegetation treatments are conducted farther from communities (Schoennagel *et al.* 2009). Beyond the WUI, point protection strategies would be used to keep fire away from isolated structures and infrastructures like cabins, communication towers, bridges, or other human assets that could be destroyed by fire. Relatively small, mobile fire crews would also use minimum impact suppression tactics (i.e., Minimum Impact Suppression Tactics [MIST]; https://www.nifc.gov/PUBLICATIONS/redbook/2003/AppendixU.pdf, accessed 12 May 2107) in backcountry areas, primarily monitoring fire spread but, when necessary, actively managing it (rather than containing and controlling wildfire as in traditional full-suppression strategies) by steering fire away from threatened social assets (Donovan and Brown 2005, 2008; Ingalsbee and Raja 2015). In municipal watersheds where fire management plans may want to avoid high-severity fires burning near water sources, more fires could be allowed to burn during moderate weather conditions. Wildfire management should be a useful tool for managing fuel loads in municipal watersheds where the use of chemicals or heavy equipment for either thinning or suppression would cause unacceptable impacts to water quality and soils. MIST could also be employed where fires in wilderness and roadless complexes, national parks, and even in roaded areas many kilometers from the nearest town pose low risk to residential areas. In sum, this approach would shift wildfire operations from limiting fire spread, size, or duration in backcountry areas to working with fire for ecosystem benefits while still effectively providing for community wildfire protection.

Sierra Nevada national forests and parks are large enough to accommodate most large fires over thousands or even tens of thousands of hectares (Appendix 1). For instance, many (>50%) of the largest forest fires from 1984 to 2014 were primarily contained within an individual national forest or national park boundary. In general, federal lands offer unique opportunities in which the maintenance of pyrodiversity for biodiversity could be emphasized in large protected areas (wilderness and roadless area complexes; Appendix 2). Coordination among agencies with similar objectives may allow for more naturally ignited fires over mixed ownerships having similar objectives (e.g., wilderness or roadless areas, other remote forests, conservation areas juxtaposed with parks) using an all-lands approach. If reserves were too small to accommodate large fires or patches of different fire severities, then complexes of multiple reserves widely distributed across a region in redundant locations would collectively help maintain the full complement of post-fire stages using the coarse-filter approach.

In the Sierra Nevada, the draft revised forest plans for the three early-adopter national forests in the southern portion of the range have included a fire-management-zoning approach similar to what we suggest here, allowing more naturally ignited fire in remote areas and suppressing fires close to communities (USDA Forest Service 2016). However, the focus in the draft plans remains on mechanical thinning and post-fire logging (USDA Forest Service 2016). We submit that an approach that allows more natural fire ignitions is advisable and warranted from the standpoint of both ecosystem integrity and public safety, as discussed herein.

Focal Species and Species of Conservation Concern (Fine Filter)

By way of example, we consider two species that could be used to monitor mixed-severity effects. The black-backed woodpecker would be an ideal focal species given its very close association with high-severity fire patches, as would the California spotted owl, a species of conservation concern. Both species are complementary to mixed-severity fire management, given that the woodpecker is mainly associated with the high-severity component, and spotted owls use a broad gradient of fire severity patches. Moreover, while there is some overlap in geographic ranges, spotted owls generally occupy low- to mid-montane forests, while the black-backed woodpecker lives in mid- to high-elevation mixed-conifer forests up to subalpine forests.

Black-backed woodpecker as focal species of high-severity fire patches. In the Sierra Nevada, black-backed woodpeckers occur across mid- to upper-montane and subalpine conifer forests from ~1200 m to 2800 m, depending on latitude. While still uncommon even in burned areas, the greatest concentrations occur in severely burned, mixed-conifer and upper montane forests with high basal area of snags (Hanson and North 2008, Saracco *et al.* 2011) where wood-boring beetle larvae are abundant (Saab *et al.* 2007). Burned areas also typically harbor high densities of medium to large dead trees >30 cm dbh (Cahall and Hayes 2009, Saab *et al.* 2009, Tingley *et al.* 2014). Black-backed woodpeckers also occur (albeit much more rarely) in dense, mature unburned forests (Bonnot *et al.* 2009, Fogg *et al.* 2014) where they have relatively larger home ranges, presumably reflecting conditions that are less than optimal (Tingley *et al.* 2014). Nevertheless, unburned forests with high levels of dead trees from drought and native bark beetles might at least slow the rate of population decline during interludes between severe fires (Rota *et al.* 2014). Only a small fraction of fires burn suitable woodpecker habitat, due to the narrow convergence of conditions that include recent (generally ≤8 years post-fire) higher-severity fire effects in dense, mature, middle- to high-elevation conifer forest (Casas *et al.* 2016). Often a single pair of birds uses hundreds of hectares (Dudley and Saab 2007, Tingley *et al.* 2014).

Black-backed woodpeckers are vulnerable to even partial post-fire logging (Hutto and Gallo 2006, Koivula and Schmiegelow 2007, Saab *et al.* 2009, Rost *et al.* 2013). Radio-telemetry studies in the Lassen and Plumas national forests of California showed that home-range sizes were significantly larger in forests in which some post-fire logging occurred, and post-fire logged patches in the Sierra Nevada were avoided (Tingley *et al.* 2014). For example, even though post-fire logging was proposed for what seems like a minor portion of the King Fire, logging was especially concentrated within the highest quality woodpecker habitat (Figure 3), where a high density of medium to large snags occurred. Notably, on national forests of the Sierra Nevada, post-fire logging decisions have typically authorized removal of 40% to 60% of high-severity patches, displacing complex early seral forest with tree plantations (e.g., USDA Forest Service 2014, 2015, 2016). Retention of dead trees in logging units generally averages ~10 trees per hectare >38 cm dbh (USDA Forest Service 2004). By comparison, to maintain habitat for this focal species, generally hundreds of medium to large snags per hectare (>30 cm dbh to 40 cm dbh, and especially snags >50 cm dbh) are needed (Hanson and North 2008, Saab *et al.* 2009, Tingley *et al.* 2014) in patches consistent with home-range size, along with an ample supply of dense, mature or old conifer forest to facilitate conditions for high quality habitat when fires do occur (DellaSala *et al.* 2014).

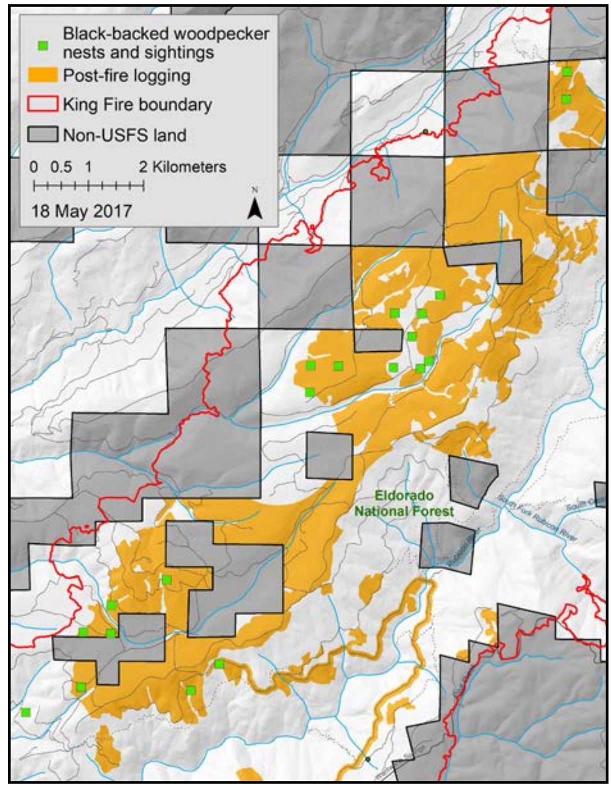

Figure 3. King Fire logging units on the Eldorado National Forest and black-backed woodpecker nests and sightings. After extensive surveys for black-backed woodpeckers were conducted for the US Forest Service throughout the fire area one year post fire, using playback recordings to detect the birds, all but one of the detections was in a relatively small area of dense, mature mid-montane conifer forest in a very large high-severity fire patch in the northern portion of the fire area (shown above). The Forest Service's decision authorized post-fire logging of ~80% of these locations.

California spotted owl as species of conservation concern. Early studies on habitat associations and reproductive success of spotted owls in the Sierra Nevada were conducted in long-unburned forests, and "non-suitable" owl habitat was typically the result of logging (e.g., Moen and Gutiérrez 1997, Blakesley *et al.* 2005). Because spotted owls are usually associated with older, dense forests, it was assumed that effects of high-severity wildfires were similar to logging (Weatherspoon *et al.* 1992). However, recent studies have demonstrated that occupancy (Roberts *et al.* 2011, Lee *et al.* 2012, Lee and Bond 2015*a*) and reproductive success (Roberts 2008, Lee and Bond 2015*b*) were similar or higher in forests burned with a mixture of fire severities compared to long-unburned forests for up to at least 15 years post fire (longer-term studies have not been conducted). Lee and Bond (2015*a*) reported higher occupancy rates than any Sierra Nevada study area for historical owl breeding sites one year after the Rim Fire. The amount of high-severity fire within an owl pair's 120 ha protected activity center, as defined by the Forest Service, had no effect on occupancy, although occupancy by single owls declined slightly as the extent of severe-fire patches increased.

Thus, even though spotted owls are not considered a fire-dependent species, they do persist after mixed-severity fires when both unburned and severely burned patches occur within historical territories (Lee *et al.* 2012; Lee and Bond 2015*a, b*). Owls foraged preferentially in high-severity patches within mature forest in the southern Sierra Nevada (Bond *et al.* 2009) and used high- and moderate-severity patches in the San Bernardino Mountains in proportion to availability (Bond *et al.* 2016). Notably, structural complexity (including high density of dead trees) is important for spotted owl foraging habitat. Bond *et al.* (2009) found that dead tree basal area and shrub cover were highest in high-severity fire patches in which owls preferentially foraged. The owls found a rich food source, in the form of small mammal prey, in post-fire habitat (Bond *et al.* 2016). California spotted owls also selected high-severity patches for foraging more than any other fire severity condition or than long-unburned forests when within 1.5 km of the nest or roost (Figures 4 and 5). Although there are reports of California spotted owls nesting in moderate-severity patches, these raptors mostly nest and roost in long-unburned or lower-severity areas within a burned landscape (Bond *et al.* 2009), underscoring the importance of the mixed-severity mosaic. In contrast, Jones *et al.* (2016) found higher rates of territory extirpation and lower rates of colonization of owl sites that experienced >50 % high-severity fire in the King Fire on the Eldorado National Forest, and reported avoidance of high-severity patches for foraging. The circumstances of their study differed greatly from others (Lee and Bond 2015*a, b*), presumably due to pre- and post-fire logging within owl territories, as well as extensive high-severity fire in pre-fire clearcuts with young plantations.

Long-term occupancy monitoring without the confounding influence of post-fire logging is especially important to understanding fire effects on spotted owls. Hence, Bond *et al.* (2009) recommend that, if managers want to maintain spotted owl habitat after fire, they should prohibit post-fire logging and pesticide and herbicide applications within at least 1.5 km of historical spotted owl nest and roost sites. Even larger areas may be needed given that owl breeding-season home ranges can extend upwards of 700 ha (Bond *et al.* 2016), and some birds expand their range or migrate during the non-breeding season (Bond *et al.* 2010). Therefore, a reasonable protected area might be within 2.4 km of nest and roost sites, which corresponds to interim spotted owl management guidelines of the Forest Service's Pacific Southwest Research Station (http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd504726.pdf).

Restoration of Degraded Forests

Land-use stressors that degrade or impair ecosystem processes are fundamentally at odds with ecosystem integrity approaches (Pimentel *et al.* 2000, USDA Forest Service 2012). Thus, restoration treatments can be used to reverse the causative agents of ecosystem degradation. One example is to limit human-set fires via: (1) seasonal closure and decommissioning of roads, or convert roads not considered essential in firefighting within the

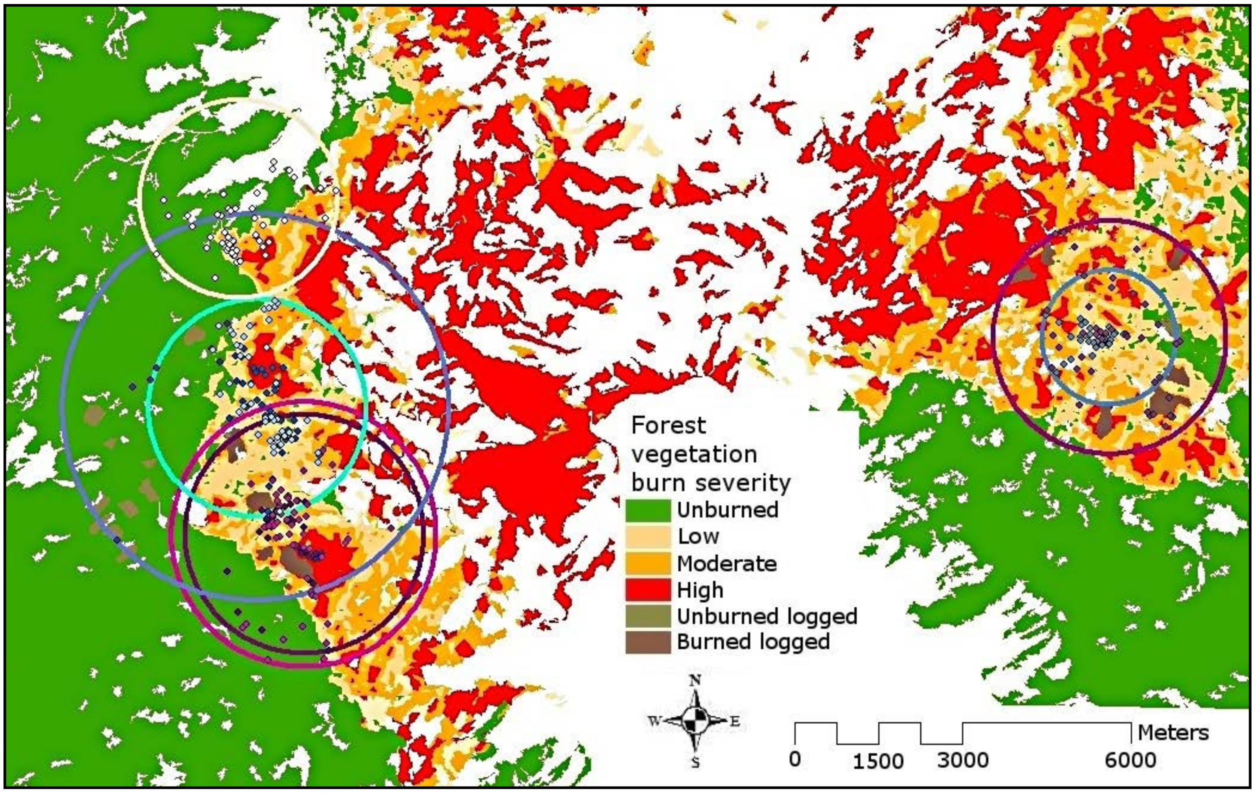

Figure 4. (A) Estimated foraging locations (obtained in 2006) of seven radio-marked California spotted owls in the 2002 McNally Fire, Sequoia National Forest, Sierra Nevada, USA. Different colored points represent each individual owl's estimated foraging location. Circles represent foraging ranges: each circle is centered on the nest with its radius extending to the farthest estimated foraging location for each individual owl. White areas are non-suitable for owls (e.g., foothill chaparral vegetation).

WUI to indefinitely closed; and (2) focused thinning and prescribed burning nearest homes, around campgrounds and other facilities, and along narrowly defined road prisms close to towns to avoid fire spread from anthropogenic ignitions. Managers could also concentrate thinning of small trees (shaded fuel breaks) along with prescribed burning nearest critical evacuation routes for communities with only one means of ingress or egress, redesign traveler stopping points along roads to avoid fire-prone settings, and concentrate visitation in fire-safe locations. Importantly, because tree plantations create unnaturally homogenized forests that lack complex structures, managers could integrate thinning with mixed-intensity prescribed burning, or naturally ignited fires, and create snags and downed logs to introduce structural complexity. Thinning small trees combined with prescribed fire (Kalies and Kent 2016) may reduce fire intensity in densely stocked tree plantations (Odion *et al.* 2004).

CONCLUSIONS

The 2012 Planning Rule provides the Forest Service with new direction for restoring and maintaining integrity and for managing focal species and species of conservation concern that can be integrated with fuels management approaches. The National Cohesive Wildland Fire Management Strategy (USDI and USDA 2014) allows managing wildfire for ecosystem benefits; hence, our findings can be applied to Department of Interior lands as well.

Figure 5. (A) General location of a California spotted owl nest territory in the 2002 McNally Fire (circle not to scale). Nest site was in a low-severity patch directly adjacent to high-severity patch (severity defined using Miller and Thode 2007). (B) Zoom-in (center snag) of general location of California spotted owl nest tree within McNally Fire burn patch shown in (A). Photos by M. Bond.

We suggest that managing for ecosystem integrity using both a coarse- and fine-filter approach centered on pyrodiverse fire effects can inform forest management in a biodiversity context. Our approach would have the added benefit of likely reducing suppression costs and some of the negative effects of mechanical vegetation removal over large areas (Dale 2006, Donovan and Brown 2008, Dunn and Bailey 2016). The complementary nature of conservation filters would allow managers to check burn severity maps with habitat associations of focal species to assess management efficacy.

Managers face substantial political and public pressure to suppress fires through the use of aggressive firefighting tactics, but such tactics do little to contain fires under extreme weather conditions (Lydersen *et al.* 2014, Moritz *et al.* 2014, Ingalsbee and Raja 2015, Carey *et al.* 2016). Instead, managers could be encouraged to use prescribed and naturally ignited fires that yield both cost savings and ecosystem benefits. Unfortunately, federal fire suppression budgets are dominated by suppression costs, causing siphoning of funds away from other essential programs (Ingalsbee and Raja 2015). To support managers in using more natural fire ignitions, conditions and certain trigger points could be more clearly defined and integrated with forest planning. This would allow flexibility to use several approaches to managing a fire, even on the same incident. Thus, in theory, a large fire could be managed in one area with general containment strategies that employ MIST (backcountry), while simultaneously in another area (near towns) with direct attack methods.

Accommodating mixed-severity fires for ecosystem benefits pertains to both ends of the fire continuum: large fires with high-severity effects that generate unique biological pulses (e.g., complex structures), and lower-severity systems that may have been homogenized through management and suppression. This suggests an important opportunity for expanding fire management beyond traditional kinds of prescribed burning to include prescriptions that benefit a broader suite of species associated with pyrodiverse landscapes (Moritz *et al.*

2014, DellaSala and Hanson 2015, Moritz and Knowles 2016). We note the conundrum of natural fire ignitions creating greater smoke emissions that may conflict with air quality objectives. Importantly, the Environmental Protection Agency (2016) recently revised policies to provide special regulatory exemptions and provisions that allow for more managed wildfires.

With proper planning and use of modern smoke management techniques, adverse effects of emissions on public health can be mitigated and fire restoration goals better accommodated. However, smoke emissions must be viewed as an unavoidable trade-off to be weighed against other potentially worse effects from attempted fire exclusion (that will eventually burn in a wildfire) or other chemical and mechanical methods for managing fuel loads that have ecosystem consequences.

There is clearly a need for research on whether natural fire ignitions can primarily provide desired mixed-severity fire effects. We suggest that studies are needed to determine the following.

(1) Specific locations and forest types best suited for mixed-severity fire effects, particularly in relation to ecological mechanisms by which pyrodiversity influences biodiversity.
(2) Current versus historical sizes and proportions of fire-severity patches and how those might be affected by climate change.
(3) Additional species that may be affected by suppression such as declining shrub-nesting birds associated with complex early-seral forests (Hanson 2014).
(4) Importance of other disturbance events (e.g., native insect outbreaks, drought) in maintaining ecosystem integrity.
(5) Effects of mechanical treatments before and after fire on the integrity and quality of mixed-severity patches including species of conservation concern and focal species.
(6) Kinds of education efforts required to implement this type of integrated disturbance ecology approach.
(7) Decision-support tools to help managers assess the costs and benefits of natural fire ignitions, along with conditions under which fires should be suppressed for human safety.

We argue that expanding natural fire ignitions for ecosystem benefits in combination with strategic use of defensible space, directed suppression, and active fuels management in appropriate areas provide untapped potential to enhance ecosystem integrity while protecting people and infrastructure with the potential for lower financial costs. Our approach is based on an ecological understanding of the importance of mixed-severity fires (DellaSala and Hanson 2015), and the need to reconsider "catastrophe" biases regarding natural disturbance processes (Lindenmayer *et al.* 2017).

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Appendix 1. Fires affecting national forests and parks within the Sierra Nevada region, California, USA, from 1984 to 2014 based on the Monitoring Trends in Burn Severity project (http://www.mtbs.gov, accessed 8 September 2015). SD = standard deviation.

Management unit	Unit area hectares	Cumulative burned area		Mean fire size hectares (SD)	Largest fire area (ha) (% of fire occurring within management unit)
		Area (ha) (number of fires)	(%)		
Sequoia and Kings Canyon National Park	350 030	31 795 (34)	9.1	1 559 (1488)	3 806 (88.5)
Lassen Volcanic National Park	43 432	12 811 (9)	29.5	1 940 (3379)	6 383 (58.5)
Yosemite National Park	301 885	102 864 (49)	34.1	4 268 (15 174)	31 841 (30.6)
Eldorado National Forest	321 290.	63 458 (9)	19.7	7 882 (12 496)	40 005 (99.6)
Inyo National Forest	834 535	47 767 (26)	5.7	4 536 (11 391)	7 995 (13.5)
Lake Tahoe Basin National Forest	80 595	1 138 (2)	1.4	1 423 (285.5)	1 083 (88.7)
Lassen National Forest	602 442	145 393 (46)	24.1	7 607 (10 801)	18 632 (75.0)
Modoc National Forest	818 852	85 022 (37)	10.4	3 221 (6 348)	15 507 (41.8)
Plumas National Forest	579 996	141 396 (37)	24.4	5 111 (8 112)	26 371 (99.0)
Sequoia National Forest	470 505	163 731 (61)	34.8	3 801 (8 563)	51 284 (86.5)
Sierra National Forest	574 583	48 785 (30)	8.5	3 261 (4373)	9 538 (100)
Stanislaus National Forest	441 366	171 391 (35)	38.8	7 647 (17 908)	71 614 (68.8)
Tahoe National Forest	476 706	54 294 (19)	11.4	5 786 (9640)	8 394 (100)
Toiyabe National Forest	731 467	63 715 (33)	8.7	2 797 (3 692)	10 163 (100)

Appendix 2. Wilderness and adjacent inventoried roadless areas (IRA) in the Sierra Nevada region, California, USA, compared to largest fire sizes.

Wilderness/IRA complex	Complex size (ha)	Largest fire within associated forest unit[1] (1984 to 2014)
Eldorado	75 255	40 005
Inyo	601 756	7 995
Lassen	99 821	6 383
Modoc	109 725	15 507
Plumas	35 987	26 371
Sequoia	266 316	51 284
Sierra	293 314	9 538
Stanislaus	143 319	71 614
Tahoe	69 519	8 394
Toiyabe	348 597	10 163
Lake Tahoe Basin	28 345	1 083

[1] Fire sizes are for national forest units with wilderness/IRA complexes. Many fires extend beyond national forest and wilderness/IRA boundaries (see Appendix 1).

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Restoring Composition and Structure in Southwestern Frequent-Fire Forests:

A science-based framework for improving ecosystem resiliency

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Abstract

Ponderosa pine and dry mixed-conifer forests in the Southwest United States are experiencing, or have become increasingly susceptible to, large-scale severe wildfire, insect, and disease episodes resulting in altered plant and animal demographics, reduced productivity and biodiversity, and impaired ecosystem processes and functions. We present a management framework based on a synthesis of science on forest ecology and management, reference conditions, and lessons learned during implementations of our restoration framework. Our framework focuses on the restoration of key elements similar to the historical composition and structure of vegetation in these forests: (1) species composition; (2) groups of trees; (3) scattered individual trees; (4) grass-forb-shrub interspaces; (5) snags, logs, and woody debris; and (6) variation in the arrangements of these elements in space and time. Our framework informs management strategies that can improve the resiliency of frequent-fire forests and facilitate the resumption of characteristic ecosystem processes and functions by restoring the composition, structure, and spatial patterns of vegetation. We believe restoration of key compositional and structural elements on a per-site basis will restore resiliency of frequent-fire forests in the Southwest, and thereby position them to better resist, and adapt to, future disturbances and climates.

Keywords: dry-mixed conifer, ecosystem services, ecosystem processes and functions, frequent-fire forests, forest structure, ponderosa pine, restoration, species composition, spatial patterns

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Executive Summary

Many forest landscapes in the Southwestern United States (Arizona, New Mexico, southwest Colorado, and southern Utah) have become increasingly susceptible to large-scale, severe wildfire, insect, and disease episodes. As a result, these areas are experiencing altered plant and animal demographics, reduced structural and spatial heterogeneity of vegetation, reduced productivity and biodiversity, and impaired ecosystem processes, functions, and services. Increased susceptibilities are most evident in frequent-fire forests—forests that historically experienced frequent, low-severity fire, which in the Southwest include ponderosa pine and dry mixed-conifer forests. Changes to these frequent-fire forests largely resulted from unregulated livestock grazing around the turn of the 20th Century, logging, and human activities such as fire suppression, resource use, and infrastructure development.

We present a management framework for improving the resistance and resiliency of frequent-fire forest ecosystems to severe disturbances. This is accomplished by restoring the characteristic vegetation composition and structure in these forests. Frequent-fire forests had a characteristic uneven-aged structure consisting of a temporally shifting mosaic of different aged tree groups and scattered individual trees in an open grass-forb-shrub matrix—a spatial and temporal pattern that provided and sustained plant and animal habitat adjacency, local biodiversity, and food webs. Hence, the key compositional and structural elements of our restoration framework are: (1) species composition (tree and understory vegetation); (2) groups of trees; (3) scattered individual trees; (4) open grass-forb-shrub interspaces between tree groups and individual trees; (5) snags, logs, and woody debris; and (6) variation in arrangements of these elements in space and time. Our framework is informed by:

- reference conditions (conditions of ecosystems before significant industrial human disturbance),
- natural ranges of variability (ranges of reference conditions for a specific ecosystem and time period),
- observed changes in disturbance regimes, and
- lessons learned during applications of our framework in frequent-fire forests in the Southwest.

The types, frequencies, and severities of disturbances (e.g., fires, insects, and diseases) played an important role in shaping the historical composition, structure, and function of frequent-fire forests. Therefore, where forest composition and its structure allow, the framework recommends that fire, the primary historical disturbance agent in these forests, play a prominent role in their restoration. The framework also emphasizes that mechanical treatments may be necessary to initiate suitable compositions and structures before reintroducing fire. Where use of fire is limited, mechanical treatments may be the only available tool to create and maintain restored forests. Conversely, fire may be the only tool in some areas. Restoration provides opportunities for the re-establishment of the characteristic disturbance regimes as well as the spatial and temporal links between pattern and process (e.g., the feedback relationship between forest structure and fire) that sustained the characteristic composition and structure of these forests. Implementation of our framework should improve overall ecosystem productivity and function and enhance ecosystem services such as soil productivity, biodiversity, wildlife habitat, clean air, water quality and quantity, wood products, and recreation.

Natural ranges of variability are considered a “best” estimate of a resilient and functioning ecosystem because they reflect the evolutionary and historical ecology of forests. Natural ranges of variability are thereby a powerful template for improving the resiliency of frequent-fire forests. Natural variability in the composition and structure across sites in these forests results from and drives spatial differences in fire effects, plant species compositions, tree establishment patterns and densities, and numbers and distribution of snags, logs, and woody debris. Managers are encouraged to recognize the natural variability in ponderosa pine and dry mixed-conifer forests and to use historical evidence, such as old trees, stumps, and logs, and biophysical site attributes (e.g., soils, slopes, aspects, and climate) to guide the restoration of variability in these forests. Studies of reference conditions in Southwestern ponderosa pine and dry mixed-conifer showed that trees occurred in a range of spatial patterns, most often aggregated but with a random distribution on certain soils. Tree groups were separated by open grass-forb-shrub interspaces of variable sizes and shapes that often contained scattered individual trees. In areas exhibiting strong tree aggregation, openness was typically higher, but on sites with less tree aggregation, openness may have been lower depending on the arrangement of trees, their sizes, and crown widths (Table 1). The distribution and abundance of snags and logs varied with site and likely coincided with the type, severity, and scale of historical disturbance (Table 1). While reference condition literature on the fine-scale (<10 acres) composition and structure in dry mixed-conifer is more limited than for ponderosa pine, studies showed many similarities—the consequence of their characteristic frequent, low-severity fire regimes. Nonetheless, ranges of reference conditions at small spatial scales showed that mean tree densities and basal areas were slightly greater in dry mixed-conifer forests than ponderosa pine, and snag and log abundances appeared similar to or slightly greater in dry mixed-conifer than in ponderosa pine forests. Compared to today's forests, characteristic dry mixed-conifer forests had higher proportions of fire-resistant/shade-intolerant tree species; lower tree densities; a more open structure comprised of higher proportions of large, old trees; and more spatial heterogeneity (groups and patches of trees).

To illustrate implementation of our framework, we describe a restoration treatment in a ponderosa pine stand in New Mexico that had experienced incidental tree cutting and no fire since the 1880s. While the stand had a characteristic component of old trees, there was a preponderance of mid-aged trees. Fire behavior modeling of pre-treatment conditions showed that 11 percent of the stand could support torching and active crown fire under dry conditions and moderate wind speeds. Our restoration treatment moved the composition and structure of the stand towards characteristic conditions—distinct tree groups, scattered single trees, and open interspaces between tree groups. Implementation of the framework resulted in predicted crown fire behavior on only 1 percent of the stand. Post-treatment abundance of logs and snags was lower than desired, but these elements will accumulate over time.

Our framework incorporates knowledge of the historical compositions, structures, functions, and processes in Southwest frequent-fire forests and how these operated through feedback mechanisms to sustain their characteristic compositions and structures. Current forest conditions are reviewed in light of historical conditions and how human-caused changes to these forests lowered their resistance and resilience to disturbance agents, which have become more intense and frequent. Our framework offers management recommendations for achieving the key compositional and structural elements for restoring frequent-fire forests. Once restored, these forests comprise a temporally shifting mosaic of groups of trees with interlocking crowns; scattered single trees; open grass-forb-shrub interspaces between tree groups; and dispersed snags, logs, and woody debris. It may not always be feasible or even desirable to restore exact reference compositions and structures. Instead, our framework's objective is to increase forest resiliency by managing forest composition and structure toward reference conditions. We believe restoration of key compositional and structural elements on a per-site basis will enhance the resiliency of frequent-fire forests in the Southwest, thereby positioning them to better adapt to future disturbances and climates. It is our intent that application of this framework be flexible and adaptive (i.e., learn-as-you-go), that it will evolve with accumulation of knowledge, and that its conceptual approach will provide a blueprint against which management plans and practices can be evaluated.

Table 1. Ranges of reference conditions for ponderosa pine and dry mixed-conifer forests in the Southwestern United States from studies detailed in Tables 3, 6, 7, and 9.

	Reference conditions by forest type	
Forest attribute	**Ponderosa pine**	**Dry mixed-conifer**
Trees / acre	11.7-124	20.9-99.4
Basal area (ft^2 / acre)	22.1-89.3	39.6-124
Openness (%)[a]	52-90	78.5-87.1
Openness on sites with strong tree aggregation (%)[a]	70-90	79-87
Spatial patterns	Grouped or random	Grouped or random
Number of trees / group	2-72	Insufficient data
Size of groups (acres)	0.003-0.72	Insufficient data
Number of groups / acre	6-7	Insufficient data
Snags / acre	1-10	≥ Ponderosa pine forests
Logs / acre	2-20	≥ Ponderosa pine forests

[a]Openness is the proportion of area not covered by tree crowns, estimated as the inverse of canopy cover. Openness data for dry mixed-conifer is limited; range of reference condition openness will likely change with additional studies.

CONTENTS

Restoring Composition and Structure in Southwestern Frequent-Fire Forests: A Science-Based Framework for Improving Ecosystem Resiliency

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Table 2. Characteristic fire regimes of Southwestern forest types. Fire frequency refers to the mean number of years between fires, and fire severity relates to the effect of the fire on dominant overstory vegetation. Infrequent-fire forests (wet mixed-conifer and spruce-fir) are included for comparison to frequent-fire forests.

Forest type (subtype)	Fire regime[a]		Fire type[b]	Forest structure	Seral species[c]	Climax species
	Fire frequency	Fire severity				
Ponderosa pine (all subtypes)	Regime I					
	0-35 years	Low	Surface	Uneven-aged, grouped, open	**Dominant:** ponderosa pine	**Dominant:** ponderosa pine Shade-intolerant species.
Dry mixed-conifer	Regime I (common)					
	0-35 years	Low	Surface	Uneven-aged, grouped, open	**Dominant:** ponderosa pine **Subdominant:** aspen, oak, Douglas-fir, Southwestern white pine, and limber pine	**Dominant:** ponderosa pine **Subdominant:** Douglas-fir and Southwestern white pine or limber pine Shade-intolerant species.
	Regime III (rare)					
	35-100+ years	Mixed	Mixed	Uneven-aged, patched, open		
Wet mixed-conifer	Regime III (common)					
	35-100+ years	Mixed	Mixed	Uneven-aged, patched, closed	**Dominant** (depending on plant association): aspen or Douglas-fir	**Dominant** (depending on plant association): white fir and/or blue spruce Shade-tolerant species.
	Regime IV (rare)					
	35-100+ years	High	Stand-replacing	Even-aged, closed		
Spruce-fir (mixed, lower subalpine)	Regime III and/or IV					
	35-100+ years	Mixed / High	Mixed/ stand-replacing	Even-aged, closed	**Dominant** (depending on plant association): aspen or Douglas-fir	**Dominant** (depending on plant association): Engelmann spruce and/or white fir Shade-tolerant species.
Spruce-fir (upper subalpine)	Regime V					
	200+ years	High	Stand-replacing	Even-aged, closed	**Dominant** (depending on plant association): aspen, Douglas-fir, or Engelmann spruce	**Dominant:** Engelmann spruce and corkbark fir or subalpine fir Shade-tolerant species.

[a]Schmidt and others (2002)
[b]Smith (2006a, 2006b, 2006c)
[c]USDA Forest Service (1997)

Introduction

There is increasing recognition that frequent-fire forests, defined as forests with fire return intervals <35 years (Table 2), have become progressively more susceptible to large-scale, severe wildfire (Covington and Moore 1994b; Steele and others 1986; Westerling and others 2006). These forests, which in the Southwestern United States include ponderosa pine and dry mixed-conifer forests (see Appendix 1 for scientific names of species referred to herein), are also increasingly prone to insect and disease epidemics and altered plant and animal habitats, all leading to reduced biodiversity, ecological function, resilience, and sustainability (Allen and others 2002; Benayas and others 2009; Carey and others 1992; Carey and others 1999; Colgan and others 1999; Covington and Moore 1994a; Kalies and others 2012; Lynch and others 2010). Reduced ecosystem resilience to disturbances is more evident in frequent-fire forests where the composition, structure (age, size, density, and spatial patterns of vegetation), processes (e.g., disturbances), and functions (e.g., food webs) have changed to a greater degree due to reductions in fire frequency than in forest types where fire was historically less frequent (Agee 2003; Covington and Moore 1994a; Crist and others 2009; Hessburg and others 1999). This reduction in fire frequency is, in part, a result of more than a century of intensive human activities, including fire suppression, livestock grazing, and logging. Important compositional and structural changes in these forests resulting from human activities, especially those that changed historical fire regimes, include:

- increased tree densities,
- reduced structural and spatial heterogeneity of vegetation,
- declines in grass-forb-shrub vegetation,
- loss of old trees, and
- reductions in the diversity and quality of plant and animal habitats and food webs (Abella 2009; Arnold 1950; Covington and others 1997; Kalies and others 2012; Larson and Churchill 2012).

In addition to increasingly frequent and uncharacteristic disturbances such as large-scale severe fire events (Allen 2007; Covington and Moore 1994b; Fitzgerald 2005; Graham and others 2004; Swetnam and others 1999) and insect epidemics (Ferry and others 1995; Hessburg and others 2005; Kolb and others 1998; Negrón 1997), these changes resulted in environments that differed from those in which the native fauna and flora evolved (Carey 2003; Carey and others 1992, 1999; Colgan and others 1999; Covington and Moore 1994b; Kalies and others 2012; Reynolds and others 1992, 2006a). Furthermore, ecosystem services such as clean air and water, water yield, wood products, recreation, aesthetic and spiritual experiences, old-growth, nutrient cycling, pollination, and carbon sequestration have been altered and are now more vulnerable to rapid degradation by uncharacteristic fire and insect epidemics (Benayas and others 2009; Ferry and others 1995; Finkral and Evans 2008; Hessburg and others 2005; Kolb and others 1998; Negrón 1997; reviewed in Evans and others 2011 and Hunter and others 2007).

Prior to human-influenced changes to the characteristic fire regime, the composition, structure, and spatial pattern in frequent-fire forests were maintained by frequent, low-severity fire through a functional relationship between pattern and process; that is, frequent low-severity fires resulted in forest structures that facilitated continued low-severity fire (Fitzgerald 2005; Graham and others 2004; Hiers and others 2009; Mitchell and others 2009; Thaxton and Platt 2006). Over time, shifting mosaics of tree groups and individual trees of varying ages were maintained within a grass-forb-shrub matrix by relationships among the severity and frequency of fire, presence of surface fuels (fuels on or near the surface of the ground), and tree regeneration sites that escaped fire (Larson and Churchill 2012). Some dry mixed-conifer forests and ponderosa pine-shrub communities experienced mixed-severity fires, which included combinations of surface and crown fires (see Table 2), sometimes resulting in larger patches of tree aggregation (Agee 1993; Arno and others 1995; Kaufmann and others 2007; Larson and Churchill 2012).

Forest restoration guided by reference conditions (conditions that characterized the status of ecosystems before significant industrial human disturbance; *sensu* Kaufmann and others 1994) provides for the approximation of the historical (i.e., natural) effects of characteristic disturbances. Restoration is the process of assisting the recovery of degraded, damaged, or destroyed ecosystems (SER 2004). Restoration initiates or accelerates ecosystem recovery with respect to ecological health (productivity), integrity (species composition, community and ecosystem structure), and sustainability (resistance and resilience to disturbance)

(SER 2004). Ecosystem resiliency is the ability of an ecosystem to absorb and recover from disturbances without altering its inherent function (SER 2004). A functioning ecosystem provides opportunities for sustaining plant and animal habitats and populations, increased biodiversity, nutrient cycling, carbon sequestration, air quality, water quality and quantity, wood products, forage, recreation, and aesthetic and spiritual experiences (Aronson and others 2007; Benayas and others 2009). Restoring forest composition and structure improves ecosystem function and resiliency (Bradshaw 1984; Cortina and others 2006).

A holistic approach to forest restoration based on appropriate science can also help meet multiple management objectives, including fuels reduction; reintroduction of characteristic disturbances; and the return of wildlife habitats, native biodiversity, and food webs (Covington and Moore 1994b; Kalies and others 2012; Reynolds and others 1992, 2006a). Management informed by reference conditions and natural ranges of variability (the range of ecological and evolutionary conditions appropriate for an area; *sensu* Landres and others 1999) allow for the restoration of the characteristic composition, structure, spatial pattern, processes, and functions of ecosystems. Managing forests guided by historical conditions also restores the evolutionary environment (Kalies and others 2012; Moore and others 1999), enhancing the capacity of organisms in ecosystems to adapt to stressors such as fire, insects, disease, and climatic variability and change.

We describe a framework, including assumptions, principles, values, concepts, and procedures, for restoring the composition, structure, and spatial pattern of ponderosa pine and dry mixed-conifer forests in the Southwest. Our framework is a science-based approach to restoration that will prove useful for developing strategic plans and management applications. The framework emphasizes vegetation composition and structure, describes expected outcomes, and presents management recommendations for implementation. Expected outcomes include: increased biodiversity, plant and animal habitats, and ecosystem services; increased resilience to insects, disease, and climate change; and reduced fuel loads and fire hazards. Key compositional and structural elements of our restoration framework are:

(1) species composition (tree and understory vegetation);

(2) groups of trees;

(3) scattered individual trees;

(4) open grass-forb-shrub interspaces;

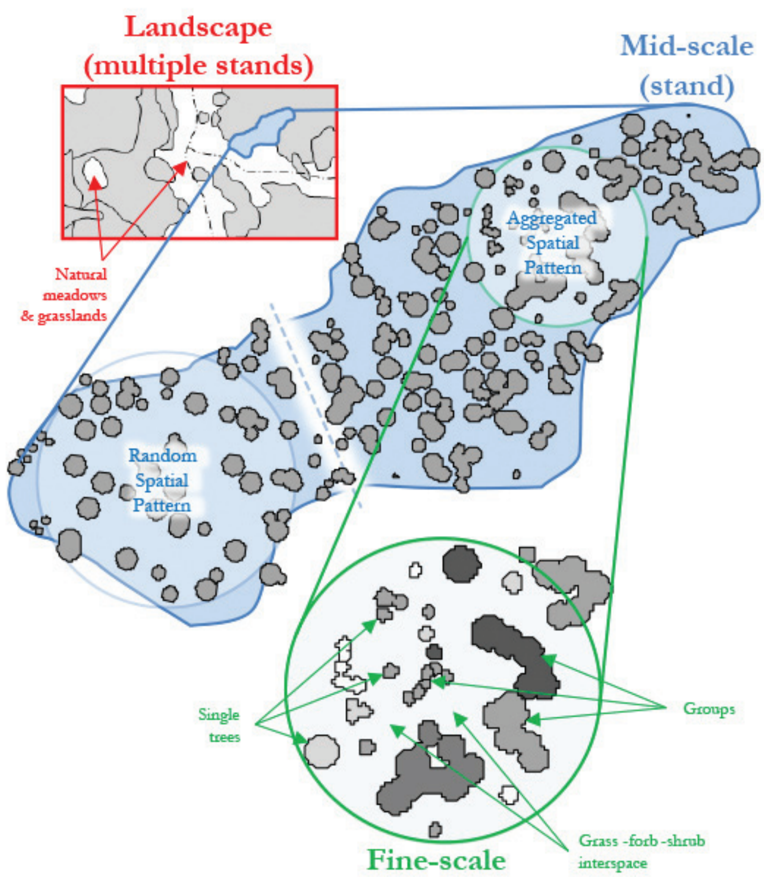

Figure 1. Characteristic vegetation patterns at three spatial scales for frequent-fire forests in the Southwest. The landscape-scale illustrates the importance of multiple stands (patches), meadows, and grasslands. The mid- and fine-scales illustrate grass-forb-shrub interspaces and uneven-aged stand conditions consisting of single, random, and grouped trees of different vegetation structural stages (from young to old) represented by different shades and sizes at the fine-scale. Also depicted are two different tree spatial patterns at the mid-scale (separated by the dashed line): trees are randomly spaced on the left side of the dashed line and are aggregated on the right (given the definition of stand as a homogenous area, both patterns could not actually be present).

(5) snags, logs, and woody debris; and

(6) variation in arrangements of these elements in space and time (Fig. 1).

Ecosystems are structured hierarchically and their composition, structure, processes, and functions are temporally and spatially dynamic. Therefore, we characterize the key compositional and structural elements at three spatial scales: fine (<10 acres), mid (10-1000 acres), and landscape (1000-10,000+ acres) (Fig. 1). These scales generally correspond with structural features in frequent-fire forests. The fine scale is an area in which the species composition—age, structure, and spatial distribution of trees (single and grouped)—and grass-forb-shrub interspaces are expressed. Aggregates of fine-scale units comprise mid-scale patches or stands, which are relatively homogeneous in vegetation composition and structure. The landscape scale is composed of aggregates of mid-scale units and usually has variable elevations, slopes, aspects, soil types, plant associations, disturbance processes, and land uses. Understanding and incorporating temporal scales (e.g., seasonal, annual, decadal, and centuries) in a restoration framework is required to sustain vegetation dynamics of forests that result from growth, succession, senescence, and the historical and anthropogenic disturbances that periodically reset the dynamics.

Management recommendations for implementing our framework are tempered by our management and research experience in frequent-fire forests, as well as by lessons learned during implementations of the framework in the Southwest. The intent of our framework is to inform management strategies that will facilitate the resumption of historical processes and functions. Managing for the framework's key elements should increase the resilience of the forests and facilitate opportunities for the resumption of characteristic function and disturbance regimes. The spatial and temporal aspects of these elements reflect the reciprocal interactions between pattern and process in these forests and are an ecological basis (Turner 1989) for incorporating spatial information in forest restoration (Larson and Churchill 2012).

Mechanisms Influencing Forest Composition

Plant species composition of a forest ecosystem is influenced by both deterministic and stochastic factors, including complex interactions among species' life histories, disturbance regimes, and chance events. The establishment, growth, and survival of under- and over-story species are affected by competition for space, light, nutrients, and moisture. For example, tree regeneration and growth is affected by species-specific shade tolerance (Fig. 2); open stand conditions favor the regeneration of shade-intolerant species while closed stands favor shade-tolerant species (Langsaeter 1944; Long 1985; USDA Forest Service 1990). Biophysical conditions, such as soils, temperature, and moisture regimes, also influence the establishment, development, and abundance of under- and over-story plant species. Disturbances (e.g., fire, insects, pathogens, drought, and wind) often interact with biophysical site characteristics to further influence composition and structure of forest ecosystems. Such disturbances have variable temporal and spatial effects on vegetation depending on their type, frequency, intensity, seasonality, and spatial scale, which collectively define a characteristic disturbance regime of an ecosystem. Species in a forest ecosystem evolved under its characteristic disturbance regime, resulting in a natural range of variability or the range of ecological and evolutionary conditions appropriate to an ecosystem (Landres and others 1999).

Fire is the primary disturbance agent in many Southwestern forests, and fire regimes are central to understanding an ecosystem's reference conditions and natural range of variability (Fig. 3; Table 2) (Fulé and others 2003). The species composition, as well as the structure and spatial pattern of vegetation in Southwestern frequent-fire forests developed in a feedback relationship with fire. Ponderosa pine and dry mixed-conifer forests are characterized by a frequent low-severity fire regime (Swetnam and Baisan 1996; Swetnam and Betancourt 1990) with historic mean fire return intervals ranging from 2-24 years (Brown and others 2001; Brown and Wu 2005; Evans and others 2011; Hunter and others 2007; Swetnam and Baisan 1996). Frequent low-severity fire favors shade intolerant and fire-resistant tree species (Fig. 2) and open forest conditions with discontinuous crowns and minimal fuels build-up, often with tree groups separated by open interspaces with grass-forb-shrub communities. In contrast, longer fire return intervals permit seedling development to larger, more fire-resistant tree sizes and favor survival of less fire-resistant species (Fig. 2) (Fulé and Laughlin 2007; Laacke 1990; Taylor and Skinner 2003).

Endemic forest insects and pathogens are important disturbance agents that do not threaten long-term

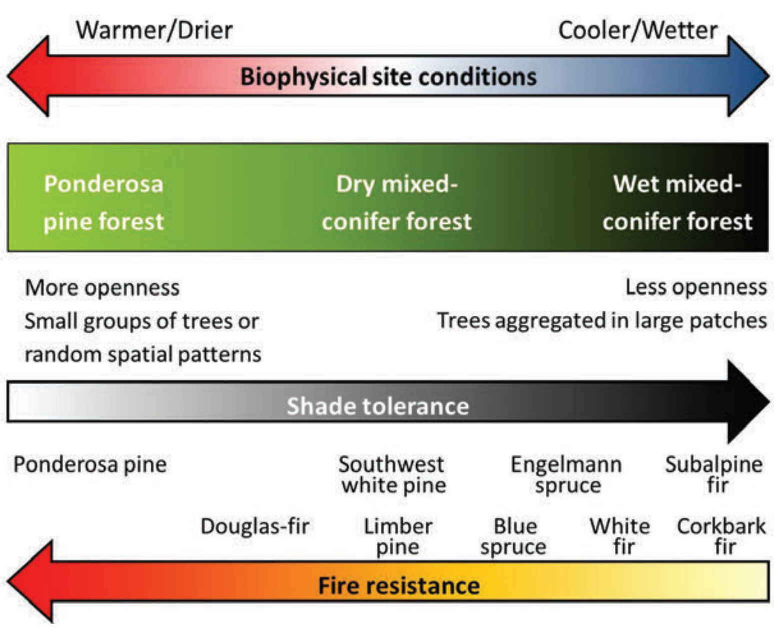

Figure 2. Dry mixed-conifer forests occupy the ecological gradient from warm/dry to cool/wet biophysical site conditions. Dry mixed-conifer is not a homogenous type, intergrading with ponderosa pine forest on warm/dry sites and wet mixed-conifer forests on cool/wet sites. Its structure and composition become more similar as it intergrades with adjacent forest. Common tree species in ponderosa pine and mixed-conifer forests also vary in their relative shade and fire tolerance.

Figure 3. Prescribed, low-severity surface fire carried by needles, cones, dried grass, and forbs on the Lincoln National Forest, 2010.

stability and productivity of forests under endemic conditions due to moderation by millions of years of evolution (Goheen and Hansen 1993). When large or uncharacteristic insect and disease outbreaks occur, profound changes to the composition, structure, processes, and functions of forests often take place. Insects and diseases affect nearly all aspects of forest stand dynamics, from seed viability to seedling survival, from bud, shoot, and leaf production to growth and maintenance, and, ultimately, the survival and distribution of mature trees (Castello and others 1995; Tainter and Baker 1996). Bark beetles, in particular, are considered primary sources of mortality in Southwestern ponderosa pine forests. In 2011 alone, bark beetles caused varying rates of ponderosa pine mortality on more than 144,000 acres in Arizona and New Mexico (USDA Forest Service 2012). Unlike bark beetles in ponderosa pine, the primary sources of mortality attributed to insects in mixed-conifer forests are typically defoliating insects. Damage from defoliators can range from large areas of widespread growth losses and infrequent mortality, as with the spruce budworm, to more localized, high levels of mortality caused by the Douglas-fir tussock moth (Wickman 1963).

While numerous species of dwarf mistletoe occur in frequent-fire forests, Southwestern (ponderosa pine) dwarf mistletoe and Douglas-fir dwarf mistletoe are the most prevalent. Dwarf mistletoes may be the most damaging of pathogens in Southwest forests with estimates of current infection being 30 percent or greater in ponderosa pine forests (Andrews and Daniels 1960; Maffei and Beatty 1988) and around 50 percent in mixed-conifer forests (Conklin and Fairweather 2010; Drummond 1982). Additionally, the presence and intensity of Southwestern dwarf mistletoe infection in ponderosa pine stands has been implicated as a source of mortality or as an exacerbating factor in bark beetle outbreaks (Negrón 1997; Stevens and Hawksworth 1984). Endemic soil fungi that cause root disease (e.g., armillaria and black-stain root diseases) also influence forest composition and structure (Rippy and others 2005). Root diseases are known to affect the ponderosa pine forests of the Southwest, with observations of mortality associated with root disease, mistletoe, and bark beetles as high as 25 percent (Wood 1983). In some locations, conifers killed by root disease are replaced by less susceptible conifers, hardwood species, or grass-forb-shrub interspaces. In the case of armillaria and related wood decay fungi, this shift in species composition can be maintained for decades due to remnant fungi in stumps and root systems (Roth and others 1980). In most situations, native root diseases do not cause irreplaceable loss of entire stands over large areas, nor do they threaten the existence of any host species. However, shifts in stand composition and other natural and human-caused disturbances have frequently resulted in increased damage from root diseases (Edmonds and others 2000).

Mechanisms Influencing Forest Structure

Frequent-fire forests typically comprise a mosaic pattern of groups of trees, scattered single trees, grass-forb-shrub interspaces, snags, logs, and woody debris (Cooper 1960; Larson and Churchill 2012; Pearson 1950; White 1985). Structural heterogeneity

Table 3. Historical spatial patterns and tree group characteristics in frequent-fire forests of the Southwest, arranged by forest type (PP: ponderosa pine, PO: pine-oak, DMC: dry mixed-conifer).

Location	Parent material	Elevation (ft)	Forest type	Reference date	Tree sizes (dbh in inch.)	Group density (groups/acre)	Group size (acres)	Trees per group[a]	Percent basal area in groups	Citation
Malay Gap, Arizona	Basalt	7200	PP	1952	≥ 4.0		0.16-0.32			Cooper 1960
Gus Pearson Natural Area, Arizona	Basalt	7398	PP	1875	Unknown		0.05-0.72	3-44		White 1985
Flagstaff, Arizona	Varying	7800	PP	1880	Unknown	1-33		2-25	28%-74%	Abella and Denton 2009
Woolsey Plots, Arizona	Basalt	7052	PP	1874	≥ 3.5	25-67	0.003-0.09	3-24	62%-75%	Sánchez Meador and others 2011
Coulter Ranch, Arizona	Basalt	7520	PO	1913	≥ 3.5		0.01-0.1			Sánchez Meador and Moore 2010
Uncompahgre Plateau, Colorado	Shale	8000	PP / DMC	1875	Unknown		0.1-0.25			Binkley and others 2008
Numerous national forests in Arizona and New Mexico	Varying	8650	PP / PO / DMC	1910	≥ 3.5	24-80	0.01-0.32	2-72	51%-85%	Sánchez Meador and others unpublished data[b]

[a]Values should be not interpreted as "strict" densities of trees within groups as authors used different definitions and methods to define and characterize "groups." We suspect that as the number of species and site productivity increase the metric of "tree group" becomes less useful than this metric at the mid- to landscape-scale. For example, when tree density is fixed and numbers of tree species varies (i.e., compare ponderosa pine vs. ponderosa-pine Gambel oak vs. dry-mixed conifer forests), the area available to a "tree group" will likely decrease.

[b]Data based on 2.47-acre plots reconstructed prior to Euro-American settlement (1876-1890) in Arizona (n = 17 plots) and New Mexico (n = 7 plots) using the same methods as Sánchez Meador and others (2010, 2011). Historical and contemporary field methods, as well as contemporary conditions, are detailed in Sánchez Meador and others (2010) and Moore and others (2004) who reported forest structural reference conditions (size distributions, tree density ranges, spatial patterns, etc.) on a subset of these same plots. In brief, all live and dead tree structures were measured, including stumps, snags, and wind-fallen trees, that grew to at least breast height (4.5 ft). All tree structures were located using historical stem-maps and measured spatial coordinates, and dendrochronological reconstructions were used to quantify structural and spatial reference conditions (Baker and others 2008; Sánchez Meador and others 2010). Spatial attributes (e.g., group size and density) were quantified using methods described in Sánchez Meador and others (2011).

Figure 4. A group of ponderosa pine trees comprised of two clumps of trees.

in these forests is a consequence of interactions among biophysical site conditions (e.g., topography, soils, climate); disturbance types, frequencies, intensities, and extent; levels of competition among species; and tree demographic rates. Variability in biophysical site conditions is a primary source of spatial and temporal variation in vegetation structure. Of studies that investigated the origin, distribution, and mortality of ponderosa pine forests, most reported uneven-aged reference conditions at the stand scale (Sánchez Meador and others 2010), but three different within-group age structures were identified. Cooper (1960) reported relatively even-aged tree groups, White (1985) and Abella (2008) reported groups of multi-aged trees, and Sánchez Meador and others (unpublished data; see Table 3 footnote) found mixtures of both types. Variation of tree ages within groups likely reflects the establishment and growth of a single, grouped cohort of trees and perhaps seedling establishment and growth of trees under, or adjacent to, tree groups (see Spatial Patterns: Formation and Maintenance) (Sánchez Meador and others 2009).

Heterogeneity of within-group tree sizes can generate from processes related to growth, competition, and disturbances and may result in a range of tree sizes irrespective of age (Mast and Veblen 1999; Pearson 1950; Sánchez Meador and others 2011; Taylor 2010; Woodall 2000). Trees on the perimeter of groups tend to have higher growth rates, attain larger sizes, lean away from the group center, and have asymmetrical crowns with larger lower limbs than interior trees (Pearson 1950). Heterogeneity in tree sizes and spacing within groups may decline over time due to mortality resulting in a gradual transition from dense to more uniform spacing of trees (Cooper 1961; Mast and Veblen 1999; Mast and Wolf 2004, 2006; Pielou 1960). However, tight clumps of trees sharing the same root ball often persist within groups (Fig. 4) (Larson and Churchill 2012). Mortality over time may also gradually reduce within-group tree density, resulting in increased variation in tree densities and ages within and among groups.

Like composition, the structure of forest vegetation is also affected by disturbances such as fire, insects, disease, wind, and drought (Brown and others 2001; Ehle and Baker 2003; Mast and others 1998, 1999). Numerous abiotic and biotic disturbances affect the composition, amount, arrangement, spatial continuity, and volatility of surface and canopy fuels (Franklin and others 2012), which in turn effects fire behavior (Van Wagner 1977). Dense forest structures can facilitate crown fire by providing a potential path for fire through tree crowns (Cruz and others 2003; Fulé and others 2001; Graham and others 2004; Stratton 2004; Van Wagner 1977, 1993). Forest density further influences surface and canopy fuels through interactions with insects and diseases. The effects of bark beetles in ponderosa pine stands are more pronounced during and following extended droughts and under dense stand conditions; both of which are conducive to the survival and reproduction of beetle populations. Negrón (1997) showed a link between roundheaded pine beetle attacks and higher densities of smaller, pole-sized trees in relatively homogenous stands of ponderosa pine in the Sacramento Mountains of New Mexico. Additionally, trees with heavy mistletoe infection are more susceptible to severe crown scorch and death from fires (Harrington and Hawksworth 1990; Hoffman and others 2007). Hawksworth and Wiens (1996) suggested that mistletoes have been important species in frequent-fire forests since fire first appeared on these landscapes.

The density and arrangement of forest canopies affects the penetration of sunlight, precipitation, humidity, and wind. In fact, dense forest structures can maintain relatively high fuel moistures and ameliorate wind effects. Forest canopies also influence the composition and abundance of surface fuels, which are essential to facilitate fire as a disturbance agent. Surface fuels also offer nutrients to soils, help reduce erosion, and influence understory vegetation productivity, density, and diversity (Kalies and others 2012; Kerns and others 2003; Moore and others 1999). In general, more fuel accumulates and persists in forests with longer fire return intervals than in those with more frequent surface fire (Brewer 2008; Minnich and others 2000). Fine fuels (grass, needles, cones, and woody material less than 0.25 inches in diameter) and small branches accumulate more rapidly under tree groups than in interspaces between tree groups (Fig. 5). This accumulation facilitates fire, in turn restricting the establishment and persistence of trees and shrubs under tree groups. The amount and composition of surface fuels interact with weather conditions to influence fire behavior. Herbaceous fuels respond quickly to relative humidity and thus carry fire less readily when humidity is high, whereas pine needles will readily carry fire under these conditions (see moisture of extinction in Anderson 1982; Scott and Burgan 2005). Furthermore, needle and twig litter will burn with higher intensity than herbaceous fuel under similar weather conditions.

Forest structure affects the distribution, density, and composition of surface and canopy fuels, which affects the behavior of fire and, ultimately, post-fire forest structure. Historically, seedling establishment was more frequent in fire-created areas of bare mineral soil where competition with other vegetation and the abundance of surface fuels were reduced (Agee 1993; Cooper 1960; Stephens and others 2008). However, regeneration is less affected by the availability of bare mineral soil in some plant associations and soil types (Hanks and others 1983; USDA Forest Service 1997). A study in the Southwest showed a high density of tree regeneration on sites with one or more of the following: fine-textured soils, understories where screwleaf muhly was the dominant graminoid, and sites with high annual precipitation (Puhlick and others 2012). Depending on seed availability, some individuals and small groups of seedlings may establish throughout the stand, including under

Figure 5. (a) Fine fuels (grasses, forbs, needles, branches, cones) beneath the crown of an individual tree and **(b)** under the canopy of a tree group.

Figure 6. A group of ponderosa pine saplings in a grass-forb interspace between mature tree groups that experienced faster growth and survived a prescribed fire. Shade-suppressed saplings in heavier fine fuel loadings under a mature group of pine did not survive the fire.

tree groups (Abella 2008; Sánchez Meador and others 2009; White 1985).

Tree seedlings that established in small forest openings are subsequently thinned by later fires and/or other sources of mortality (Fig. 6) (Cooper 1960, 1961; Sánchez Meador and others 2010; Stephens and Fry 2005; White 1985). Young tree groups in open areas reach fire-resistant sizes more rapidly than those beneath closed canopies (Fitzgerald 2005; Sackett and Hasse 1998; York and others 2004). Fire-caused thinning of young tree groups was more substantial if the group was overtopped by older trees due to suppressed seedling growth and increased litter accumulation (Agee 1993; Cooper 1960). Fire-spread through young tree groups may also be influenced by microclimate and fuel moisture in these groups (Harrington 1982). As trees grow, increasing needle and twig accumulations facilitate the spread of surface fire. Seedlings that establish some distance away from mature older trees are also more likely to survive fires due to less rapid accumulation of fine fuels and small branches from overstory trees (Fig. 5, 6), likely leading to less intense and severe fire (Cooper 1960) and variable spacing of tree groups. The seasonality and burning conditions of fire occurrence also result in variable outcomes.

Spatial Patterns: Formation and Maintenance

Spatial patterns of vegetation are a component of forest structure. The historical spatial mosaic of tree groups, scattered individual trees, and openings in frequent-fire forests was maintained by interactions among the locations and types of fuels, the frequency and severity of fire, and tree regeneration and mortality patterns. A landscape mosaic of tree groups and scattered individual trees within an open grass-forb-shrub matrix, along with snags, logs, and woody debris, provides for the predominance of surface fire mixed with small-scale, variable fire behavior (e.g., torching). An open or grouped spatial structure reduces canopy continuity, decreasing a stand's vulnerability to active crown fire (Fitzgerald 2005; Fulé and others 2004; Roccaforte and others 2008; Stephens and others 2009). These interactions were mediated by small-scale variability in fire behavior and effects and often resulted in sites with aggregated tree regeneration that were temporarily "free" or "safe" from fire (Larson and Churchill 2012). The location of some safe-sites for tree regeneration appeared to be related to local areas of previously more intense fire associated with accumulations of coarse woody debris (logs and other dead woody material greater than 3 inches in diameter) originating from the death of individual trees (Sánchez Meador and Moore 2010; West 1969; White 1985) or tree groups (Cooper 1960; Stephens and Fry 2005; Taylor 2010; West 1969). Death of individuals or groups of old trees create new snags and logs that, when consumed by fire, result in "safe" sites for tree regeneration. Extended fire-free periods may allow tree regeneration in areas not typically fire "safe" (Fig. 7) (Fulé and others 2009), resulting in temporal shifting of tree locations where new cohorts develop to fire-resistant sizes. The cyclic repetition of forest vegetation dynamics stemming from disturbances and tree regeneration perpetuates a shifting mosaic of tree groups and individual trees in different stages of development in a grass-forb-shrub matrix (Fig. 8).

Figure 7. Ponderosa pine regeneration under a group of snags. This site is not currently fire "safe" due to the accumulation of surface fuels over an extended fire-free period. In the absence of fire, these seedlings could grow to fire-resistant sizes. If fire occurs prior to the trees attaining fire-resistant size, the seedlings would likely not survive. However, the reduction of surface fuels post-fire may create a temporary fire-safe site for future regeneration.

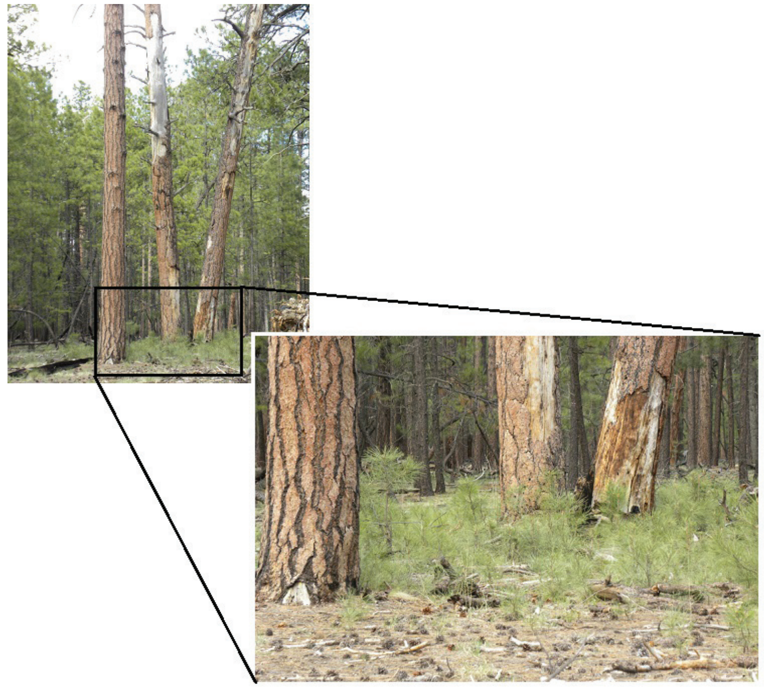

Figure 8. Tree groups and a single individual tree on the right in a grass-forb-shrub interspace.

Insects and diseases also shape spatial patterns of forested landscapes. Due to the slow spread of infection, it has been suggested that the current distribution of mistletoe throughout the Southwest is likely similar to its historical distribution, although spatial continuity and levels of infection may have changed (Conklin and Fairweather 2010). Under historical forest conditions, it is likely that large-scale, contiguous insect and disease outbreaks would have been rare. It is more likely that mistletoe would have thrived in denser multistoried portions of stands that escaped fire pruning and thinning (see Conklin and Geils 2008 for additional discussion). In such portions, periodic tree deaths would have occurred directly from mistletoe, or infected trees would have had increased the likelyhood of succumbing to bark beetles or root disease. Localized mistletoe infections would have created pockets of tree death that could eventually serve as regeneration sites. In cases where regeneration occurred in larger openings between trees, trees may have escaped mistletoe infection altogether. Other scenarios can be envisioned. For instance, in cases of stands with relatively homogenous age and spacing, bark beetles may have had periodic population increases, causing high rates of local mortality. Localized beetle outbreaks likely occurred in stands with severe crown damage following fire (Breece and others 2008), and these infestations may have spilled over into undamaged trees nearby, creating larger openings. Root diseases also create scattered mortality, small openings, and increased volume of snags and downed large woody debris (Rippy and others 2005).

An understanding of forest processes and their effects at different spatial scales is important because landscapes are spatially dependent (Turner 1989). Inferences about patterns and processes in forests are contingent upon the scale at which they are investigated. For example, a fine-scale model for ponderosa pine regeneration showed that the majority of the variance (76 percent) in seedling density was explained by properties such as soil texture and pH, precipitation, seed tree proximity, and composition of the plant community (Puhlick and others 2012). However, at the mid- to landscape-scale, models including abiotic conditions and tree density at this broader scale accounted for less of the variability in observed seedling densities (only 13 percent) (Puhlick and others 2012). Fire further shapes tree spatial patterns at varying scales through its influence on seedling survival, with variability in the severity, seasonality, and frequency of fire (Cooper 1960; Pearson 1950; Stephens and Fry 2005; Taylor 2010; West 1969; White 1985). An overall aggregated (grouped) historical tree pattern separated by openings has been frequently reported in Southwestern frequent-fire forests (Fig. 8) (Larson and Churchill 2012). However, Abella (2008), Binkley and others (2008), and Sánchez Meador and others (unpublished data, see Table 3 footnote) observed grouped and random (no aggregation) historical tree spatial patterns (Fig. 9). Schneider (2012) observed only random historical tree spatial patterns in Southwestern ponderosa pine.

Spatial heterogeneity can exist at any scale, and the value of metrics used to assess forest conditions varies in usefulness with scale. At mid- and landscape scales, elements such as single tree and group density become less useful as a metric and elements such as patches, the grass-forb-shrub matrix, stand density, canopy cover, and basal area become more appropriate. Patches are roughly synonymous with stands, being defined as an area of relatively homogeneous vegetation composition and structure differing from its surroundings (Forman 1995). Patches are the basic unit of the landscape, and their sources of variability are attributed to scale-appropriate factors such as elevation, topography, climate, and land use. Our restoration framework describes forest composition, structure, and spatial patterns at fine-, mid-, and landscape-scales (Fig. 1).

Figure 9. Random (i.e., not aggregated) distribution of ponderosa pine trees in a patch of old trees. Also displayed are snags, logs, and coarse woody debris.

Southwestern Frequent-Fire Forests

The natural range of variability is a "best" estimate of a resilient and functioning ecosystem because it reflects the evolutionary ecology of these forests. Natural range of variability is therefore a powerful science-based foundation for developing a framework for restoring the composition and structure of forests (Kaufmann and others 1994; Keane and others 2009; Moore and others 1999). The natural range of variability can be estimated by pooling reference conditions across sites within a forest type. Reference conditions for a forest type typically vary from site to site due to differences in factors such as soil, elevation, slope, aspect, and micro-climate and manifests as differences in fire effects, tree densities, patterns of tree establishment and persistence, and numbers and dispersion of snags and logs. When pooled, these sources of variability comprise the natural range of variability of a site or forest type.

Our estimates of natural ranges of variability are derived from multiple lines of evidence based on historical ecology techniques (Egan and Howell 2001) such as written and oral historical records, historical photographs, early forest inventories, and dendrochronological studies (Table 4). While cultural accounts and early inventories provide a general context of historical conditions, they do not fully characterize forest structure by today's statistical standards. More recently, dendrochronological techniques for quantifying historical conditions, including spatial and temporal variation, have been developed (e.g., Covington and Moore 1994a; Covington and others 1997; Fulé and others 1997; Mast and others 1999; Sánchez Meador and others 2010; White 1985). Nonetheless, there is a clear need for additional reference condition data sets, including sites from a wider spectrum across environmental gradients (e.g., soils, moisture, elevations, slopes, aspects) occupied by frequent-fire forests in the Southwest, especially in dry mixed-conifer. While the quantity of reference data sets is increasing, existing data represent a largely unbalanced sampling across gradients (e.g., most data sets are from basaltic soils and on dry to typic plant associations), and there have been few studies quantitatively examining and reporting spatial patterns of trees and the sizes and shapes of grass-forb-shrub interspaces.

Ponderosa Pine

Woolsey (1911) described Southwestern ponderosa pine forests as having "…pure park-like stand(s) made up of scattered groups of 2-20 trees, usually connected by scattering individuals. Openings are frequent and vary in size. Because of the open character of the stand and the fire-resisting bark, often 3 inches thick, the actual loss in yellow (ponderosa) pine by fire is less than with other more gregarious species." Others also described historical ponderosa pine forests as having low density, open stands consisting of groups of pine trees interspersed with grassy or shrubby openings (Dutton 1882; Lang and Stewart 1910; Pearson 1923; White 1985).

Tree density, structure, spatial pattern, and ecological functions in today's ponderosa pine forests of the Southwest are greatly altered from their historical conditions. Most Southwest ponderosa pine forests are at much greater risk of high-intensity, severe fire than they were prior to Euro-American settlement (Covington 1993; Fulé and others 2004; Moore and others 1999; Roccaforte and others 2008). Historical ponderosa pine forests had widely spaced, large trees, typically occurring in small groups with scattered single trees, and open forest conditions with a productive grass-forb-shrub understory (Cooper 1960; Dutton 1882; Lang and Stewart 1910; Pearson 1923, 1950; Sánchez Meador and others 2009, 2011; White 1985). The grass-forb-shrub vegetation and other fine fuels and branches carried fires started by lightning and, to an uncertain extent, by Native Americans (Allen and others 2002; Kaye and Swetnam 1999). Forest composition, structure, and spatial patterns were maintained by low-severity surface fires that occurred every 2-26 years (Fig. 3), rarely killing large trees, thinning regeneration, and maintaining an open forest structure (Dieterich 1980; Fiedler and others 1996; Fitzgerald 2005; Pearson 1950; Swetnam and Dieterich 1985; Weaver 1951; Woolsey 1911). Fire chronologies in Western U.S. frequent-fire forests are

Table 4. Citations informing our restoration framework for frequent-fire forests arranged by information type.

Information type	Citations (arranged alphabetically)
Reference conditions from old-growth, natural areas, and other restoration studies	Abella (2008); Abella and Denton (2009); Abella and others (2011); Agee (2003); Binkley and others (2008); Biondi (1996); Biondi and others (1994); Boyden and others (2005); Cocke and others (2005); Cooper (1960, 1961); Covington and Moore (1994a, 1994b); Covington and Sackett (1986); Covington and others (1997); Fornwalt and others (2002); Friederici (2004); Fulé and others (1997, 2002a, 2003, 2009); Harrod and others (1999); Heinlein and others (1999, 2005); Hessburg and others (1999); Johnson (1994); Larson and Churchill (2012); Madany and West (1983); Mast and others (1999); Menzel and Covington (1997); Moore and others (2002, 2004); Pearson (1950); Roccaforte and others (2010); Romme and others (2009); Sánchez Meador and Moore (2010); Sánchez Meador and others (2009, 2010, 2011); Schneider (2012); Smith (2006a, 2006b, 2006c); Taylor (2010); Waltz and Fulé (1998); West (1969); White (1985); White and Vankat (1993); Williams and Baker (2011, 2012); Youngblood and others (2004)
Reference conditions from observations of early explorers, scientists, and managers	Beale (1858); Dutton (1882); Greenamyre (1913); Lang and Stewart (1910); Leopold (1924); Liebeg and others (1904); Meyer (1934); Pearson (1923); Plummer (1904); Rasmussen (1941); Wheeler (1875); Woolsey (1911)
Disturbance histories	Agee (1993); Allen (2007); Andrews and Daniels (1960); Brown and others (2001); Brown and Wu (2005); Dieterich (1980); Ehle and Baker (2003); Ferry and others (1995); Fulé and others (2003); Fulé and others (2004); Grissino-Mayer and others (1995, 2004); Hart and others (2005); Heinlein and others (2005); Hessburg and others (1994); Hessburg and others (2005); Kaye and Swetnam (1999); Korb and others (2013); Littell and others (2009); Lynch and others (2010); Maffei and Beatty (1988); Minnich and others (2000); Scholl and Taylor (2010); Stephens and others (2008); Swetnam and Baisan (1996); Swetnam and Bentacourt (1990); Swetnam and Dieterich (1985); Taylor (2010); Taylor and Skinner (2003); Touchan and others (1996); Weaver (1951); Williams and Baker (2012)

Table 4. *Continued.*

Information type	Citations (arranged alphabetically)
Disturbance effects (fires, insects, and diseases)	Arno and others (1995); Barton (2002); Bentz and others (2009); Conklin and Geils (2008); Castello and others (1995); DeLuca and Sala (2006); Dhillon and Anderson (1993); Drummond (1982); Edmonds and others (2000); Fettig (2012); Fitzgerald (2005); Franklin and others (2012); Fulé and Laughlin (2007); Goheen and Hansen (1993); Harrington and Hawksworth (1980); Hawksworth and Wiens (1996); Hessburg and others (1994); Hoffman and others (2007); Jenkins and others (2008); Lundquist (1995); Madany and West (1983); Miller and Keen (1960); Miller (2000); Moeck and others (1981); Naficy and others (2010); Negrón (1997); Negrón and others (2009); Parsons and DeBenedetti (1979); Rippy and others (2005); Savage and Mast (2005); Stevens and Hawksworth (1984); Tainter and Baker (1996); Von Schrenck (1903); Wickman (1963); Wood (1983)
Effects of forest management on ecosystem functions and processes	Arnold (1950); Baker (1986, 2003); Benayas and others (2009); Beier and others (2008); Boerner and others (2009); Breece and others (2008); Carey (2003); Carey and others (1999); Cocke and others (2005); Colgan and others (1999); Conklin and Geils (2008); Cortina and others (2006); Covington and others (1997); Covington and Sackett (1986, 1992); Cram and others (2007); Dodd and others (2006); Douglass (1983); Feeney and others (1998); Fettig and others (2007); Ffolliott and others (1989); Finkral and Evans (2008); Fulé and others (2001); Harr (1983); Honig and Fulé (2012); Kolb and others (1998); Koonce and Roth (1980); Korb and others (2003); Long and Smith (2000); Mitchell and others (2009); Moore and others (2006); Pilliod and others (2006); Roccaforte and others (2008); Stephens and others (2009); Stratton (2004); Strom and Fulé (2007); Troendle (1983); Waltz and Covington (2003); Wightman and Germaine (2006)
Climate change projections and impacts	Bentz and others (2010); Breshears and others (2005); Brown and others (2004); Harris and others (2006); Honig and Fulé (2012); Karl and others (2009); McKenzie and others (2004); Millar and others (2007); Miller and others (2009); Overpeck and others (2012); Parker and others (2000); Price and Neville (2003); Seager and others (2007); Shafer and others (2001); Smith and others (2008); Spittlehouse and Stewart (2004); Spracklen and others (2009); Westerling and others (2006)
Approaches to restoration and/or monitoring	Allen and others (2002); Aronson and others (2007); Block and others (2001); Bradshaw (1984); Busch and Trexler (2003); Clewell and others (2005); Covington (1993, 2003); Covington and others (1997); Crist and others (2009); Egan and Howell (2001); Falk (2006); Fiedler and others (1996); Fitzgerald (2005); Fulé and others (2002b); Graham and others (2004); Kaufmann and others (1994); Keane and others (2009); Landres and others (1999); Laughlin and others (2006); Lindenmayer and Likens (2010); Long and others (2004); Moore and others (1999); Morgan and others (1994); Mulder and others (1999); Murray and Marmorek (2003); Noon (2003); Palmer and Mulder (1999); Reynolds and others (1992, 2006a); Roccaforte and others (2010); SER (2004); Sitko and Hurteau (2010); Swetnam and others (1999); Wagner and others (2000); Walters (1986); Williams and others (2009)
Science syntheses and tools for forest management	Abella (2008); Abella and others (2006); Anderson (1982); Brewer (2008); Brown and others (2003); Clary (1975); Conklin and Fairweather (2010); Cruz and others (2003); Evans and others (2011); Graham and others (1994); Hunter and others (2007); Long (1985); Noss and others (2006); Patton and Severson (1989); Pearson (1950); Schmidt and others (2002); Schubert (1974); Scott and Burgan (2005); Triepke and others (2011); USDA Forest Service (1990)
Vegetation classifications	Comer and others (2003); DeVelice and others (1986); Hanks and others (1983); USDA Forest Service (1997); Winthers and others (2005)

reviewed in Evans and others (2011), Hunter and others (2007), Smith (2006b), and Swetnam and Baisan (1996).

Bark beetles also influenced pre-Euro-American ponderosa pine structure. Various sources indicate that bark beetle outbreaks occurred periodically in the Western United States since at least the 1750s (Bentz and others 2009) and likely much longer. Current forested landscapes are experiencing outbreaks that are larger and more frequent than previously recorded (Lynch and others 2010). For example, bark beetles caused variable amounts of mortality on more than 700,000 acres in Arizona and New Mexico in 2003 (Fettig and others 2007; USDA Forest Service 2004). Although there is no direct evidence linking the effects of bark beetles to the structure of pre-Euro-American frequent-fire forests, evidence from today's beetle population dynamics suggests that homogenous, dense, even-aged stands are highly susceptible to beetle outbreaks (Fettig and others 2007; Negrón 1997). However, historical observations suggest that high-density, even-aged stand structures were infrequent or rare in frequent-fire forests (Woolsey 1911; reviewed in Covington and Moore 1994a, 1994b). Alternatively, spatial heterogeneity would have been promoted and maintained at the fine scale by bark beetle attacks on single or small groups of trees, or perhaps in high density groups or patches, which would have created growing space for regeneration or surviving trees (Fettig 2012; Lundquist 1995; Von Schrenck 1903). During droughts, it was likely that many more trees would have succumbed to bark beetles (Bentz and others 2010; Negrón and others 2009). Bark beetles evolved under the range of natural variability where there would have been sufficient hosts (e.g., fire-stressed, lightning struck, and broken top trees) to maintain endemic beetle populations (reviewed in Jenkins and others 2008 and Moeck and others 1981).

Ponderosa Pine: Species Composition: Ponderosa pine is the dominant seral and climax tree species in Southwest ponderosa pine forests. Depending on locale, ponderosa pine forests may also have a mix of Gamble oak, evergreen oaks, junipers, pinyon pines (DeVelice and others 1986), with occasional presence of quaking aspen, New Mexico locust, Douglas-fir, or southwestern white pine. Ponderosa pine is one of the most fire-adapted conifer species in the West, and its resistance to surface fire increases as trees age (Miller 2000).

Composition of the grass-forb-shrub community in ponderosa pine forests is typically diverse, especially in open interspaces between trees (e.g., Fig. 8) (Abella and others 2011; Laughlin and others 2006; Moir 1966; Naumburg and DeWald 1999). Ponderosa pine plant associations (classified by understory plant assemblages, plant succession, and co-dominant tree species) are variable and are reflective of local biophysical site and climate conditions that both influence the type of disturbances and vegetation responses to disturbances (Table 5) (USDA Forest Service 1997). Southwestern ponderosa pine plant associations range from pure ponderosa pine to mixed tree species overstories with understories ranging from bunchgrass/forb to shrub-dominated types, and these can be broadly grouped into four forest subtypes: (1) ponderosa pine-bunchgrass, (2) ponderosa pine-Gambel oak, (3) ponderosa pine-evergreen oak, and (4) ponderosa pine-shrub (Appendix 2). The most mesic sites are the ponderosa pine-Gambel oak and some ponderosa pine-bunchgrass plant associations; the most xeric sites are the ponderosa pine-evergreen oak and some ponderosa pine-shrub plant associations. Bunchgrass plant associations generally occupy the mid-range of the moisture gradient for ponderosa pine forests in the Southwest.

Understory composition includes various combinations of grasses, forbs, shrubs, ferns, and cacti depending upon plant associations (Korb and Springer 2003; USDA Forest Service 1997), all of which contribute to the biodiversity found in frequent-fire forests (Laughlin and others 2006). The growth habit (e.g., bunchgrass, sod, or shrub) and spatial patterns of the understory influence the establishment and growth of trees (Biondi 1996; Boyden and others 2005; Sánchez Meador and others 2009; Youngblood and others 2004) and provide wildlife habitats (Dodd and others 2006; Reynolds and others 1992; Waltz and Covington 2003; Wightman and Germaine 2006; USDA Forest Service 1997). Variation in species composition among plant associations within forest subtypes influences forest dynamics. For example, within the ponderosa pine bunchgrass subtype, tree regeneration establishes rapidly following disturbance on sites with screwleaf muhly plant associations (the most mesic associations in the bunchgrass subtype), episodically on Arizona fescue plant associations (the typic associations in the bunchgrass subtype), and sparsely on blue grama plant associations (the most xeric associations in the bunchgrass subtype) (USDA Forest Service 1997). Tree establishment often occurs differently in shrub-dominated plant associations than in bunchgrass types, where rapid re-sprouting of shrub species (e.g., shrub live oak) following disturbances may inhibit pine regeneration. Other re-sprouting shrubs (e.g., New Mexico locust) are nitrogen-fixers and have been shown to facilitate pine seedling establishment (Fisher and

Table 5. Plant associations for the ponderosa pine series in the Southwestern United States sorted by ponderosa pine subtype, temperature-moisture gradient, dominant season of precipitation, and parent material type (USDA Forest Service 1997).

Plant association (common name)	Ponderosa pine subtype	Temperature-moisture gradient[a]	Climate[b]	Dominant season of precipitation[c]	Parent material type[d]
Ponderosa pine plant association series					
Ponderosa pine/screwleaf muhly-Arizona fescue	Bunchgrass	Cool-wet	Cold	Summer	Variable
Ponderosa pine/screwleaf muhly-Arizona fescue/blue grama	Bunchgrass	Cool-wet	Cold	Summer	Variable
Ponderosa pine/screwleaf muhly-Arizona fescue/Gambel oak	Bunchgrass	Cool-wet	Cold	Summer	Variable
Ponderosa pine/screwleaf muhly	Bunchgrass	Cool-wet	Cold	Winter	Sed./rhy./tuff
Ponderosa pine/screwleaf muhly/Gambel oak	Bunchgrass	Cool-wet	Cold	Winter	Variable
Ponderosa pine/Arizona fescue	Bunchgrass	Typic	Cold	Winter	Variable
Ponderosa pine/Arizona fescue/Parry's oatgrass	Bunchgrass	Typic	Cold	Winter	Volcanic
Ponderosa pine/Arizona fescue/blue grama	Bunchgrass	Typic	Cold	Winter	Variable
Ponderosa pine/Arizona fescue/Gambel oak	Bunchgrass	Typic	Cold	Winter	Variable
Ponderosa pine/blue grama	Bunchgrass	Warm-dry	Cold	Summer	Variable
Ponderosa pine/blue grama/gray oak	Bunchgrass	Warm-dry	Cold	Summer	Variable
Ponderosa pine/blue grama/Gambel oak	Bunchgrass	Warm-dry	Cold	Summer	Variable
Ponderosa pine/mountain muhly	Bunchgrass	Warm-dry	Cold	Summer	Variable
Ponderosa pine/Arizona walnut	Bunchgrass	Warm-dry	Cold	Summer	Alluvium
Ponderosa pine/Indian ricegrass	Bunchgrass	Warm-dry	Cold	Winter	Aeolian
Ponderosa pine/rockland	Bunchgrass	Variable	Variable	Variable	Variable
Ponderosa pine/Gambel oak	Gambel oak	Cool-wet	Cold	Winter	Variable
Ponderosa pine/Gambel oak/Arizona fescue	Gambel oak	Cool-wet	Cold	Winter	Variable
Ponderosa pine/Gambel oak/mountain muhly	Gambel oak	Cool-wet	Cold	Winter	Variable
Ponderosa pine/Gambel oak/New Mexico locust	Gambel oak	Cool-wet	Cold	Winter	Variable
Ponderosa pine/Gambel oak/longtongue muhly	Gambel oak	Warm-dry	Cold	Winter	Sed./gran.
Ponderosa pine/Gambel oak/two needle pinyon	Gambel oak	Warm-dry	Cold	Winter	Variable
Ponderosa pine/Gambel oak/blue grama	Gambel oak	Warm-dry	Cold	Winter	Variable

Table 5. *Continued.*

Plant association (common name)	Ponderosa pine subtype	Temperature-moisture gradient[a]	Climate[b]	Dominant season of precipitation[c]	Parent material type[d]
Ponderosa pine plant association series					
Ponderosa pine/netleaf oak	Evergreen oak	Cool-wet	Mild	Summer	Volcanic
Ponderosa pine/silverleaf oak	Evergreen oak	Typic	Mild	Summer	Volcanic
Ponderosa pine/gray oak/mountain muhly	Evergreen oak	Warm-dry	Mild	Summer	Variable
Ponderosa pine/gray oak/longtongue muhly	Evergreen oak	Warm-dry	Mild	Summer	Variable
Ponderosa pine/Arizona white oak	Evergreen oak	Warm-dry	Mild	Summer	Variable
Ponderosa pine/Arizona white oak/blue grama	Evergreen oak	Warm-dry	Mild	Summer	Variable
Ponderosa pine/Emory oak	Evergreen oak	Warm-dry	Mild	Winter	Alluvium
Ponderosa pine/blue grama/big sagebrush	Shrub	Cool-wet	Cold	Winter	Sed.
Ponderosa pine/kinnikinnik	Shrub	Typic	Cold	Winter	Rhy./tuff/gran.
Ponderosa pine/black sagebrush	Shrub	Warm-dry	Cold	Winter	Sed.
Ponderosa pine/pointleaf manzanita	Shrub	Warm-dry	Mild	Summer	Variable
Ponderosa pine/wavyleaf oak	Shrub	Warm-dry	Cold	Summer	Variable
Ponderosa pine/Stansbury cliffrose	Shrub	Variable	Cold	Winter	Limestone

[a]Typic refers to modal, mid-gradient temperature-moisture types.
[b]Climate refers to mean annual soil temperatures, with cold climates having frigid soils (mean annual soil temperatures <8 °C) and mild climates having mesic soils (mean annual soil temperatures >8 °C).
[c]Dominant season of precipitation refers to the 6-month period (winter = October-March, summer = April-September) that typically has higher average precipitation levels. Most ponderosa pine and dry mixed-conifer sites in the Southwestern United States receive bimodal precipitation, but the season listed in the table experiences higher average precipitation levels.
[d]Variable = multiple parent materials; sed. = sedimentary; rhy. = rhyolites; gran. = granites

Fulé 2004; USDA Forest Service 1997). Fire may also facilitate establishment of tree regeneration on sites with non-sprouting shrub species (e.g., black or big sagebrush species) by removing competition. Together, trees and the grass-forb-shrub community affect below- and aboveground microclimates (i.e., soil moisture, nutrients, etc.) as well as ecological processes and functions such as biodiversity, trophic interactions, food webs, disturbances, and hydrology (Abella 2009; Arnold 1950; Barth 1980; Covington and others 1997; Kalies and others 2012; Moir 1966; Parker and Muller 1982; Scholes and Archer 1997) (see Expected Outcomes of Framework Implementation). Environmental variables such as light intensity, soil pH, soil and litter depth, and percent litter cover are directly influenced by the presence of tree canopy cover (Evenson and others 1980). For example, Abella (2009) reported that understory species richness was greater and plant cover was up to eight times greater in openings than under tree canopies in a ponderosa pine/Gamble oak forest.

Mycorrhizal fungi are important species in ponderosa pine and play an important role in plant nutrition, nutrient cycling, soil structure, and food webs (Carey 2003; Johnson and others 1997). Two Arizona studies reported higher densities of mycorrhizal propagules in areas where grass cover was greater and tree cover was less, such as in areas following mechanical treatments and burning, and that increased light and soil moisture in restored stands likely increased photosynthesis and mycorrhizal infection (Korb and others 2003; Korb and Springer 2003). Other studies show that abundant arbuscular mycorrhizae can increase plant diversity and overall community structure (Klironomos and others 2000; van der Heijden and others 1998). Arbuscular mycorrhizae are particularly important in grass-dominated ecosystems (Dhillion and Anderson 1993; Koske and Gemma 1997), but little is known of their status in the grass-forb-shrub community in ponderosa forests (Korb and Springer 2003).

Ponderosa Pine: Forest Structure: Structure in ponderosa pine forests emanates from the vertical and horizontal arrangement of trees and grass-forb-shrub species. Specifically, the vertical and horizontal architecture of a forest arises from variations in tree and grass-forb-shrub species and their ages, heights, crown spreads, densities, and spatial heterogeneity. Human activities since the late 19th Century resulted in changes to forest structure due to a reduction in fire frequency causing tree density and surface fuel load increases (Moore and others 2004; Naficy and others 2010; Parsons and DeBenedetti 1979; Scholl and Taylor 2010). For example, Moore and others (2004) reported a mean tree density increase by a factor of almost 7 (32-208 trees per acre) between 1909 and the 1990s. Tree encroachment into grass-forb-shrub forest openings has resulted in a decline in percent cover, abundance, and biodiversity of open grass-forb-shrub communities (Abella 2009; Bogan and others 1998; Clary 1975; Covington and Moore 1994b; Moore and others 2006; Moore and Deiter 1992; Swetnam and others 1999).

Differences in reference conditions for tree densities have been reported for fine- versus coarse-textured soils (Abella and Denton 2009; Puhlick 2011). Average plot-level reference conditions in ponderosa pine on basalt soils ranged between 0-220 trees per acre and 33-83 square ft per acre of basal area while sites on coarse-textured soils (primarily limestone) ranged between 8 and 262 trees per acre and 22 and 89 square ft per acre of basal area (Table 6; Fig. 10). In general, ranges reported for reference tree densities on coarse-textured soils were higher than those reported on fine-textured soils (Table 6). The minimum diameters reported in Table 6 may also result in a source of error that can lead to small underestimates of historical tree densities reported in studies. Additional error may result from missing fully decomposed structures at time of measurement and reconstruction (Fulé and others 1997; Mast and others 1999; Moore and others 2004).

To date, only six studies report tree spatial reference conditions in the Southwestern ponderosa pine forests. Based on these studies, the historical conditions in ponderosa pine exhibited as many as 67 tree groups per acre. Tree groups ranged between 0.003 and 0.72 acres in size and were composed of 2-72 trees (Table 3; Fig. 4). Tree groups were separated by grass-forb-shrub openings of variable sizes and shapes that contained scattered individual trees (Fig. 8). The proportion of the stand or mid-scale area not covered by vertical projections of tree crowns (referred to as "openness") has received little attention. However, several studies have reported the inverse of openness—canopy cover (Table 7); White (1985), Covington and Sackett (1986), and Covington and others (1997) reported 21.9, 19.0, and 17.3 percent canopy cover for ponderosa pine stand reference conditions on the Fort Valley Experimental Forest, Arizona, respectively. A nearby study of a reconstructed ponderosa pine/Gambel oak site on the Coconino National Forest, Arizona, reported a range of 10.2-18.8 percent canopy cover (Sánchez Meador and others 2011). Fulé and others (2002) reported an average canopy cover of 48.3 percent for the Rainbow Plateau, an area in the Grand Canyon National Park-North Rim

Table 6. Historical forest structure of ponderosa pine (pine-oak shaded) forests of the Southwest, arranged by parent material and mean tree density.

Location	Parent material	Elevation (ft)	Size/age reported	Reference date	Trees per acre			Basal area (ft^2/acre)			Citation
					Range	Mean	Std Err	Range	Mean	Std Err	
Gus Pearson Natural Area, Arizona	Basalt	7398	Age	1875		15.0					White 1985
Coconino National Forest, Arizona (avg)[a]	Basalt	6907	Size	1910		16.0			38.1		Woolsey 1911
Gus Pearson Natural Area, Arizona[b]	Basalt	7400	Size	1925		21.8			56.6		Pearson 1950
Gus Pearson Natural Area, Arizona	Basalt	7300	No	1876		22.8			46.2		Covington and others 1997
Bar M Canyon, Arizona	Basalt	7000	No	1867	21-24	23.0			65.0		Covington and Moore 1994b
Flagstaff, Arizona[c]	Basalt	7355	No	1880	1-58	23.7	4.0				Abella and Denton 2009
Gus Pearson Natural Area, Arizona	Basalt	7300	Age	1876		24.0					Mast and others 1999
San Francisco Peaks, Arizona	Basalt	8594	Age	1876		24.8	2.6		33.0	4.9	Cocke and others 2005
Mt. Trumbull, Arizona	Basalt	7740	Age/Size	1870		25.2	3.5		38.8	6.1	Heinlein and others 1999
Coconino National Forest, Arizona (max)[a]	Basalt	6907	Size	1910		34.5			81.2		Woolsey 1911
Mt. Logan, Arizona[b]	Basalt	7483	Age/Size	1870		38.3	5.8		46.2	7.8	Waltz and Fulé 1998
Mt. Trumbull, Arizona	Basalt	6970	Size	1870	0-220	39.2	3.9	0-143	41.6	4.1	Roccaforte and others 2010
Chimney Spring, Arizona[a]	Basalt	7380	Size	1920		42.8					Biondi and others 1994
Coulter Ranch, Arizona[a]	Basalt	7520	Size	1913	30-66	51.5	10.8	67-120	83.0	19.5	Sánchez Meador and Moore 2010
Camp Navajo, Arizona	Basalt	7592	Age/Size	1883		59.9	5.8		56.2	6.1	Fulé and others 1997
Malay Gap, Arizona[b]	Basalt	7200	Age/Size	1952		124.0			70.1		Cooper 1960
Woolsey Plots, Arizona[a]	Basalt	7052	Size	1874	18-51	33.1	4.6	40-79	61.5	5.6	Sánchez Meador and others 2010
Flagstaff, Arizona[c]	Cinders	7355	No	1880	7-74	22.5	6.2				Abella and others 2011
Mt. Logan, Arizona[c]	Cinders	7115	Age/Size	1870	34-38	29.9	6.4	60-64	60.3	9.1	Waltz and Fulé 1998
Red Cinder, Arizona	Cinders	7631	Age/Size	1885		74.1			65.3		Abella 2008
Prescott National Forest, Arizona (avg)[a]	Granitic	5320	Size	1910		27.7			25.5		Woolsey 1911

Location	Parent material	Elevation (ft)	Size/age reported	Reference date	Trees per acre			Basal area (ft2/acre)			Citation
					Range	Mean	Std Err	Range	Mean	Std Err	
Tusayan, Arizona (avg)[a]	Limestone	7075	Size	1910		10.7			22.1		Woolsey 1911
Zion National Park, Utah	Limestone	7096	Age	1881	3-25	14.0					Madany and West 1983
Flagstaff, Arizona[a]	Limestone	7355	No	1880	14-34	22.0	2.2				Abella and others 2011
Walnut Canyon National Monument, Arizona[d]	Limestone	6808	Size	1876		29.1			39.2		Menzel and Covington 1997
North Kaibab, Arizona	Limestone	7300	No	1881		55.9					Covington and Moore 1994a
Kaibab Plateau, Arizona[c]	Limestone	7500	No	1929	40-55						Rasmussen 1941
Grandview, Arizona	Limestone	7422	Age	1887	4-247	64.6	10.4	19-281	74.1	12.6	Fulé and others 2002
Fire Point, Arizona	Limestone	7671	Age	1887	16-126	61.8	61.8	28-132	89.3	9.1	Fulé and others 2002
Rainbow Plateau, Arizona	Limestone	7612	Age	1879	8-228	56.7	5.7	1-99	39.6	2.6	Fulé and others 2002
Powell Plateau, Arizona	Limestone	7533	Age	1879	8-262	63.6	9.4	20-337	78.0	10.9	Fulé and others 2002
Zuni, New Mexico (max)[a]	Rhyolite	6557	Size	1910		22.6			52.8		Woolsey 1911
Cibola National Forest, New Mexico[a]	Rhyolite	8382	Age/Size	1890	47-61	54.2	6.9				Moore and others 2004
Carson, New Mexico (max)[a]	Shale	6983	Size	1910		38.4			79.9		Woolsey 1911
Mogollon Plateau, Arizonac	Mixed	Mixed	No	1890	33-76	57.3	30.7	48-79			Williams and Baker 2011, 2012
Uncompahgre Plateau, Colorado	Shale	7500	Size	1875	30-90	55		20-90	55		Binkley and others 2008

[a]Minimum tree DBH recorded = 3.5 inches
[b]Minimum tree DBH recorded = 6 inches
[c]Minimum tree DBH recorded = 4 inches
[d]Minimum tree DBH recorded = 10 inches

Table 7. Historical canopy cover and openness estimates of frequent-fire forests of the Southwest, arranged by forest type (PP: ponderosa pine, PO: pine-oak, DMC: dry mixed-conifer).

Location	Parent material	Forest type	Method	Reference date	Canopy cover (%)	Openness[a] (%)	Citation
Gus Pearson Natural Area, Arizona	Basalt	PP	Standing age class	1875	21.9	78.1	White 1985
Gus Pearson Natural Area, Arizona	Basalt	PP	Dendro-reconstruction	1876	19.0	81.0	Covington and others 1997
Chimney Springs, Arizona	Basalt	PP	Standing size class	1876	17.3	82.7	Covington and Sackett 1986
Woolsey Plots, Arizona	Basalt	PP/PO	Dendro-reconstruction	1874	10.2-18.8	81.2-89.8	Sánchez Meador and others 2011
Rainbow Plateau, Arizona	Limestone	PP/PO	Dendro-reconstruction	1879	48.3	51.7	Fulé and others 2002a
Cheesman Lake, Colorado	Granitic	PP/DMC	FVS[b]-reconstruction	1900	12.9-21.5	78.5-87.1	Fornwalt and others 2002

[a]Openness is the proportion of area not covered by tree crowns, estimated as the inverse of canopy cover.
[b]Forest Vegetation Simulator

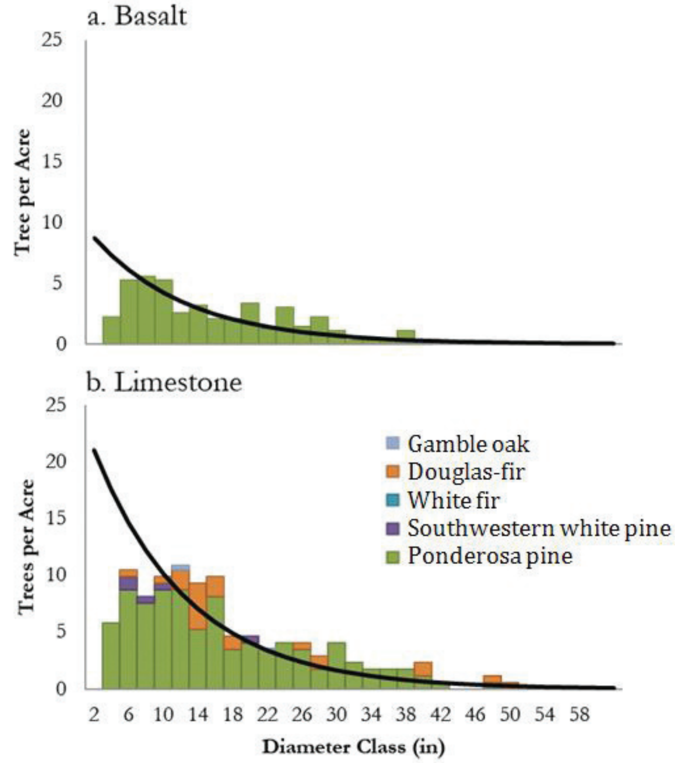

Figure 10. Theoretical diameter distributions representing reference conditions illustrating a superimposed basal area-diameter distribution (BDq) (where q = 1.2); (a) pure ponderosa pine present on basalt soils, (b) dry mixed-conifer on limestone soils. Seedling and sapling-sized tree distribution (i.e., trees in the 2-inch DBH class) on both sites may not be fully represented.

where the authors suggested that contemporary conditions were statistically similar to historical reference conditions as determined by basal area comparisons. A reference condition study conducted in ponderosa pine near Cheeseman Lake, Colorado, reported a range of 12.9-21.5 percent canopy cover (Fornwalt and others 2002). Overall, the range of canopy cover for ponderosa pine for these studies was about 10-50 percent, giving reference conditions for openness (i.e., inverse of canopy cover) of 50-90 percent. If areas with strong tree aggregation (i.e., with interlocking crowns; Fig. 11) exhibit lower mid-scale canopy cover (10.2-21.9 percent; Table 7), then it stands to reason that sites with less tree aggregation would have higher mid-scale canopy cover due to tree arrangement and reduced crown overlap (Christopher and Goodburn 2008).

Snags, logs, and woody debris are important structural and functional elements in frequent-fire forests (Figs. 12 and 13), yet little is known about volumes of coarse woody debris under historical fire regimes. Nonetheless, studies using extensive, historical stem-maps and/or locations of historical evidences (e.g., logs,

Figure 11. Interlocking or nearly interlocking crowns are components of groups of mid-aged to old trees.

Figure 12. Snags, logs, and woody debris are important components of frequent-fire forests. They provide structural diversity, nutrient cycling, and wildlife habitat.

stumps, and snags) reported a mean of 2.3 snags and 2.7 logs per acre (Moore and others 2004), 1-8 snags and 3-23 logs per acre (Sánchez Meador and others 2010), and 10 snags and 20 logs per acre as reference conditions for southwestern ponderosa pine (Abella 2008). These densities suggest that the distribution and

Figure 13. Litter, logs, and coarse woody debris contribute to fire spread and intensity. Old logs also provide local evidence of historical forest composition and structure. The excessive quantity of litter is a result of the lack of fire in this frequent-fire forest.

abundance of snags and logs varied with site and likely coincided with the type, severity, and scale of historical disturbance.

Dry Mixed-Conifer

Mixed conifer forests can be divided into two subtypes: a warm-dry (dry mixed-conifer) type and a cool-moist (wet mixed-conifer) type. Dry mixed-conifer forests are similar to ponderosa pine forests in general stand structure, but Douglas fir, white fir, white pines, and, occasionally, blue spruce are also important components of these forests (Fig. 14). Wet mixed-conifer forests typically lack ponderosa pine, have a greater abundance of Douglas-fir and white fir, and, on some sites, include other fire-intolerant and shade-tolerant species such as blue spruce, subalpine/corkbark fir, and Engelmann spruce (Fig. 2). Dry mixed-conifer forests typically occupy the lower, warmer, and drier end of the elevation zone occupied by mixed-conifer forests. They intergrade with the cool/moist ponderosa pine types on warmer/drier sites at the lower end of the mixed-conifer zone and with wet mixed-conifer forests on cooler/moister sites at the upper end of the zone (Korb and others 2013; Romme and others 2009; Smith and others 2008).

Dry mixed-conifer forests intergrade with or are adjacent to pure ponderosa pine forests and experience similar site conditions and ecological disturbances (types and frequencies) (Grissino-Mayer and others 1995). Romme and others (2009) suggested that the stand structure of dry mixed-conifer was maintained in part by recurrent fires of relatively low to moderate

Figure 14. Groups of dry mixed-conifer are similar to groups in ponderosa pine forests but often have more diverse assemblages of species and higher tree densities.

severity, although small areas of higher-severity crown fire were likely. While only a few studies report the extent of mixed-severity fires (Romme and others 2009), Fulé and others (2009) found no areas of high-severity fire larger than 158 acres as inferred by the current extent and presence of even-aged structures or early seral species.

Dry mixed-conifer forests occur on relatively warm sites at lower elevations or on southerly aspects at higher elevations and are characterized by historical frequent surface-fires synchronized by climate (approximately 9-30 years) (Brown and others 2001; Brown and Wu 2005; Fulé and others 2003, 2009; Grissino-Mayer and others 2004; Heinlein and others 2005). In contrast, wet mixed-conifer is typified by mixed-severity fire regime (Fulé and others 2003). Many studies based on fire-scarred trees show that dry mixed-conifer forests had frequent but variable fire return intervals. Some studies report fire return intervals that were similar to ponderosa pine, as frequently as every 4-14 years (Brown and others 2001; Touchan and others 1996; reviewed in Evans and others 2011), whereas other dry mixed-conifer forests experienced fires as infrequently as every 18-32 years (Fulé and others 2003; Korb and others 2013; Touchan and others 1996; reviewed in Evans and others 2011). A recent study in Southwestern Colorado warm/dry mixed conifer forests found a mean fire return interval ranging from 9-30 years on three different sites at similar latitude and elevation. Korb and others (2013) also showed significant influence of local site factors (e.g., topography, forest structure, and species composition) on fire frequency and severity. Departures from historical compositions, structures, and spatial patterns are likely greater on the warmer/drier than the cooler/wetter portion of the mixed-conifer environmental gradient due to a more severe disruption of the characteristic fire regime (Fulé and others 2002).

When direct evidence of historical fire regime is lacking (i.e., fire scars not present), plant associations that classify seral and climax species composition relative to the shade and fire tolerance of tree species and biophysical site conditions may assist in making inferences regarding fire regimes (see Tables 2 and 8). Openings in dry mixed-conifer include grasses, forbs, shrubs, ferns, and cacti (Korb and Springer 2003), but the specific assemblage of understory plants varies greatly by plant association, being broadly grouped as dominated by bunchgrasses or by forbs/shrubs (Table 8). Bunchgrass-dominated plant associations in dry mixed-conifer forests generally occur in warmer/drier conditions than sites dominated by forbs and shrub understories (e.g. white fir/Arizona fescue [warm/dry] compared to white fir/forest fleabane [cool/moist]; Table 8). For example, the U.S. Forest Service, Southwestern Region utilizes plant association classifications for mapping the spatial extent of dry and wet mixed-conifer forests on National Forest Lands.

Dry Mixed-Conifer: Species Composition: Due to a predominance of frequent, low-severity fire, historical species composition in dry mixed-conifer forests was dominated by fire-resistant, shade-intolerant conifers such as ponderosa pine, Southwestern white pine, and Douglas-fir (Fig. 2) (Evans and others 2011; Fulé and others 2003). Dry mixed-conifer forests occur in environments that are wet enough to support trees such as white fir and aspen. However, these species are also more susceptible to death from fire than fire-resistant pines and Douglas-fir (Fig. 2) (Evans and others 2011; Fulé and others 2003). Consequently, species composition in dry mixed-conifer forests was historically regulated by the balance between climate and disturbance agents such as fire. Periods of frequent fire in mixed-conifer gave fire-resistant species a competitive advantage, allowing them to establish dominance. During "fire-free" or less frequent-fire periods, ponderosa pine persisted

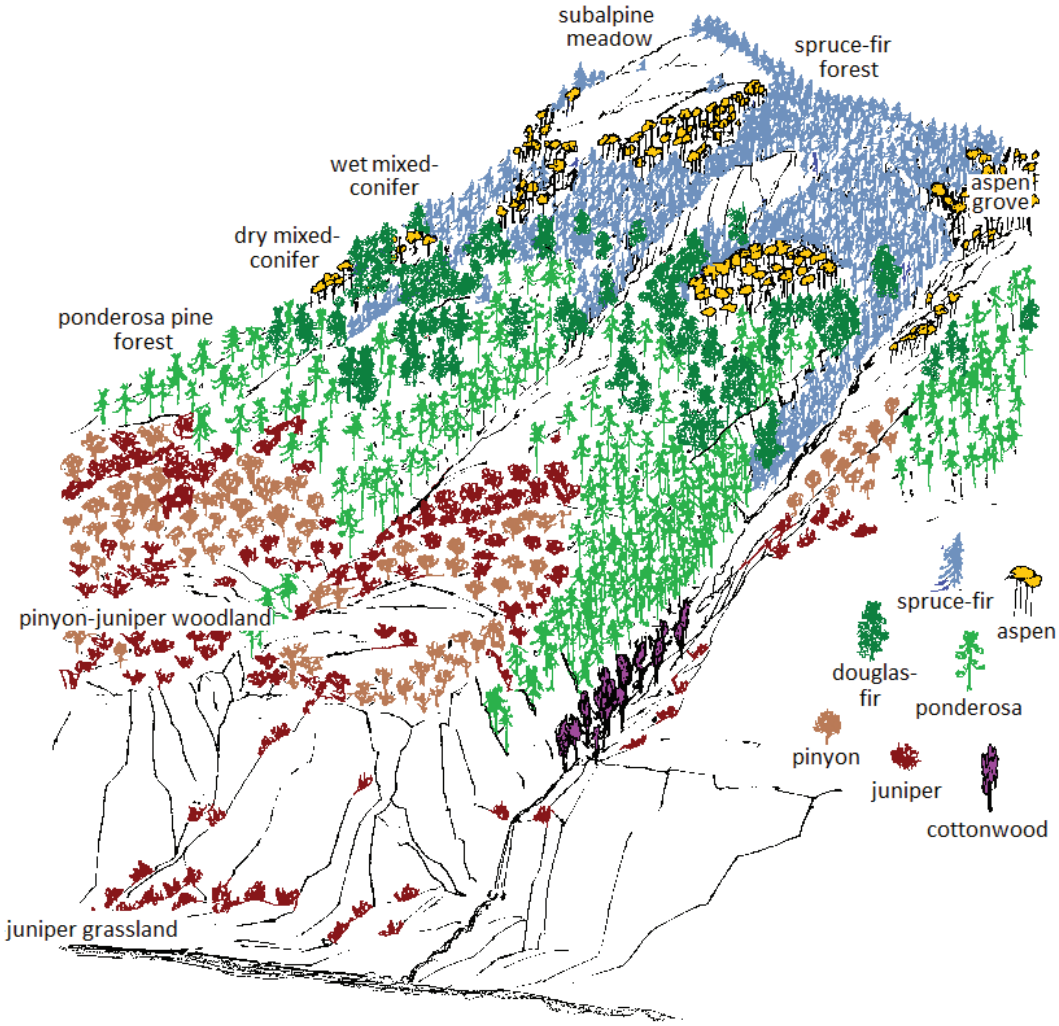

Figure 15. Illustration of changes in forest type by elevation and aspect (adapted from LANL 2011).

due to its dominant positions in the forest canopy (Fulé and others 2009). As a result, shade-tolerant, less fire-resistant species were historically minor components on drier sites, such as ridge tops and southwest-facing slopes, and likely more frequent on cooler and/or more mesic sites in frequent-fire forests, such as drainages and north-facing slopes (Fig. 15) (Romme and others 2009).

Compared to early 1900s Southwestern forest inventories, the current species composition of dry mixed-conifer forests has shifted toward more shade-tolerant, less fire-resistant species (Fulé and others 2009; Johnson 1994; Romme and others 2009). For example, one study in northern Arizona found that ponderosa pine represented an average 64 percent of basal area in the 1870 forest (range 54-69 percent) but only 36 percent in the same forest in 2003 (range 27-46 percent) (Fulé and others 2009). A recent study in Southwestern Colorado found that species composition prior to the last fire record on two different sites (1861 and 1878) was dominated by ponderosa pine, but white fir and Douglas-fir increased in dominance since the cessation of fire (Korb and others 2013). Other studies similarly concluded that extended fire exclusion in dry mixed-conifer forests resulted in substantial increases in stand-level tree density, especially by shade-tolerant white fir and Douglas-fir (Fulé and others 2004; Heinlein and others 2005). These increases resulted in forests with greater homogeneity in species composition across landscapes (Cocke and others 2005; White and Vankat 1993). Furthermore, early selective logging of ponderosa pine and intensive grazing exacerbated the compositional shift toward mesic species (Cocke and others 2005). The combination of fire exclusion, grazing, selective logging, and favorable climatic conditions for young tree establishment in the early 20th Century has created atypical stand compositions and structures in many of today's dry mixed-conifer forests

Table 8. Plant associations for dry mixed-conifer forests in the Southwestern United States sorted by plant association series, dry mixed-conifer subtype, temperature-moisture gradient, dominant season of precipitation, and parent material type (USDA Forest Service 1997).

Plant association (common name)	Dry mixed-conifer subtype	Temperature-moisture gradient[a]	Climate[b]	Dominant season of precipitation[c]	Parent material type[d]
Douglas-fir plant association series					
Douglas-fir/Arizona fescue	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/Arizona fescue/bristlecone pine	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/Arizona fescue/limber pine	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/Arizona fescue/quaking aspen	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/fringed brome	Bunchgrass	Cool-wet	Cold	Winter	Variable
Douglas-fir/Gambel oak/Arizona fescue	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/Gambel oak/screwleaf muhly	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/mountain muhly/limber pine	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/mountain muhly/two needle pinyon	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/screwleaf muhly	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir (scree)	Forb-shrub	Variable	Variable	Summer or winter	Igneous
Douglas-fir/creeping barberry	Forb-shrub	Warm-dry	Cold	Winter	Variable
Douglas-fir/Gambel oak	Forb-shrub	Warm-dry	Cold	Winter	Variable
Douglas-fir/Gambel oak/rockspirea	Forb-shrub	Warm-dry	Cold	Winter	Variable
Douglas-fir/kinnikinnik	Forb-shrub	Warm-dry	Cold	Winter	Rhy./tuff
Douglas-fir/mountain ninebark	Forb-shrub	Warm-dry	Cold	Winter	Variable
Douglas-fir/silverleaf oak/Chihuahua pine	Forb-shrub	Warm-dry	Mild	Summer	Variable
Douglas-fir/silverleaf oak/netleaf oak	Forb-shrub	Warm-dry	Mild	Summer	Rhy./tuff
Douglas-fir/silverleaf oak/ponderosa pine	Forb-shrub	Warm-dry	Mild	Summer	Variable
Douglas-fir/Arizona white oak	Forb-shrub	Warm-dry	Mild	Summer	Variable
Douglas-fir/bigtooth maple	Forb-shrub	Typic	Cold	Summer	Sed.
Douglas-fir/rockspirea	Forb-shrub	Warm-dry	Cold	Winter	Variable
Douglas-fir/wavyleak oak	Forb-shrub	Warm-dry	Cold	Summer	Sed.

Table 8. *Continued.*

Plant association (common name)	Dry mixed-conifer subtype	Temperature-moisture gradient[a]	Climate[b]	Dominant season of precipitation[c]	Parent material type[d]
White fir plant association series					
White fir/Arizona fescue	Bunchgrass	Cool-wet	Cold	Winter	Variable
White fir/Arizona fescue/Gambel oak	Bunchgrass	Cool-wet	Cold	Winter	Variable
White fir/Arizona fescue/muttongrass	Bunchgrass	Cool-wet	Cold	Winter	Variable
White fir/screwleaf muhly	Bunchgrass	Cool-wet	Cold	Summer	Variable
White fir/Arizona walnut	Forb-shrub	Cool-wet	Cold	Winter	Alluvium
White fir/creeping barberry	Forb-shrub	Typic	Cold	Winter	Variable
White fir/creeping barberry/common juniper	Forb-shrub	Typic	Cold	Winter	Variable
White fir/creeping barberry/New Mexico locust	Forb-shrub	Typic	Cold	Winter	Variable
White fir/Gambel oak	Forb-shrub	Typic	Cold	Winter	Variable
White fir/Gambel oak/Arizona fescue	Forb-shrub	Typic	Cold	Winter	Variable
White fir/Gambel oak/pine muhly	Forb-shrub	Typic	Cold	Winter	Cong./tuff/and.
White fir/Gambel oak/rockspirea	Forb-shrub	Typic	Cold	Winter	Variable
White fir/Gambel oak/screwleaf muhly	Forb-shrub	Typic	Cold	Winter	Variable
White fir/kinnikinnik	Forb-shrub	Typic	Cold	Winter	Rhyolite
White fir/mountain snowberry/limber pine	Forb-shrub	Cool-wet	Cold	Winter	Variable
White fir/mountain snowberry/ponderosa pine	Forb-shrub	Typic	Cold	Winter	Variable
White fir/Nevada pea	Forb-shrub	Typic	Cold	Winter	Variable
White fir/New Mexico locust	Forb-shrub	Typic	Cold	Winter	Variable
White fir/New Mexico locust/dryspike sedge	Forb-shrub	Typic	Cold	Winter	Variable
Blue spruce plant association series					
Blue spruce/Arizona fescue	Bunchgrass	Typic	Cold	Winter	Rhy./basalt
Blue spruce/dryspike sedge	Bunchgrass	Typic	Cold	Winter	Rhy./basalt

[a]Gradient based on all mixed conifer forests (dry and wet mixed-conifer types). Typic refers to modal, mid-gradient temperature-moisture types.
[b]Climate refers to mean annual soil temperatures, with cold climates having frigid soils (mean annual soil temperatures <8 ˚C) and mild climates having mesic soils (mean annual soil temperatures >8 ˚C).
[c]Dominant season of precipitation refers to the 6-month period (winter = October-March, summer = April-September) that typically has higher average precipitation levels. Most ponderosa pine and dry mixed-conifer sites in the Southwestern United States receive bimodal precipitation, but the season listed in the table experiences higher average precipitation levels.
[d]Variable = multiple parent materials; sed. = sedimentary; rhy.= rhyolites; cong. = conglomerate; and. = andesite

(Moore and others 2004). In many locations, large, dominant ponderosa pine and Douglas-fir trees have been reduced to few or none, leaving today's stands dominated by young ponderosa pine, Douglas-fir, and white fir (Fulé and others 2003).

Dry mixed-conifer plant associations are highly variable and reflective of local biophysical site conditions that influence the type of disturbances and vegetation responses to disturbances (Table 8) (USDA Forest Service 1997). These plant associations are classified by forest series representing the most shade-tolerant conifer species that can establish and grow on a given site, absent disturbance. However, ponderosa pine typically dominates the species mix in dry mixed-conifer forest series under the characteristic fire regime. Dry mixed-conifer forest series include: (1) Douglas-fir, (2) white fir, and (3) those blue spruce plant associations that do not classify as wet mixed-conifer. These series can be subdivided by understory plant composition into the general subtypes of bunchgrass and forb-shrub. The most mesic dry mixed-conifer sites are the forb-shrub plant associations, and the most xeric are the bunchgrass plant associations. These subtypes differ in their relative fire frequencies; bunchgrass understories support more frequent surface fire, while forb-shrub understories facilitate less frequent surface fire and greater fuel accumulation (Anderson 1982; LANDFIRE 2007; Scott and Burgan 2005; USDA Forest Service 1997).

Dry Mixed-Conifer: Forest Structure: Compared to ponderosa pine, there is considerably less literature on fine-scale forest structure and spatial pattern reference conditions in dry mixed-conifer forests. However, there are some historical references to similarities between structure and spatial pattern of these two forest types. Due to its frequent fire regime, the historical fine-scale structure and spatial pattern of dry mixed-conifer forests were similar to ponderosa pine in having a more open structure (Muldavin and Tonne 2003; Swetnam and Baisan 1996) and a similar aggregated arrangement of trees in some stands (Binkley and others 2008; Sánchez Meador and others unpublished data, see Table 3 footnote). Lang and Stewart (1910; p. 19) noted that "evidence indicates light ground fires over practically the whole forest, some of the finest stands of yellow pine show only slight charring of the bark and very little damage to poles and undergrowth." Dutton (1882) observed that within both the ponderosa pine and mixed ponderosa pine/Douglas-fir forest types "the trees are large and noble in aspect and stand widely apart, except in the highest parts of the [Kaibab] plateau where the spruces predominate. Instead of dense thickets where we are shut in by impenetrable foliage, we can look far beyond and see the tree trunks vanishing away like an infinite colonnade." These observations are consistent with statements that "pure ponderosa pine forests and warm-dry mixed conifer forests were affected primarily by frequent, low-severity fires that maintained an open stand structure with a broad range of tree sizes and ages" (Romme and others 2009).

Empirical evidence also indicates that historical dry mixed-conifer forests had lower tree densities and a more open structure comprised of a higher proportion of old and large trees, were more spatially heterogeneous (having groups and patches of trees), and were more uneven-aged compared to current conditions (Fig.16) (Binkley and others 2008; Fulé and others 2002a, 2003, 2009; Heinlein and others 2005; Moore and others 2004). However, as mixed conifer forests transition toward cooler and wetter site conditions, less frequent and more severe fires result in mixtures of even- and uneven-aged forest structures. At the landscape scale, wet mixed-conifer forests were historically more spatially heterogeneous than ponderosa pine forests because of a mixed-severity fire regime affected by topography, soils, land use, and vegetation (Binkley and others 2008; Fulé and others 2002a, 2003, 2009; Muldavin and Tonne 2003; Smith 2006a; Romme and others 2009; Touchan and others 1996). Variable forest structures and spatial patterns across landscapes

Figure 16. Aerial photo of a dry mixed-conifer forest on a north-facing slope in the Cibola National Forest. In this stand, about 60-70 percent of the area is under mid- to old-age tree cover and 30-40 percent is in grass-forb-shrub interspaces.

resulted, in part, from variation among sites on the temperature/moisture continuum and their species compositions, successional status, and disturbance regimes. Warm, dry mixed-conifer sites likely experienced more frequent and less severe surface fire, resulting in more open forests with a mixture of small tree groups and areas with random tree spatial patterns. In contrast, cool, moist sites experienced mixed or high-severity fires at longer fire return intervals, resulting in relatively closed forests with tree cohorts distributed in larger patches (Fig. 14) (Fulé and others 2003; Romme and others 2009).

Studies of reference conditions for dry mixed-conifer forests reported mean tree densities and basal areas similar to those in ponderosa pine stands but with slight increases at the fine scale (Table 9; Fig. 17). For example, pre-Euro-American settlement dry mixed-conifer forests on limestone soils ranged between 36 and 100 trees per acre and 34 and 124 square ft of basal area per acre on sites in Arizona and New Mexico, respectively (Table 9; Fig. 10). For dry mixed-conifer forests on the Uncompahgre Plateau in Colorado, Binkley and others (2008) reported reference conditions for canopy cover ranging from 12.0-21.5 percent in areas that exhibit fine-scale aggregation; openness was therefore 78.5-88.0 percent in these areas. Fornwalt and others (2002) modeled reference canopy cover conditions of 13-22 percent (78-87 percent openness) for forests with fine-scale tree aggregation on the Colorado Front Range (Table 7). Based on reported studies, historical dry mixed-conifer forests were structurally similar to ponderosa pine with respect to tree groups with small meadows between them (Binkley and others 2008).

Abundance of snags, logs, and woody debris in dry mixed-conifer was likely similar to or slightly greater than that of ponderosa pine. Moore and others (2004) reported 4.9-34.9 snags per acre for dry mixed-conifer reference conditions as determined from extensive, historical stem-maps and relocation of historical evidences (e.g., logs, stumps, and snags). While the historical amount of these structural elements in dry mixed-conifer has received little attention, contemporary studies suggest that more productive dry mixed-conifer sites had higher fuel loads than ponderosa pine sites (Brown and others 2003; Graham and others 1994).

Despite the above similarities, dry mixed-conifer forests occur on a diverse range of sites and have more diversity in species composition, structure (Fig. 17), spatial pattern, processes (i.e., fire regimes and other disturbances), and functions than ponderosa pine forests. While studies demonstrate considerable similarity between dry mixed-conifer and ponderosa pine disturbance processes and forest structures, we point again to the limited numbers and geographical locations of studies of historic structural conditions in dry mixed-conifer and call for additional research to increase our understanding of historical ranges of conditions for these forests (see Monitoring, Adaptive Management, and Research Needs).

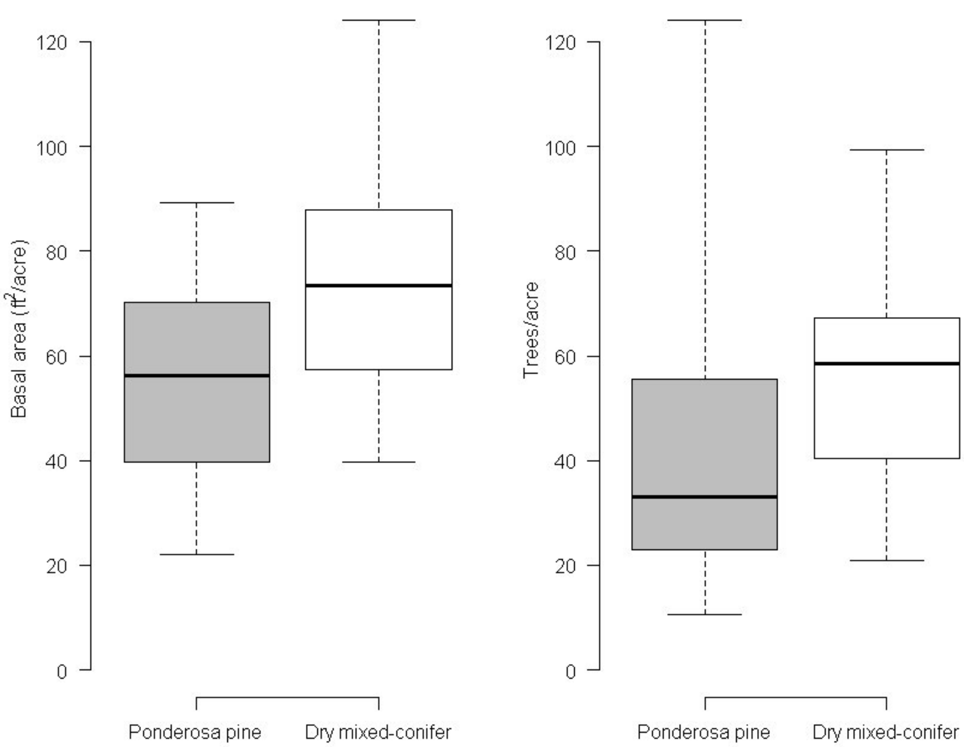

Figure 17. Distribution of reference conditions reported in Tables 6 and 9 for basal area and trees per acre in ponderosa pine and dry mixed-conifer forests. Lines bisecting boxes represent median values; lower and upper borders of boxes represent first and third quartile values; and whiskers (i.e., endpoints of dashed lines) represent maximum and minimum reported values.

Table 9. Historical forest structure of dry mixed-conifer forests of the Southwest, arranged by parent material and average tree density.

Location	Parent material	Elevation (ft)	Size/age reported	Reference date	Trees per acre			Basal area (ft^2/acre)			Citation
					Range	Mean	Std Err	Range	Mean	Std Err	
San Francisco Peaks-East, Arizona	Basalt	8318	Age	1892		20.9	3.4		39.6	3.9	Heinlein and others 2005
San Francisco Peaks-West, Arizona	Basalt	8318	Age	1876		21.0	1.7		54.0	6.1	Heinlein and others 2005
Sitgreaves National Forest, Arizona (max)[a]	Basalt	6300	Size	1910		31.0			66.9		Woolsey 1911
San Francisco Peaks, Arizona	Basalt	9200	Age	1876		65.1	6.8		77.9	12.8	Cocke and others 2005
Blue and White Mountains, Arizona[b]	Basalt	8950	Size	1912		68.7			84.4		Greenamyre 1913
Middle Mountain, Colorado	Granitic	8520	Size	1870	51-59	57.3	4.0	43-60	47.9	4.6	Fulé and others 2009
Jemez, New Mexico (max)[a]	Limestone	7013	Size	1910		35.6			91.2		Woolsey 1911
Kaibab Plateau, New Mexico[c]	Limestone	[illegible]	Size	1909		45.3			60.7		Lang and Stewart 1910
Alamo, New Mexico (max)[a]	Limestone	[illegible]	Size	1910		46.5			97.9		Woolsey 1911
Gila, New Mexico[a]	Limestone	9055	Age/Size	1890		65.6					Moore and others 2004
Jemez, New Mexico[a]	Limestone	[illegible]	Age/Size	1890	66-112	88.8	23.2				Moore and others 2004
Little Park, Arizona	Limestone	8640	Age	1880		98.3	5.8		76.7	9.1	Fulé and others 2003
Swamp Ridge, Arizona	Limestone	8143	Age	1879	36-151	99.4	5.2	65-235	124.1	7.8	Fulé and others 2002a
Black Mesa, Arizona[c]	Sedimentary	Mixed	No	1890		58.4	27.3				Williams and Baker 2011; 2012
Uncompahgre Plateau, Colorado	Shale	8000	Size	1875	30-110	60		25-130	70		Binkley and others 2008

[a]Minimum tree DBH recorded = 3.5in.
[b]Minimum tree DBH recorded = 4 in.
[c]Minimum tree DBH recorded = 6 in

The Restoration Framework

Here, we describe our framework for restoring resiliency to frequent-fire forests in the Southwest. We first provide an overview of our framework, including its ecological foundation, its key elements, and the sources of its science base. We then discuss the spatial and temporal scales at which forest structures are described, and follow this with a description of the desired key compositional and structural elements of a restored forest at those scales for ponderosa pine and then dry mixed-conifer forests. Finally, we provide recommendations for implementing the framework in these forests and finish with brief before and after descriptions of the composition and structure in a ponderosa pine area in New Mexico where we implemented our framework.

The framework is organized around key compositional and structural elements at three spatial scales and is based on a synthesis of reference conditions, literature on the ecology of frequent-fire forests (Table 4) (see Science Review: Forest Ecology), our understanding of the ecology of these forests, decades of collective experience of forest managers and researchers (e.g., Schubert 1974), and lessons learned during applications of our framework in Southwestern frequent-fire forests. Our framework is informed by the ranges of mean forest characteristics from reference conditions research plots, which were typically <10 acres and therefore best describe variability at the fine scale (Tables 3, 6, 7, and 9). Means across plots are more representative of mid-scale conditions than means reported for individual sample plots. Therefore, we point out that any point estimates with a range of mean values may not be appropriate for a given site and we recommend using local, site-specific biophysical conditions and historical evidences to inform specific treatments.

Forest ecology, historical (reference) conditions, and the natural range of variability are frequently used to define restoration goals, to estimate the restoration potential of sites, and to evaluate the success of restoration efforts. Natural range of variability is useful for understanding the natural variability in composition, structure, processes, and functions among sites and for understanding the dynamic nature of ecosystems. It is also a useful reference for establishing limits of acceptable change for ecosystem components and processes (Morgan and others 1994). Our framework is not intended to re-create specific reference conditions. Rather, the framework identifies key elements that characterized frequent-fire forests before industrial logging and the disruption of historical disturbance regimes. These key compositional and structural elements are:

(1) species composition (tree and understory vegetation);

(2) groups of trees;

(3) scattered individual trees;

(4) open grass-forb-shrub interspaces;

(5) snags, logs, and woody debris; and

(6) variation in arrangements of these elements in space and time.

The key elements provide inferences about species compositions, structural conditions, and the cumulative effects of disturbances on processes and functions that provide frequent-fire forests with resistance and resilience to disturbance.

Citations supporting our restoration framework appear mostly in the Science Review: Forest Ecology section but in other sections as needed. We recognize the limited number and geographic extent of scientific studies of reference conditions for ponderosa pine and especially for dry mixed-conifer, not only in the Southwest but throughout the western United States. Nonetheless, our framework is timely because of the growth in knowledge over the past decades regarding current and historical ecology of these forests. It is also timely because of increased frequencies, intensities, and extents of uncharacteristic disturbances, which may worsen under climate change (Littell and others 2009; Millar and others 2007; Miller and others 2009; Westerling and others 2006). We believe that moving current forest conditions toward their characteristic compositions, structures, and spatial patterns will increase their resistance and resilience to future disturbances and will result in outcomes as varied as fire fuels reduction, restoration of wildlife habitats, biodiversity, diverse food webs, and increased ability of these forests to provide ecosystem services.

Spatial and Temporal Scales

Ecosystems are structured hierarchically and their composition, structure, process, and function are temporally and spatially dynamic. Therefore, we characterize the key compositional and structural elements at three spatial scales: the fine-scale (<10 acres), mid-scale (10-1000 acres), and landscape-scale (1000-10,000+ acres) (Fig. 1). These scales generally correspond with structural features in frequent-fire forests. For example, the fine scale is an area in which the species composition, age, structure, and spatial distribution of trees (single and grouped), and open grass-forb-shrub interspaces are expressed. Aggregates of fine-scale units comprise mid-scale units, which are referred to as patches (i.e., stands) and are relatively homogeneous in vegetation composition and structure that differ from their surroundings. The landscape scale is composed of aggregates of mid-scale units and usually has variable elevations, slopes, aspects, soil types, plant associations, disturbance processes, and land uses. Understanding and incorporating temporal scales (seasonal, annual, decadal, and centuries) in a restoration framework is required to sustain vegetation dynamics of a forest that result from growth, succession, senescence, and the natural and anthropogenic disturbances that periodically reset the dynamics.

Key Elements by Forest Type: Ponderosa Pine

Southwest ponderosa pine forests occur at elevations ranging from approximately 5000-9000 ft and typically intergrade with woodland types on warm/dry sites (typically at lower elevations) and mixed-conifer types on cool/moist sites (typically at higher elevations). The characteristic fire regime for ponderosa pine is frequent, low-severity fires (Fire Regime 1; Table 2). Surface fuels (fine fuels, branches, and coarse woody debris) and small trees facilitate this fire regime. Fires burn primarily on the forest floor and rarely spread to tree crowns and canopies. Individual trees or tree groups may occasionally torch during fires. Based on plant associations, a system for classifying plant communities on their potential climax species compositions (Table 5) (USDA Forest Service 1997), we differentiated four Southwestern ponderosa pine forests subtypes: (1) ponderosa pine-bunchgrass, (2) ponderosa pine-Gambel oak, (3) ponderosa pine-evergreen oak, and (4) ponderosa pine-shrub (Appendix 2).

Ponderosa Pine: Fine-Scale Elements (<10 acres):

Species composition: Overstories are dominated by ponderosa pine but may occasionally contain other conifer or hardwood species. Herbaceous understories are typically grasses and forbs at the mid-point within the temperature/moisture gradient over which ponderosa pine occurs. At the warm/dry end of the gradient, ponderosa pine forest intergrades with pinyon-juniper or evergreen oak woodlands (e.g., juniper, pinyon, Emory oak, Arizona white oak, silverleaf oak, and grey oak) with a shrub component (e.g., manzanita, shrub live oak, sumac, or mountain mahogany). In the cool/moist portion of the gradient, Gambel oak is often a component of ponderosa pine forests, and grass and forb understories may include shrubs (e.g., ceanothus, and currants) (Table 5). At the cool/moist end of the gradient, ponderosa pine intergrades with dry mixed-conifer forests where there may be a minor presence of quaking aspen, Douglas-fir, Southwestern white pine, white fir, and blue spruce. Variation in overstory species composition influences forest structure, disturbance types and intensities, tree mortality rates, and the composition and structure of the grass-forb-shrub community.

- Trees typically occur in irregularly shaped, small groups with interlocking or nearly interlocking crowns when in the mid- to old-aged structures (Fig. 11), range in size from 2-72 trees, and occupy between 0.003 and 0.72 acres each (Table 3; Fig. 4). Groups can be even- or uneven-aged. Size, shape, number of trees per group, and number of groups per area are variable (see Science Review: Forest Ecology). If trees are aggregated (i.e., grouped), more productive sites will have more trees per group, and if not aggregated, will support more individual trees per acre. Where groups are even-aged, a high level of interspersion of groups of differing ages constitutes the desired uneven-aged structure at the fine- and mid-scale. Trees within groups are variably spaced with some tight clumps.
- Where reference conditions show the presence of scattered individual trees, their ages are variable (young to old) and they can comprise 15-70 percent of total stand basal area, with the remaining stand basal area being trees in groups (Table 3). Variability in number of individual trees is associated with various factors, such as soils, plant associations, climate, and disturbances.
- Grass-forb-shrub interspaces surround tree groups and individual trees (Fig. 8) and are variably shaped and sized.
- Snags, top-killed, lightning- and fire-scarred trees, and coarse woody debris (logs and other dead woody material greater than 3 inches in diameter) are generally large in diameter and height, scattered throughout the mid-scale, and concentrated in past disturbance sites in abundances of 1-10 snags and 3-10 tons per acre (Figs. 12 and 13). Overall, snags, logs, and coarse woody debris are spatially and temporally variable.

Figure 18. Uneven-aged forest comprised of an interspersion of tree groups of different ages.

Ponderosa Pine: Mid-Scale Elements (10-1000 acres):

The mid-scale is an aggregate of fine-scale units (i.e., tree groups, scattered individual trees, and grass-forb-shrub interspaces) and is collectively referred to as a patch or stand. Mid-scale patches are relatively homogeneous in vegetation composition and structure and differ from surrounding patches.

- Tree species composition is relatively homogenous within patches and is a function of disturbance, time since disturbance, tree density and size/age structure, topography, soils, local climate, site history, ecological legacy, and stochasticity (e.g., mass seeding and weather events).
- Average total tree densities and basal areas generally range from 11-124 trees per acre and 22-90 square ft of basal area per acre (Table 6).
- More productive sites may have more trees per area. Aggregates of many randomly distributed trees (areas >10 acres) function as patches.
- For sustainability and biodiversity purposes, it is desirable that patches comprise uneven-aged forests with an approximate balance of age classes ranging from young to old (Fig. 18). Infrequently, patches of even-aged forest structure may be present.
- All age classes of appropriate hardwood species (e.g., Gambel and evergreen oaks and other hardwoods) are present depending on a site's plant association (Table 5).
- "Openness" (estimated as the inverse of canopy cover) ranges from 52-90 percent. In areas exhibiting fine-scale aggregation of trees, mid-scale openness is typically high (78-90 percent; Table 7), and on more productive sites, especially where tree arrangement is random, openness may be less (see discussion of openness in Science Review: Forest Ecology).

Ponderosa Pine: Landscape-Scale Elements (1000-10,000+ acres):

- The landscape scale is an aggregate of mid-scale units and includes areas with variable topography (i.e., elevation, slope, and aspect), soils, plant associations, disturbance types, and land use legacies. The landscape is a functioning ecosystem that contains all components, processes, and functions that result from characteristic disturbances, including snags, downed logs, and old trees.
- Old-growth structural features occur throughout the landscape as tree groups or single trees within uneven-aged patches (stands) or occassionally as small even-aged patches. Old-growth structural features include old trees, snags, downed wood (coarse woody debris), and horizontal and vertical structural diversity in a grass-forb-shrub matrix (Table 10; Fig. 9). The location of old-growth structural features may shift on the landscape over time as a result of succession and disturbance.

Table 10. Essential structural features of old growth in frequent-fire forests. Note that whether or not a feature is essential may depend on scale—fine-, mid-, and landscape-scale. For example, age variability is possible at all scales, but snags and large dead and downed fuels may not exist in some groups and patches (adapted from Kaufmann and others 2007).

Structural feature	Essential structural feature?	Comment
Large trees	No	Tree size depends on species and site characteristics (moisture, soils, and competition). Young trees may be large, and old trees may be small.
Old trees	Yes	Trees develop unique structural characteristics when old (e.g., dead tops, flattened crowns, branching characteristics, bark color and texture).
Age variability	No	An important feature in some old-growth forest types. Some forests regenerate episodically (even-aged) with most trees establishing in a few years to a decade, probably in conjunction with wet years and large seed crops and in concurrence with relatively long intervals between fires. Others may regenerate over decades (uneven-aged).
Snags and large dead and downed fuels	Yes	Snags and large logs are essential for old growth, but forests with more frequent fires may have fewer logs. Densities and sizes of snags and logs vary depending on forest type, precipitation, and other factors. Snags, logs, and woody debris typically distributed unevenly in landscapes.
Between-patch structural variability	Yes	High variability is a critical feature. Within-patch variability may be low, but variation among patches may be high. Proportions of patches with different developmental stages vary depending on forest type, climate, etc.

- Plant associations vary across environmental gradients (e.g., changes in slope, aspect, climate, and soil type) and reflect their historical species composition, structure, and spatial aggregations.
- Denser tree conditions may exist as patches in locations such as north-facing slopes and canyon bottoms.
- Natural and anthropogenic disturbances such as fire or tree thinning treatments are sufficient to maintain desired overall species composition, tree density, age structures, snags, coarse woody debris, and nutrient cycling.

Key Elements by Forest Type: Dry Mixed-Conifer

Southwest dry mixed-conifer forests generally occur at elevations ranging from 5500-9500 ft. At lower elevations within this range, dry mixed-conifer forests commonly occur on north-facing slopes or canyon bottoms and ponderosa pine forests on south-facing slopes and ridgetops. At the upper elevation range, dry mixed-conifer forests typically occupy south and west slopes, with wetter forest types (e.g., wet mixed-conifer) on north aspects. Dry mixed-conifer forests are dominated by shade-intolerant trees such as ponderosa pine, Douglas-fir, Southwestern white pine, limber pine, quaking aspen, and other hardwoods, with a lesser presence of shade-tolerant species such as white fir and blue spruce depending on biophysical site conditions and the frequency of low-severity fire. Aspen may occur individually or in groups of variable size. While less is known about historical conditions in dry mixed-conifer than in ponderosa pine, available information shows a similarity in the structure and spatial pattern of these two forest types.

Characteristic fire regimes for Southwestern dry mixed-conifer are frequent low-severity fires (Fire Regime 1) with infrequent mixed-severity fires (Fire Regime 3; Table 2) operating at all spatial scales. Surface fuels and small trees facilitate this fire regime. While fires burn primarily on the forest floor, occasionally individual trees or tree groups may torch. Crown fires rarely spread from tree group to tree group.

Dry Mixed-Conifer: Fine-Scale Elements (<10 acres)

- Species composition: Overstories are dominated by fire-resistant, shade-intolerant trees such as ponderosa pine, Douglas-fir, Southwestern white pine, and limber pine, with occasional inclusion of aspen and other hardwoods. Shade-tolerant conifers, such as white fir and blue spruce, may be present but are subdominant in abundance. At the warm/dry end of the temperature/moisture gradient occupied by dry mixed-conifer types, this forest type intergrades with ponderosa pine-bunchgrass and ponderosa pine-Gambel oak subtypes. At the cool/moist end of the gradient, dry mixed-conifer intergrades with the wet mixed-conifer type typified by a mixed-severity fire regime. Differences in overstory species composition influences structure (tree density, tree group size, number of individuals, regeneration), disturbance events (species-specific insect and diseases, fuel type and quantity), distribution of snags and coarse woody debris, and species composition of the grass-forb-shrub community.

- Where dry mixed-conifer forests occur at the warmer/drier end of the environmental gradient (Fig. 2), trees typically occur in irregularly shaped groups, trees within groups are variably spaced, and group sizes generally range from a few trees up to an acre (Fig. 14), similar to ponderosa pine forest types. Reference conditions show tree group sizes ranging from 0.01-0.33 acres (Table 3) (see Science Review: Forest Ecology). Trees within groups are of similar or variable ages and groups are composed of one or more species. Crowns of trees within the mid-aged to old groups are interlocking or nearly interlocking (Fig. 11). Size, shape, number of trees per group, and numbers of groups per area are variable (see Science Review: Forest Ecology). If aggregated, more productive sites will have more trees per group, or if not aggregated will support more trees per acre.
- No data are available on the proportion of stand basal area in individual trees verses tree groups. More research is needed (see Monitoring, Adaptive Management, and Research Needs).
- Grass-forb-shrub interspaces surround tree groups and individual trees (Figs. 14 and 16) and are variably shaped and sized.
- Snags, top-killed, lightning- and fire-scarred trees, logs, and coarse woody debris (>3 inches diameter) are generally large in height and diameter, scattered throughout, and concentrated at past disturbance events in abundances of 5-35 snags and 8-16 tons per acre (see Science Review: Forest Ecology). Overall, snags, logs, and coarse woody debris are spatially and temporally variable.

Dry Mixed-Conifer: Mid-Scale Elements (10-1000 acres)

- The mid-scale is an aggregate of fine-scale units (i.e., tree groups, scattered individual trees, and grass-forb-shrub interspaces) and is collectively referred to as a patch or stand. At the mid-scale, patches can be relatively homogeneous in vegetation composition and structure and differ from surrounding patches. Vegetation is typically characterized by variation in the sizes and numbers of tree groups and the density and extent of patches of trees, each typically varying by elevation, soil type, aspect, and site productivity. Occasionally, patches may be composed of randomly arranged trees.
- In general, tree densities range from 20-100 trees per acre and 40-125 square ft basal area per acre (Table 9) (see Science Review: Forest Ecology). Stand density is likely to increase as site conditions transition toward the cooler/moister end of the environmental gradient for dry mixed-conifer forests and on more productive soil types.
- For sustainability and biodiversity purposes, it is desirable that patches have an uneven-aged forest structure with an approximate balance of age classes ranging from young to old. Infrequently, patches of even-aged forest structure may be present.
- Species composition may be variable within patches and is a function of disturbance, tree density, tree size and age structure, topography, soil, local climate, site history, ecological legacy, and stochasticity (e.g., weather events, mass seeding).
- It is desirable that all age classes of appropriate hardwood species (e.g., aspen, Gambel oak, and maple) are present depending on a site's plant association (Table 8).
- "Openness" is similar to ponderosa pine at the warmer/drier end of the environmental gradient occupied by dry mixed-conifer forests (Table 7) but is likely to decrease from the warmer/drier site conditions to the cooler/wetter end of the environmental gradient due to moister conditions, higher productivity, and less frequent low-severity fire.

Dry Mixed-Conifer: Landscape-Scale Elements (1000-10,000+ acres)

- The landscape scale is an aggregate of mid-scale units and includes areas with variable topography, soils, plant associations, disturbance types, and land use legacies. The landscape is a functioning ecosystem that contains all its components, processes, and functions that result from characteristic disturbances, including snags, downed logs, and old trees.
- Old-growth structural features occur throughout the landscape as tree groups or single trees within uneven-aged patches (stands) or occassionally as small even-aged patches. Old-growth structural features

include old trees, dead trees (snags), downed wood (coarse woody debris), and horizontal and vertical structural diversity in a grass-forb-shrub matrix (Table 10). The location of old-growth may shift on the landscape over time as a result of succession and disturbance (tree growth and mortality).

- Plant associations vary across environmental gradients (e.g., changes in slope, aspect, climate, and soil type) and reflect their historical species composition, structure, and spatial aggregations.
- Denser tree conditions may exist as patches in some locations such as north-facing slopes and canyon bottoms.
- Natural and anthropogenic disturbances such as fire or tree thinning treatments are sufficient to maintain desired overall species composition, tree density, age structures, snags, coarse woody debris, and nutrient cycling.

Implementation Recommendations

Here, we offer recommendations for implementing our framework. These were developed from our understanding of the body of forest ecology and management literature (see Science Review: Forest Ecology), our research and management experience, and lessons learned during implementations of our restoration framework. At the end of this section we present an overview of a case study on the implementation of our framework that illustrates its success in moving current forest conditions toward uneven-aged forest mosaics comprised mostly of fire-adapted species; tree groups; scattered individual trees; grass-forb-shrub interspaces; snags, logs, woody debris; and the spatial arrangement of these elements.

Classification of Site Variability

Ecological classification of a site indicates its biological capabilities regarding species composition, structure, processes, and functions. Ecological classification is useful for implementing our restoration framework because classification depends on variability of local climate, soil, vegetation, geology and geomorphology, and a site's characteristic disturbances and vegetation responses (USDA Forest Service 1997). The variability within and among sites across landscapes is the basis for describing the range of variation in forest conditions in our restoration framework. Recognition of within- and among-site variability is paramount for developing localized restoration objectives. Example classification systems include the U.S. Forest Service Terrestrial Ecosystem Unit Inventory (Winthers and others 2005), which classifies land units by soil, climate, slope, geology, geomorphology, and plant associations, and NatureServe's Ecological Systems (Comer and others 2003). The biotic and abiotic variables used in these classification systems describe a site's biophysical characteristics.

Recommendations by Key Elements

Species Composition

- Manage for percent species composition as indicated by local historical evidence (live trees and snags and logs from trees that originated prior to 1880), biophysical site conditions, and other management objectives (e.g., favoring scarce species; preserving genetic diversity; enhancing wildlife habitat; resilience to climate change; or achieving other resource objectives, social values, and regulatory requirements).

Tree Groups and Individual Trees

- Use a site's historical spatial patterns to inform restoration targets and treatments. Where information on reference conditions is not available, fine-scale spatial patterns may be informed by reference data in Table 3, 6, 7 and 9 and combined with local historical evidence (see Friederici 2004) such as grouped and individual old trees, large logs, and stumps, and a site's biophysical conditions.
- Evaluate current conditions in relation to desired conditions to develop management prescriptions. Avoid arbitrary constraints such as diameter limits for tree cutting (see Abella and others 2006; Triepke and others 2011).
- Where spatial heterogeneity is desired, consider combinations of burns, intermediate and free thinning, and individual tree or small group selection cutting methods to create a heterogeneous structure of groups, single trees, and grass-forb-shrub interspaces. Once heterogeneity is established, consider maintaining the desired structure and spatial pattern with fire and/or single tree and small group selection.
- Where trees are spatially aggregated, maintain interlocking or nearly interlocking crowns in mature and old groups and provide for variable tree spacing within groups; avoid thinning old tree groups.
- Manage young tree groups to create future variable tree spacing and interlocking crowns. Thin young tree-groups to facilitate development of desired within-group characteristics (e.g., variable tree spacing and interlocking or nearly interlocking crowns) in mid- to old-aged tree groups.
- Tree groups generally are small (2-72 trees per group, see Science Review: Forest Ecology) (Fig. 4). Use historical evidence and biophysical capabilities to determine a site's mean and range (minimum, maximum) of trees per group and numbers and spacing of tree groups per area.
- Mid-scale patches (stands) of less-aggregated or randomly arranged trees may be appropriate where

historical evidences do not exhibit spatial aggregation or for achieving other resource objectives.

- Where appropriate, retain or regenerate scattered individual trees between groups.
- Use historical evidence, biophysical site conditions, plant associations, and current conditions (e.g., competition from brush on certain plant association types, degree of disease or insect infestation) to inform regeneration treatments.
- Where management objectives are to maintain conifer dominance and where post-treatment dominance by shrub understories is undesired (e.g., in some ponderosa pine-evergreen oak, ponderosa pine-shrub, and dry mixed conifer-forb/shrub forest types), consider smaller interspaces to avoid excessive shrub response and increased ladder fuel accumulation.
- Consider temporary deviations from uneven-aged management to even-aged cutting methods to initiate recovery on sites damaged by epidemic (severe) insect or disease infestation or other disturbances.
- Manage fire (wildfire or prescribed) frequency and severity towards achieve desired forest structures, spatial arrangements, regeneration patterns, and fuel consumption objectives.
- Design and place regeneration treatments to favor recruitment of shade-intolerant, fire-resistant species.
- Vary treatment prescriptions (cutting and/or fire) to create a mosaic of groups of trees, scattered single trees, and grass-forb-shrub interspaces.

Grass-Forb-Shrub Interspaces

- The grass-forb-shrub community is the matrix in which tree groups and scattered individual trees are arranged (Fig. 8).
- The size and arrangement of grass-forb-shrub interspaces reflect local site conditions and historical evidence. Where trees are grouped, interspaces may be as wide as 1-2 mature tree heights from nearest drip lines of adjacent tree groups. Binkley and others (2008) reported approximately 150 ft between historic groups of trees in dry mixed-conifer in Southwest Colorado; Pearson (1923) reported 100-150 ft diameter openings (interspaces) between historic tree groups in ponderosa pine forests in northern Arizona.
- Sizes of grass-forb-shrub interspaces are a less useful metric for tree spacing in areas where trees are more randomly spaced (i.e., not aggregated). Use a site's historical vegetation spatial patterns as a guide for restoration.
- Grass-forb-shrub interspaces are generally larger on dry sites. Interspaces provide rooting space to support grouped trees.
- Meadows, grasslands, and other non-forested areas may be present as inclusions in forested landscapes; these areas are not considered interspaces.

Snags, Logs, and Woody Debris

- Manage for the continuous presence of snags, logs, and woody debris, especially large snags in various stages of decay throughout the landscape (Figs. 12 and 13). Frequent fires both recruit and consume these elements.

Arrangement of Key Elements in Space and Time

- Recognize the importance of spatial and temporal heterogeneity in forest composition and structure to ecological processes and functions.
- Where objectives include sustainability of wildlife habitat, biodiversity, and wood products, manage for a balance of age classes from cohort establishment (seedling/saplings) to old forest structure, and for grass-forb-shrub interspaces (Figs. 18 and 19).
- Where threatened, endangered, or other rare species are a concern, alternative composition and structures may be needed.

Management Feasibility

Our key elements focus on the compositional and structural features of frequent-fire forests with the goal of creating opportunities for the resumption of characteristic ecological processes and functions and to re-establish the pattern-process link. In some cases, fire can be used to develop the desired composition and structure, while in other cases, it may be more effective when it follows the restoration of forest composition and structure through mechanical treatments. Some of the recent wildfire events in the Southwest may present opportunities to initiate the post-fire "reset" of composition and structure toward desired conditions through broad-scale application of managed fires. In many Southwestern areas, restoration of frequent-fire forests will be labor intensive and costly. In other areas, implementation, or certain implementation tools, may be constrained by logistic, economic, social considerations, and special land designations (e.g., wilderness and protected areas). For example, degraded conditions in current forests may limit the use of fire. In such areas, mechanical treatments may be necessary before introducing fire. In areas where silvicultural treatments

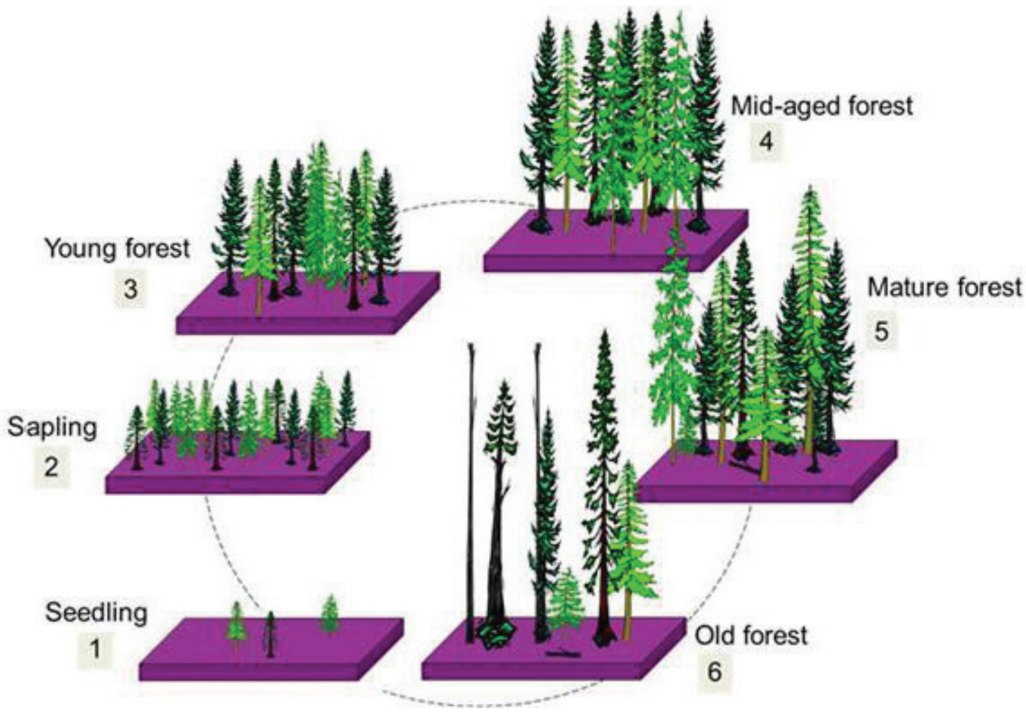

Figure 19. Illustration of the development of tree groups from seedlings to old forest at the fine scale.

are constrained by operational feasibility (e.g., access, slope, or economics) or in wilderness areas, fire may be the only management tool.

- It may not be feasible for management to approximate historical composition and structure patterns and/or fully restore characteristic ecological processes and functions everywhere.
  - o Socio-economic considerations (e.g., smoke, operational capacity, and public safety) may limit the use of fire and prescribed cutting. Some areas may require combinations of treatments to create and maintain desired compositions, structures, processes, and functions.
- Existing conditions influence treatment prescription and choice of tools.
  - o Fire alone can be used where there may be less need for precise outcomes. Fire may result in more variable forest density, numbers, and sizes of groups, and greater distribution of age classes.
  - o Where sustained production of ecosystem services is desired, managing at the extremes of the natural range of variability may be desired. For example:
    - ▪ Higher forest density and a balance of forest structural stages may be desirable to ensure economic sustainability (i.e., to maintain some level of sustained wood products) and for maintaining denser tree habitat conditions for some wildlife species.
    - ▪ Lower forest density and open forest structure may be desirable to facilitate additional reductions in fire hazard and for maintenance of more open habitat for some wildlife species.
  - o Depending on existing conditions, achieving the key elements may require multiple treatments (e.g., prescribed cutting and fire) over long time periods.
- Past disturbances, such as those resulting from fire and insects, may provide early management opportunities (i.e., reforestation and fire management) to put recovering forests on trajectories toward development of key compositional and structural elements.
- Consider strategic placement of restoration treatments to capitalize on the use of wildfire, under appropriate conditions, across broad landscapes.

Implementation of the Framework: Bluewater Demonstration Site

One of several implementations of our restoration framework was on the Cibola National Forest (Bluewater demonstration project) in New Mexico in 2010. Objectives of this project were to:

(1) create resilient forest composition and structure;

(2) move a predominately mid-aged forest toward uneven-aged conditions with an approximate balance of tree age classes;

(3) restore grass-forb-shrub interspaces;

(4) reduce fuels and fire hazard; and

(5) promote wildlife habitat, biodiversity, and wood products.

Our attempt to achieve the key compositional and structural elements in one treatment on the Bluewater site was limited by existing conditions; a portion of the mature and old trees had been harvested in prior treatments, there was little existing regeneration, and the site had a preponderance of mid-aged ponderosa pine trees. A comparison of pre- and post-treatment conditions (Figs. 20 and 21; unit 5A) attests to on-the-ground feasibility and utility of our framework recommendations for restoring the key elements in Southwestern ponderosa pine forests. Details for this project are available from the Forestry Staff with the USDA Forest Service Southwestern Region in Albuquerque, New Mexico.

Pre-Treatment Conditions

The Bluewater demonstration site is a 73-acre ponderosa pine stand (Fig. 22) that contained three different plant associations: ponderosa pine/mountain muhly, ponderosa pine/Arizona fescue, and ponderosa pine/blue grama, all of which are characterized as bunchgrass plant associations. The ponderosa pine site index is 72 for a base age of 100 years (Minor 1964). Soils are moderately productive and variable throughout the unit, comprised of alluvium and residuum from granite, and residuum derived from sandstone and claystone. The climate is temperate, with an average 180-day frost-free growing season from mid-May through mid-September and annual precipitation ranging from 17-25 inches, with greater than half occurring during the growing season.

Sanitation and improvement harvests occurred in the stand in the mid-1980s to remove diseased, dying, and poorly formed trees and, with the exception of piled slash burning in that treatment, the site had not experienced fire since the early 1900s. Prior to treatment, stand density averaged 216 trees and 125 square ft of basal area per acre. The stand was uneven-aged but had a predominance of mid-aged trees (Table 11). Fire behavior modeling demonstrated that 11 percent of the area had potential to support torching and active crown fire under dry conditions (i.e., completely dried fuel) and 15-mile/hr unobstructed wind speed.

Prescription Description

Tree marking occurred in spring 2010, tree cutting occurred in summer 2010, and prescribed burning is scheduled for fall and winter 2013. Treatment prescriptions were developed to produce the composition, structure, and spatial pattern identified in our framework for ponderosa pine: a predominant composition of ponderosa pine; re-establishment of a grass-forb-shrub community in interspaces between trees; groups of trees with interlocking or nearly interlocking crowns in the older age-classes; scattered individual trees; and retention of snags, logs, and woody debris.

The objective was to adjust stocking and spatial arrangement of residual trees (i.e., leave trees) to create or move the forest toward an uneven-aged and aggregated stand structure with a balance of age classes. Treatment prescriptions allowed within-site flexibility in numbers of trees per group and numbers and dispersions of groups per area as informed by historical evidence (i.e., old trees, logs, stumps with establishment date <1880) and existing forest structure. Treatment prescriptions used group selection to create grass-forb-shrub interspaces and regeneration sites and free thinning in immature leave tree groups to develop/retain interspersion of tree groups of different age classes and group sizes. Tree marking crews were instructed not to thin mature and old groups of trees except to remove young trees within these groups to reduce ladder fuel. Our intent was to have about 40 percent of the forested area occupied by mature-to-old tree groups, both of which meet old-growth objectives.

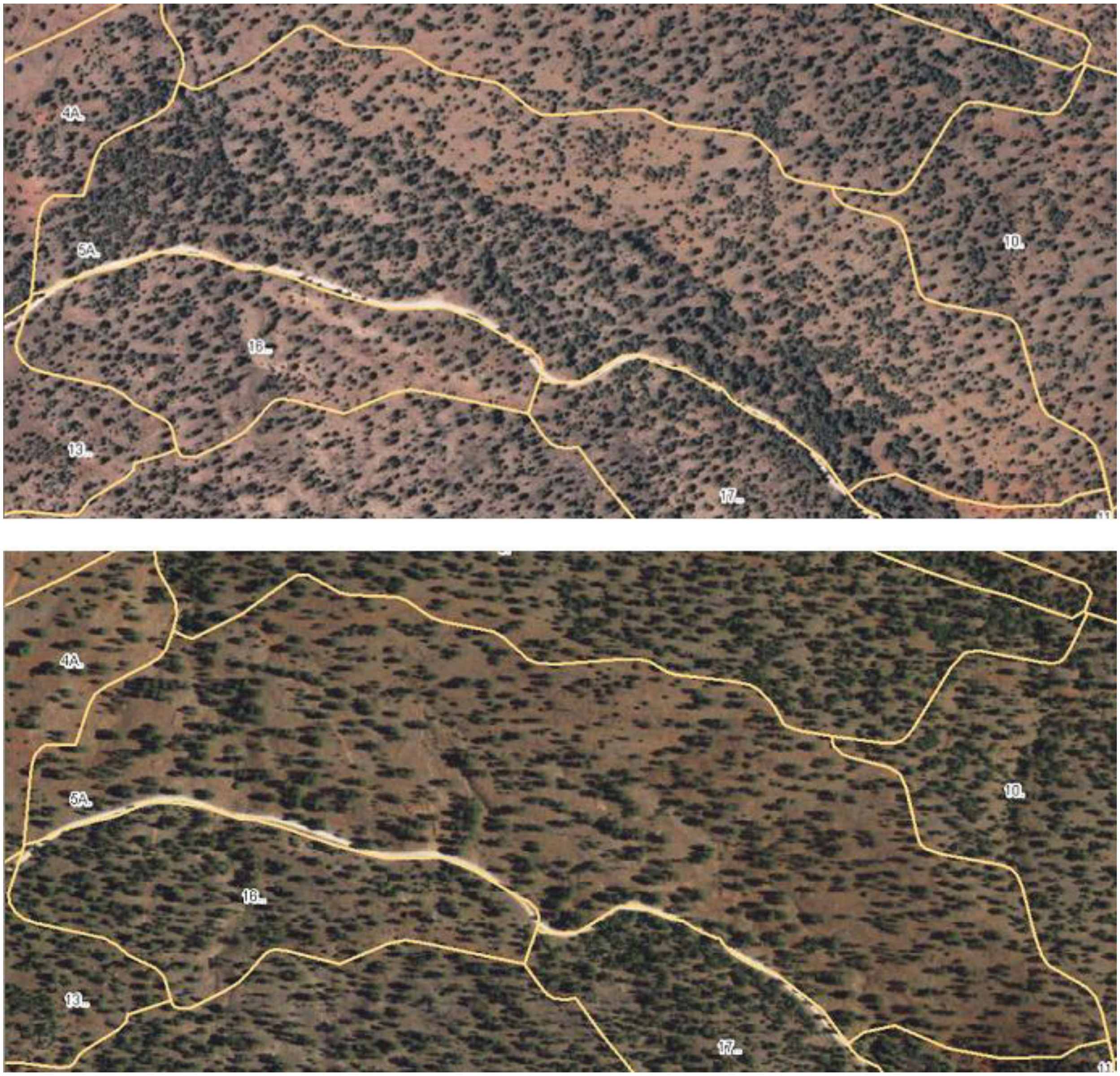

Figure 20. Aerial views of unit 5a on the Bluewater demonstration site in the Cibola National Forest, New Mexico. Prior to treatment (top image), forest density was substantially greater and more spatially homogenous than after the 2010 restoration treatment (bottom image) that applied the principles of our restoration framework.

Objectives were to favor retention of Southwestern white pine, ponderosa pine, and Douglas-fir; maintain minor components of pinyon pine and some juniper species; and favor Gambel oak and Rocky Mountain juniper trees for wildlife habitat. Leave-tree marking identified tree groups and single trees for retention. Leave trees were selected based on tree vigor and ages, with the objective of retaining an approximate balance of age classes. Special emphasis was also placed on within-group structure, including the retention of subdominant, dead-topped, and lightning-struck trees for wildlife habitat. Because no snags were present on the site, trees with declining vigor were retained for snag recruitment. Leave tree groups were either a single size or a blend of variably-sized trees. Trees within young groups were selected to encourage the development of future interlocking crowns. Overly dense young tree groups were thinned to facilitate vigor and future crown growth. Leave tree groups were generally 0.25-0.75 acres, but groups as small as a few trees and as large as 2 acres were also desired. After an initial training period, the marking crew successfully created the desired pattern of groups, scattered single trees, and grass-forb-shrub interspaces. However, they tended to mark numerous small-sized groups instead of a range of group sizes. To establish group size variability, we revisited the treatment area and added trees to some groups.

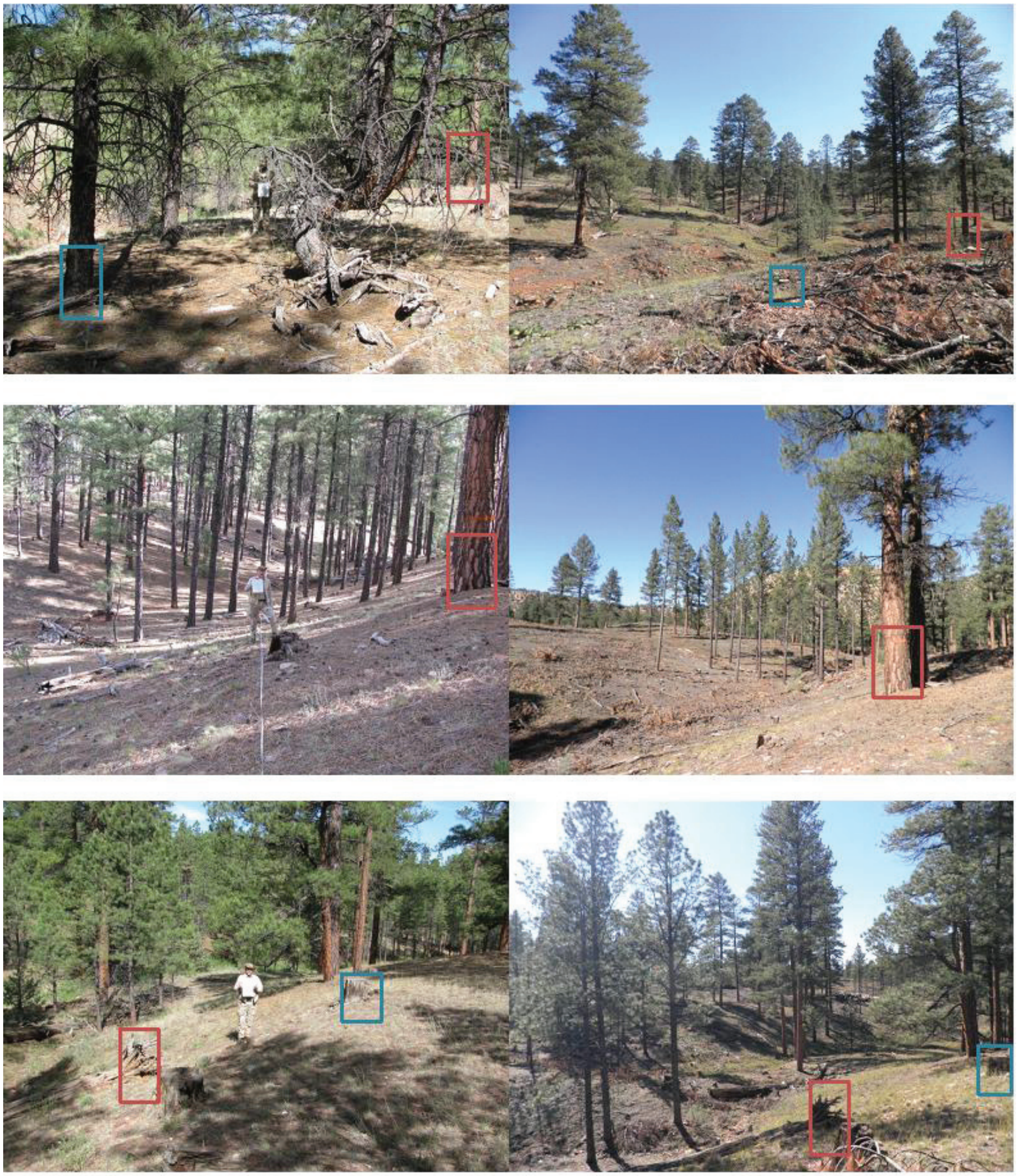

Figure 21. Paired photos from the same point before (left) and after (right) treatment in the Bluewater demonstration site, Cibola National Forest, New Mexico, USA. Colored boxes identify the same trees, cut stumps, or logs in before and after photos.

Interspaces between tree groups were created to provide for grass-forb-shrub vegetation and areas for root development. Desired interspace distances between leave groups ranged from 20-100 ft (drip line to drip line), with most distances ranging from 50-70 ft. To remedy a deficit of seedlings and saplings, regeneration sites ranging from 0.33-1.0 acre were created.

Treatment prescriptions specified the desired abundance of snags, logs, and woody debris: averages of 2 snags per acre with diameter at breast height (dbh) >12 inches and 3 downed logs per acre with dbh >12 inches. Where existing snag density was less than 2-3 per acre, live trees with broken tops or defects or fading green trees were retained for future snag and log recruitment.

The northern goshawk, tassel-eared squirrel, and Merriam's turkey were given special consideration. The treatment prescription was consistent with the restoration of habitats of plants and animals in the northern goshawk's food web (Reynolds and others

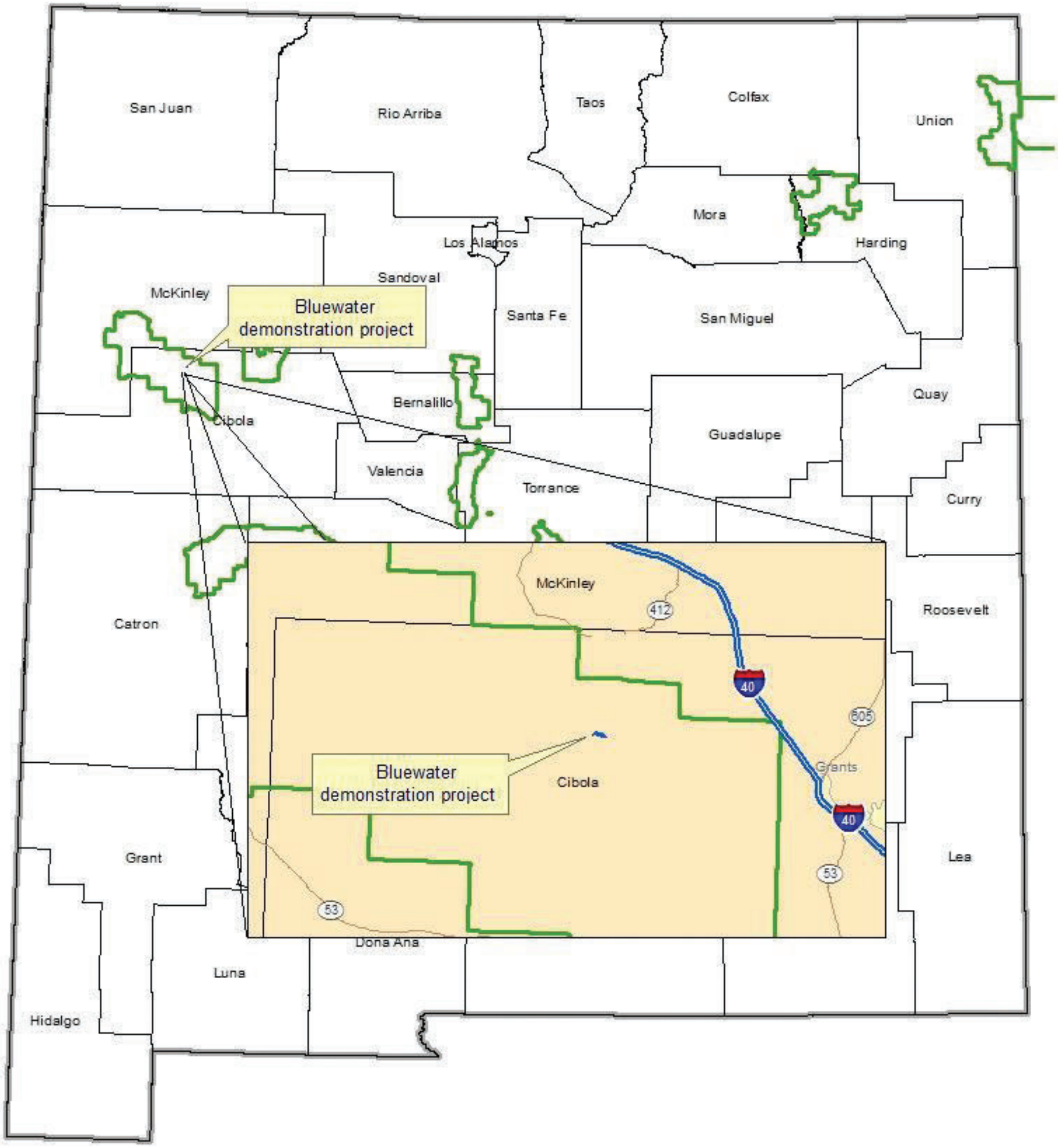

Figure 22. Location of the Bluewater demonstration project (108.45555° W, 38.45461° N) on the Cibola National Forest (green outline) in New Mexico, USA.

Table 11. Estimated proportion of stand area represented by different tree ages and sizes pre- and post-treatment on the Bluewater demonstration site.

Tree structural classes		Proportion of stand area under tree canopy	
Tree age[a]	dbh[b] range (inches)	Pre-treatment conditions	Post-treatment conditions
Seedling/sapling	0-4.9	5%	22%
Young	5-11.9	35%	26%
Mid-aged	12-17.9	40%	32%
Mature	18-23.9	10%	10%
Old	>24	10%	10%

[a]Tree ages are assumed to be related to sizes of dominant /co-dominant trees
[b]dbh = diameter at breast height

1992, 2006a, 2006b), including older tree groups with interlocking crowns for tassel-eared squirrels (Dodd and others 2003, 2006; Reynolds and others 1992) and high interspersion of grass-forb-shrub interspaces (foraging and brood habitat), closed-canopied tree groups (nesting and hiding cover), and large, old trees (roosting habitat) for Merriam's turkey (Hoffman and others 1993; Porter 1992).

Post-Treatment Conditions

This restoration treatment succeeded in creating the key compositional and structural elements identified in our framework (Figs. 21 and 23). The treatment retained the uneven-aged structure in the stand, increased the degree of interspersion of age classes, and is on a trajectory toward an approximate balance of age classes. The stand still had fewer seedling-saplings and mature and old trees than desired due to deficits in pre-treatment conditions (Tables 11 and 12). Approximately 28 percent of the area in the post-treatment stand was under the crowns of mid- to old-aged trees and 72 percent was open with no tree cover (Fig. 20). Approximately 20 percent of the post-treatment open area is designated for future tree recruitment, which will result in a desired 52 percent openness and 48 percent under tree cover. Open interspaces between tree groups were created for grass-forb-shrub communities and fire-safe sites were created for tree regeneration (Fig. 23). Post-treatment stand densities averaged 57 trees and about 40-80 square ft of basal area per acre. Most leave trees were arranged in groups with interlocking crowns, but scattered individual trees were retained across the site. Tree group sizes ranged from a few trees to 0.47 acres based on the area covered by tree crowns estimated from aerial photographs.

The post-treatment composition, structure, and spatial pattern of the stand reduced the risk of crown fire from pre-treatment conditions. Post-treatment FlamMap simulations predicted surface fires across 99 percent of the area and passive crown fire on 1 percent. Post-treatment abundance of small diameter woody debris was higher than intended, but prescribed burning will consume much of this material. Post-treatment abundance of logs and snags was lower than desired; however, these key structural features are expected to accumulate over time and with maintenance treatments. Mechanical treatments moved this forest stand toward restored conditions, but many years and multiple follow-up treatments (fire, mechanical, or combinations of these) will be needed to produce and maintain the desired key elements.

Future Management

Future plans are to broadcast burn the Bluewater site in the fall and winter of 2013 in order to initiate nutrient cycling and maintain fuels at desired levels. Subsequent entries will involve either tree felling, fire, or combinations of these to maintain or enhance the restoration treatment and manage for the desired mix and balance of tree age structures. Post-treatment conditions are being monitored at fixed photo-plots (Fig. 21) to determine whether compositional and structural objectives are being met and to inform future management.

Figure 23. Implementation of our framework in a ponderosa pine forest on the Bluewater demonstration site created groups of trees of a variety of vegetation structural stages (Table 11). The mechanical treatment also created open areas that will support grass-forb-shrub communities and tree regeneration.

Table 12. Post-treatment stocking level for the Bluewater demonstration site. All tree species are included in these estimates.

dbh[a] range (inches)	**Trees/acre**	**Basal area (ft^2/acre)**
1-4.9	3	0.4
5-8.9	17	4.6
9-12.9	23	16.2
13-16.9	5	6.1
17-20.9	5	10.2
21-24.9	2	4.3
>25	2	6.1
	Total 57	47.9

[a]dbh = diameter at breast height

Expected Outcomes of Framework Implementation

Our restoration framework is intended to promote ecosystem resilience by using fire and prescribed cutting treatments to restore the species compositions, structures, and spatial patterns of Southwestern frequent-fire forests. Restoring these features should allow re-establishment of characteristic processes such as disturbance regimes, nutrient cycling, food webs, hydrologic function, and ecosystem services such as biodiversity, old-growth, wood products, aesthetics, and recreation. Restoring characteristic compositions, structures, processes, and functions should also re-establish the evolutionary environment to which plants and animals native to these forests were adapted. Having intact, self-regulating, productive, and adaptive ecosystems is a compelling strategy for allowing species in the ecosystem to adapt to changing environments and facilitate their migration in the face of uncertain climate changes and disturbances. The following description of expected outcomes from restoring forest composition, structure, and spatial pattern in Southwestern frequent-fire forests is intended as an overview of some important outcomes from the restoration of these forests; this overview is not a comprehensive review of the literature. Improved understandings of these and other outcomes will require additional research (see Monitoring, Adaptive Management, and Research Needs).

Ecosystem Resilience to Climate Change

Restoring ecosystem resilience based on historical conditions has been a central concept in ecosystem management (Covington 2003; Folke and others 2004; Scheffer and others 2001). However, the relevance of historical conditions as reference points and targets for restoration has been questioned on the basis of uncertainty of future ecological conditions due to global climate change (Harris and others 2006; Millar and others 2007; Wagner and others 2000). Specific challenges for restoring and sustaining frequent-fire forests in the face of climate change are uncharacteristically rapid alterations of environments and combinations of disturbances and non-native biotic factors producing conditions never before documented in evolutionary time—conditions that may overwhelm characteristic ecological processes (Fulé 2008). In light of these challenges, we review the evolutionary history of these forests.

Over the past several million years, forests and woodlands in the Southwest, including their associated microbial, plant, and animal communities, have tracked favorable habitats and climates whose migrations across geographical and elevational ranges were driven by major climate fluctuations (Bonnicksen 2000; Covington 2003; Delcourt and Delcourt 1988). Since the end of the last major glacial period (14,000 years ago), ponderosa pine returned to the high elevation plateaus and mountains of Arizona about 10,000 years ago and to the central Rocky Mountains only about 5000 years ago (Baker 1986; Covington 2003; Latta and Milton 1999; Millar 1998). In the last 50 million years, frequent-fire forests survived wide swings in environmental conditions (Moore and others 1999). Component species of frequent-fire forests adapted over evolutionary time to arid environments that have been characterized by variable wet and dry periods, including prolonged droughts, and disturbances such as fire, insects, and diseases. These disturbances varied in frequency, intensity, and extent (Covington and Moore 1994b); served as checks on the demographic rates of component species; and resulted in self-regulating processes of nutrient cycling, productivity, and regeneration (Allen and others 2002; Cooper 1960; Covington and others 1997; Covington and Moore 1994b; Falk 2006).

The highest confidence in future climates is associated with projections that are consistent among climate change models and observed climate changes. Surface temperatures in the Southwest are predicted to increase substantially, with more warming in the summer and fall than in winter and spring; summer heat waves will become longer and hotter, with reductions of late winter/spring mountain snowpack due mostly to warmer temperatures (Overpeck and others 2012). Observed Southwest droughts have been exacerbated by warmer summer temperatures and are projected to become hotter, more severe, and more frequent, suggesting an increased drying in the Southwestern United States and that historical drought levels may become the norm (Overpeck and others 2012; Seager and others 2007). Such droughts will directly increase tree mortality and vulnerability to pathogen attacks (Breshears and others 2005) and enhance the size and severity of wildfires (Fulé 2008). Thus, current conditions in frequent-fire forests (i.e., high stand densities, accumulations of fuels on the forest floor, and encroachment of fire-susceptible species; Cocke and others 2005; Cooper 1960) will increase the susceptibility to stand-killing fire (Fulé 2008). It is also likely that on some sites, fire-caused changes in vegetation (e.g., forest to grasslands or shrublands) may not at all resemble those of historical forests (Barton 2002; Savage and Mast 2005; Strom and Fulé 2007). Predicted changes to warmer climates in the American Southwest are expected to affect forests via geographical shifts in suitable environments for the dominate forest species. Shifts are expected to be to higher elevations and northward (Fulé 2008; Shafer and others 2001).

Uncertainties associated with future climate changes make the development of restoration strategies increasingly complex and challenging. The scenario of future hotter, more severe, and more frequent droughts in the Southwest (see Karl and others 2009) includes increased competition for water and increased frequency and extent of high-severity fire, insect, and disease disturbances. Restoring the characteristic composition, structure, and spatial pattern in frequent-fire forests would thereby:

- reduce tree densities and canopy continuity;
- recreate grass-forb-shrub plant communities;
- reduce competition for space, water, and nutrients (Covington and others 1997); and
- provide for the re-establishment of characteristic disturbance regimes (Covington and others 1997; Fulé and others 2002b; Kolb and others 1998).

Nonetheless, restoration strategies should account for an ecosystem's current condition as they may influence an ecosystem's development under future climate. Alternative successional pathways under future climactic variability may invalidate reference conditions as baselines for restoration (Clewell and others 2005; Pilliod and others 2006).

While climate forecasting remains imperfect, fire predictions for Western North America suggest substantial increases in occurrences, spread, and intensity (Brown and others 2004; Honig and Fulé 2012; McKenzie and others 2004; Spracklen and others 2009). Thus, managing frequent-fire forests toward the historical composition, structure, and spatial pattern is consistent with a reduced vulnerability to catastrophic loss (Allen and others 2002; Falk 2006; Honig and Fulé 2012). While we recognize that uncertainties in how species and communities can and will respond to rapid climate change, we agree with Fulé (2008) that it makes sense to restore fire and fire-related composition, structures, and spatial patterns to enhance resistance to catastrophic loss. Restoring the composition, structure, and spatial pattern of these forests should increase their resistance and resilience to climate changes, thereby providing opportunities for species to migrate or develop local adaptations. In fact, Fulé (2008) suggested a restoration strategy that focuses on mesic areas at higher latitudes and elevations (i.e., upper portions of the ponderosa zone and the transitional dry mixed-conifer zone) where forests are more likely to survive climate change. Fulé (2008) recommended using reference conditions from low and southerly areas to guide management in higher-elevation ecosystems to provide for the migration of species as climate warms.

In summary, both reference conditions and natural range of variability are useful guides for management because Southwest frequent-fire forests were historically resilient to drought, insect pathogens, and severe wildfire. Our restoration framework should therefore increase the resistance (by forestalling impacts), resilience (through improved recovery after disturbance), and response (allowing transitions or migrations to new conditions) of frequent-fire forests to climate change (Millar and others 2007; Parker and others 2000; Price and Neville 2003; Spittlehouse and Stewart 2003).

Disturbance Regimes

Restoring the composition, structure, and spatial patterns of frequent-fire forests will provide for the re-establishment of feedback relationships between pattern and disturbance processes in these forests (Larson and Churchill 2012). Disturbances are temporary changes in environmental conditions that cause changes in ecosystem composition and structure. Restoring the composition and structure of frequent-fire forests will result in a more open forest structure and decrease the potential for epidemic outbreaks of insects and diseases and stand-replacing fire (Fitzgerald 2005; Fulé and others 2002, 2004; Graham and others 2004; Roccaforte and others 2008; Strom and Fulé 2007). The restoration of grass-forb-shrub interspaces and resultant separation of tree canopies will increase herbaceous plant development and provide fuels to carry frequent surface fires. In turn, restoration of characteristic fire regimes should sustain forest composition, structure, processes, and functions. Reduced tree densities result in reduced competition for resources, increased tree vigor, and reduced insect and disease infestations (Hessburg and others 1994; Kolb and others 1998).

The intent of our framework is not to eliminate insects and diseases but to return populations and their effects to an endemic, low background level of tree mortality (Miller and Keen 1960). In areas with higher tree densities that may have escaped repeated surface fire, bark beetles can be a significant agent for shaping forest structure and fine-scale spatial heterogeneity. Increasing the spacing between groups of trees can reduce the continuity of mistletoe occurrence across the landscape and reduce mistletoe spread between groups, creating the opportunity for groups of trees that are free of mistletoe (Hawksworth 1961). Frequent surface fires can elevate tree crown bases and increase tree spatial heterogeneity, both of which can slow mistletoe spread (Conklin and Geils 2008). Frequent surface fire can also reduce the severity of mistletoe infection by killing heavily infected trees (Conklin and Geils 2008; Koonce and Roth 1980).

Nutrient Cycling

A restored fire regime can also improve soil nutrient conditions. Intense heat from fire volatilizes nitrogen from the soil and surface fuels, often causing the total nitrogen concentration of forest soils to decline (Boerner and others 2009; DeLuca and Sala 2006). However, nitrogen concentrations tend to recover and even increase two to four years following fire as soil microbes decompose ash and plant litter (Boerner and others 2009). Fire can also cause an immediate pulse of inorganic nitrogen due to the combustion of organic matter and mortality of soil microbes (DeLuca and Sala 2006). Soil ammonium concentrations in ponderosa pine forests may increase as much as 20-fold following fire followed by dramatic increases in nitrate levels after the first year (Covington and Sackett 1992). Frequent burning can maintain elevated levels of inorganic nitrogen in forest soils by depositing charcoal, which binds to inorganic nitrogen and slows its leaching, and by promoting the establishment of grasses and herbaceous vegetation (DeLuca and Sala 2006; Hart and others 2005). Grasses and herbaceous vegetation produce litter with higher nitrogen-to-carbon ratios than conifer vegetation; thus, the presence of herbaceous vegetation may stimulate decomposition and enhance the availability of inorganic nitrogen in forest soils (Hart and others 2005). Fires also kill large trees, creating snags that ultimately become coarse woody debris that plays an important role in nutrient cycling (Brown and others 2003; Cram and others 2007; Graham and others 1994; Harvey and others 1988; Lowe 2006).

Biodiversity and Food Webs

Many ecosystem processes influence plant productivity, soil fertility, water availability, and other local and global environmental conditions. These processes are often controlled by the diversity and composition of plant, animal, and microbial species native to an ecosystem, and recent studies suggest that losses in biodiversity can alter the magnitude and stability of ecosystem processes (Naeem and others 1999). As a dominant species in frequent-fire forests, ponderosa pine influences the understory vegetation, soils, and plant and animal habitats and communities (Moore and others 1999). A community is a group of organisms that interact and share an environment. Organisms in a community may compete for resources, profit from presence of other organisms, or use other organisms as a food source. In the Southwest, ponderosa pine forests are occupied by over 250 species of vertebrates, invertebrates, soil organisms, and plant species (Allen and others 2002; Patton and Severson 1989), many of which adapted to high levels of the spatial heterogeneity and biodiversity that characterized historical frequent-fire forests. A compositionally and structurally diverse understory provides food and cover for many species of vertebrates and invertebrates, each contributing to ecological functioning and food webs. For example, the dispersion of mycorrhizal fungi, a root symbiont critical to the growth and health of trees, is likely reliant on small mammal transfer via feces (Johnson 1996).

Current frequent-fire forests are uncharacteristically homogeneous in composition and structure with reduced plant and animal habitats and lowered biodiversity (Allen and others 2002; Kalies and others 2012; Laughlin and others 2006; Patton and Severson 1989; Villa-Castillo and Wagner 2002; Waltz and Covington 2003). Achieving our restoration framework's key elements restores habitats at multiple spatial scales, especially through the re-establishment of species-rich grass-forb-shrub communities and the productivity, biodiversity, and trophic interactions they support (Abella 2009; Clary 1975; Kalies and others 2012; Oliver and others 1998; Reynolds and others 1992, 2006a; Rieman and Clayton 1997). Dense tree conditions in current frequent-fire forests favor plants and animals that do better in more close-canopied forests. Restoration to more open forest conditions may result in the decline of these species but should increase abundance of more open forest species (Kalies and others 2012). Nonetheless, because our framework creates a variety of forest age and structural stages, including groups and patches with dense forest structures, declines of denser-forest obligates may be minimized (e.g., tassel-eared squirrel; Dodd and others 2003, 2006; Kalies and others 2012), resulting in higher overall species diversity (Noss and others 2006).

Another concern is that the fine-scale structural heterogeneity of forests resulting from restoration of frequent-fire forests may lower the abundance and viability of large-area-dependent species (e.g., spotted owl; Holthausen and others 1999; Prather and others 2008). These concerns might be ameliorated by developing specific desired conditions for breeding sites (e.g., on denser north slopes) and feeding sites with prey habitats (Prather and others 2008; Reynolds and others 1992). It is worth noting that breeding sites or entire refugia for imperiled species may receive protection from loss by encircling them with restored forests, lowering risk of catastrophic loss through fire or insects (Prather and others 2008). This indicates that restoration of these forests and the habitats they contain may provide for the historical distribution and abundance of plants and animals in Southwestern frequent-fire forests.

Restoration of frequent-fire forests should lead to more robust food webs by re-creating diverse habitats across landscapes. Species diverse and productive grass-forb-shrub communities in interspaces between tree groups support broad-based food webs that many invertebrates, birds, mammals, and their predators depend upon (Abella 2009; Dodd and others 2003; Ganey and others 1992; Kalies and others 2012; Linkhart and others 1998; Reynolds and others 1992, 2006a; Rosenstock 1998). The importance of diverse tree and grass-forb-shrub habitats and robust food webs at multiple spatial scales was demonstrated by temporal variations in the vital rates of northern goshawk (Reynolds and others 1992, 2005, 2006a, 2006b), a sensitive species that has been the subject of extensive research in the Southwest (Beier and Drennan 1997; Beier and others 2008; Boal and Mannan 1994; Ingraldi 2005). In the Southwest, goshawk reproduction typically varied extensively year-to-year and was strongly associated with the abundance and availability of food; in years when prey numbers were low, goshawk population reproduction was a fraction of reproduction in years when prey was abundant (Beier and others 2008; Reynolds and others 2005; Salafsky and others 2005, 2007). Goshawks typically feed on a broad suite of prey—from robins, jays, woodpeckers, doves, and grouse to tree squirrels, ground squirrels, rabbits, and hares, each occupying different habitats (Reynolds and others 1992, 2006a). Annual population highs and lows of each prey species are not always in phase; a year's population low of one or more prey is often compensated by higher abundances of other species (Salafsky and others 2005). Due to compensation, forest management strategies that provide a fine- to mid-scale interspersion of habitats are more likely to successfully maintain an entire suite of prey at higher total abundance through both good and poor prey years in individual goshawk home ranges (Reynolds and others 1992, 2006a). For the goshawk and the many other avian and mammalian predators (e.g., raptors, weasels, bobcats, and coyotes) in Southwestern frequent-fire forests, the grass-forb-shrub prey community is particularly important because it is occupied by a large proportion of the birds and mammals native to these forests as well as many important prey species, including rabbits, grouse, ground squirrels, mice, and voles. Prey species in this vegetation layer had larger body masses than most other species occurring in frequent-fire forests (Reynolds and others 1994; Salafsky and others 2005). Furthermore, several of these species are known to attain high population abundance in response to grass-forb-shrub productivity and biodiversity (Ernest and others 2000; Gross and others 1974; Hernández and others 2011; Hostetler and others 2012; McKay 1974). Others of our framework's key elements also create important habitats in Southwestern frequent-fire forests, including:

- dense groups and patches of older-aged trees with interlocking crowns for tree squirrels and species requiring denser forest conditions;
- snags for woodpecker foraging and nesting;
- snags for secondary-cavity nesters, bark gleaning birds, and hunting and sallying perches;
- logs for many invertebrate species (spiders, ants), woodpeckers, mice, rabbits, ground squirrels, grouse, and wild turkey; and
- woody debris for many small mammals.

Old-Growth

The key elements described in the restoration framework provide and sustain old-growth tree components at all spatial scales. Old-growth components provide a number of ecosystem services—plant and animal habitat, biodiversity, carbon sequestration, hydrologic function, high-quality wood products, aesthetics, and spiritual values. Old-growth structure includes old trees, dead trees (snags), downed wood (coarse woody debris), and structural diversity (Figs. 9, 12, and 13) (Franklin and Spies 1991; Helms 1998; Kaufmann and others 2007). The concept of old-growth includes multiple spatial and temporal scales, ranging from individual trees to tree groups and patches to landscapes and their development overtime. Definitive characteristics of old growth in the Southwest vary by forest type as a consequence of differences in species composition, tree longevities and sizes, and the characteristic types, frequencies, and severities of disturbances (Harmon and others 1986). Old-growth forests in the Southwest have been partitioned into three groups based on different fire regimes and resultant compositional and structural features (Table 10): frequent, low-severity fire; mixed-severity fire; and infrequent, high-severity fire (Table 2).

Old-growth in frequent-fire forests occurs as old trees in groups and as scattered individuals within uneven-aged forests. These forests are less dense and have fewer logs and woody debris than high-severity infrequent-fire forests. Old-growth structural features typically occur at the fine scale (Meyer 1934; Weaver 1951) and are composed of small, old tree groups interspersed with similarly sized groups of younger trees, seedlings to mid-aged (Table 10) (Cooper 1961; Harrod and others 1999; Morgan and others 2002; Pearson 1950; Woolsey 1911). The fine-scale age diversity through growth and development sustained the old-growth tree components. Our framework's key restoration elements in frequent-fire forests include all the essential structural features of old growth distributed throughout the uneven-aged forest (Kaufmann and others 2007).

In contrast to frequent-fire forests, old-growth in forests with a mixed-severity fire regime (Table 2) is characterized by adjacent forest patches burned by either low- or high-severity fire (Fulé and others 2003; Grissino-Mayer and others 1995). This results in landscapes with patches of old-growth intermixed with patches of different forest ages. Under an infrequent, high-severity regime (Table 2), old-growth forests are driven by mid- to landscape-scale, high-severity fire followed by vegetation recovery and succession occurring over long periods between fires. Infrequent, high-severity fire regimes typically have large (>100 acres) patches of forests dominated by large, old trees with multiple canopy layers with similar times since disturbance and vegetation origin dates.

Hydrologic Function

We found no published studies that evaluated the long-term effects of restoration on hydrologic function and water yield in Southwestern frequent-fire forests (see Monitoring, Adaptive Management, and Research Needs). However, studies on the effects of different tree harvest prescriptions on hydrologic function and water yield offer insights into the probable effects of reducing current high tree densities through restoration of frequent-fire forests in the Southwest. Hydrologic function and water yield in forests are greatly influenced by the amount and distribution of vegetation, precipitation, snow melt, basin physiography, and soil type. In dense (92-140 ft^2/acre) ponderosa pine forests, reduction of residual basal area to less than 100 ft^2 per acre resulted in increased water yield, although large variations in yield are typical. In addition, initial mean increases in water yield of 15-45 percent can be realized in ponderosa pine forests on basalt-derived soils when high basal area in current forests is reduced. However, increases can be expected to decline with time as vegetation establishes and develops (Baker 1986; Douglas 1983; Harr 1983; MacDonald and Stednick 2003; Troendle 1983). Removal or reduction of forest cover can increase soil water storage, which then becomes available for groundwater recharge (Baker and others 2003). Soil water content was reported to be higher in thinned and thinned-and-burned areas than in untreated-control areas on basalt soils in northern Arizona. However, observed annual variation in water yield showed that the amount and timing of precipitation had a greater overall effect on water yield than did the removal of trees (Feeney and others 1998).

From the above it seems reasonable that restoring our framework's key elements will benefit hydrologic function by reducing stand density and creating open grass-forb-shrub interspaces, decreasing canopy transpiration and interception losses, concentrating snow in interspaces, and increasing soil infiltration, water storage, and stream and spring flow (Baker 1986; Ffolliott and others 1989). While an objective of increasing water yield may not be a sufficient justification for forest restoration, increases in water yield are a significant incidental benefit (Baker 2003).

Wood Products

The re-establishment of frequent, low-severity fire is critical to the success of our restoration framework. However, because of limitations such as proximity to human developments, air quality restrictions, and workforce capacity, the use of fire will probably continue to be limited. Therefore, mechanical-only treatments, or perhaps combinations of fire and mechanical treatments, are likely to be the restoration tools of choice in much of the Southwestern landscape. Another limitation to restoration is the economic viability of treatments; can treatments generate revenue to fund restoration or must they be subsidized? In the initial stages of forest restoration, an abundant supply of lower-valued wood products could help create local products, industries, and enterprises and generate some revenue. Establishment of small-diameter tree markets, followed by shifts to markets targeting the use of restoration by-products (e.g. traditional and emerging products utilizing a wider range of tree sizes), will be essential to long-term restoration and stable local industries. Yields between 400 and 700 cubic ft per acre seem reasonable from a cutting cycle of 25 to 30 years once restoration achieves an approximate balance of structural stages in frequent-fire forests (Youtz and Vandendrieshe 2012). Such yields would help offset costs of achieving multiple objectives.

Aesthetics and Recreation

The public often judges the ecological health of a forest by appearance. Hill and Daniel (2008) found that acceptance of restoration activities may be contingent on public perceptions of aesthetics and knowledge of ecological benefits. People prefer landscapes with large trees, openings, and varied spatial distribution of vegetation that provide views through the site and into the landscape (Brush 1979). Recreational campers preferred camp-sites that were about 60 percent shaded (James and Cordell 1970), while others preferred uneven-aged forest landscapes over even-aged, dense stands (Brown and Daniel 1984, 1986, 1987; Ryan 2005). Restored forests meet these scenery preferences, suggesting greater public acceptance and support.

Monitoring, Adaptive Management, and Research Needs

Frequent-fire forests in the Southwest are complex and dynamic, and our understanding of how they function and respond to disturbances is limited by available data. Knowledge gaps and unexpected events inevitably make forest management and restoration inherently challenging. Key to meeting restoration challenges are the conduct of ecological monitoring, adaptive management, and additional research. This framework and its application are intended to be dynamic and adaptive and will evolve with accumulations of new monitoring and research information.

Ecological monitoring is the means by which managers evaluate whether the current conditions of an ecological system match, or are on a trajectory to match, some desired condition (Noon 2003). Monitoring provides feedback on the impacts of management treatments (Lindenmayer and Likens 2010; Palmer and Mulder 1999) and is typically divided into three categories: implementation, effectiveness, and validation (Busch and Trexler 2003). Implementation monitoring occurs during implementation and determines whether treatments were carried out as intended. Effectiveness monitoring determines the extent to which treatments achieved their ultimate objectives. Validation monitoring assesses the degree to which underlying assumptions about ecosystem relationships are supported (Block and others 2001; Busch and Trexler 2003) and functions to identify knowledge gaps or research needs.

Adaptive management requires feedback obtained from monitoring regarding the success or failure of treatments (Walters 1986). Adaptive management is the "rigorous approach for learning through deliberately designing and applying management actions as experiments" (Murray and Marmorek 2003). In contrast to simply measuring treatment effects and making slight adjustments to future treatments, adaptive management depends on structured, adaptive decision making (Williams and others 2009). It is most useful when managers and scientists identify threshold values for triggering management actions (Noon 2003). A clear description in a plan of how monitoring will be used in decision-making is essential (Noon 2003; Williams and others 2009). This could be achieved administratively (Mulder and others 1999; Sitko and Hurteau 2010), legally via the National Environmental Policy Act process (Buckley and others 2001), or through collaborative agreements (Gori and Schussman 2005). Monitoring data should be compiled, analyzed, and reported in a timely manner so that managers are provided information to improve decision-making (Mulder and others 1999) and to identify knowledge gaps.

Although much is known about historical forest composition, structure, and disturbance in frequent-fire forests, our knowledge of the mechanisms of spatial pattern formation and maintenance is limited, indicating a research need (Larson and Churchill 2012). A limited understanding of reference conditions on different parent material, especially in dry mixed-conifer, is an important data limitation for designing and implementing appropriate resource management. While the number of reference data sets is increasing, existing data have focused largely on tree density. There is a clear need for studies on spatial patterns and the sizes and shapes of grass-forb-shrub interspaces, as well as the mechanisms for the formation and maintenance of spatial patterns. Additional research needs are:

- Increased understanding of reference conditions and the natural range of variation across ecological gradients such as latitude and longitude, soils, topography, and climate in Southwest frequent-fire forests, especially in dry mixed-conifer.
- Increased understanding of differences between ponderosa pine and dry mixed-conifer forests in reference conditions and the historical types, frequencies, severities of disturbances, and responses of vegetation. Of particular need are:
  (1) A greater understanding of variation of reference conditions (composition, structure, and spatial pattern) in forest subtypes and different plant associations.
  (2) How reference conditions influenced the effects of fire on tree regeneration and mortality in forest subtypes and in the transition zones between subtypes.
  (3) The effectiveness of restoration treatments at achieving desired objectives, especially on avoiding the conversion of these subtypes to alternative plant associations.
- Increased understanding of ecosystem processes and functions as they respond to restoration of the composition and structure of frequent-fire forests.

- Increased understanding of the mechanisms of spatial pattern formation (e.g., aggregated and random tree distributions) within- and among-groups, including the presence, abundance, and dispersion of individual trees.
- An understanding of historical roles of insect and disease in shaping forest composition, structure, and spatial pattern, and the effects of restoration on the frequency and severity of insect and disease disturbances at all scales.
- An understanding of the effects of exotic insect, disease, plant, and animal species, and how these may alter forest composition, structure, processes, and functions.
- Increased understanding of the efficacy of fire versus tree cutting only and cutting combined with fire at achieving the desired composition, structure, processes, and functions in frequent-fire forests at all scales.
- Identification of management strategies for restoring composition and structure in transitional zones between forest types and future directions given climate change.
- Development and refinement of new and existing tools and metrics for measuring spatial heterogeneity at ecologically meaningful scales.
- Improved understanding of wildlife habitat and wildlife uses of restored composition and structure of frequent-fire forests.
- Improved understanding of long-term effects of restoration and maintenance treatments (mechanical, fire, and a combination of the two) on water yield and quality.
- Assessment of ecological, economic, and social benefits and costs (e.g., invasive species) of different restoration methodologies and implementation practices, such as methods for treating slash, tree marking approaches, spatial scales of treatment, and frequency of maintenance treatments.
- Exploration of management applications to implement our framework on broad landscapes in an economically efficient manner.

Summary

Our forest restoration framework provides managers and researchers a review of existing knowledge regarding the historical compositions and structures in Southwest frequent-fire forests and how these operated through feedback mechanisms that sustained their characteristic compositions, structures, and functions. Current forest conditions, the cumulative consequences of various human activities that altered historical conditions, are reviewed in light of historical conditions with a focus on how human-caused changes lowered the resistance and resilience of these forests to historical disturbance agents that themselves have become more intense and frequent. Guided by our understanding of how the composition, structure, and spatial pattern of historical frequent-fire forests affected their resistance, resilience, and responses to disturbances, our restoration framework identifies desired key compositional and structural elements of these forests and provides management recommendations for restoring those key elements. We believe implementation of our framework provides opportunities for re-establishing characteristic processes such as frequent, low-severity fire and ecological functions such as habitat, biodiversity, and food webs.

The key compositional and structural elements of historical frequent-fire ponderosa pine and dry mixed-conifer forests in the Southwest can be envisioned over time as a shifting mosaic of groups of trees with interlocking crowns; single trees; open grass-forb-shrub interspaces; and dispersed snags, logs, woody debris (Larson and Churchill 2012; Long and Smith 2000; Reynolds and others 1992). Research shows that the degrees of tree aggregation; sizes and numbers of tree groups; numbers and dispersion of single trees; sizes and shapes of grass-forb-shrub interspaces; and numbers, sizes, and dispersions of snags, logs, and woody debris in reference conditions varied among sites by soil, topography, climate, disturbance regime, and past stochastic events. Our restoration framework recognizes this site-to-site variability and articulates the importance of restoring that variability by using existing evidence (e.g., old trees, snags, stumps, and logs) and biophysical site indicators as guides for restoring local variability. In our view, restoration of spatial and non-spatial elements of forest structure on a per-site basis is the most practical, science-based strategy to return frequent-fire forest ecosystems in the Southwest to resistant, resilient, and responsive conditions that will best position them to adapt to future disturbance regimes and climates (Larson and Churchill 2012; Millar and others 2007). We intend this framework and its application to be flexible and adaptive (i.e., learn-as-you-go) and to evolve with accumulation of knowledge, and for its conceptual approach to provide a blueprint against which management plans and practices can be evaluated.

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Glossary

Age class is defined as trees that originated within a relatively distinct range of years. Typically, the range of years is considered to fall within 20 percent of the average maturity (e.g., if 100 years is required to reach maturity, then there would be five 20-year age classes) (Helms 1998).

Basal area is the cross-sectional area of all stems of a species or all stems in a stand measured at breast height (4.5 ft above the ground) and expressed per unit of land area.

Biodiversity is the variety and abundance of life forms, processes, functions, and structures of plants, animals, and other living organisms, including the relative complexity of species in communities, gene pools, and ecosystems at spatial scales from local to regional to global (Helms 1998).

Canopy cover (*see* forest canopy cover)

Canopy fuels are all burnable materials, including live and dead foliage, lichen, stems, and branch wood located in the forest canopy.

Characteristic (natural) conditions (e.g., vegetation composition and structure), processes (e.g., disturbance regimes), and functions (e.g., habitat, biodiversity, and food webs) of a forest type that are present under the natural range of variability.

Clump refers to (1) the aggregate of stems issuing from the same root, rhizome system, or stool; or (2) an isolated generally dense group of trees (Helms 1998). A clump is relatively isolated from other clumps or trees within a group of trees, but a standalone clump of trees can function as a tree group or a single structure (Fig. 4).

Coarse woody debris is dead woody material on the ground greater than 3 inches in diameter, including logs (Figs. 12 and 13).

Composition is the array of species present in an ecosystem. In forestry, this term often refers to the proportion of each tree species in a stand expressed as a percentage of the total number, basal area, or volume of all tree species in the stand (Helms 1998).

Diameter at breast height (DBH) is the diameter of a tree typically measured at 4.5 ft above ground level.

Disturbance (characteristic and uncharacteristic): Any relatively discrete event in time that disrupts ecosystems, communities, or population structure and changes resources, substrate availability, or the physical environment (Helms 1998). Characteristic disturbances are those whose extent, frequency, and severity fall within the natural range of variability. Uncharacteristic disturbances are outside the natural range of variability and interrupt characteristic processes and functions.

Dry mixed-conifer forests occupy the warmer and drier sites between elevations of 5000 and 10,000 ft and are characterized by a relatively frequent historic fire regime (<35 years fire return interval), resulting in surface fire and infrequently, mixed-severity fire effects. This forest type is typically dominated by shade-intolerant species such as ponderosa pine, with minor association of aspen, Douglas-fir, and Southwestern white pine during early seral stages. More shade-tolerant conifers such as Douglas-fir, white fir, and blue spruce are dominant at climax stages. In the Southwestern United States, this type is primarily described by the Society of American Foresters cover types interior Douglas-fir and white fir.

Ecological (ecosystem) health (*see* forest health)

Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. Restoration initiates or accelerates ecosystem recovery with respect to its health (productivity), processes, and functions (biodiversity, food webs, and sustainability) (adapted from SER 2004).

Ecosystem integrity is the state or condition of an ecosystem that displays the biodiversity characteristic of the reference, such as species composition and community structure, and is fully capable of sustaining normal ecosystem functioning (SER 2004).

Ecosystem resiliency is the ability of an ecosystem to absorb and recover from disturbances without altering its inherent functions (SER 2004).

Ecosystem services are the benefits people obtain from ecosystems, including provisioning services such as food and water; regulating services such as flood and disease control; cultural services such as spiritual, recreational, and cultural benefits; and supporting services such as nutrient cycling, that maintain the conditions for life on Earth (Millennium Ecosystem Assessment 2005).

Ecosystem stability is the ability of an ecosystem to maintain its given trajectory (SER 2004).

Ecosystem sustainability is the capacity of ecosystems to maintain ecosystem services in perpetuity without degradation of its productivity and function at all scales. For example, in the context of our restoration framework, sustainability results in maintaining the key elements in space and time.

Even-aged forests are forests that are comprised of one or two distinct age classes of trees.

Evolutionary environment refers to the range of abiotic and biotic conditions that have exerted selection pressure on and are critical to the survival of species or groups of species (Kalies and others 2012; Moore and others 1999).

Fine fuels are fast-drying dead or live fuels, generally characterized by a comparatively high surface area-to-volume ratio, that are less than 0.25 inch in diameter and have a time-lag of one hour or less. These fuels (grass, leaves, needles, etc.) ignite readily and are consumed rapidly by fire when dry (NWCG 2012).

Fire regime refers to the patterns of fire occurrences, frequency, size, severity, and sometimes vegetation and fire effects in a given area or ecosystem. A fire regime is a generalization based on fire histories at individual sites (McPherson and others 1990).

Fire return interval is the number of years between two successive fires in a specified area (McPherson and others 1990).

Forest canopy cover is the proportion of ground or water covered by a vertical projection of the outermost perimeter of tree canopies, regardless of tree spatial arrangement.

Forest health is the state or condition of forest ecosystems in which its attributes (i.e., productivity) are expressed within "normal" ranges of activity relative to its ecological stage of development. A restored ecosystem expresses health if it functions normally relative to its reference ecosystem (adapted from SER 2004).

Frameworks provide a set of assumptions, concepts, values, and practices that constitute a way of viewing reality (American Heritage Dictionary 2011).

Free thinning is the removal of trees to control stand spacing and favor desired trees using a combination of thinning criteria without regard to crown position (Helms 1998).

Frequent-fire forests are forests with fire regime 1, those forests with fire frequency <35 years (Schmidt and others 2002).

Functions (ecological functions) are the outcomes of ecosystem components and processes (e.g., interactions within and among species). Examples include primary and secondary production and mutualistic relationships. Ecosystem functions are broadly categorized as regulation functions, habitat functions, production functions (e.g., genetic and medicinal resources), and information functions (e.g., spiritual and historic information) (De Groot and others 2002).

Group refers to a cluster of two or more trees with interlocking or nearly interlocking crowns (Fig. 4 and 12) at maturity surrounded by grass-forb-shrub interspaces (Fig. 8). Size of tree groups is typically variable depending on forest type and site conditions and can range from fractions of an acre (i.e., a two-tree group), such as in ponderosa pine or dry mixed-conifer forests, to many acres, as is common in wet mixed-conifer and spruce fir forests. Trees within groups are typically non-uniformly spaced, some of which may be tightly clumped.

Group cutting (selection) is the removed of small groups of trees to establish of new age classes (Helms 1998).

Improvement harvests involve the removal of poorly formed or low-vigor trees to improve stand productivity and/or quality (Helms 1998).

Interspaces are areas not currently under the vertical projection of the outermost perimeter of tree canopies (Fig. 8). They are generally composed of grass-forb-shrub communities but could also be areas with scattered rock or exposed mineral soil. Interspaces do not include meadows, grasslands, rock outcroppings, and wetlands (i.e., exclusions adjacent to and sometimes within forested landscapes).

Leave trees or snags (*see* residual (leave) trees or snags)

Matrix refers to the background cover type of an area. In frequent-fire forests, grass-forb-shrub communities form the background matrix upon which tree groups and individual trees are spatially arranged. It is the most extensive and connected landscape element that plays the dominant role in landscape functioning. The expression of this matrix between tree groups and individual trees is referred to as interspace. The location of tree groups and individual trees on the matrix and the proportion of patches represented by the matrix will change over time due to disturbance.

Mixed-severity fire regimes are characterized by closely juxtaposed forest patches affected by low- and high-severity burning (Fulé and others 2003).

Natural (historical, characteristic) range of variation describes the variability of ecological conditions (e.g., reference compositional and structural conditions) and the spatial and temporal variation in these conditions during a period of time specified to represent characteristic conditions (i.e., conditions relatively unaffected by people) for an ecosystem in a specific geographical area (Kaufmann and others 1994; Landres and others 1999).

Old growth in Southwestern forested ecosystems is defined differently than the traditional definition based on Northwestern infrequent-fire forests. Due to large differences among Southwest forest types and their characteristic disturbances, old growth forests vary extensively in tree size, age classes, presence and abundance of structural elements, stability, and presence of understory. Important structural features of old growth in frequent-fire forests are large trees, old trees, age variability, snags, large dead and downed fuels, and between-patch structural variability (Fig. 9 and Table 10) (Kaufmann and others 2007).

Openness is estimated as the inverse of forest canopy cover for a given area. For example, a forest with 70 percent canopy cover would have openness of 30 percent.

Patches are areas larger than tree groups in which the vegetation composition and structure are relatively homogeneous (sensu Forman 1995). Patches can be composed of randomly arranged trees or multiple tree groups, and they can be even-aged or uneven-aged. Patches comprise the mid-scale, ranging in size from 10-1000 acres. Patches and stands are roughly synonymous.

Pattern (*see* spatial pattern)

Plant associations are plant community types based on land management potential, successional patterns, and species composition (Helms 1998).

Ponderosa pine forests are widespread in the Southwest occurring at elevations ranging from 6000-7500 ft and occupying warmer and drier sites within the montane forest life zone. These forests are characterized by a relatively frequent historic fire regime resulting in surface fire effects. Ponderosa pine is the dominant tree species in this forest type, but other tree species may be present, including Gambel oak, pinyon pine, and juniper species. This forest type often has a shrubby understory mixed with grasses and forbs but sometimes occurs as savannah with extensive grasslands interspersed between widely spaced clumps or individual trees. The ponderosa pine type is distinguished from dry mixed-conifer types by the plant community successional stages. The ponderosa pine forest type is dominated at all successional stages from seral to climax by ponderosa pine. Ponderosa pine often dominates early seral stages of dry mixed-conifer forests also, but these types are not considered to be ponderosa pine forest types because the climax species composition is dominated by other conifer species or ponderosa pine in mixtures with other conifer species.

Processes (ecological processes) are the dynamic attributes of ecosystems in terms of matter and energy, including interactions among organisms and interactions between organisms and their environment (De Groot and others 2002; SER 2004). Examples of processes are: evolution, fire and insect disturbances, photosynthesis, seed dispersal, decomposition, and soil formation.

Reference conditions are conditions existing prior to the suppression or exclusion of the primary processes and mechanisms influencing a system along a natural trajectory (sensu Kaufmann and others 1994). The reference can consist of one or several specified locations that contain model ecosystems, a written description, or a combination of both. Information collected on the reference includes both biotic and abiotic components (SER 2004)

Regeneration sites are tree-free areas created by group cutting for the purpose of establishing tree regeneration.

Residual (leave) trees or snags are those remaining after an intermediate or partial cutting of a stand (Helms 1998).

Resilience (*see* ecological resiliency)

Resiliency (*see* ecological resiliency)

Restoration (*see* ecological restoration)

Sanitation harvests involve the removal of trees to improve stand health by stopping or reducing the actual or anticipated spread of insects and disease (Helms 1998).

Safe zones (fire-free zones) are microsites where seedlings can establish and grow above the lethal flaming zone. Safe zones can be created by fire, such as the ash bed of a consumed log.

Single tree selection cutting is removal of individual trees of all size classes more or less uniformly throughout the stand to promote growth of remaining trees and to provide space for regeneration (Helms 1998).

Site index is an indicator of site quality expressed in terms of the average height of trees (defined as a certain number of dominants, codominants, or the largest and tallest trees per unit area) of a given species at a specified index or base age (Helms 1998).

Snags are standing dead or partially dead trees (snag-topped), often missing many or all limbs. They provide essential wildlife habitat for many species and are important for forest ecosystem function (Fig. 12).

Spatial pattern is the spatial arrangement of elements at the fine-, mid-, and landscape-scales that determine the function of a landscape as an ecological system (adapted from Helms 1998).

Stand density index is a widely used measure that expresses relative stand density based on some standard condition such as the relationship of number of trees to the stand quadratic mean diameter (Helms 1998) or the biological maximum density for a specific species (Long 1985).

Stands are areas in which the biophysical site conditions and the vegetation composition and structure are relatively homogeneous. Stands comprise the mid-scale, thus ranging in size from 100-1000 acres. Stands and patches are roughly synonymous

Structure is the physiognomy or architecture of an ecosystem with respect to the density, horizontal stratification, spatial pattern, and frequency distribution of vegetation (i.e., overstory, understory, etc.) size, age, and/or life form (adapted from SER 2004).

Surface fuel includes all fuels lying on or near the surface of the ground, consisting of leaf and needle litter, dead branch material, downed logs, bark, tree cones, and low stature living and dead plants (adapted from NWCG 2012).

Sustainability (*see* ecosystem sustainability)

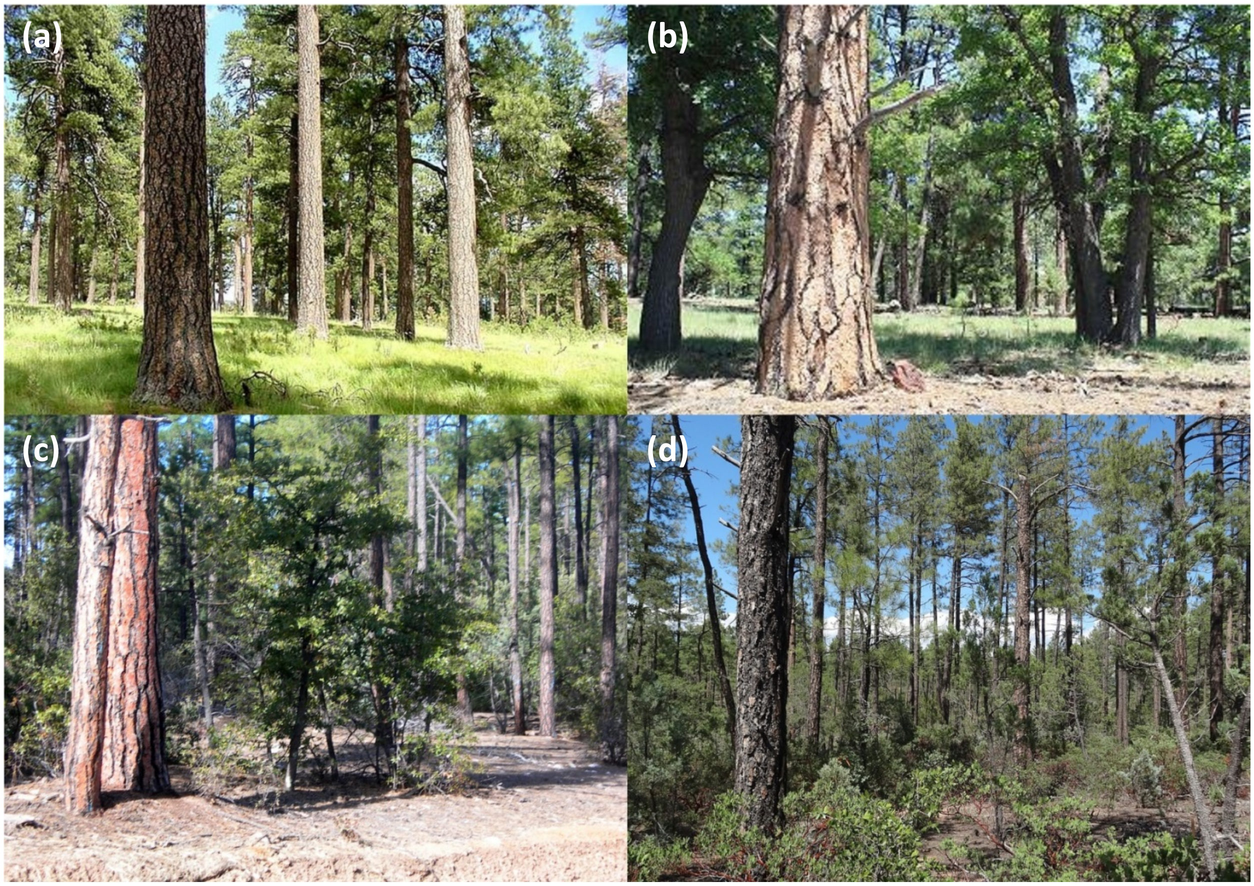
Uneven-aged forests are forests that are comprised of three or more distinct age classes of trees, either intimately mixed or in small groups (Fig. 18) (Helms 1998).

Appendix 1. Common and Scientific Names for Species Referenced in This Document.

Common name	Scientific name
Tree species	
Arizona walnut	*Juglans major*
Arizona white oak	*Quercus arizonica*
Bigtooth maple	*Acer grandidentatum*
Blue spruce	*Picea pungens*
Bristlecone pine	*Pinus aristata*
Chihuahua pine	*Pinus leiophylla*
Corkbark fir	*Abies lasiocarpa* var. *arizonica*
Douglas-fir	*Pseudotsuga menziesii* var. *glauca*
Emory oak	*Quercus emoryi*
Evergreen oaks	*Quercus* spp.
Gamble oak	*Quercus gambelii*
Grey oak	*Quercus grisea*
Junipers	*Juniperus* spp.
Limber pine	*Pinus flexilis*
Pinyon pines	*Pinus* spp.
Ponderosa pine	*Pinus ponderosa*
Quaking aspen	*Populus tremuloides*
Silverleaf oak	*Quercus hypoleucoides*
Southwest white pine	*Pinus strobiformis*
Subalpine fir	*Abies lasiocarpa*
Two-needle pinyon	*Pinus edulis*
White fir	*Abies concolor*
Shrub species	
Big sagebrush	*Artemisia tridentata*
Black sagebrush	*Artemisia nova*
Ceanothus	*Ceanothus* spp.
Common juniper	*Juniperus communis*
Creeping barberry	*Mahonia repens*
Currant	*Ribes* spp.
Kinnikinnik	*Arctostaphylos uva-ursi*
Manzanita	*Arctostaphylos* spp.
Mountain mahogany	*Cercocarpus montanus*
Mountain ninebark	*Physocarpus monogynus*
Mountain snowberry	*Symphoricarpos oreophilus*
Netleaf oak	*Quercus rugosa*
New Mexico locust	*Robinia neomexicana*
Pointleaf manzanita	*Arctostaphylos pungens*
Rockspirea	*Holodiscus dumosus*
Shrub live oak	*Quercus turbinella*

Stansbury cliffrose	*Purshia stansburiana*
Sumac	*Rhus* spp.
Wavyleaf oak	*Quercus undulata*
Grass and sedge species	
Arizona fescue	*Festuca arizonica*
Blue grama	*Bouteloua gracilis*
Dryspike sedge	*Carex siccata*
Fringed brome	*Bromus ciliatus*
Indian ricegrass	*Achnatherum hymenoides*
Longtongue muhly	*Muhlenbergia longiligula*
Mountain muhly	*Muhlenbergia montana*
Muttongrass	*Poa fendleriana*
Parry's oatgrass	*Danthonia parryi*
Screwleaf muhly	*Muhlenbergia virescens*
Forb species	
Forest fleabane	*Erigeron eximius*
Nevada pea	*Lathyrus lanszwertii*
Parasitic plant species	
Douglas-fir dwarf mistletoe	*Arceuthobium douglasii*
Southwestern (Ponderosa pine) dwarf mistletoe	*Arceuthobium vaginatum* subsp. *cryptopodum*
Fungus species	
Armillaria root disease	*Armillaria* spp.
Black stain root disease	*Leptographium* spp.
Insect species	
Bark beetles	*Dendroctonus* spp. and *Ips* spp.
Douglas-fir tussock moth	*Orgyia pseudotsugata*
Roundheaded pine beetle	*Dendroctonus adjunctus*
Spruce budworm	*Choristoneura occidentalis*
Mammal species	
Ground squirrels	*Callospermophilus* spp.
Coyote	*Canis latrans*
Tassel-eared squirrel	*Sciurus aberti*
Hares	*Lepus* spp.
Bobcat	*Lynx rufus*
Rabbits	*Sylvilagus* spp.
Bird species	
Northern goshawk	*Accipiter gentilis*
Merriam's turkey	*Meleagris gallopavo* var. *merriami*

Appendix 2. Major Ponderosa Pine Forest Subtypes: (a) Ponderosa Pine/Bunchgrass, (b) Ponderosa Pine/Gambel Oak, (c) Ponderosa Pine/Evergreen Oak, and (d) Ponderosa Pine/Evergreen Shrub.



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Planning for Connectivity

A guide to connecting and conserving wildlife within and beyond America's national forests

Defenders of Wildlife

The Center for Large Landscape Conservation

Wildlands Network
Reconnecting Nature in North America

Yellowstone to Yukon Conservation Initiative

ACKNOWLEDGEMENTS

Planning for Connectivity is a product of The Center for Large Landscape Conservation, Defenders of Wildlife, Wildlands Network and Yellowstone to Yukon Conservation Initiative. This guide focuses on requirements established under the National Forest System land management planning rule to manage for ecological connectivity on national forest lands and facilitate connectivity on planning acrosss land ownerships. The purpose of the guide and its parent publication, *Planning for Diversity*, is to help people inside and outside of the Forest Service who are working on forest plan revisions navigate these complex diversity and connectivity requirements.

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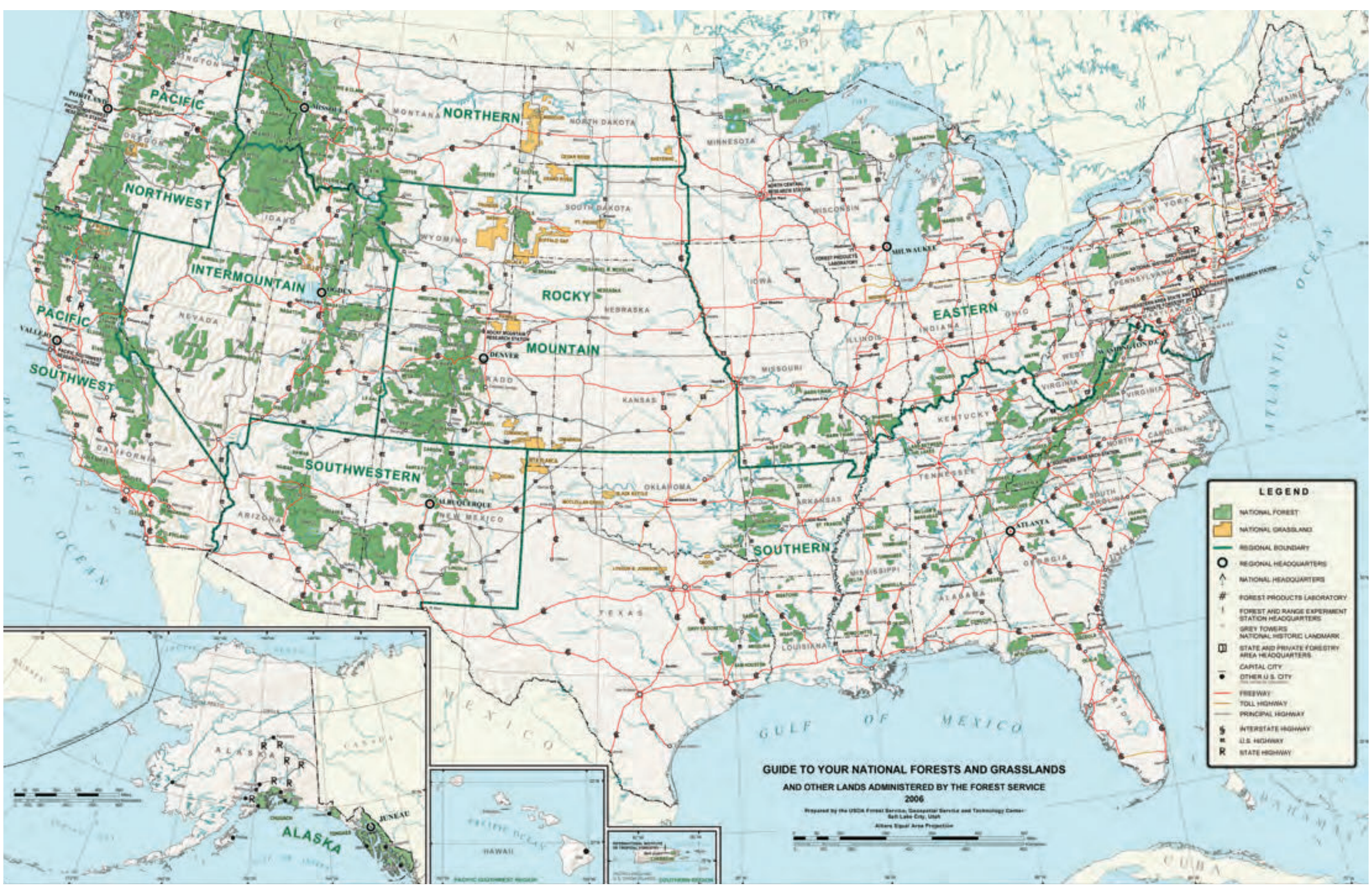
INTRODUCTION

The United States Forest Service manages more than 193 million acres—over 8 percent of all U.S. lands—an area about the size of Texas and twice the size of the National Park System. The National Forest System comprises 154 national forests and 20 national grasslands and one national prairie (collectively referred to as "national forests" in this guide). Located in 42 states, Puerto Rico and the U.S. Virgin Islands, these public lands are essential to the conservation of wildlife habitat and diversity. National forests encompass three-quarters of the major U.S. terrestrial and wetland habitat types—including alpine tundra, tropical rainforest, deciduous and evergreen forests, native grasslands, wetlands, streams, lakes and marshes. This variety of ecosystems supports more than 420 animals and plants listed under the Endangered Species Act (ESA) and an additional 3,250 other at-risk species.

To guide the management of each national forest, the Forest Service is required by law to prepare a land management plan (forest plan). Forest plans detail strategies to protect habitat and balance multiple uses to ensure the persistence of wildlife, including at-risk and federally protected species.

In April 2012, the Forest Service finalized regulations implementing the National Forest Management Act (NFMA). These regulations, commonly referred to as the "2012 Planning Rule" established a process for developing and updating forest plans and set conservation requirements that forest plans must meet to sustain and restore

The National Forest System



the diversity of ecosystems, plant and animal communities and at-risk species found on these public lands (36 C.F.R. §§ 219.1-219.19, abbreviated throughout this report by omitting “36 C.F.R. §”).

The forest planning rule includes explicit requirements for managing for ecological connectivity on national forest lands and facilitating connectivity planning across land ownerships—the first such requirements in the history of U. S. public land management. The pending revisions of most forest plans provide a significant opportunity to protect and enhance the diversity of habitat and wildlife on national forest lands by developing forest plans that promote the conservation and restoration of ecological connectivity.

This guide is designed to help people, working within and outside of the Forest Service, develop effective connectivity conservation strategies in forest plans developed under the 2012 Planning Rule. It summarizes the role of connectivity within the conservation framework of the rule and offers guidance and examples of how to conduct connectivity planning in the land management planning process.

The guide is a collaboration of Defenders of Wildlife, The Center for Large Landscape Conservation, Wildlands Network and Yellowstone to Yukon Conservation Initiative and is our collective interpretation of the connectivity requirements of the 2012 Planning Rule. The guide is intended to add value to official agency policies developed to support implementation of the rule. In January 2015, the Forest Service published Final Agency Directives for Implementation of the 2012 Planning Rule (FSM 1900 Planning, FSH 1909.12). While this guide and those directives may in some cases describe different approaches to implementing the connectivity requirements of the planning rule, we believe our interpretations are consistent with the planning rule and NFMA and hope the guide is viewed as a useful companion set of recommendations from the perspective of conservation organizations experienced in national forest planning, connectivity science and policy.

The guide covers the unique connectivity aspects of the planning rule, a rule that addresses complex ecosystem and species conservation processes and has many specific requirements.

How to Use This Guide

Planning for Connectivity presents guidance and best practices for connectivity planning, including examples from case studies in forest planning. Resources associated with the case studies are listed in the references section. We suggest using this guide in tandem with *Planning for Diversity,* a companion publication that addresses the overarching conservation framework of the 2012 Planning Rule. *Planning for Diversity,* additional resources on diversity and connectivity science and planning and a collection of forest planning case studies are available online at www.defenders.org/forestplanning.

THE IMPORTANCE OF CONNECTIVITY

It is useful to think of connectivity contributing to both the structure and function of ecosystems and landscapes. Structural connectivity is the physical relationship between patches of habitat or other ecological units; functional connectivity is the degree to which landscapes actually facilitate or impede the movement of organisms and processes of ecosystems (Ament et al. 2014).

The structure or pattern of an ecosystem or landscape can be defined as the arrangement, connectivity, composition, size and relative abundance of patches that occur within an area of land at a given time. Patches are surface areas that differ from their surroundings in nature or appearance (Turner et al. 2001). They can be characterized by vegetation type, seral stage, habitat type or other features relevant to a species and also by the condition of surrounding lands, which can significantly affect the biological character of a habitat patch.

Fragmentation, the breaking up of habitat or cover type into smaller disconnected patches (Turner et al. 2001), may result from natural or anthropogenic disturbances that introduce barriers to connectivity. In natural landscapes, patches that differ from the surrounding area would likely be areas disturbed by fire, flood, blowdown or other natural processes. In managed landscapes, habitat or cover can be fragmented by human caused disturbances such as road-building or removal of vegetation. In natural and managed fragmented landscapes, patches can be thought of as the remaining undisturbed areas. The greatest conservation needs are usually associated with maintaining or restoring connectivity among patches.

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The arrangement of patches of vegetation defines the pattern of a landscape like this one in Medicine Bow National Forest.

Other terms related to connectivity and wildlife movements include (Ament et al 2014):

- **Corridor.** A distinct component of the landscape that provides connectivity (think of it as a linear patch).
- **Linkage area or zone**. Broader regions of connectivity important to maintain ecological processes and facilitate the movement of multiple species.
- **Permeability.** The degree to which landscapes are conducive to wildlife movement and sustain ecological processes.

The 2012 Planning Rule defines connectivity as:

> *Ecological conditions that exist at several spatial and temporal scales that provide landscape linkages that permit the exchange of flow, sediments, and nutrients; the daily and seasonal movements of animals within home ranges; the dispersal and genetic interchange between populations; and the long distance range shifts of species, such as in response to climate change* (219.19).

The planning rule definition reflects both structural and functional aspects of connectivity. The rule's reference to spatial scales and "landscape linkages" suggests a structure of connected patches and ecosystems. Functional connectivity is also part of the definition: water flows, sediment exchange, nutrient cycling, animal movement/dispersal, species climate adaptation and genetic interchange are all ecological processes that are sustained by connectivity.

Any comprehensive strategy for conserving biological diversity requires maintaining habitat across a variety of spatial scales and includes the maintenance of connectivity, landscape heterogeneity and structural complexity (Lindenmayer and Franklin 2002). Connectivity is especially important for enabling adaptation to changing stressors, including climate change. The challenge of climate change was a driving factor in the development of the 2012 Planning Rule (77 Fed. Reg. 21163). A review of 22 years of recommendations for managing biodiversity in the face of climate change found improving landscape connectivity is the most frequently recommended strategy for allowing biodiversity to adapt to new conditions (Heller and Zaveleta 2009).

Wildlife species are becoming increasingly isolated in patches of habitat surrounded by a human-dominated landscape. Exacerbating this fragmentation is the effect of exurban development that continues to encroach on Forest Service lands (Hansen et al. 2005; Stein et al. 2007). The distribution of many wildlife populations continues to shrink as a result. Aquatic and terrestrial landscape patterns have been substantially altered, reducing or eliminating ecological connectivity for many wildlife populations. Physical barriers with human development further reduce connectivity. Changes in habitat, such as the simplification of complex forest vegetation, can also make critical areas for movement less permeable to some species. Scientists recognize that preserving or enhancing connectivity can be a practical tool for conserving biodiversity in such circumstances (Worboys et al. 2010).

THE 2012 FOREST PLANNING RULE

The 2012 Planning Rule is a federal regulation implementing NFMA (1600 U.S.C. § 1600 et seq.). NFMA was enacted in 1976 in large part to elevate the value of ecosystems, habitat and wildlife on our national forests to the same level as timber harvest and other uses. NFMA codified an important national priority: forest plans must provide for the diversity of habitat and animals found on national forests.

NFMA established a process for integrating the needs of wildlife with other multiple uses in forest plans. Most importantly, the law set a substantive threshold Forest Service actions must comply with for sustaining the diversity of ecosystems, habitats, plants and animals on national forests. However, the law gave discretion to the Forest Service, through the development of forest planning regulations and forest plans, to define that threshold.

THE PLANNING PROCESS

According to NFMA, forest plans are to be revised on a 15-year cycle. The planning rule provides a process for developing, revising or amending plans that is adaptive and science-based, engages the public and is designed to be efficient, effective and within the agency's ability to implement (77 Fed. Reg. 21162).

The planning rule establishes a three-phase process:

1. **Assessment**. The assessment identifies and evaluates information relevant to the development of a forest plan. The assessment is used during plan revision to evaluate what needs to change in the current plan, including what is needed to meet the requirements of the planning rule.
2. **Development.** During the plan development stage, the Forest Service develops and finalizes the forest plan and plan monitoring program. A draft proposal is developed and management alternatives are evaluated through the process established by the National Environmental Policy Act (42 U.S.C. § 4321 et seq.).
3. **Implementation/monitoring.** After finalizing the forest plan, the agency begins to implement the plan, including the development and implementation of management projects. Projects must be consistent with the forest plan and implementation of the plan must be evaluated through a monitoring program. Monitoring information is then evaluated to determine if aspects of the forest plan should be changed.

In addition, the Forest Service must use the best available scientific information to inform the planning process (219.3) throughout all three phases.

The planning rule describes these phases as iterative, complementary and sometimes overlapping. The intent is to provide a planning framework that is responsive to new information and changing conditions.

FOREST PLAN COMPONENTS

Forest plans guide subsequent project and activity decisions, which must be consistent with the forest plan. Forest plans do this through the use of plan components, the basic building blocks of forest plans. Plan components (Table 1) shape implementation of the forest plan and are the means of meeting the requirements of the 2012 Planning Rule.

Two fundamental types of plan components are associated with the diversity requirements of the rule: landscape components and project components.

Landscape components relate to the vision and priorities for the plan area, a landscape larger than individual project areas. These components are outcome-oriented, describe how the Forest Service would like the plan area to look and function and include desired conditions and objectives. Projects to be initiated under the forest plan are designed to contribute to achieving one or more of these outcomes. It is important that desired conditions and objectives be specific enough to establish a purpose and need for the projects designed to help achieve them.

Project components pertain to how individual projects are designed and implemented under the forest plan. They include standards, guidelines and suitability determinations that prohibit specific uses. They can preclude or regulate particular management options, dictate the outcome specifications for project areas or establish procedures

Table 1. Plan components under the 2012 Planning Rule

Plan Component	Description (219.7(e))
Desired Conditions (Landscale-level)	A description of specific social, economic and/or ecological characteristics of the plan area (or a portion of the plan area) toward which management of the land and resources should be directed. Desired conditions must be described in terms specific enough to allow progress toward their achievement to be determined, but do not include completion dates.
Objectives (Landscape-level)	A concise, measurable and time-specific statement of a desired rate of progress toward a desired condition or conditions. Objectives should be based on reasonably foreseeable budgets.
Standards (Project-level)	A mandatory constraint on project and activity decision-making established to help achieve or maintain the desired condition or conditions, to avoid or mitigate undesirable effects or to meet applicable legal requirements.
Guidelines (Project-level)	A constraint on project and activity decision-making that allows for departure from its terms as long as the purpose of the guideline is met. Guidelines are established to help achieve or maintain a desired condition or conditions, to avoid or mitigate undesirable effects or to meet applicable legal requirements.
Suitability of Lands (Project-level)	Specific lands within a plan area are identified as suitable for various multiple uses or activities based on the desired conditions applicable to those lands. The plan also identifies lands within the plan area as not suitable for uses that are not compatible with desired conditions for those lands.

that must be followed in preparing projects. It is very important to note that project plan components—especially standards—are most useful when greater certainty is important, such as in meeting diversity requirements necessary to protect at-risk species. Under the planning rule, every action proposed on Forest Service lands must comply with standards and guidelines and may not occur on lands unsuitable for that action.

DIVERSITY

NFMA requires that the Forest Service's planning regulations "provide for diversity of plant and animal communities based on the suitability and capability of the specific land area in order to meet overall multiple-use objectives" (16 U.S.C. § 1604(g)(3)(B)). This diversity requirement has been interpreted by the agency in the NFMA planning regulations and by the courts.

The Forest Service has interpreted the diversity requirement in NFMA through the development of the 2012 Planning Rule, which offers an approach to meeting the diversity requirement described in more detail in the following section on the ecosystem-species approach. A pivotal piece of the diversity interpretation is the persistence of individual species on national forest lands. Maintaining viable populations of native species is the scientifically accepted method of achieving the conceptual goal of maintaining species diversity. According to a 1999 Committee of Scientists report commissioned for the purposes of forest planning, "[d]iversity is sustained only when individual species persist; the goals of ensuring species viability and providing for diversity are inseparable" (Committee of Scientists 1999: 38).

The federal judiciary's interpretation of the diversity requirement in the rule include a ruling that the NFMA diversity mandate not only imposes a substantive standard on the Forest Service, it "confirms the Forest Service's duty to protect [all] wildlife" (*Seattle Audubon Society v. Moseley*, 1489). Courts have also recognized that the Forest Service's "statutory duty clearly requires protection of the entire biological community" (*Sierra Club v. Espy*, 364).

THE ECOSYSTEM-SPECIES APPROACH

Three overarching substantive requirements (Table 2) in the planning rule pertain to NFMA's diversity requirement:

1. Maintain or restore the ecological integrity of terrestrial and aquatic ecosystems (219.9(a)).
2. Maintain or restore the diversity of ecosystems and habitat types (219.9(a)).
3. Provide the ecological conditions necessary for at-risk species (219.9(b)).

The fundamental premise of the planning rule for meeting the NFMA diversity requirement is that plan components for ecosystem integrity and diversity will provide the ecological conditions to both maintain the diversity of plant and animal communities and support the persistence of most (but not all) native species in a

Table 2. Ecological concepts and requirements of the 2012 Planning Rule[1]

Ecological Concept	Definition and Requirement from the Planning Rule (219.9, if applicable)
Ecosystem	*Definition:* A spatially explicit, relatively homogeneous unit of the Earth that includes all interacting organisms and elements of the abiotic environment within its boundaries. An ecosystem is commonly described in terms of its composition, structure, function and **connectivity**. *Requirement:* The plan must include plan components, including standards or guidelines, to maintain or restore the diversity of ecosystems and habitat types throughout the plan area. In doing so, the plan must include plan components to maintain or restore key characteristics associated with terrestrial and aquatic ecosystem types, rare aquatic and terrestrial plant and animal communities, and the diversity of native tree species similar to that existing in the plan area.
Ecological Integrity	*Definition:* The quality or condition of an ecosystem when its dominant ecological characteristics (e.g., composition, structure, function, **connectivity**, species composition and diversity) occur within the natural range of variation and can withstand and recover from most perturbations imposed by natural environmental dynamics or human influence. *Requirement:* The plan must include plan components, including standards or guidelines, to maintain or restore the ecological integrity of terrestrial and aquatic ecosystems and watersheds in the plan area, including plan components to maintain or restore their structure, function, composition and **connectivity**.
At-risk Species ▪ Threatened and Endangered ▪ Candidate and Proposed ▪ Species of Conservation Concern	*Definition:* Threatened and endangered species are federally listed under the ESA; proposed and candidate species have been either formally proposed or are being formally considered for listing under the ESA. Species of conservation concern are species for which the regional forester has determined that the best available science indicates substantial concern over the species' capability to persist over the long-term in the plan area. *Requirement:* The responsible official shall determine whether or not the (ecosystem) plan components provide the ecological conditions necessary to contribute to the recovery of federally listed threatened and endangered species, conserve proposed and candidate species, and maintain a viable population of each species of conservation concern within the plan area. If the responsible official determines that the (ecosystem) plan components are insufficient to provide such ecological conditions, then additional, species-specific plan components, including standards or guidelines, must be included in the plan to provide such ecological conditions in the plan area.
Ecological Conditions	*Definition:* The biological and physical environment that can affect the diversity of plant and animal communities, the persistence of native species and the productive capacity of ecological systems. Ecological conditions include habitat and other influences on species and the environment, e.g., the abundance and distribution of aquatic and terrestrial habitats, **connectivity**, roads and other structural developments, human uses and invasive species.
Viable Population	*Definition:* A population of a species that continues to persist over the long term with sufficient distribution to be resilient and adaptable to stressors and likely future environments.
Focal Species	*Definition:* A small subset of species whose status permits inference to the integrity of the larger ecological system to which it belongs and provides meaningful information regarding the effectiveness of the plan in maintaining or restoring the ecological conditions to maintain the diversity of plant and animal communities in the plan area. Focal species would be commonly selected on the basis of their functional role in ecosystems.

plan area (219.9). To meet the rule's requirements for at-risk species (which include federally listed threatened and endangered species, proposed and candidate species, and species of concern (SCC)), additional "species-specific" plan components may be necessary. The rule's two-tiered conservation approach (alternatively called the "ecosystem-species" or "coarse-fine filter" planning method) relies on the use of surrogate measures, or key characteristics,

1. Ecological "conditions" are defined broadly to include human structures and uses, while "ecological integrity" stresses dominant "characteristics" that suggest natural conditions and should not include human structures and uses. The term "key ecosystem characteristics" is commonly used in discussions of ecological integrity, but should not be understood to apply to human structures and uses in that context. Human structures and uses are nevertheless relevant to species viability and persistence, and therefore to diversity.

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Connectivity is an ecological condition that pronghorn and other species need to persist within and beyond the boundaries of national forests and grasslands.

to represent the condition of ecosystems, as well as the identification of at-risk species and evaluation of whether those species will be sustained through ecosystem-level plan components, or whether they require specific management attention in the form of species-level plan components.

At the ecosystem scale, the rule requires forest plans to have plan components to maintain or restore the integrity of terrestrial and aquatic ecosystems in the plan area (219.9(a)(1)) and the diversity of ecosystems and habitat types (219.9(a)(2)). Essentially this requires forest plans to maintain or restore the variety of ecosystems and habitat types found on the forests (e.g., conifer forests, wetlands, grasslands), as well as the condition of the ecosystems themselves. If the ecosystem-scale plan components are not sufficient to provide ecological conditions (i.e., meet the conservation needs) for at-risk species, additional plan components to do so are required (219.9(b)(1)). In some cases, the Forest Service may determine that it is beyond its authority or "not within the inherent capability of the plan area" to provide those conservation conditions and thus other requirements apply (219.9(b)(2)).

Connectivity plays a key role in the rule's conservation approach (see Table 2). As a key characteristic of ecosystems, connectivity should be addressed through ecosystem-scale plan components in order to maintain or restore "ecological integrity." Connectivity may also be an "ecological condition" needed by individual species, and so forest plans may need to address connectivity at the species level. For example, a recent amendment to forest plans in Wyoming protects migration corridors between seasonal habitats for pronghorn (Ament et al. 2014).

The rule's approach to conservation planning relies on the use of key characteristics in assessments, planning and monitoring to represent the condition of ecosystems, as well as the identification of at-risk species, some of which may require connectivity conditions to persist. It will be necessary for forest plans to identify key characteristics of ecosystem connectivity, as well as structure, function and composition (Table 3).

The concept of ecological integrity is used to represent the status of an ecosystem. An ecosystem is considered to have integrity when its key ecosystem characteristics occur within the natural range of variation (NRV) (219.19). NRV can be thought of as a reference condition reflecting "natural" conditions. Those conditions can be estimated using information from historical reference ecosystems or by other science-based methods. For example, many current forest ecosystems exhibit landscape connectivity patterns that differ from historical or reference conditions. For the purpose of sustaining ecosystems and wildlife, the 2012 Planning Rule directs the Forest Service to manage key characteristics of ecosystems, including their connectivity characteristics, in light of these reference conditions.

It is therefore important that forest plans have plan components, including desired conditions, to move landscapes toward a more natural range of connectedness.

ISSUES OF SCALE

The definition of connectivity in the planning rule intends for it to be provided at appropriate ecological scales. Strategies for managing connectivity in forest plans will vary based on the relevant species and their particular requirements for connectivity. The planning process must consider the habitat needs of target species and the nature of their movements. Forest plans should provide for habitat connectivity to address localized movements, as well as landscape-scale linkages between larger blocks of habitat.

Land managers must look at the broader landscape context when addressing connectivity in forest plans (219.8(a)(1)). They should consider what they are connecting and be alert to connecting specific watersheds or other geographic areas identified as being relatively more important for a particular species. Aquatic species provide a good example of large-scale connectivity needs because the existence of a connected network of aquatic ecosystems is known to be critically important to migratory aquatic species, especially when disturbances occur.

Chinook salmon and other migratory fishes need a connected network of aquatic ecosystems to survive. Forest plans must consider the large-scale connectivity needs of these species.

For many species, persistence within a national forest depends on connectivity that extends beyond forest boundaries. While the Forest Service has no authority to regulate land uses outside national forests, it can influence conservation on adjacent lands by how it chooses to manage its own lands. A forest plan should consider

Table 3. The use of key characteristics in forest planning

Ecosystem Character	Definition (219.19)	Examples of Key Characteristics
Connectivity	Ecological conditions that exist at several spatial and temporal scales that provide landscape linkages that permit the exchange of flow, sediments and nutrients; the daily and seasonal movements of animals within home ranges; the dispersal and genetic interchange between populations; and the long-distance range shifts of species, such as in response to climate change.	Structural: size, number and spatial relationship between habitat patches, mapped landscape linkages and corridors. Functional: measure of ability of native species to move throughout the planning area and cross into adjacent areas.
Composition	The biological elements within the different levels of biological organization, from genes and species to communities and ecosystems.	A description of major vegetation types, patches, habitat types, soil types, landforms and wildlife populations.
Structure	The organization and physical arrangement of biological elements such as snags and down woody debris, vertical and horizontal distribution of vegetation, stream habitat complexity, landscape pattern and connectivity.	Arrangement of patches within a landscape, habitat types within a forest, trees within a forest stand, wildlife within a planning area.
Function	Ecological processes that sustain composition and structure such as energy flow, nutrient cycling and retention; soil development and retention; predation and herbivory; and natural disturbances such as wind, fire and floods.	Types, frequencies, severities, patch sizes, extent and spatial pattern of disturbances such as fires, landslides, floods and insect and disease outbreaks.

connectivity when prioritizing lands for acquisition or conservation easements on adjacent ownerships. At a finer scale, a forest plan's requirements for size and arrangement of patch characteristics may be sufficient to produce an appropriately structured landscape for connectivity.

CONNECTIVITY INFORMATION

The scientific literature includes many connectivity and corridor studies and analyses. Peer-reviewed connectivity information pertaining to all regions of the country is readily available to inform national forest planning. In recent years, the Forest Service Research and Development Branch itself has produced numerous materials on various aspects of connectivity that can be used to support analyses of conditions, trends and sustainability. The available literature includes general publications about the science of connectivity and research on specific locations and/or species.[2] Examples include Cushman and others' analysis of corridors (2012) and McKelvey and others' (2011) identification of wolverine corridors.

Independent analyses of connectivity are also now available for many areas. The nationwide system of Landscape Conservation Cooperatives (LCC) has prioritized managing for connectivity across the country. For example, the South Atlantic LCC is completing a project titled "Identifying and Prioritizing Key Habitat Connectivity Areas for the South Atlantic Region." The Western Governors Association spearheaded the development of databases and mapping systems in the western states to identify important habitat and corridors region-wide.

The planning rule also cites other governmental management plans as sources of information to consider in assessing and planning for connectivity (219.6(a)(1)). It is critical that forest plans take into account land uses on adjacent lands and the importance of such lands to connectivity. The Forest Service should engage with highway departments, state wildlife agencies, tribal governments and county planning organizations that might affect connectivity on adjacent or intervening landscapes. These entities may have identified potential corridors that should be recognized in the forest planning process.

CONNECTIVITY COORDINATION

There is an additional requirement in NFMA that is particularly important to developing plan components for connectivity. It is a procedural requirement that the planning process be "coordinated with the land and resource management planning processes of State and local governments and other Federal agencies" (16 USC § 1604(a)). One of the purposes of the planning rule was to "[e]nsure planning takes place in the context of the larger landscape by taking an 'all-lands approach'" (77 Fed. Reg. 21164).[3] To accomplish this, forest plans should consider how habitat is connected across ownership boundaries.

The planning rule accounts for this type of "all lands" connectivity by:

- Requiring assessments to evaluate conditions, trends and sustainability "in the context of the broader landscape" (219.5(a)(1)).
- Recognizing that sustainability depends in part on how the plan area influences, and is influenced by, "the broader landscape" (219.8(a)(1)(ii), (iii)).
- Requiring coordination with other land managers with authority over lands relevant to populations of species of conservation concern (219.9(b)(2)(ii)).
- Requiring coordination with plans and land-use policies of other jurisdictions (219.4(b)).
- Requiring consideration of opportunities to coordinate with neighboring landowners to link open spaces and take joint management objectives into account (219.10(a)(4)).

Achieving the broader scale "all-lands" goals of the planning rule requires partnerships and compatible management across landscapes among multiple landowners and jurisdictions. In particular, there is a need for a landscape-scale strategic approach to conserving connectivity. NFMA has established that the way to communicate a long-term and reliable management commitment for National Forest System lands is through forest plan decisions for specific areas.

There is a significant commitment to connectivity conservation within Forest Service policy and from many agency partners. Examples of coordinated multi-agency planning efforts that specifically address connectivity and can guide the Forest Service as it seeks to implement the new rule are summarized in Appendix A.

2. Forest Service research publications on the topic may be found by entering the search term "connectivity" at www.treesearch.fs.fed.us/.

3. The planning rule defines landscape as "[a] defined area irrespective of ownership or other artificial boundaries, such as a spatial mosaic of terrestrial and aquatic ecosystems, landforms, and plant communities, repeated in similar form throughout such a defined area" (219.19).

BEST PRACTICES FOR CONNECTIVITY PLANNING

The following sections present guidance and best practices for connectivity planning, including examples from case studies in forest planning. Resources associated with the case studies are listed in the references at the end of the guide. Additional forest planning case studies are available online at www.defenders.org/forestplanning.

ASSESSING CONNECTIVITY

The planning rule requires that assessments be conducted prior to plan revisions to determine what needs to be changed in the existing forest plan, to serve as the basis for developing plan components and to inform a monitoring program. The Forest Service must review all relevant existing information and then determine the best available scientific information about conditions, trends and sustainability for connectivity in relationship to the forest plan within the context of the broader landscape (219.5(a)(1)). The Forest Service must document in the assessment report "how the best available scientific information was used to inform the assessment" (219.6(b)).

For connectivity, the assessment should address both ecosystem and species-level connectivity issues. At the ecosystem-scale, the assessment needs to identify the ecosystems and habitat types within the planning area, and then evaluate the diversity and integrity of those based on information related to their structure, function, composition and connectivity.

We recommend including the following in an assessment of connectivity at the ecosystem level:

- The selection of key characteristics for connectivity (see Table 3, page 10).
- A discussion of the NRV or "reference conditions" for the characteristics (e.g., historical pattern and connectivity).
- An evaluation of system drivers (e.g., climate change) and stressors (e.g., barriers to connectivity) on the characteristics.
- A discussion of the future status of the characteristic under current management and the current plan.

The end result should be a connectivity assessment that can be used to determine:

- How the current plan needs to change to maintain or restore connectivity.
- What plan components may be necessary to achieve the ecosystem-based connectivity requirements in the rule.

Connectivity must also be assessed as a potential condition necessary to sustain individual species. In the assessment, the Forest Service will present information on the ecological needs of species so that plan components can be developed to meet the rule's requirements for species. Particular attention should be paid to the connectivity needs of all at-risk species. To demonstrate that plan components will be effective in maintaining a "viable population" in the plan area, the assessment must provide a means of determining a "sufficient distribution" (see Table 2, page 8). The assessment should describe the relationship between connectivity and the distribution of species necessary for persistence, especially with regard to stressors like climate change. It is important that the assessment evaluate how species move, what barriers to those movements may exist and how the Forest Service can reduce the impact of those barriers within the context of recovery, conservation and viability.

The Flathead National Forest plan revision (assessment, 2014), which is being conducted under the 2012 Planning Rule, offers an example of assessing connectivity needs. The Flathead assessment includes a significant discussion of connectivity for terrestrial habitat, views connectivity from both an ecosystem and species perspective and considers both shorter term vegetation barriers on the forest and longer term human barriers between national forest lands. The example below shows how the Flathead National Forest presented a key ecosystem characteristic, description and data source for connectivity (adapted from Flathead 2014: 103, Table 26):

Key Ecosystem Characteristic: Horizontal Patterns and Landscape Connectivity

Description: The horizontal pattern of forest size/structure classes across the landscape and the spatial linkages between them, which is influenced both by human activities, such as harvesting and development, and natural processes, such as wildland fire.

Data Source for Current Condition: Montana Natural Heritage Program databases; Flathead National Forest VMap; Flathead National Forest NRV analysis.

The assessment provides a description of current and reference (NRV) conditions and expected trends for this key characteristic, as well as an evaluation of the impact of stressors (e.g., from timber harvest and developments) on habitat. The following is a key finding from the assessment:

> *Significant departures from historical conditions in patch sizes and density was noted in the NRV analysis for nearly all forest structural classes forest-wide. This trend mirrored that occurring at the larger Northern Rocky Mountain ecoregion, where drastically increased forest fragmentation was noted. The analysis found a decrease in patch size and corresponding increase in patch density, resulting in a trend of increasing forest fragmentation. The changes were most dramatic for the early successional forest patches and found to be outside the range of historical variability, which is of particular concern to ecological integrity* (Flathead 2014: 137, internal citations omitted).

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The Flathead National Forest connectivity assessment for the fisher specifies that this at-risk species requires mature forests arranged in connected, complex shapes with few isolated patches.

The Flathead assessment also presented connectivity information for an at-risk species, the fisher. This information can be used to determine the effectiveness of the current plan in providing for habitat connectivity for the species or to develop new plan components:

> *At the scale of 50–100 km2 (12,355–24,710 acre) landscapes, fishers in northern Idaho and west-central Montana selected for home ranges with greater than 50 percent mature forest arranged in connected, complex shapes with few isolated patches, and open areas comprising <5 percent of the landscape. Jones and Garton (1994) stated that preferred habitat patches should be linked by travel corridors of closed canopy forest and that riparian areas make excellent corridors provided they are large enough to enable fishers to avoid predation* (Flathead 2014: 197).

CONNECTIVITY MANAGEMENT AREAS

For connectivity, it is especially important to determine where plan components will apply. While it may be relatively easy to state desired forest-wide conditions related to connectivity, this approach by itself fails to focus efforts on areas with known connectivity values (e.g., roadless areas) and may not effectively promote integration with other uses that can lead to recognition of conflicts.

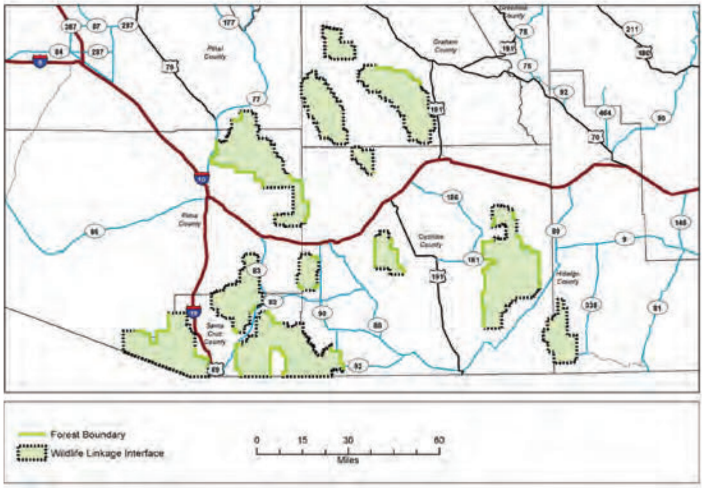
The planning rule states that the plan must indicate to which part of the plan area each plan component applies (219.7(e)). It defines "management areas" as parts of the plan area that have "the same set of applicable plan components" (219.19). Desired conditions and other plan components should be specified for particular linkage areas or corridors where they can be identified and the assessment finds them to be important to the persistence of target species in the plan area. Where connectivity is constrained, it may be necessary to identify specific areas to be managed as patches and their connecting corridors. Identifying specific management area(s) for connectivity provides clear forest plan direction on the importance of these areas and clarity for future projects.

The following case studies are examples of spatially recognizing connectivity in forest planning. An additional example is provided in the section on "Barriers to Connectivity" on page 18.

CASE STUDY: Wildlife Linkages in the Sky Islands

The mountainous "sky islands" of the Coronado National Forest in Arizona are made up of forested ranges separated by valleys of desert and grassland plains. They are among the most diverse ecosystems in the world because of their topographic complexity and location at the convergence

Figure 1. Wildlife linkages on the Coronado National Forest



Source: Coronado 2013: 64, Figure 3

U.S. FISH AND WILDLIFE SERVICE

A remote camera captured this image of an ocelot in the Huachuca Mountains of Arizona, an area where the proliferation of highways has affected connectivity among ocelot populations. To address the problem, the Coronado National Forest plan designated linkage areas on the boundary of the forest to coordinate connectivity management with other jurisdictions.

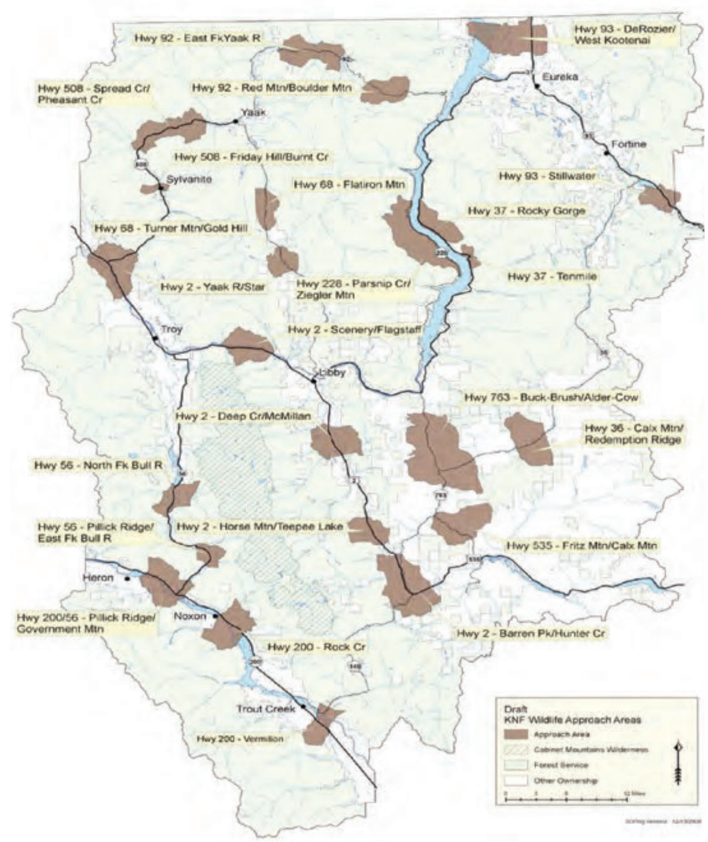
of several major desert and forest biological provinces. The valleys act as barriers to the movement of certain woodland and forest species. Species such as mountain lions and black bears depend on movement corridors between mountain islands to maintain genetic diversity and population size. Ocelots and jaguars at the northern end of their range here depend on connectivity to source populations in Mexico. The proliferation of highways and resulting increase in the number of road deaths among dispersing ocelots has affected connectivity among ocelot populations and colonization of new habitats. Movement corridors for jaguars in the American Southwest and northern Mexico are not well known but probably include a variety of upland habitats that connect some of the isolated, rugged mountains, foothills and ridges in this region.

The revised plan for the Coronado (draft, 2013) designates "wildlife linkages interface" areas, based on a statewide interagency effort that produced *Arizona's Wildlife Linkages Assessment* (Arizona Wildlife Linkages Workgroup 2006). The forest plan recognized that land management outside of the national forest boundaries affects biological resources on the national forest. Using data from the interagency group, the plan designates linkage areas on the boundary of the national forest (see Figure 1). These designated areas have management direction to maintain and reduce connectivity barriers and to coordinate connectivity management with other jurisdictions.

CASE STUDY: Grizzly Bear Approach Areas

The Kootenai National Forest in Idaho and Montana provided an excellent example of how to plan strategically for connectivity that has been confined to identifiable corridors and linkage areas. In 2008, the Kootenai identified and mapped locations of 24 approach areas important for grizzly bear connectivity using the best available scientific information from existing government and nongovernmental organizations, criteria for barriers (land ownership, topography, forest cover, land development) and wildlife use (Figure 2). Approach areas were defined as places where corridors or linkage zones cross what are termed "fracture zones" (e.g., valley bottoms

Figure 2. Grizzly bear approach areas on the Kootenai National Forest[4]



Source: Brundin and Johnson 2008: 3, Figure 1

4. The approach areas were not carried forward into the final, revised forest plan.

NATIONAL PARK SERVICE

The Kootenai National Forest plan identified "approach areas"—places where roads and other barriers to connectivity may hinder grizzly bear movement.

with highways and railways) where animal movements may be hindered and mortality risk elevated. The Kootenai also identified conservation measures that could be included in the forest plan as plan components for the approach areas and identified private lands where land exchanges, conservation easements or direct acquisition may be appropriate to improve management for one or more wildlife species (IGBC Public Lands Wildlife Linkage Taskforce 2004).

CASE STUDY: Blue Mountains Wildlife Corridor Management Area

The draft Blue Mountains National Forests plan (proposed plan, 2014), which covers the Malheur, Umatilla and Wallowa-Whitman national forests (the three forests span the states of Oregon, Washington and Idaho), establishes a management area identified as a "wildlife corridor" to connect wilderness areas and provide for landscape connectivity and defined as follows:

> *Wildlife corridors are areas designed to maintain habitat linkages between wilderness areas. Although disagreement exists regarding the utility of corridors, this management area emphasizes management for landscape connectivity, which is "the degree to which the landscape facilitates or impedes movement among resource patches," [sic] (Taylor et al. 1993) or "the functional relationship among habitat patches, owing to the spatial contagion of habitat and the movement responses of organisms to landscape structure," [sic] (With et al. 1997). A wide variety of vegetation structure and composition is present, with some showing evidence of past human disturbance and others showing affects primarily from natural disturbances, such as wildfires. Both summer and winter motor vehicle travel is restricted to designated routes. Recreation users can expect to find evidence of human activity in the form of vegetation management, mining, and road building. However, many of the roads that are closed to motor vehicle travel occur in these areas* (Blue Mountains 2014: 90).

The plan also provides a "strategy" for each management area. While the draft forest plan has drawn some criticism over unrelated issues, establishing a management area for corridors based on landscape function and structure allows for the design of habitat linkages in a variety of forms other than just simple linear connection between habitat patches.

LANDSCAPE PLAN COMPONENTS FOR CONNECTIVITY

Forest plan connectivity assessments should indicate if plan components are necessary to maintain or restore connectivity, either as an important contribution to ecological integrity or to provide conditions necessary for an at-risk species. An early consideration in forest plan connectivity planning should be the desired structure and pattern of the planning area landscape and the development of landscape plan components—desired conditions and objectives, where the desired condition describes how the connected landscape should look, and the objectives describe the timeframe and steps for achieving the desired condition.

Forest plans should include desired conditions and objectives for the sizes and distribution of habitat patches and other key characteristics of connectivity. It is also important to show the general areas where connectivity will be emphasized on a map and that the identification and management of these areas take into account the role and contribution of national forest lands to connectivity across other land ownerships.

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The Canada lynx, a species listed as threatened under the Endangered Species Act, requires connected habitat across wide areas. Forest plan standards are in place to ensure that the connectivity and other habitat needs of lynx are met on national forests.

Table 4. Examples of landscape connectivity plan components in forest plans

Landscape Plan Components	Case Study and Comments
▪ Forest boundaries are permeable to animals of all sizes and offer consistent, safe access for ingress and egress of wildlife. In particular, segments of the national forest boundary identified in [the wildlife linkages interface] remain critical interfaces that link wildlife habitat on both sides of the boundary. Fences, roads, recreational sites and other man-made features do not impede animal movement or contribute to habitat fragmentation.	The Coronado National Forest consists of isolated mountain ranges, leading the draft plan to explicitly recognize the importance of connectivity and the value of coordinated planning with adjacent jurisdictions. This is especially important to ocelots and jaguars, which occur here at the northern end of their range and depend on connectivity to source populations in Mexico (Coronado 2013). This is direction for a specific management area.
▪ Retain natural areas as a core for a regional network while limiting the built environment to the minimum land area needed to support growing public needs. ▪ Reduce habitat loss and fragmentation by conserving and managing habitat linkages within and, where possible, between the national forests and other public and privately conserved lands. ▪ Preserve wildlife and threatened, endangered, proposed, candidate and sensitive species habitat and connecting links between the San Diego River Watershed and San Dieguito/Black Mountain.	The forest plan for the Cleveland National Forest was revised in conjunction with three other California national forests. The forests face a common management challenge of collaborating in nontraditional formats with local communities and governments to maintain and restore habitat linkages between the national forests and other open space reserves. This is forest-wide direction, but also refers to specific locations.
Landscape patterns are spatially and temporally diverse and have a positive influence on overall ecological function and scenic integrity. Landscape patterns provide connectivity, allowing animals to move across landscapes. Landscape patterns are resilient and sustainable, considering the range of possible climate change scenarios. The plans include a forest-wide desired condition that mentions “the ability of species and individuals to interact, disperse, and find security within habitats in the planning area” (Blue Mountains 2014: 30).	The Blue Mountains National Forests provide an important wildlife corridor connecting habitats and wildlife migration routes between the Rocky Mountains and central Oregon (Blue Mountains 2014). This is forest-wide direction about landscape patterns, in addition to the specific management area direction described above.
Federal ownership is consolidated when opportunities arise to improve habitat connectivity and facilitate wildlife movement.	This is forest-wide direction in the proposed action for the Nez Perce-Clearwater plan revision for use in subsequent land adjustment planning. Identifying priority locations in the plan would be more helpful (Nez Perce-Clearwater 2014).

Table 4 presents examples of landscape connectivity plan components in forest planning. (The language of the plan components is either verbatim or summarized. See the “References” section for source materials.) It should be noted that these examples (drawn from older forest plans) would need to be worded more explicitly under the 2012 Planning Rule, which requires desired conditions to be “specific enough to allow progress toward their achievement to be determined” (219.7(e)(1)(i)).

PROJECT PLAN COMPONENTS FOR CONNECTIVITY

Project components pertain to how projects are designed and implemented under the forest plan. Standards and guidelines, and suitability determinations for connectivity should be designed to promote achievement of the desired conditions and objectives for connectivity. Connectivity standards should be developed when greater certainty is important, such as in meeting diversity requirements necessary to protect at-risk species.

Table 5 provides examples of standards and guidelines for connectivity in forest planning. (The language of the plan components may be verbatim or summarized. See the “References” section for source materials.)

AQUATIC ECOSYSTEM CONNECTIVITY

Forest Service lands are most often found in the higher elevations of watersheds where streams provide clear, high-quality water. Management of aquatic ecosystems often centers on providing habitat that will support important fisheries.

Plan components for ecosystem integrity (including connectivity) must take into account the interdependence of terrestrial and aquatic ecosystems (219.8(a)(1)). There

Table 5. Examples of connectivity standards and guidelines in forest plans

Project Connectivity Plan Component	Case Study and Comments
▪ Retain connections of at least 400 feet in width to at least two other [late-successional/old growth] stands. ▪ Connections should occur where medium diameter or larger trees are common, and canopy closures are within the top one-third of site potential. ▪ The length of connecting corridors should be as short as possible. ▪ Understory should be left in patches or scattered to assist in supporting stand density and cover.	The Eastside Screens are rules for logging adopted as amendments to forest plans east of the Cascade crest in Washington and Oregon in 1996. They are intended to protect remaining late-successional and old-growth forests and to retain "connectivity corridors" between them (USFS 1995).
▪ When highway or forest highway construction or reconstruction is proposed in linkage areas, identify potential highway crossings. ▪ [National forest] lands in lynx linkage areas shall be retained in public ownership. ▪ New permanent roads should not be built on ridge-tops or saddles, or in lynx linkage areas.	The Canada lynx was listed as a threatened species in March 2000, largely due to a lack of adequate regulatory mechanisms in existing land management plans for federal lands. Lynx are known to disperse over wide areas, therefore it was important to add conservation measures to forest plans for lynx connectivity, which the Forest Service did in 2007 (USFS 2007) .

is an additional requirement in the planning rule to maintain or restore the ecological integrity of riparian areas, "including plan components to maintain or restore structure, function, composition, and connectivity …" (219.8(a)). This must be done by establishing "riparian management zones" and applying plan components to them that address riparian management issues. In particular, plan components for riparian management areas must specifically address ecological connectivity, blockages of watercourses, and aquatic and terrestrial habitats (219.8(a)(3)).

Many connectivity issues become intertwined in riparian areas, and plans can address them in conjunction with either terrestrial or aquatic connectivity or both. At a broad scale, management of riparian zones contributes to overall ecological integrity by providing connectivity between watersheds for both terrestrial and aquatic species. Riparian zones also provide connectivity that contributes to the terrestrial and aquatic integrity of individual watersheds. At a fine scale, the integrity of riparian areas themselves depends on the quality of aquatic and terrestrial habitat and often requires connectivity within and from riparian areas to other systems, including the hydrologic connectivity of a water body to floodplains or groundwater (floodplain connectivity can be a limiting factor for fish).

Sophisticated conservation strategies for salmonid species have been included in forest plans in the inland Pacific Northwest for two decades. The "PACFISH" and "INFISH" conservation strategies (1995) developed by the Forest Service and the Bureau of Land Management address connectivity in two primary ways. At the broader scale, they designate watersheds where management will emphasize water quality and fish habitat. This includes existing stronghold populations of fish and, importantly, additional watersheds that can be connected to those strongholds and restored. This will create a network of connected high-quality habitat that allows recolonization after a disturbance event such as a wildfire, flood or drought has rendered an area temporarily unsuitable.

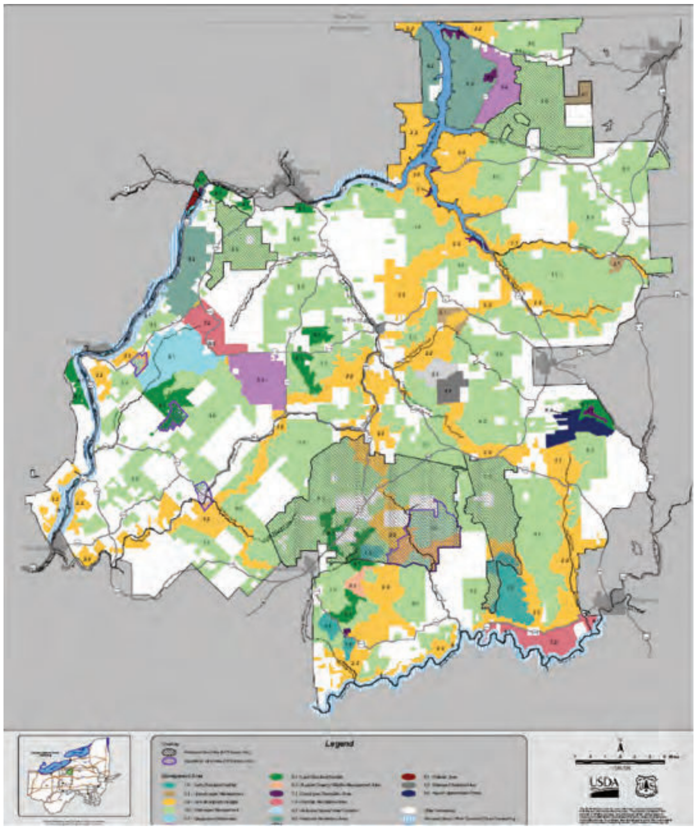
The Eastern Brook Trout Joint Venture, a partnership of state and federal agencies, nongovernmental organizations, and academic institutions, used a similar approach with the eastern brook trout in its native habitat (Maine to Georgia).According to its publication, *Conserving the Eastern Brook Trout: Action Strategies*, restoration should focus on habitat supporting populations that are doing relatively well, and then extend to adjacent habitats. An important part of this strategy is to "[i]dentify barriers to fish passage and re-establish habitat connectivity where possible" (Eastern Brook Trout Joint Venture 2008: 26).

The combination of designating watersheds and identifying connectivity barriers should lead to objectives that prioritize locations for restoration, such as the following connectivity objectives:

- Increase aquatic habitat connectivity through replacement of 90 culverts.
- Restore stronghold watersheds connectivity for aquatic species in four to six subwatersheds or on 80 to 120 stream miles.
- Establish self-sustaining brook trout populations in 10 percent of known extirpated key watersheds by 2025.

Existing forest plans also define riparian management areas, where standards and guidelines to protect aquatic resources apply to various management activities. While

Figure 3. Old forest connectivity management



Source: Allegheny National Forest Management Area Map (2007)

these plan components are primarily for the purpose of protecting resident fish, they also facilitate migration. The following type of standard would specifically address this connectivity issue: Construction or reconstruction of roads shall provide for passage of fish at all stream crossings.

BARRIERS TO CONNECTIVITY

National forest lands encompass a variety of permanent developments such as roads, railways, energy and mineral development infrastructure, recreation infrastructure and fencing. Evaluation and management of connectivity require determining the nature and effect of barriers on permeability and providing direction to reduce the effects of existing barriers and to avoid the creation of new ones. The more confined and unique the corridors or linkage zones are, the more attention should be paid to how barriers are managed. Forest plans must address barriers to connectivity that are relevant to ecological diversity and the persistence of species in a plan area.[5]

One key aspect of barriers that must be considered in relation to national forest management is their cause and degree of permanence. If barriers to wildlife movement and connectivity are due to natural disturbance (e.g., a forest opening caused by a fire or landslide), they can be viewed as transitory barriers that can be expected to "move" from place to place as new openings are created and then closed by natural succession. However, if the movement barrier for a particular species of wildlife is a lack of habitat that is difficult to restore, such as old-growth forest, the connectivity problem may be longer term and the need to protect existing patches using project plan components may be greater.

The Allegheny National Forest in Pennsylvania provides an example of old forest connectivity management, where habitat diversity was one of the key issues identified at the beginning of the plan revision process. The forest plan paid specific attention to "providing late structural and old growth forests and habitat connectivity across the landscape" (ROD, 2007: B-3). The revised plan established a management area for "late structural linkages" based on

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Forest plans should recognize the value of rare habitats, such as old-growth forest like this in the Siuslaw National Forest, in providing for connectivity.

5. While the effectiveness of habitat corridors providing connectivity is no longer disputed (Gilbert-Norton et al. 2010), potential negative consequences may result from movement of invasive, exotic, and otherwise harmful species or diseases, especially in aquatic habitats. This has been noted especially for inland trout species, where enhancing connectivity could do more harm than good by promoting competition or hybridization with non-native species, or introducing diseases. These kinds of risks should be identified and mitigated to the extent possible when designing landscape connections. Moreover, efforts to connect landscapes that have not historically been connected should be avoided.

existing core blocks of wilderness areas, research natural areas, national recreation areas and other protected areas. It was also designed to specifically include areas of known goshawk nest sites and rattlesnake dens, thus affording additional protection for these species (see Figure 3).

ROADS AND CONNECTIVITY

Roads and their associated human uses are one of the most common, persistent and obstructive barriers to terrestrial and aquatic wildlife connectivity. The National Forest System has approximately 375,000 miles of roads.[6] Decisions to build, decommission, open or close roads can affect connectivity in significant ways. Recognition of the role of unroaded (i.e., roadless) areas for the purposes of connectivity planning is equally important. Forest plans provide the overall guidance for how many roads there will be on a forest and how they are to be used.

Use of roads by the public is also governed by the Forest Service "Travel Management Rule," regulations published in 2005 to establish a nationally consistent approach to local determinations of where excluding motorized use is necessary to protect other resources or, conversely, where such use is desirable and ecologically acceptable. The regulations require each national forest to identify and designate roads, trails, and areas that are open to motor vehicle use. Motorized use is prohibited anywhere that is not so designated. These decisions are part of travel management plans, and these plans must be consistent with forest plans.

Clearly, decisions to have a road or to allow motorized use should take into account the effect of that particular road on connectivity. To fully understand the effects, it is necessary to know what role an area or corridor is expected to play in providing connectivity and what else is likely to happen there that will affect its connectivity value. The forest plan is the place to provide answers to those questions.

Where motorized use is inconsistent with the desired condition for an area, including desired connectivity conditions, a forest plan can identify the area as one that is not suitable for motorized use. This precludes the establishment of motorized routes in the area. It should also lead to eliminating any existing motorized use through road or area closures.

Site-specific desired conditions for connectivity are helpful in deciding where to manage for motorized use. The Gallatin National Forest Travel Plan Final

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Roads and their associated human uses are one of the most common, persistent and obstructive barriers to connectivity on national forest lands. The National Forest System has about 375,000 miles of roads.

6. See www.fs.fed.us/eng/transp/.

Environmental Impact Statement (2006) includes a site-specific goal for identified "wildlife corridors," which provides a good example of a desired condition that should be included in a forest plan:

> *Provide for wildlife movement and genetic interaction (particularly grizzly bear and lynx) between mountain ranges at Bozeman Pass (linking the Gallatin Range to the Bridger/Bangtails); across highway 191 from Big Sky to its junction with highway 287 (linking the Gallatin and Madison Mountain Ranges); the Lionhead area (linking the Henry's Lake Mountains to the Gravelly Mountains and areas west); Yankee Jim Canyon (linking the Absoroka Mountains to the Gallatin Range); and at Cooke Pass (linking the Absoroka/Beartooth Range to areas south)* (Gallatin 2006: 3-88 – 3-89).

A connectivity characteristic commonly used in forest plans to protect wildlife and fish habitat is road density. Road density limits are especially useful for protecting big game hunting opportunities. The presence and use of roads have also been found to create risks to movement of large carnivores such as grizzly bears, a federally listed threatened species. To comply with the ESA, forest plans in grizzly bear range include restrictions on road density.

The Flathead National Forest provides some of the most important grizzly bear habitat in the National Forest System. As a result of ESA consultation on the forest plan, the Forest Service adopted Amendment #19 in 1995 that applied objectives and standards for each of 70 grizzly bear management subunits across the Flathead (where national forest ownership is greater than 75 percent) (Flathead 1995). For example, an objective was developed stating that within five years total road density of greater than two miles per square mile would occur on less than 24 percent of the grizzly bear management unit and in 10 years that would be further reduced to less than 19 percent. Similarly, standards were used to ensure there would be no net increases in road densities above a certain threshold and to maintain the security of core grizzly bear habitat areas. These types of connectivity and security plan components have been successful in reducing the number of roads forest-wide by approximately 700 miles and increasing secure core area from 63 percent to 70 percent (Flathead 2012: unpaginated, Tables 16b-9 and 16b-10).

For terrestrial species, it is often the use of the road that is more of a barrier to connectivity than the physical presence of the road. Many current plans address the need to limit motorized access during big game hunting season or to protect sensitive big game habitat such as winter range.

CONCLUSION

The connectivity planning direction found in the 2012 Planning Rule provides a significant opportunity to develop and implement landscape- and project-scale connectivity strategies on Forest Service lands and to coordinate connectivity planning across land ownerships. To be successful, forest planning stakeholders—including Forest Service planners, conservation advocates, scientists and other agencies and governments—must collaborate to devise innovative approaches.

Connectivity planning also requires forward thinking to execute the vision of a connected landscape. There is no one way to develop and implement connectivity strategies within forest plans. We hope this guide stimulates innovative ideas and is a starting point for developing effective approaches to connectivity planning within forest plans.

Share Your Experiences

Please share your forest planning experiences with us and let us know if this guide was useful. Your input will help us build our list of case studies and improve the effectiveness of this planning tool. Send your feedback to Pete Nelson (pnelson@defenders.org).

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APPENDIX:

EXAMPLES OF COORDINATED CONNECTIVITY PLANNING

Multi-Organization Initiatives, including the Forest Service

America's Great Outdoors Initiative

www.doi.gov/americasgreatoutdoors/index.cfm

One of the goals of the President's America's Great Outdoors Initiative is "the conservation of land, water, wildlife, historic, and cultural resources, creating corridors and connectivity across these outdoor spaces, and for enhancing neighborhood parks." The "Large Landscapes Initiative" seeks to "improve collaboration across federal agencies and with state and local partners, especially given the inherent cross-jurisdictional nature of restoring large landscapes." It currently includes a study of specific wildlife linkage locations across major highways in the "Crown of the Continent" ecosystem in Montana.

Department of the Interior, Landscape Conservation Cooperatives

www.fws.gov/landscape-conservation/lcc.html

LCCs provide a forum for federal agencies (including the Forest Service), states, Tribes, non-governmental organizations, universities and others to work together to coordinate management response to climate change at the landscape level. "New wildlife corridors" was one of the specific needs identified nationally. The Great Northern LCC partners, for example, agreed to conservation goals that prominently feature connectivity as an important element of ecosystem integrity, and they also identified "target species" that depend on connectivity. Land management plans would be the vehicle for the Forest Service to incorporate broader landscape conservation goals.

Western Governors' Association Wildlife Corridors and Crucial Habitat Initiative

www.westgov.org/wildlife-corridors-and-crucial-habitat

The Western Governors' Association's initial policy stated that federal land management agencies should identify key wildlife migration corridors in their land management plans. The Forest Service is participating in implementing this connectivity guidance. In November 2012, the Forest Service encouraged forest supervisors conducting forest planning to consider information compiled by states for this initiative as part of implementing the 2012 Planning Rule.

Grizzly Bear Recovery Planning

www.igbconline.org/index.php/population-recovery/grizzly-bear-linkage-zones

The Recovery Plan for Grizzly Bear identifies the need to evaluate potential linkage areas within and between recovery areas. The Interagency Grizzly Bear Committee (IGBC, which includes the Forest Service) determined that "… linkage zone identification and the maintenance of existing linkage opportunities for wildlife between large blocks of public lands in the range of the grizzly bear are fundamental to healthy wildlife." Maps of linkage areas have been developed by the U.S. Fish and Wildlife Service and sanctioned by the IGBC.

Forest Service Initiatives

Properly addressing connectivity in land management plans will also promote coordination and integration within the Forest Service and advance other agency prerogatives.

The Forest Service Strategic Framework for Responding to Climate Change includes "development of wildlife corridors to facilitate migration" as a strategy to address climate change effects (www.fs.fed.us/climatechange/pdf/Roadmapfinal.pdf). One of the "immediate initiatives" in the roadmap is connecting habitats to improve adaptive capacity by:

- Collaborating with partners to develop strategies that identify priority locations for maintaining and restoring habitat connectivity. Seeking partnerships with private landowners to provide migration corridors across private lands.
- Removing or modifying physical impediments to species movement most likely to be affected by climate change.
- Managing forest and grassland ecosystems to reduce habitat fragmentation.
- Continuing to develop and restore important habitat corridors for fish and wildlife.

The Forest Service Open Space Conservation Strategy states that "[o]ur vision for the 21st century is an interconnected network of open space across the landscape that supports healthy ecosystems and a high quality of life for Americans" (www.fs.fed.us/openspace/national_strategy.html).

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THE CENTER FOR LARGE LANDSCAPE CONSERVATION

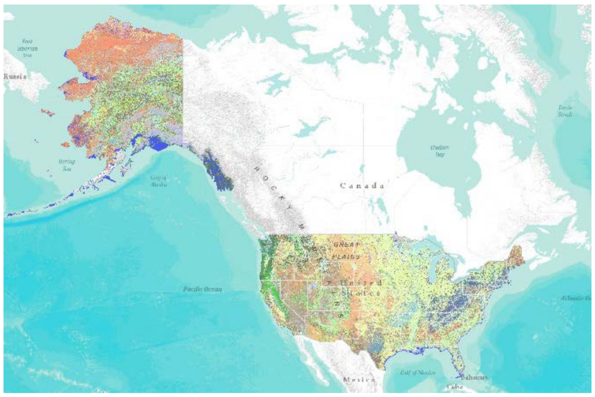
WILDLANDS NETWORK
RECONNECTING NATURE IN NORTH AMERICA

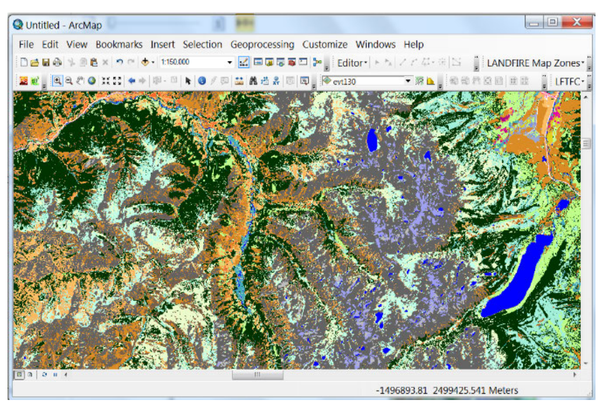
Yellowstone to Yukon
Conservation Initiative

Modifying LANDFIRE Geospatial Data for Local Applications

Version 1

September 2016





TEAMS

Integrating science, technology and fire management.
Wildland Fire Management RD&A

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Introduction

The LANDFIRE Program provides "wall-to-wall" geospatial data of vegetation, wildland fuel, fire regime, disturbance, and topographic characteristics for the United States (Rollins 2009). LANDFIRE data are often an excellent choice for wildland fire and land management planning applications due to their consistent mapping methodologies across land ownership boundaries and relevancy to common conservation and land management questions. LANDFIRE data are distributed free of charge through the Program's website at www.landfire.gov.

This guide will focus on LANDFIRE data for the conterminous United States, Alaska, and Hawaii. A subset of LANDFIRE data products is available for the Pacific and Caribbean U.S. insular areas; however, the mapping methodologies for these areas vary substantially enough from those for the conterminous U.S., Alaska, and Hawaii that we do not include discussion of these data in this version of the guide. We also focus primarily on LANDFIRE versions 1.0.5 (LANDFIRE 2001) through 1.3.0 (LANDFIRE 2012) as some major changes to mapping methodology occurred between version 1.0.0 (LANDFIRE National) and LANDFIRE 2001.

Although developed for sub-regional to national-scale planning applications, the utility of LANDFIRE data at finer scales has been demonstrated. The data are commonly applied on active wildland fire incidents (Noonan-Wright et al. 2011) and in landscape-level land management planning (Helmbrecht et al. 2012, Price et al. 2012, Scott et al. 2012, Tuhy et al. 2010). However, the applicability of LANDFIRE data at finer scales varies by the data product in question, its intended use, and location of interest. The LANDFIRE Program states that:

> *"Managers and planners must evaluate LANDFIRE data according to the scale and requirements specific to their needs (for example, habitat requirements for the species being considered or requirements by community leaders and interagency partners)…. It is the responsibility of the user to be familiar with the value, assumptions, and limitations of LANDFIRE products"* (USFS 2015).

It is within this context that we present this guide, with the purpose of providing direction on the critique and modification of LANDFIRE geospatial data products for local applications. This guide builds upon previous work on this topic by others (Stratton 2006, 2009; The Nature Conservancy 2011a; The Nature Conservancy 2011b; The Nature Conservancy 2013). It is not so much a "cookbook" or "how-to" guide, as the specifics vary greatly by data product, intended use, scale, and location. Rather, we present primary considerations for using and modifying the data for use in local applications and provide examples and demonstrations of available tools and methods for completing common critique and modification tasks.

This guide is presented in seven chapters:

Chapter 1 provides a brief overview of LANDFIRE data products; discusses general considerations of scale, accuracy, and resolution in the critique of LANDFIRE geospatial data; and provides examples of common reasons for modifying LANDFIRE geospatial data.

Chapter 2 presents a conceptual framework for critiquing and modifying geospatial data, emphasizing the importance of framing analysis objectives and an iterative approach.

Chapter 3 describes the LANDFIRE disturbance data mapping process and discusses considerations specific to data currency, disturbance causality, and modifying data to reflect changes due to new disturbances.

Chapter 4 defines LANDIFRE potential and existing vegetation data products; describes the LANDFIRE vegetation mapping process; and discusses considerations specific to application of the NatureServe Ecological Systems classification, map zone boundaries, and succession class mapping rules.

Chapter 5 defines LANDFIRE fuel data products; describes the LANDFIRE fuel mapping process; and discusses considerations specific to map zone boundaries, application scale, disturbance updates, and modeling.

Chapter 6 describes the LANDFIRE vegetation dynamics models and their role in developing fire regime and vegetation departure products and discusses considerations specific to the integrated nature of LANDFIRE vegetation products, knowledge uncertainty, map zone boundaries, differences between data versions, and conducting local vegetation departure analysis.

Chapter 7 presents two interpreted examples of critiquing and modifying LANDFIRE data for local applications. The first example focuses on using LANDFIRE data for wildfire hazard analysis in the Rogue Basin of southwest Oregon. The second example focuses on using LANDFIRE data for vegetation departure analysis in the southern Sierra Nevada Mountains.

Chapter 1: Background

LANDFIRE Product Overview

LANDFIRE produces more than 20 geospatial data layers, a suite of vegetation dynamics models representing pre-Euro-American settlement vegetation conditions, and databases with vegetation plot and management activities information. The geospatial data, which are the focus of this guide, cover all lands in the United States and are developed using methods that utilize Landsat imagery, plot data, and biophysical gradient modeling (Rollins 2009). The mapping methodology is generally consistent by version across all regions of the country. LANDFIRE periodically updates its data products to incorporate changes over time (Nelson et al. 2013, Table 1).

Table 1: Comparison of LANDFIRE versions 1.0.0 (LANDFIRE National) through 1.3.0 (LANDFIRE 2012).

LANDFIRE Version	Currency	Distribution Date	Version Information
National (1.0.0)	Circa 2001	2008	The first full iteration of LANDFIRE data based on Landsat imagery from 1999-2001.
2001 “Refresh” (1.0.5)	Circa 2001	2011	Enhanced National by improving biophysical setting and existing vegetation type, cover and height mapping.
2008 “Refresh” (1.1.0)	Circa 2008	2011	Updated 2001 products for disturbance and succession. Landsat 1984-2008 imagery analyzed for change.
2010 (1.2.0)	Circa 2010	2013-14	Products updated for disturbance and succession. Landsat 2007-2011 imagery analyzed for change. Refined urban, agriculture, and wetlands mapping.
2012 (1.3.0)	Circa 2012	2014-15	Products updated for disturbance and succession. Landsat 2010-2013 imagery analyzed for change.

LANDFIRE geospatial data can be divided into five primary categories: vegetation, wildland fuels, fire regime, disturbance, and topography (Table 2). The vegetation data layers characterize both existing and potential vegetation type, vegetation structure, and vegetation development stage, and are primary inputs for developing other data layers. The wildland fuel data layers depict surface and canopy fuel characteristics that are inputs to various geospatial fire modeling systems. The fire regime data layers estimate the fire frequency and severity prior to European-American settlement as well as the current condition of the vegetation within the context of the historical disturbance regime. The disturbance data layers depict landscape changes that result from natural disturbances (e.g. wildfires and hurricanes) and management activities (e.g. prescribed fire and timber harvest), and are used to update the vegetation and fuel data layers over time (Nelson et al. 2013). Finally, the topographic data layers are required inputs to common geospatial fire behavior modeling systems and are used as base data for developing other LANDFIRE data layers. Modification of topographic data (elevation, slope, and aspect) is uncommon and therefore not discussed in this guide.

Table 2: LANDFIRE data products organized by data category.

Data Category	Abbreviation	Data Products
Vegetation	EVT	Existing Vegetation Type
	EVC	Existing Vegetation Cover
	EVH	Existing Vegetation Height
	SCLASS	Succession Class[a]
	ESP	Environmental Site Potential
	BpS	Biophysical Setting
	--	Vegetation Dynamics Models[b]
	LFRDB	LANDFIRE Reference Database[c]
Fuel	FBFM13	13 Fire Behavior Fuel Models
	FBFM40	40 Fire Behavior Fuel Models
	CFFDRS	Canadian Forest Fire Danger Rating System (AK only)
	FCCS	Fuel Characteristics Classification System Fuelbeds
	FLM	Fuel Loading Models
	CC	Forest Canopy Cover
	CH	Forest Canopy Height
	CBD	Forest Canopy Bulk Density
	CBH	Forest Canopy Base Height
Fire Regime	FRG	Fire Regime Groups
	MFRI	Mean Fire Return Interval
	PLS	Percent Low-severity Fire
	PMS	Percent Mixed-severity Fire
	PRS	Percent Replacement-severity Fire
	VCC	Vegetation Condition Class[d]
	VDEP	Vegetation Departure[e]
Disturbance	DYEAR	Disturbance 1999-2012
	FdistYEAR	Fuel Disturbance
	VdistYEAR	Vegetation Disturbance
	Events	Public Events Geodatabase[f]
Topography	ASP	Aspect
	DEM	Elevation
	SLP	Slope

[a]LANDFIRE groups succession class with its fire regime datasets because it is used to assess current vegetation condition, but in this guide it is grouped with the vegetation datasets because it is created from a compilation of existing vegetation datasets.
[b]The Vegetation Dynamics Models are non-spatial products used as primary inputs for mapping the fire regime datasets, to provide rulesets for mapping succession classes and as an ancillary data source for mapping existing vegetation type, biophysical settings and fire behavior fuel models.
[c]A database with geo-reference plot information used for mapping the vegetation datasets.
[d, e]Vegetation condition class and vegetation departure were not created for LANDFIRE 2010.
[f]A geo-referenced collection of disturbance and management information used to create the disturbance datasets.

The remainder of this chapter presents general considerations about the critique and modification of LANDFIRE geospatial data. Subsequent chapters will provide greater detail about specific considerations in the vegetation, fuels, fire regime, and disturbance categories.

Considerations of Scale

A primary consideration when evaluating a geospatial data layer is its scale. Traditionally map, or cartographic, scale is defined as the mathematical relationship between a given feature on a map and that same feature on the ground. For example, a typical topographic map from the U.S. Geological Survey's 7.5-minute quadrangle series has a map scale of 1 to 24,000 (1:24,000) meaning that one map unit is equivalent to 24,000 of the same units on the ground. Geospatial data do not have a map scale in the traditional sense. A geographic information system (GIS) stores the exact coordinates of every feature in a geospatial data layer, allowing users to zoom in and out on the monitor to view data at nearly any map scale, regardless of the precision of the underlying source data. This does not mean that geospatial data do not have a scale; rather, the scale of geospatial data may be difficult to discern.

In a more general sense scale may be defined as the spatial (or temporal) dimension of an object or a process, and is characterized by grain and extent (Turner et al. 2001). Grain is the finest level of resolution in geospatial data. For raster data, grain refers to cell size and for polygon data (i.e., vector data), grain refers to the minimum mapping unit. LANDFIRE raster data have a 30m x 30m cell size—that is, each data cell, or pixel, represents a $900m^2$ (approximately 0.22 acre) area on the ground. LANDFIRE data therefore have a 30-meter spatial resolution, or grain size. However, LANDFIRE data are not intended to be accurate or useful at the extent of an individual pixel or small group of pixels. The scale at which LANDFIRE data are applicable varies by product, intended use, and the location of interest.

With geospatial data there are no concrete rules that specify the required scale for a given application. Different analyses require data at different scales. For example, the scale needed to identify threatened and endangered species habitat is different than the scale needed to distinguish forests from grasslands. The critical question is, are the data good enough to meet the analysis needs? Evaluating the data accuracy and resolution requirements of your analysis will help answer this critical question.

Considerations of Accuracy and Resolution

Evaluating the accuracy and resolution of LANDFIRE data will help determine its suitability for a given use. Two types of accuracy to evaluate are **positional accuracy**, or the ability of the data to reflect the true or accepted geographic location of features in space, and **content accuracy**, or the agreement between mapped units and the true or accepted value of those units. Likewise, evaluation of resolution includes **spatial resolution**, or the amount of ground area represented by a pixel, and **thematic resolution**, or the level of detail in the classification of map units. Issues with accuracy and resolution are not mutually exclusive—problems with one may result in problems with the other. Ultimately, the goal of understanding these issues is to evaluate the strengths and weaknesses of a geospatial data layer to determine its suitability for a particular analysis. Next we discuss considerations of accuracy and resolution relevant to LANDFIRE data.

Positional Accuracy

Positional accuracy refers to the ability of the data to reflect the true or accepted geographic location of features. There is little the end user of LANDFIRE data can do to improve issues of positional accuracy. Boundaries or distinctions between feature types (e.g., vegetation types) may not be precisely located solely due to the raster format of LANDFIRE data. The spatial resolution of raster data has a direct effect on positional accuracy: the larger the cell size the less accurate the location (Figure 1). However, these should be minor issues if applying LANDFIRE data at an appropriate scale, one in which the data meet the analysis needs. It is also worth noting that vector data, such as the LANDFIRE event polygons, and plot data from the LANDFIRE Reference Database, are not immune to error in the location of features. Issues of positional accuracy may arise due to errors in source data, precision of field measurements, or errors in data entry.

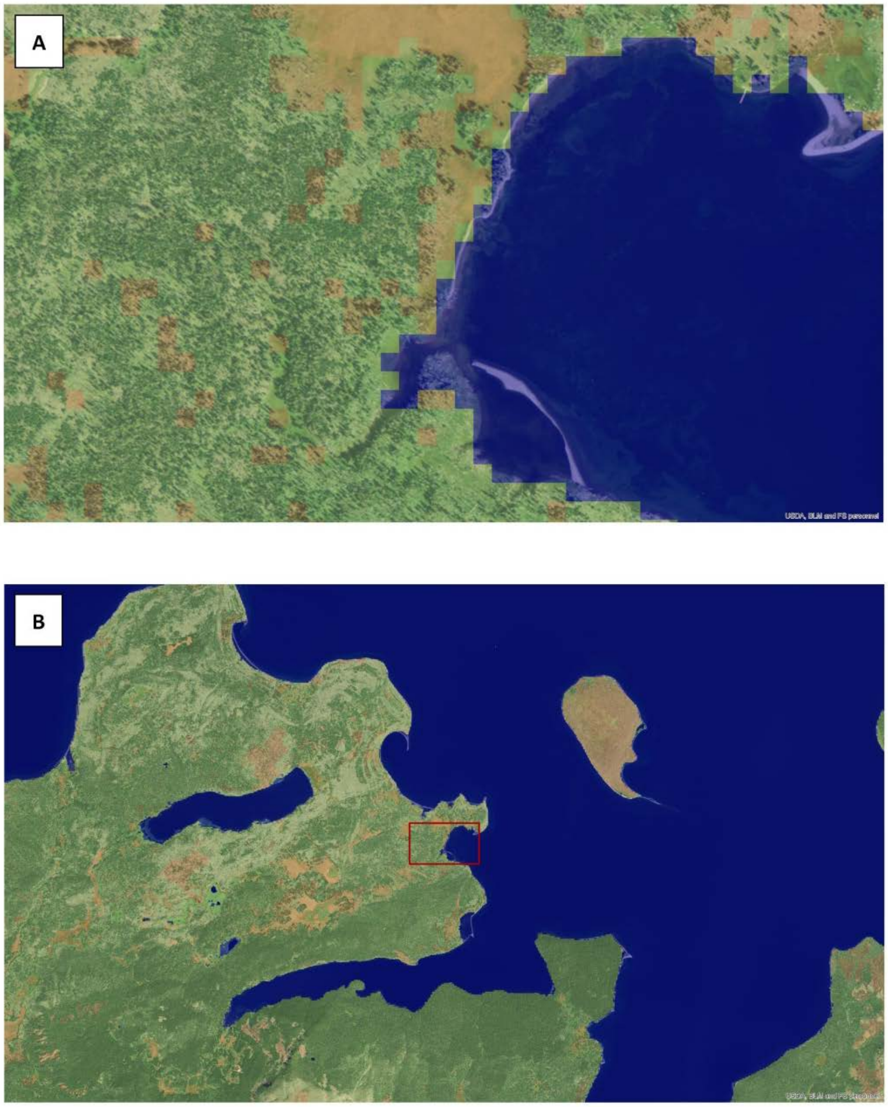


Figure 1. An example of how the spatial resolution of raster data has a direct effect on positional accuracy. LANDFIRE 30-meter resolution data does not precisely depict the shoreline or the boundary between grass (yellow shade) and forest (green shade) when viewed at a small extent (A), but at a broader extent (B), these differences are less apparent and less significant. The red rectangle in panel B shows the extent of panel A.

Content Accuracy

Content accuracy refers to the agreement between mapped units and the true or accepted value of those units. In other words, are the pixel values correct? There is much that can be done by end users of LANDFIRE data to improve issues of content accuracy based on local knowledge, additional data sources, and an understanding of the LANDFIRE data development process. This is the primary focus of this guide. Different types of errors that may affect content accuracy are described next.

Map Unit Errors

Errors in map unit assignments in LANDFIRE data may arise through errors or limitations in remote sensing data, field plots, statistical modeling, processing logic, or a combination of these and other factors. Due to variation in data sources, this error is typically not systematic geographically. For example, the abundance and quality of plot data is inconsistent across the U.S., and cloud-free imagery is more difficult to acquire in certain areas of the country (e.g., Alaska) than others.

Data Currency Errors

One of the most obvious sources of error in vegetation and fuel data is the currency, or age, of the data. Vegetation and fuels change over time due to disturbance and vegetation growth. Disturbance may include the development of previously undeveloped land, natural disturbances, such as windthrow or wildfire, and management activities such as forest thinning or prescribed fire. Vegetation growth over time may result in changes to species composition, structure, and associated dead wood and surface matter. LANDFIRE updates its products accounting for both disturbance and vegetation growth every two years (Nelson et al. 2013), but by the time the data are delivered to the user, they are typically three or more years out-of-date. For example, LANDFIRE 2010 existing vegetation and fuel data were not available for the northwest and southwest United States geographical areas until May 2013.

The importance of updating for these temporal changes should ultimately be determined by the analysis objectives, but the need will also be influenced by the geography and vegetation dynamics of the analysis area. In areas where disturbance is uncommon or where vegetation growth is slow, less frequent updating will be required than in areas that experience frequent disturbances or rapid vegetation growth. For example, vegetation and fuel maps likely need more frequent updating in the south-eastern United States where vegetation growth is more rapid and human and natural disturbances are more frequent than in the desert portions of the southwest. In more mesic life zones, such as mid-elevation forest, the geospatial data layers likely need more frequent updating than in drier low-elevation shrub or grassland zones. Even within local areas there are typically management areas with higher wildland fire or other disturbance activities that require updating as compared to adjacent areas with low activity. Other factors to consider when assessing data currency are the type and size of disturbances that need to be reflected in the data to meet analysis objectives.

Processing or Logic Errors

In some cases, content accuracy issues are introduced during data processing. Unintentional or accidental errors may be difficult to find and correct, but a common source of content error in LANDFIRE products is the result of applying generalized mapping rule sets—a pixel's value is determined by a combination of values from other data as specified in a rule. For example, a rule may assign fire behavior fuel model TU5 (Very High Load, Dry Climate Timber-Shrub), when vegetation type equals mesic mixed conifer and canopy cover is less than 60%. Rule sets are developed and applied at the map zone level (Figure 2). While these rules may be appropriate at the scale of an entire map zone (LANDFIRE map zones range between 12 and 60 million acres in size in the conterminous U.S. and Alaska; Hawaii is a single map zone of 4 million acres), they may need to be refined for application at finer scales. In other words, the "best fit" for an entire map zone may be a compromise between different parts of the map zone. There are also often inconsistencies in mapping rules between adjacent map zones resulting in an "artificial edge" in the data. Many analysis areas often extend across two or even three map zone boundaries. On the ground these changes would start gradually near the boundary between map zones displaying continuous change across the boundary. However, accurately mapping this type of gradual transition is very difficult to achieve in a large national mapping program such as LANDFIRE.

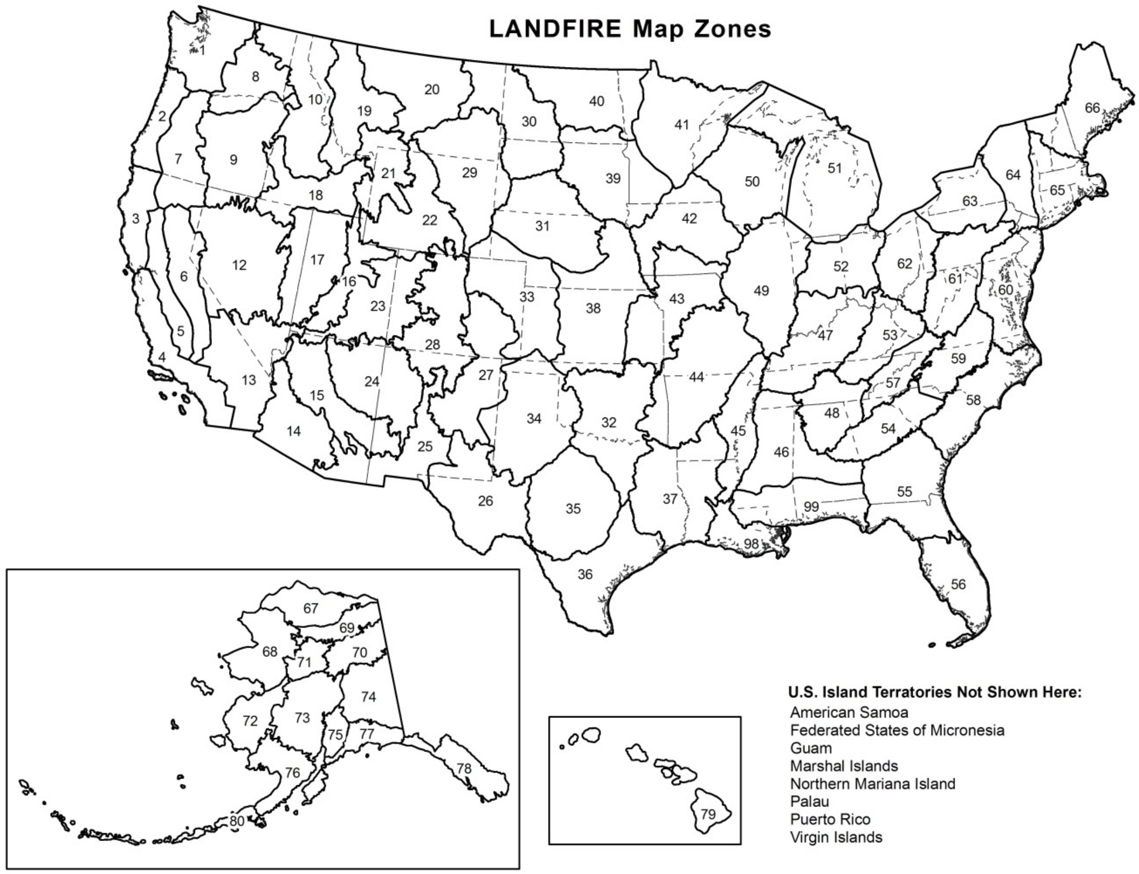


Figure 2. LANDFIRE map zones. There are 80 LANDFIRE map zones across the continental U.S., Alaska, and Hawaii, ranging in size from 4 to 60 million acres.

Content error may also arise due to incomplete knowledge and uncertainty. For example, LANDFIRE's pre-Euro-American fire frequency and severity data are created using a lookup table that links a biophysical setting with the results of a model used to simulate vegetation dynamics and estimate the mean fire frequency and fire severity distribution. The models are created using the best available literature and expert knowledge, but for many biophysical settings, the available information is far from complete. For example, there is considerably more information available to create vegetation dynamics models for biophysical settings that have economic value (e.g. productive forests) than biophysical settings that are rare or traditionally have had little economic value (e.g. arid shrublands; Blankenship et al. 2012). Greater uncertainty about historical fire regime characteristics is also associated with biophysical settings where evidence of historical fires is sparse, non-existent, or just harder to acquire, such as in stand-replacement or very long-interval fire regimes.

Spatial Resolution

As mentioned above, the spatial resolution of raster data is defined as the amount of ground area represented by a pixel. LANDFIRE data are based on Landsat satellite imagery, which have a 30m x 30m pixel size. In other words, each individual pixel represents an area of 30m x 30m, or 900m^2 (about .22 acres), on the ground.

Spatial resolution can be adjusted if necessary to meet analysis objectives. Decreasing spatial resolution by increasing pixel size (e.g., resampling 30m resolution data to 270m resolution) is sometimes done to: reduce processing time for computationally intensive analyses; decrease file storage space requirements; speed up display time; and/or, reduce the "pixelated" look of a map by absorbing isolated cells into larger patches. While it is possible to adjust resolution the other way, that is to change from coarser to finer resolution, greater map detail can only be achieved if finer resolution geospatial data are incorporated into the resampling process. That is, resampling to a finer resolution without additional finer-scale information gives a false sense of accuracy (see sidebar).

Resampling Raster Data Layers

Resampling is the process of changing the resolution of a dataset. Raster data may be made coarser by aggregating adjacent pixels. Some users of LANDFIRE data who perform national summaries of the data have resampled LANDFIRE grids from 30m to 270m. At this broad extent, resampling may have little impact on the results but can greatly increase computer processing efficiency.

Resampling to a finer resolution is sometimes referred to as downscaling and is often associated with the process of obtaining local level climate data from global climate models. Resampling to a finer resolution is possible using the resample techniques available in ArcGIS, but these techniques will not change the accuracy of the underlying data.

There are several resampling methods available in ArcGIS software, and the resampled raster values will differ depending on the method used.

Thematic Resolution

Thematic resolution refers to the level of detail in the map units. The thematic resolution of LANDFIRE data varies by data product. The most common reason that an end user of LANDFIRE data might change thematic resolution is to ensure that the level of detail in the map units aligns with the level of detail needed to achieve the analysis objectives.

Thematic resolution can be changed to achieve either coarser or finer map units by grouping or splitting map units respectively. Grouping map units is accomplished by aggregating similar map units or by choosing a higher or coarser level within a classification hierarchy (Table 3). One advantage of grouping map units is that it may improve the content accuracy because fewer and more broadly defined units can be mapped, thus minimizing potential error. Splitting map units to achieve higher thematic resolution requires more detailed ancillary data such as maps, plots, higher resolution imagery, or other geospatial data that can be used to distinguish units at a finer level than the original geospatial data layer.

Table 3. Hierarchy of LANDFIRE biophysical setting and existing vegetation type map units. Users can choose the level that best fits their needs or create a hybrid classification by aggregating units. Note that the Society of Americana Foresters and Society of Range Management map units that are provided in the existing vegetation type data layer attribute table is for reference only. This "cover type based" map unit classification is not equivalent to the NatureServe ecological systems classification used by LANDFIRE (see Chapter 4).

Data Layer	Map Unit Level	Example
Biophysical Settings	BpS Name	Central Mixed Grass Prairie
	Group Name	Bluebunch Wheatgrass-Big Bluestem-Little Bluestem-2
	Group Vegetation	Grassland
Existing Vegetation Type	EVT Name	Laurentian-Acadian Northern Hardwoods Forest
	System Group Physiognomy	Hardwood
	System Group Name	Yellow Birch-Sugar Maple Forest
	Society of American Foresters & Society of Range Management Cover Type	SAF 27: Sugar Maple
	National Vegetation Classification System Physiognomic Order	Tree-dominated
	National Vegetation Classification System Physiognomic Class	Closed tree canopy
	National Vegetation Classification System Physiognomic Subclass	Deciduous closed tree canopy

Reasons for Modifying LANDFIRE Data

The above considerations should be helpful in determining whether LANDFIRE data are appropriate for specific objectives and whether modifications are necessary. LANDFIRE geospatial data are commonly modified for the following reasons:

1. update for landscape changes that have occurred since the LANDFIRE version,
2. calibrate to local data and knowledge,
3. improve the thematic agreement (accuracy),
4. change the spatial or thematic resolution (e.g. lump or split map units),
5. modify the map unit classification,
6. create additional data versions that reflect temporal variability (e.g. peat soils being available for burning in drought situations, or exotic annual grasses being present in wet years but not dry years),
7. facilitate comparative analysis by creating data versions (e.g. analyzing pre- and post-treatment effects or comparing treatment alternatives),
8. analyze future conditions (e.g. modifying data to represent future conditions under a climate change scenario).

Conclusion

This chapter provided an overview of LANDFIRE data products, general considerations for critiquing LANDFIRE geospatial data products, and a list of common reasons why these geospatial data are modified for local applications. LANDFIRE's suite of products provides a rich set of data that have proven useful for addressing sub-regional, regional, and national level land management issues and research questions (e.g. Aycrigg et al. 2013, Cochrane et al. 2012, Reeves and Mitchell 2011, Swaty et al. 2011, Zhu et al. 2010). Through proper critique and modification by local natural resource and geospatial professionals, LANDFIRE data may also be appropriately applied to finer-scale, local applications. (e.g., Helmbrecht et al. 2012, Price et al. 2012, Scott et al. 2012, Tuhy et al. 2010). The importance of issues and the time and effort spent addressing them should be determined by the analysis objectives.

Chapter 2: Framework for Data Critique and Modification

This chapter presents a five-step conceptual framework for data critique and modification (Figure 3). The framework begins with defining objectives. Having a clear understanding of objectives will provide a foundation for the remaining steps of the framework. The process is iterative, as findings in one step of the framework may require reevaluation of a previous step. The framework is meant to be flexible and some steps may be combined, depending on the analysis objectives, processes being performed, and experience of the analyst. Certain tools may facilitate the integration of steps. For example, the LANDFIRE Total Fuel Change tool (LFTFCT 2011) allows the analyst to critique, modify, and analyze certain aspects of fuel mapping simultaneously. The framework is typically applied by a team, wherein specialists with expertise in various disciplines (e.g., fire/fuels, silviculture, ecology, and GIS) are involved in the process.

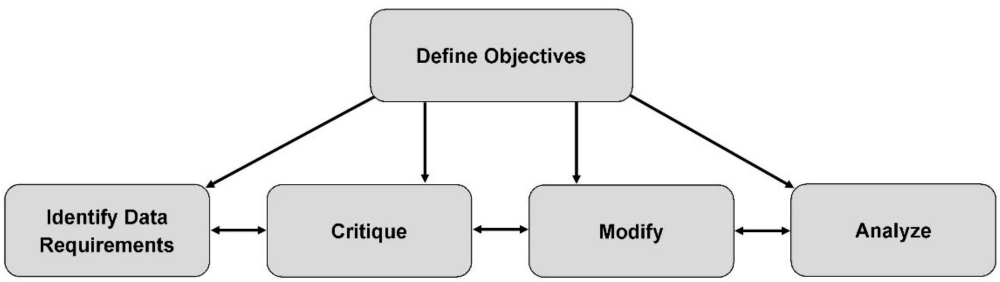


Figure 3. A conceptual framework for data critique and modification. The five-step framework begins with defining clear project objectives. The objectives will dictate the data requirements, influence the type of critique performed, dictate the types of modifications that are needed and determine the analysis performed. The framework is meant to be flexible and in some cases the process may be iterative.

Define objectives

The first step in the data critique and modification process is to define the team's objectives. Clear objectives will be a guiding principle for every other step in this process. For a given analysis determine what is needed from the data (and why), and its intended use. Defining objectives will help determine the data used, the landscape extent, the type of critique to do, and the type and level of modifications necessary.

Identify data requirements

With clear objectives in mind, the next step is to identify the data required to achieve those objectives. For example, if the objective is to assess vegetation departure from a historical reference condition, data is required that characterizes both the historical and current vegetation condition. If the objective is to assess potential wildland fire behavior, data is required that characterizes the fuels and topography of the area of interest.

As will be discussed in subsequent chapters, it is important to understand the linkages among LANDFIRE datasets, as well as the dataset creation method. Resolving issues with data that are mapped using a rule-based methodology, such as fire behavior fuel model or succession class, may require critiquing the data from which those data are derived, such as vegetation type, cover, and height or biophysical setting, thereby increasing the data requirements.

Critique

After identifying data requirements, the critical question is: are the data are good enough to meet the analysis objectives? Data need not be perfect to be useful. Ask what the important characteristics of the data are, and answer this question being mindful of the considerations discussed in Chapter 1. For the given objective: is the scale appropriate, are the data current, are the map units appropriate, and is the spatial resolution (pixel size) too coarse, or too fine? This is an iterative step; the critique may identify the need for additional data and that data will also need to be critiqued. For example, if the data are obsolete due to a recent disturbance, and that disturbance needs to be represented in the data to meet the objectives, then acquire and critique the disturbance data as it will be used to update the original data set.

Modify

Modification of data is the technical step and where GIS skills are mandatory. Subsequent chapters will provide examples of methods for conducting common modification tasks. This is also an iterative step. After modifying the data, critique it once again to be sure the desired result is achieved.

Analyze

The type of analysis performed is determined by the analysis objectives. Common analyses with LANDFIRE data include fire behavior modeling, vegetation departure assessment, and comparative analysis between land management alternatives. It is not uncommon for the results of a particular analysis to reveal data issues or requirements overlooked the first time through the framework. This step may be integrated with the previous step depending on both the analysis type and the experience of the analyst (Chapter 7).

Conclusion

This chapter presented a conceptual framework for critiquing LANDFIRE data for use in local applications. The following chapters discuss specific considerations for critiquing and modifying data from four of the five LANDFIRE data categories: disturbance, vegetation, fuels, and fire regime. Modification of topographic data (elevation, slope, and aspect) is uncommon and therefore not discussed in this guide; however, know that errors may still exist in these data. Having a thorough understanding of the assumptions and limitations of the data is of primary importance in data critique. Therefore, each of the following chapters begins with an overview of how LANDFIRE develops the data products of each category. Next are primary considerations for critiquing the data in each category and examples of why these considerations are important to local applications. Chapter 7 introduces common tools and techniques used for critiquing and modifying LANDFIRE data through interpreted examples.

Chapter 3: Disturbance

Landscape change due to planned and unplanned disturbances is continuously occurring across the United States. Updating LANDFIRE geospatial data for recent disturbances to vegetation and fuels is therefore a common modification task users of LANDFIRE data will encounter: this discussion of data critique and modification considerations thus begins with the disturbance data category. Additional considerations about updating for disturbance as it pertains specifically to vegetation, fuels, and fire regime data will be discussed in subsequent chapters.

LANDFIRE Disturbance Mapping Process

LANDFIRE maps the location, extent, type, and severity of both planned and unplanned disturbances. These data are used for determining vegetation transitions over time, and subsequently updating vegetation and fuel data products. As of LANDFIRE version 1.3.0 (LANDFIRE 2012), yearly geospatial disturbance data are available from 1999 through 2012. The yearly disturbance data are also compiled into two composite disturbance data layers—vegetation disturbance and fuel disturbance—representing disturbances occurring over the previous ten year time period. A time-since-disturbance attribute is recorded in the composite disturbance layers (Figure 4).

Yearly Disturbance Value Attribute Table

Rowid	VALUE *	COUNT	DIST_YEAR	DIST_TYPE	TYPE_CONFI	SEVERITY	SEV_CONFID	SOURCE1	SOURCE2	SOURCE3	SOURCE4	
3	13	13953	2009	Wildfire	High	Medium	High	MTBS				MTBS mapped wildfire.
4	14	6430	2009	Wildfire	High	High	High	MTBS				MTBS mapped wildfire.
5	15	156	2009	Wildfire	High	Increased Green	High	MTBS				MTBS mapped wildfire.
6	21	711	2009	Wildfire	High	Unburned/Low	High	BARC				BARC mapped wildfire.
7	22	94	2009	Wildfire	High	Low	High	BARC				BARC mapped wildfire.
8	23	32	2009	Wildfire	High	Medium	High	BARC				BARC mapped wildfire.
9	31	37	2009	Wildfire	High	Unburned/Low	High	RAVG				RAVG mapped wildfire.
10	32	2	2009	Wildfire	High	Low	High	RAVG				RAVG mapped wildfire.
11	33	1	2009	Wildfire	High	Medium	High	RAVG				RAVG mapped wildfire.
12	413	1	2009	Development	High	High	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Development Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
13	421	2834	2009	Clearcut	High	Low	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Clearcut Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
14	422	424	2009	Clearcut	High	Medium	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Clearcut Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
15	423	1749	2009	Clearcut	High	High	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Clearcut Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
16	431	6951	2009	Harvest	High	Low	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Harvest Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
17	432	1118	2009	Harvest	High	Medium	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Harvest Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
18	433	1793	2009	Harvest	High	High	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Harvest Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
19	441	19638	2009	Thinning	High	Low	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Thinning Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.
20	442	1535	2009	Thinning	High	Medium	High	Refresh Events	MIICA	dNBR		MIICA identified disturbance within Thinning Refresh Event perimeter. Severity determined by dNBR standard deviation breakpoints.

1 (0 out of 101 Selected)

Composite Disturbance Value Attribute Table

Rowid	VALUE *	COUNT	D_TYPE	D_SEVERITY	D_TIME	R	G	B	RED	GREEN	BLUE
0	0	2877793	No Disturbance	NA	NA	0	0	0	0	0	0
1	111	483654	Fire	Low	One Year	25	0	0	1	0	0
2	112	290323	Fire	Low	Two to Five Years	25	0	0	1	0	0
3	113	352163	Fire	Low	Six to Ten Years	25	0	0	1	0	0
4	121	186765	Fire	Moderate	One Year	25	0	0	1	0	0
5	122	105493	Fire	Moderate	Two to Five Years	25	0	0	1	0	0
6	123	121960	Fire	Moderate	Six to Ten Years	25	0	0	1	0	0
7	131	51118	Fire	High	One Year	25	0	0	1	0	0
8	132	64489	Fire	High	Two to Five Years	25	0	0	1	0	0
9	133	49453	Fire	High	Six to Ten Years	25	0	0	1	0	0
10	211	14359	Mechanical Add	Low	One Year	25	10	0	1	0.4	0
11	212	31130	Mechanical Add	Low	Two to Five Years	25	10	0	1	0.4	0
12	213	38477	Mechanical Add	Low	Six to Ten Years	25	10	0	1	0.4	0
13	221	263	Mechanical Add	Moderate	One Year	25	10	0	1	0.4	0
14	222	840	Mechanical Add	Moderate	Two to Five Years	25	10	0	1	0.4	0
15	223	4543	Mechanical Add	Moderate	Six to Ten Years	25	10	0	1	0.4	0

1 (0 out of 37 Selected)

Figure 4. Yearly and composite disturbance data attribute tables. The yearly disturbance data layers are attributed with the year, type, and severity of the disturbance as well as up to four input data sources, type and severity confidence levels, and a synopsis of the data and reasoning used to determine the map unit classification. The yearly disturbance data are compiled into a composite disturbance data layer. The

disturbance year (dist_year) is classified into a time-since-disturbance category (d_time) in the composite layer.

LANDFIRE disturbance data are developed through a multistep process that incorporates Landsat satellite imagery, disturbance polygons derived from local agencies, and other ancillary data. The remainder of this section provides a general overview of the LANDFIRE disturbance mapping process (Figure 5). More detailed information is available on the LANDFIRE website and in the literature cited below.

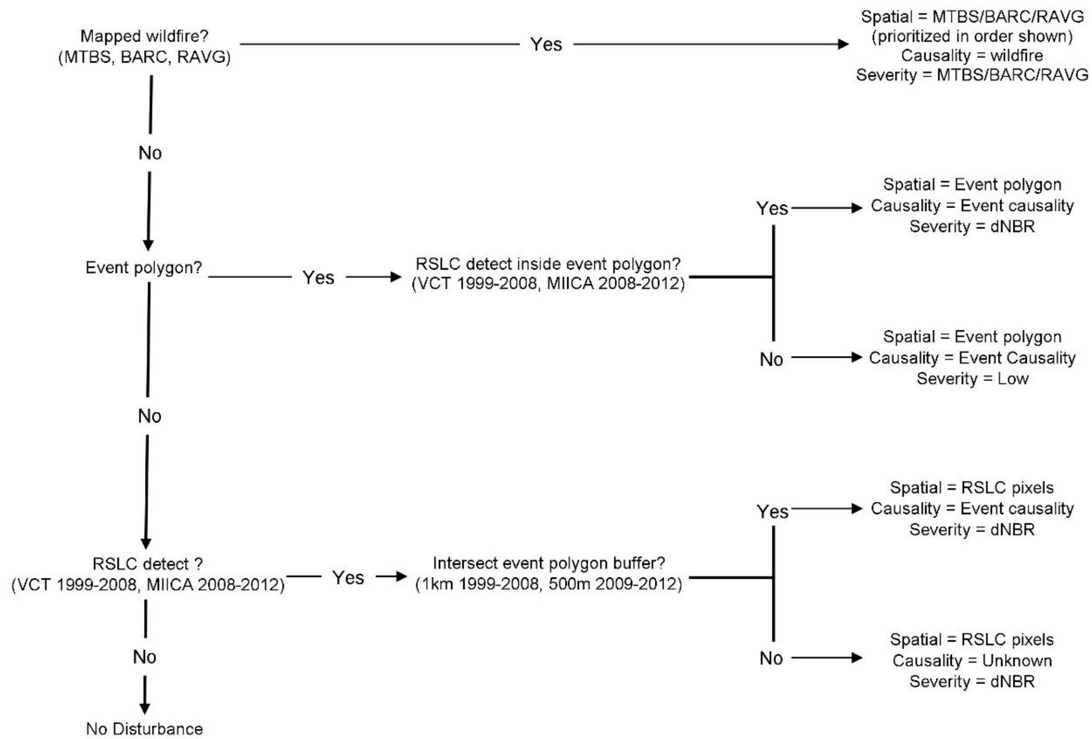


Figure 5. The LANDFIRE disturbance mapping process. LANDFIRE disturbance data are developed through a multistep process that incorporates Landsat satellite imagery, local agency derived disturbance polygons, and other ancillary data.

The first step in this process is to detect when and where disturbances have occurred. Three sources of information are used to accomplish this task: wildfire severity data from the Forest Service Remote Sensing Applications Center (RSAC), event polygons from the LANDFIRE events geodatabase, and change detection data derived from Landsat satellite imagery.

Wildfire Severity Data

RSAC manages three wildland fire severity mapping programs: Monitoring Trends in Burn Severity, Burned Area Emergency Response, and Rapid Assessment of Vegetation Condition after Wildfire. The data from these programs differ in the date of post-fire imagery and/or the severity mapping methodology used to create them. LANDFIRE uses all three datasets to map the extent and severity of wildfires.

LANDFIRE Events Data

Polygon data of vegetation and fuel management activities comprise the LANDFIRE events geodatabase. These data are obtained from federal, state, local, and private organizations and are compiled by LANDFIRE analysts. Events on national forest system lands rely heavily on data from the USDA Forest Service, Forest Activities Tracking System (FACTS). Regardless of the source, all events are crosswalked to one of 22 (including the exotic plants map unit) LANDFIRE event types (USFS 2013).

Change Detection

Lastly, LANDFIRE has incorporated two landscape change detection methodologies that apply Landsat satellite imagery in the development of the disturbance data. In the LANDFIRE 2008 mapping effort, a vegetation change and tracking process referred to as the Vegetation Change Tracker (VCT; Huang et al. 2010) was used. Beginning with the LANDFIRE 2010 mapping effort, the program adopted a new process called Multi-Index Integrated Change Analysis (MIICA; Jin et al. 2013). The MIICA process improves detection of disturbances in non-forest vegetation types, whereas VCT primarily identified disturbances in forested vegetation (D. Long, personal communication, July 23, 2013). MIICA was used to detect 2008 disturbances not identified through the VCT process, and all disturbances in 2009 through 2012. MIICA was not retroactively applied to the individual year disturbance data prior to 2008.

The second step in the disturbance mapping process is to assign causality, or disturbance type, to an identified disturbance. If the causality is know, that is, it is a mapped wildfire or LANDFIRE event, the causality is recorded in the yearly disturbance data attribute table. If the disturbance is identified through the change detection process, two additional sources of information are used to assign the likely causality: the National Gap Analysis Program's Protected Area Database and the USDA Forest Service, Pacific Northwest Research Station's SmartFire information system. Yearly disturbance layers are attributed with up to 19 of the 22 LANDFIRE event types plus an "unknown" class. This class indicates a disturbance occurred but the causality is uncertain (Table 4).

Table 4: Comparison of disturbance type attributes between LANDFIRE individual year and composite disturbance data layers. The composite vegetation disturbance (VDist) information is used to inform updates to the existing vegetation type data layer. The composite fuel disturbance (FDist) information is a subset of the VDist used to inform updates to fuel data layers.

LANDFIRE Event Type	Yearly Disturbance	VDist	Description	FDist	Applicable FDist Lifeforms
Wildfire	Wildfire	Fire	A catch-all term used to describe any non-structure fire that occurs in the wildland.	Fire	Herbaceous, Shrub, Tree
Wildland Fire Use	Wildland Fire Use				
Prescribed Fire	Prescribed Fire				
Wildland Fire	Wildland Fire				
Mastication	Mastication	Mechanical Add	A mechanical activity by which fuel is added to the natural fuelbed or in which the natural fuel structure is changed from a vertical to horizontal arrangement (e.g., mastication).	Mechanical Add	Shrub, Tree
Other Mechanical	Other Mechanical				
Clearcut	Clearcut	Mechanical Remove	A mechanical activity in which fuel is not added to the natural fuelbed (e.g., whole-tree harvesting) or in which natural fuels are removed.	Mechanical Remove	Shrub, Tree
Harvest	Harvest				
Thinning	Thinning				
Weather	Weather	Windthrow	Weather related event that results in loss of vegetation such as blowdown, hurricane, or tornado.	Windthrow	Tree
Insects	Insects	Insects-Disease	Infestations of insects and/or disease that can affect vegetative health and structure.	Insects-Disease	Shrub, Tree
Disease	Disease				
Insects/Disease	Insects/Disease				
Insecticide	Insecticide	Chemical	Application of a chemical substance such as herbicide.	NA	NA
Herbicide	Herbicide				

LANDFIRE Event Type	Yearly Disturbance	VDist	Description	FDist	Applicable FDist Lifeforms
Chemical	Chemical				
Biological	Biological	Biological	The use of living organisms, such as predators, parasites, and pathogens, to control weeds, pest insects, or diseases.	NA	NA
Development	Development	Development	Conversion of natural lands into housing, commercial, or industrial building sites. Involves permanent land clearing.	NA	NA
Exotic Plants	Exotics	Exotics	The presence of non-native species.	Exotics	Herbaceous, shrub
Planting	NA	NA	NA	NA	NA
Reforestation	NA	NA	NA	NA	NA
Seeding	NA	NA	NA	NA	NA
NA	Unknown	NA	Sources indicate that a disturbance occurred but causality is uncertain.	NA	NA

The final step in the disturbance mapping process is to map the disturbance severity. Information for determining disturbance severity may come from any one of the three data sources described above: RSAC wildfire severity, LANDFIRE events geodatabase, or remotely sensed change detection methods. Disturbance severity is assigned to one of three classes: low, moderate, or high (Table 5).

Table 5: LANDFIRE disturbance severity classes.

Severity	Description
Low	Less than 25% above-ground biomass removed.
Moderate	25 – 75% above-ground biomass removed.
High	Greater than 75% above-ground biomass removed.

The flow chart shown in Figure 5 may be used as an aid to understand this process. Where a wildfire has been mapped by one or more of the RSAC wildfire severity mapping programs, the information is used to determine the extent, year, causality (i.e., wildfire), and severity of the fire. In areas where a wildfire has not been mapped by one of the RSAC programs, but a LANDFIRE event has been mapped using other methods, the extent and causality of the event polygon are used. If the change detection process also detected the disturbance, severity is derived from the remote sensing data using the differenced Normalized Burn Ratio methodology (Key and Benson 2005). If no disturbance was detected via change detection, the year attributed to the event polygon is used and severity is set to low. Finally, if neither a wildfire or event is mapped to an area but a change is detected via remote sensing, the extent, year, and severity are determined by inference. This is done through analysis of the change detection data and assignment of causality based on proximity to event polygons and other ancillary data such as the National Gap Analysis Program's Protected Area Database and buffered SmartFire points. In addition to year, type, and severity, the yearly disturbance data layers are attributed with input data sources, type and severity confidence levels, and a synopsis of the data and reasoning used to determine the map unit classification (Figure 4).

The yearly disturbance data layers are then integrated into two composite data layers representing disturbances occurring over the previous ten year time period. In instances where multiple disturbances from different years overlap, the type and severity of the most recent disturbance is used in the composite data layer. An exception to this rule is in the case of a fire disturbance type (prescribed or wildfire) which overrides other disturbance types and is assigned to the composite layers regardless of when the fire occurred in the series of events. The disturbance type attribute of the yearly disturbance layers is reclassified into one of nine disturbance type map units in the final composite *vegetation* disturbance layer (Table 4). The year of the disturbance is classified into one of three time-since-disturbance classes: one year, two to five years, or six to ten years.

The composite *vegetation* disturbance layer is used to inform updates to the existing vegetation type, cover, and height data layers (Chapter 4). Both the yearly disturbance and composite vegetation disturbance layers are compiled from "raw" disturbance data. As such, direct comparison with existing vegetation data may reveal illogical combinations (e.g., fire and water mapped to the same pixel). When vegetation transition rules are applied to update the vegetation data layers, illogical combinations are filtered out.

The composite *fuel* disturbance layer is used to inform updates to fuel data layers (Chapter 5). The composite *fuel* disturbance layer is a subset of the composite *vegetation* disturbance layer and does not include chemical, biological, or development map units (see comparison in Table 4). The reasoning for this is that the composite fuel disturbance data layer is only applied in cases where both the post-disturbance vegetation characteristics *and* the disturbance that created those characteristics influence the post-disturbance fuels. For example, an herbicide application may cause a transition in vegetation type, cover, and/or height; and a fire behavior fuel model would be assigned based on these post-disturbance vegetation characteristics. The fact that the change was caused by the application of an herbicide does not factor into the assignment of the fuel model. This is in contrast to what would occur in a forested vegetation type after a wildfire, for example, where the post-disturbance vegetation characteristics *and* the fact that fire consumed dead wood and surface organic matter would both need to be taken into consideration in assigning the post-disturbance fuel model (Chapter 5). The composite fuel disturbance layer undergoes additional filtering to remove inconsistent disturbance/lifeform combinations (e.g., windthrow in herbaceous- or shrub-dominated landscapes, Table 4).

Considerations

Time-Since-Disturbance

LANDFIRE periodically updates the geospatial data products it develops to represent change due to disturbances; however, the update process itself takes two to three years to complete. Under the current update schedule, LANDFIRE data are typically 3–5 years out of date for any given year. In regards to the vegetation and fuel disturbance data layers, the time-since-disturbance attribute may therefore be out of date. For example, LANDFIRE 2012 data reflects conditions through the end of 2012. Thus, a disturbance that occurred in 2012 would be assigned to the one year time-since-disturbance class in the LANDFIRE 2012 data. However, the LANDFIRE 2012 data were released in the later months of 2014. For application in 2015, the original 2012 disturbance is 3 years old putting it in the 2–5 year time-since-disturbance class (Table 6). Likewise, disturbances that occurred in 2008 and 2009 should be shifted from the 2–5 year time-since-disturbance class to the 6–10 year class in 2014.

Table 6: Comparison of time-since-disturbance (TSD) between currency and release dates. For application in 2015, LANDFIRE 2012 disturbance data in the one year TSD class should be updated to the two to five year class. Likewise, disturbances that occurred in 2008 and 2009 should be updated to the six to 10 year TSD class.

Disturbance Year:	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
Data TSD (Years)	10	9	8	7	6	5	4	3	2	1	--	--
Current (TSD) Years	12	11	10	9	8	7	6	5	4	3	2	1

Whether this is of concern or not depends on the particular data products, their intended use, and the location of your assessment. For example, the assignment of fire behavior fuel model for use in fire behavior simulation is sensitive to the time-since-disturbance attribute. This is especially true in areas of the country where vegetation growth and fuel accumulation are rapid.

Disturbance Type

Disturbance type, or causality, is assigned to the vegetation disturbance and fuel disturbance data layers by pairing remote sensing data with information from the LANDFIRE Events Geodatabase. Individual disturbances are first classified into one of 22 LANDFIRE event types. Nineteen of these event types, plus an "unknown" class, are used to attribute the yearly disturbance data layers. The event types of the yearly layers are then reclassified into one of nine disturbance types in the composite vegetation disturbance layer and six types in the composite fuel disturbance layer (Table 4). Two disturbance types in particular—mechanical add and mechanical remove—can be especially challenging to assign from the information available in the events data but are very important for determining post-disturbance fuel. Whether the surface fuels (e.g., branches, needles, bark) generated from a forest management activity are added, rearranged, or removed from a site is highly dependent on factors such as site characteristics, management techniques, and management objectives. The management techniques and objectives are strongly influenced by law, regulations, and policies (local through national). These factors are highly variable in both location and time. For example, in more humid areas of the United States where downed wood decomposes quickly, activity fuel may be left on site to decompose and provide valuable nutrients to the soil. Conversely, in drier climates where this fuel takes years to decades to decompose, local, regional and/or national regulations or policy may dictate that activity fuel be removed from the site.

The information in the events data is typically not specific enough to discern these differences and LANDFIRE updates must therefore resort to the broad definitions of mechanical add/remove shown in Table 4. For local applications however, local resource professionals often have the institutional knowledge and/or ancillary information to critically critique, and update if necessary, disturbance type attributes.

Most Recent Disturbance Rule

As discussed above, in instances where multiple disturbances from different years overlap, the composite disturbance data layer is assigned the attributes of the most recent disturbance. The only exception to this rule is if fire is one of the disturbances, in which case the severity and time-since-disturbance of the fire is assigned to the composite layer regardless of when it occurred in the series of events. Multiple entries in the same treatment unit are quite common (e.g., a thinning treatment followed by treatment of activity fuels). In areas where timber harvesting is common, four or more entries may be found in short succession (e.g., a pre-treatment, one or more harvest entries, a fuel treatment, and site-preparation for planting or natural regeneration). A harvest treatment is also common, as timber salvage, after a fire.

In these situations the "most recent disturbance" rule, or "fire overrides other disturbances" rule, can lead to issues of content accuracy in the composite disturbance layers. For example, consider a high-severity harvest, such as a clearcut or shelterwood cut, followed by a low-severity disturbance, such as site-preparation or piling activity fuels. If these subsequent activities are at least a year apart, the composite data layer will be assigned "low-severity," even though all or most of the overstory vegetation was removed.

New Disturbances

The above considerations about time-since-disturbance and disturbance type attributes were presented in the context of critiquing disturbances that were already mapped and included in the LANDFIRE disturbance data products. As discussed previously, the composite disturbance data may be 3–5 years out of date upon time of version release. Updates are therefore often necessary, especially in actively managed landscapes or landscapes in which natural disturbances have occurred after the currency date of the latest LANDFIRE version. New disturbances may be added to the vegetation and fuel disturbance data layers using a variety of geospatial techniques and tools. The most appropriate technique may be influenced by the availability of recent disturbance data, the thematic and spatial detail of the data, and the experience of the analyst. For example, recent disturbance data may be in the form of a polygon shapefile depicting the location, extent, and type of disturbance without information on severity (e.g., locally developed prescribed fire burn unit map), or in the form of a raster data layer representing multiple classes of severity (Figure 6 e.g., RSAC wildfire severity data). Regardless of the techniques applied, new disturbances must be attributed with type, severity, and time-since-disturbance to be added to the vegetation disturbance and fuel disturbance data layers.

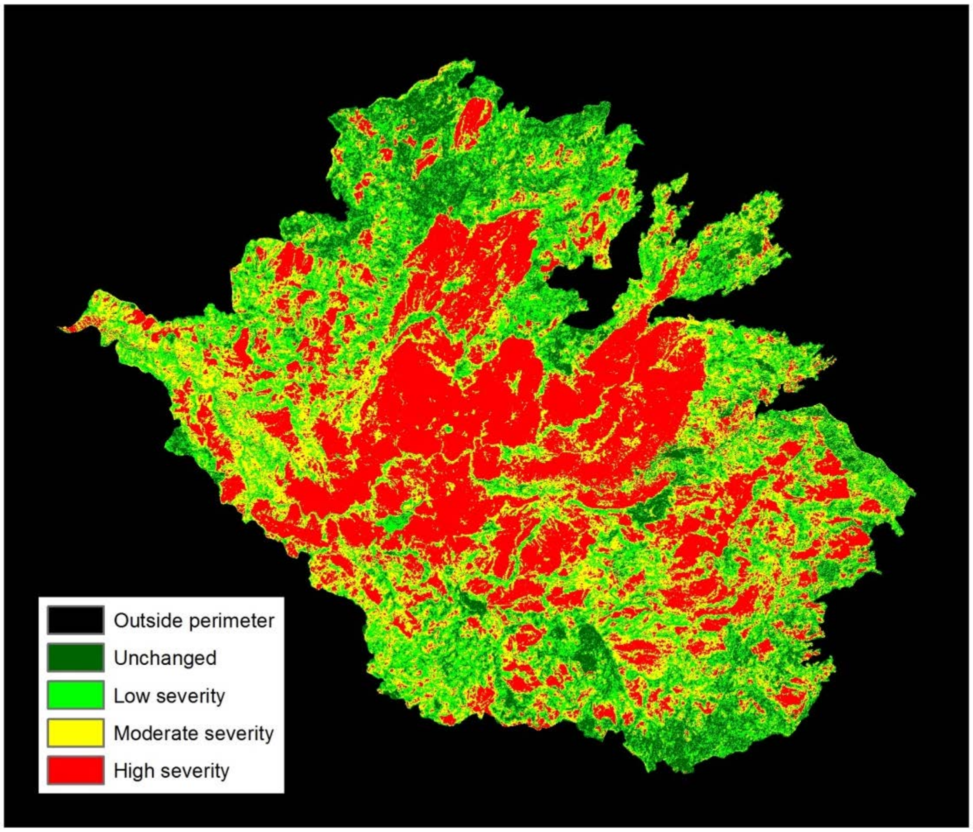


Figure 6. Four class severity classification of the 2013 Rim fire in California. Data were acquired from the U.S. Forest Service Remote Sensing Applications Center, Rapid Assessment of Vegetation Condition after Wildfire program.

Chapter 4: Vegetation

LANDFIRE develops geospatial data of potential and existing vegetation. The potential vegetation products include environmental site potential and biophysical setting. In contrast to the environmental site potential, the biophysical setting reflects potential for the historically dominant vegetation. The existing vegetation products include existing vegetation type, existing vegetation cover, existing vegetation height, and succession class. These six vegetation products are foundational to the development of other LANDFIRE geospatial data depicting fuel and fire regime characteristics.

This chapter presents an overview of the LANDFIRE vegetation mapping process, common considerations for critiquing LANDFIRE vegetation data, and examples of common pitfalls.

Vegetation Mapping Process

Potential Vegetation

Potential vegetation refers to the vegetation that could be supported at a given site based on the site's biophysical environment. LANDFIRE maps two representations of potential vegetation: environmental site potential and biophysical setting. Environmental site potential represents the late successional vegetation community that may become established at a site in the absence of disturbance. Biophysical setting represents the vegetation community that may have been dominant at a site prior to Euro-American settlement based on both the biophysical environment and an approximation of the historical disturbance regime.

Potential vegetation is mapped by LANDFIRE using a predictive modeling approach referred to generally, as *classification and regression tree* (CART; Figure 7) analysis, in conjunction with rule-based mapping techniques. First, field plot data (available in the LANDFIRE Reference Database; LFRDB [n.d.]), are keyed to environmental site potential classes based on the presence and abundance of indicator plant species that identify the biophysical conditions of the site. These plots are then intersected and attributed with information from biophysical gradient data layers (e.g., soil depth, average temperature, and average daily precipitation). The gradient layers are modeled from climate, soil, and topographic data and indirect topographic gradients such as elevation, slope, and indices of slope position. The information gathered from plot locations is then used as training data to develop the CART model—a statistical model used to predict a dependent variable (environmental site potential class) based on correlation with the independent variables (biophysical gradients). The CART model is then applied spatially to create a draft map of the environmental site potential of every pixel across the landscape based on combinations of the biophysical gradient data. The draft product is then refined using rule sets derived from the Nature Serve

Ecological Systems map unit descriptions and expert review.

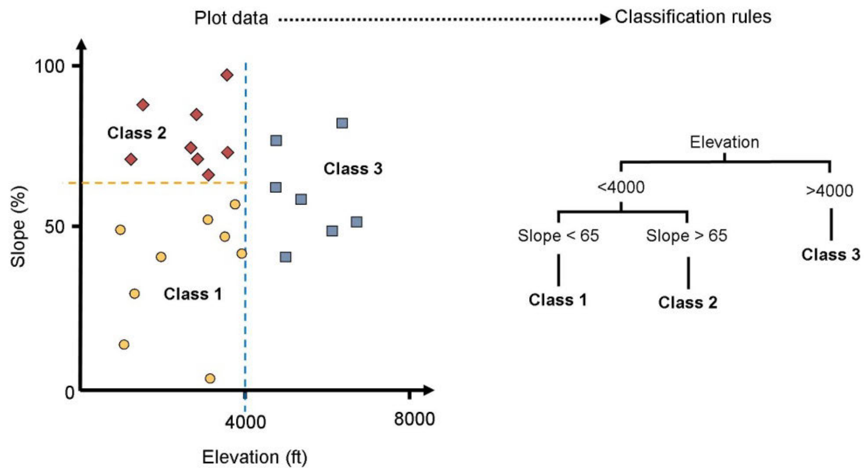


Figure 7. Classification tree conceptual diagram. In this simplified example, three classes of vegetation are plotted in respect to two biophysical gradients: elevation and slope (left side of figure). The relationship between the three vegetation classes and two biophysical gradients are then translated into classification rules (right side of figure), which are then in turn used to build spatial data layers. Approximately 40 biophysical gradients are used in the creation of the LANDFIRE potential vegetation data layers**.**

The environmental site potential data layer becomes the starting point for mapping Biophysical Settings. Environmental site potential units are associated with biophysical setting units using rule sets based on assumptions pertaining to vegetation dynamics and disturbance regimes. For example, an environmental site potential that is dominated by shade-tolerant species such as Douglas-fir or grand fir in the absence of disturbance may be mapped as a ponderosa pine- or western larch-dominated biophysical setting in an area with a frequent low-severity fire regime that would favor species that are less shade-tolerant and more fire-adapted. In other cases, alternate CART models were built to map biophysical settings from General Land Office survey data and Natural Resource Conservation Service Ecological Site Descriptions.

Existing Vegetation

Existing vegetation refers to the vegetation that is currently present on a given site. LANDFIRE maps four characteristics of existing vegetation: type, cover, height, and succession class. Existing vegetation is mapped using a predictive modeling approach similar to that used for potential vegetation; the primary difference is the input data. Like potential vegetation, methods for mapping existing vegetation type apply geospatial data of biophysical gradients and information from field plots. Because plot data can sometimes be many years old and vegetation characteristics may change rapidly, an additional filtering process is applied to ensure that current data are being used to develop the CART models. The existing vegetation type mapping process also includes data derived from Landsat satellite imagery as input. The base Landsat imagery used by LANDFIRE to derive existing vegetation products was acquired in the years 1999–2003, with newer imagery brought in to detect changes over time due to disturbance during the disturbance update process (Chapter 3).

Existing vegetation cover represents the area of the ground covered by a vertical projection of the canopy: in other words, the area of the ground covered if one were to look straight down from above (Figure 8). This is not to be confused with canopy closure, which is the proportion of the sky hemisphere obscured by vegetation when viewed from a single point (Jennings et al. 1999). Cover is mapped separately for herbaceous, shrub, and tree lifeforms using a predictive modeling approach based on plot data, satellite imagery, and biophysical gradient data layers. The canopy cover of each lifeform is binned into ten-percent classes[1] and then merged into a composite data layer in which the upper-layer lifeform's cover is assigned to the pixel. The training data for each lifeform are based on plot-level, ground assessments. However for the tree lifeform, plot canopy cover is modeled using a stem-mapping approach developed by Toney et al. (2009). This method was applied to the LANDFIRE 2001 data and is being applied to subsequent versions as an improvement over the canopy cover mapping in LANDFIRE National, which tended to over-predict tree canopy cover (Nelson et al. 2013, USFS [n.d.], Forest Canopy Cover…).

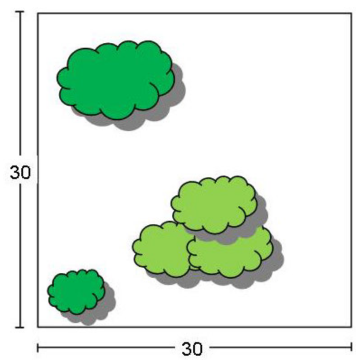


Figure 8. Vertically projected canopy cover. Existing vegetation cover represents the vertically projected canopy cover of the dominant lifeform for a pixel. In this example, the canopy cover within a 30-by-30 meter pixel is approximately 25%.

The existing vegetation height product represents the average height of the dominant lifeform. Like canopy cover, canopy height is mapped separately for herbaceous, shrub, and tree lifeforms using plot data, satellite imagery, and biophysical gradient data layers in a predictive modeling approach. The height of each lifeform is binned into classes and then merged into a composite data layer in which the upper-layer lifeform's height is assigned to the pixel (Table 7). For forests, a Shuttle Radar Topography Mission (SRTM) derived vegetation height product (Kellendorfer et al. 2004) is added to the other data sources for predictive modeling in LANDFIRE 2001 (Toney et al. 2012). The addition of the SRTM data provides a vertical structure measurement unavailable from the two-dimensional Landsat imagery which improved forest height mapping (Nelson et al. 2013, LANDFIRE 2008). Existing vegetation height for forests represents the average height of the dominant and co-dominant trees (weighted by basal area) for the pixel (Toney et al. 2012). In other words, the height value does not represent the average height of all individual trees, nor does it represent the average height of only the dominant trees. For non-forest areas, existing vegetation height represents the average height of the dominant lifeform. This is determined from species height weighted by species cover composition.

[1] For Alaska, tree and shrub cover is binned into three classes: 10%-25%, 26%-60%, and > 60%; herbaceous cover is binned into two classes: 10%-60% and > 60%.

Table 7: LANDFIRE height classes by lifeform and geographic area.

Lifeform:	Height Class (m) CONUS and HI	Height Class (m) Alaska
Herbaceous	0 - 0.5	0 - 0.5
	0.5 - 1	> 0.5
	> 1	
Shrub	0 - 0.5	0 - 0.5
	0.5 - 1	0.5 – 1.5
	1 – 3	> 1.5
	> 3	
Tree	0 - 5	No difference
	5 - 10	
	10 - 25	
	25 - 50	
	> 50	

The final characteristic of existing vegetation mapped by LANDFIRE is succession class. Succession class represents the current stage of vegetation development within an individual biophysical setting. It is very important to understand that the succession class should not be used independent of its biophysical setting. **Without its biophysical setting the succession class has no definition**. LANDFIRE maps up to seven succession classes using a rule-based approach—for each biophysical setting, a succession class is assigned based on rules that define specific combinations of existing vegetation type (primarily lifeform), existing canopy cover, and existing canopy height (Figure 9). Up to five of the seven succession classes are used to represent development stages characteristic of those found under the historical disturbance regime. Two classes are used to represent uncharacteristic conditions. Uncharacteristic native identifies native vegetation conditions that would be unlikely to occur under the historical disturbance regime, such as excessive canopy cover for a biophysical setting succession class with a frequent low-severity fire regime. Uncharacteristic exotic identifies areas in which exotic species have partially or completely replaced the native species.

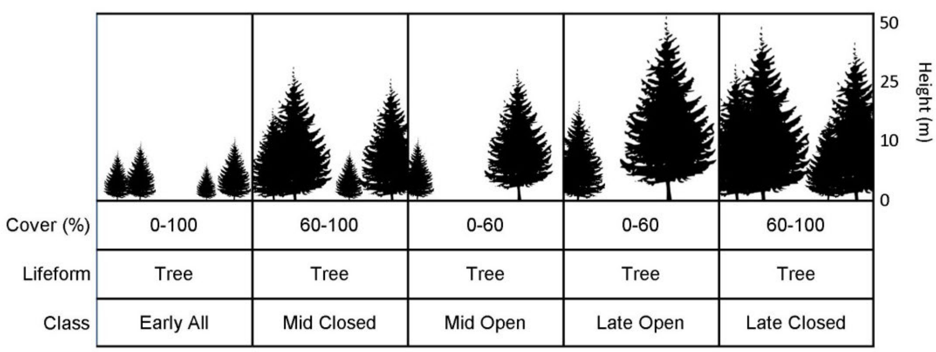


Figure 9. Typical five-class model for a forested biophysical setting, demonstrating succession class assignment based on vegetation characteristics. LANDFIRE maps up to seven succession classes using a rule-based approach—for each biophysical setting, a succession class is assigned based on rules that define specific combinations of existing vegetation type (primarily lifeform), existing canopy cover, and existing canopy height.

Updating Existing Vegetation

The existing vegetation products are periodically updated for changes due to disturbances and growth. The disturbance updating process was discussed in Chapter 3. Changes due to growth are incorporated in the mapping process after the disturbance update through a series of transition rules. Rules for updating the non-forest vegetation type for growth are developed based on the vegetation dynamics development models and the judgment of LANDFIRE analysts and other regional experts. Transition rules for forest vegetation type, cover, and height were developed based on forest growth simulations for Forest Inventory and Analysis plots modeled in the Forest Vegetation Simulator (FVS, Dixon 2002; Nelson et al. 2013). In Hawaii and Alaska (except southeast AK), where Forest Inventory and Analysis data are not available, forest transitions were developed by LANDFIRE analysts and other regional experts (Nelson et al. 2013). In LANDFIRE 2008 and 2010 the vegetation products were updated for both disturbances and growth. In LANDFIRE 2012, the vegetation products were updated for disturbance only. The transition rules are documented in databases available from the LANDFIRE Program website.

Considerations

One Classification, Three Interpretations

LANDFIRE uses the same map unit classification and naming system for the environmental site potential, biophysical setting, and existing vegetation type data layers. However, each of these layers must be interpreted differently, since they have different definitions and processing methods. A first step in identifying and mitigating possible vegetation type mapping issues (existing or potential) is to have a thorough understanding of this map unit classification and naming system and how it is used in the LANDFIRE existing vegetation type, environmental site potential, and biophysical setting data layers.

LANDFIRE uses NatureServe's Ecological Systems classification (Comer et al. 2003) as the primary map units and naming system for its existing vegetation type, environmental site potential, and biophysical setting products. The Ecological Systems classification units are intended to represent *existing* vegetation communities that persist for anywhere from 50 to hundreds of years, but LANDFIRE applies this concept in three different ways. In the existing vegetation type data layer, Ecological Systems are used as they were designed—to classify existing vegetation communities. For the environmental site potential data layer, LANDFIRE uses Ecological Systems to classify potential vegetation communities that could exist on a site given its biophysical characteristics in the absence of disturbance. Environmental site potential classes are modified to map LANDFIRE's biophysical setting concept which represents vegetation communities that may have been present prior to European-American settlement based on the biophysical environment and the historical disturbance regime. These are major differences and can have substantial effects on interpreting the data. For example, the same pixel classified as a Douglas-fir/grand fir forest environmental site potential based on the physical environment could be classified as a ponderosa pine forest biophysical setting, because of its historical fire regime, and a grass- or shrub-existing vegetation type due to a recent high-severity fire event. In rangeland, the same pixel classified as pinyon-juniper environmental site potential could be classified as a grassland biophysical setting, because of its historical fire regime, and a shrub existing vegetation type due to reduction in fire frequency.

Another important consideration specific to LANDFIRE existing vegetation type is that the NatureServe Ecological Systems map units represent vegetation communities that are typically comprised of groups of species. Most existing vegetation map users are more familiar with the concept of cover types. Cover types, in contrast to Nature Serve Ecological Systems map units, represent one or more dominant species at a single point in time. NatureServe Ecological Systems map units are not equivalent to cover types. The LANDFIRE existing vegetation type attribute table provides a crosswalk to the Society of American Foresters (SAF) and Society for Range Management (SRM) cover types classes as a guide to help users better understand LANDFIRE’s map units. However, because SAF and SRM map units represent cover types and LANDFIRE’s units represent systems, the crosswalk should not be interpreted as an exact match.

Potential vs. Existing Vegetation Type Rectification

The LANDFIRE potential vegetation data layers (environmental site potential and biophysical setting) were mapped using a predictive modeling approach based on plot data and biophysical gradient data layers, but did not incorporate imagery or use the existing vegetation type to modify the mapping process. This results in the potential vegetation data layers being inherently coarser in concept than the existing vegetation type data layer, which integrates Landsat satellite imagery. However, due to time and budgetary constraints, the LANDFIRE program has not been able to rectify either environmental site potential or biophysical setting with existing vegetation as to the inclusion or exclusion of specific existing vegetation types that would better depict site potential, thus improving content accuracy. Therefore, the user may find illogical combinations of these data layers and existing vegetation type for the same pixel, such as an existing vegetation type mapped to the same pixels as a biophysical setting that would not support the vegetation type due to moisture or topo-edaphic (i.e., soil) constraints. An example of this would be a riparian existing vegetation type, such as upper montane riparian, mapped to a non-riparian biophysical setting, such as sagebrush steppe. In the vegetation departure data products (Chapter 6) this situation may falsely indicate ecological departure. In these situations it can be difficult to determine which data layer is correct, but it is typically assumed that the existing vegetation type data layer is more likely to accurately depict the site because it integrates satellite imagery, and plot data go through additional filtering in its development.

Map Zone Boundaries

Because LANDFIRE vegetation data were mapped independently by map zone (Figure 2), differences or abrupt changes are sometimes found along map zone boundaries. For example, where map zone boundaries coincide with ecological division boundaries (Comer et al. 2003), there may be a change in the existing vegetation type map unit for similar vegetation types, such as Colorado Plateau pinyon-juniper woodland (Intermountain Basins ecological division) to Southern Rocky Mountain pinyon-juniper woodland (Rocky Mountain ecological division) (Figure 10). This does not necessarily indicate a mapping issue; however, secondary data products for which existing vegetation type is a variable in their mapping methodology—such as succession class (below), fire behavior fuel model (Chapter 5), and the fire regime and vegetation departure products (Chapter 6)—may be influenced by the difference in vegetation type map unit.

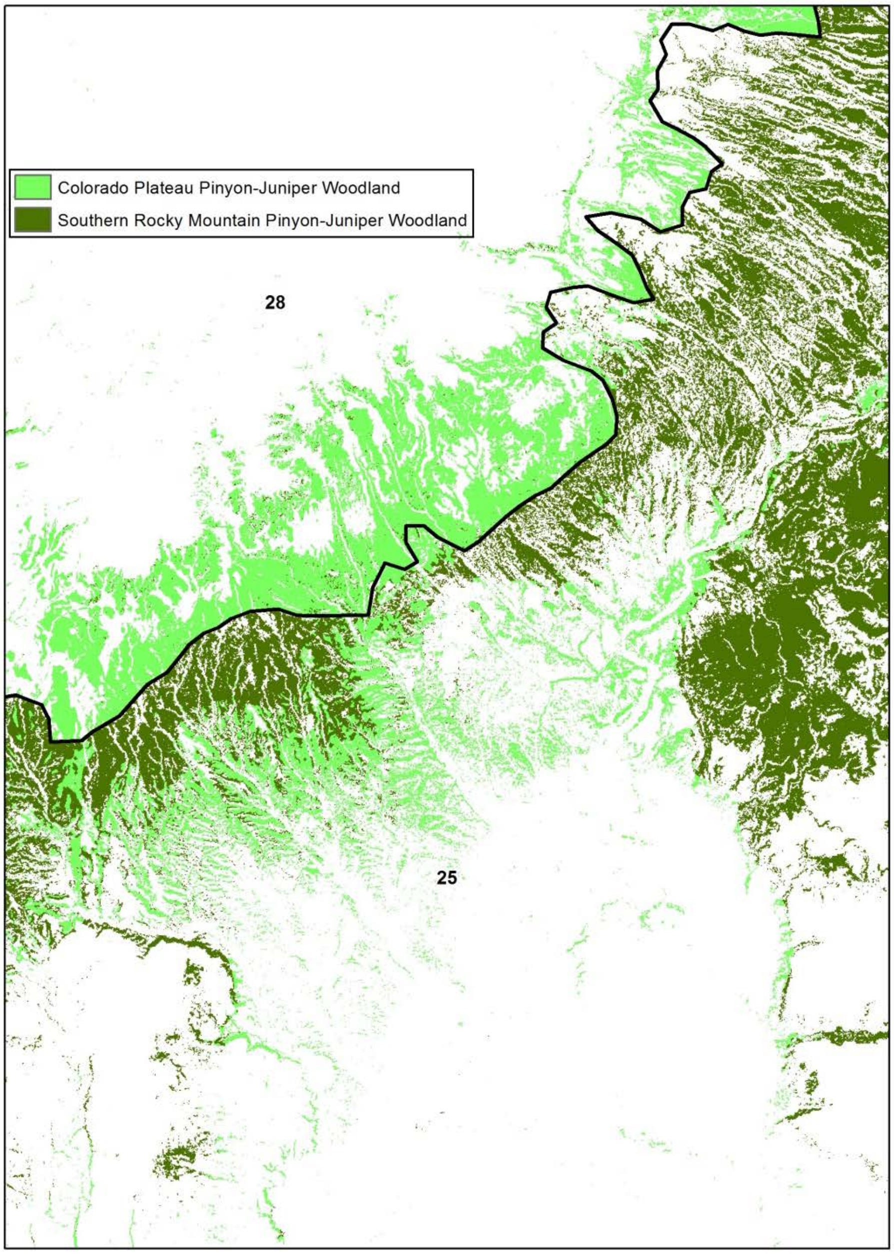


Figure 10. Comparison between the Colorado Plateau pinyon-juniper woodland existing vegetation type (Intermountain Basins ecological division) and the Southern Rocky Mountain pinyon-juniper woodland existing vegetation type (Rocky Mountain ecological division) at the map zone 25 and 28 map zone boundary. Secondary data products for which existing vegetation type is a variable in their mapping methodology may be influenced by the difference in vegetation map units at the boundary.

Existing vegetation cover is a primary variable in mapping secondary data products (i.e., succession class, fire behavior fuel model, and vegetation departure products). An abrupt change in vegetation cover within the same existing vegetation type is sometimes found at map zone boundaries (Figure 11). This may occur if the satellite imagery used for the adjacent zones was collected in different years and those years received significantly different amounts of precipitation, especially in dry southwestern ecosystems, or if different configurations of plot data were used between the zones (D. Long, personal communication, July 6, 2015). This may lead to an artificial demarcation in secondary data products and subsequently the results of analyses that use these products such as fire behavior and vegetation departure.

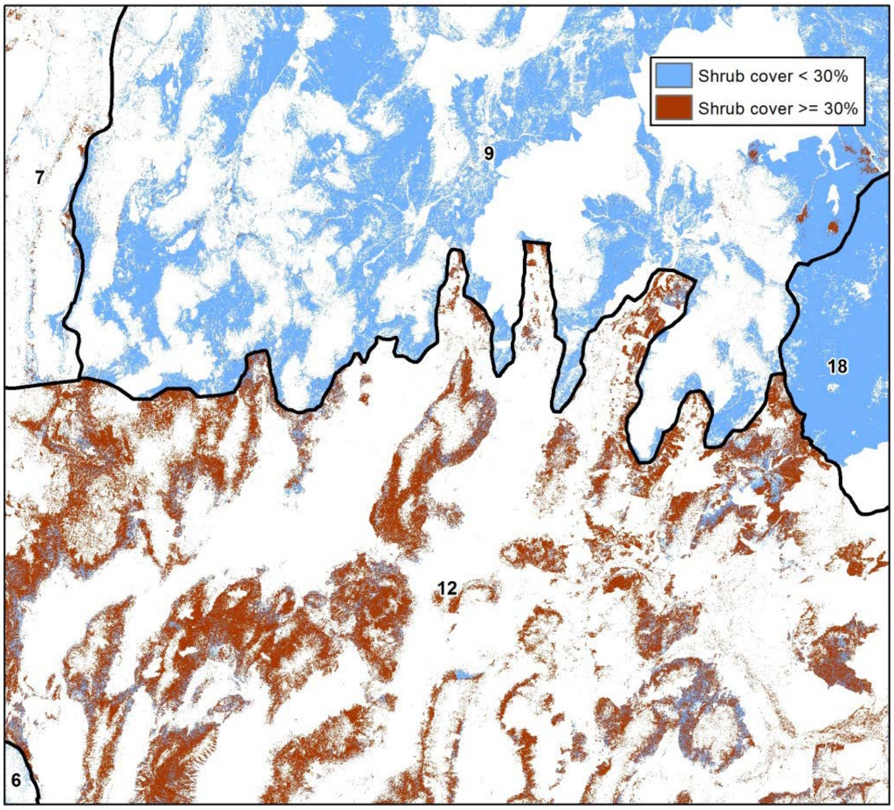


Figure 11. Abrupt change in canopy cover in the Inter-Mountain Basins Big Sagebrush Shrubland existing vegetation type at the boundary between map zones 9 and 12. This can have a profound effect on secondary data layers that use existing canopy cover as a mapping variable.

Succession Class Mapping Rules

LANDFIRE succession class is mapped using a rules-based approach. The mapping rules are based on unique combinations of biophysical setting and existing vegetation type, existing vegetation cover, and existing vegetation height. The rules were developed through a series of workshops by regional experts in vegetation dynamics and fire ecology (Rollins 2009) and are described in both the LANDFIRE vegetation dynamics model descriptions and vegetation dynamics model tracker database available for download from the LANDFIRE website.

One primary consideration in critiquing succession class mapping rules is that the modelers who developed the vegetation dynamics models sometimes emphasized species composition and structure in the definition of classes, while the mappers relied primarily on lifeform and structure to map the classes. As a result, in cases where species composition differentiates between classes of the same structure (Figure 12), LANDFIRE may not have mapped the succession classes appropriately. Post-processing in a GIS may be required to refine the succession class map based on species composition.

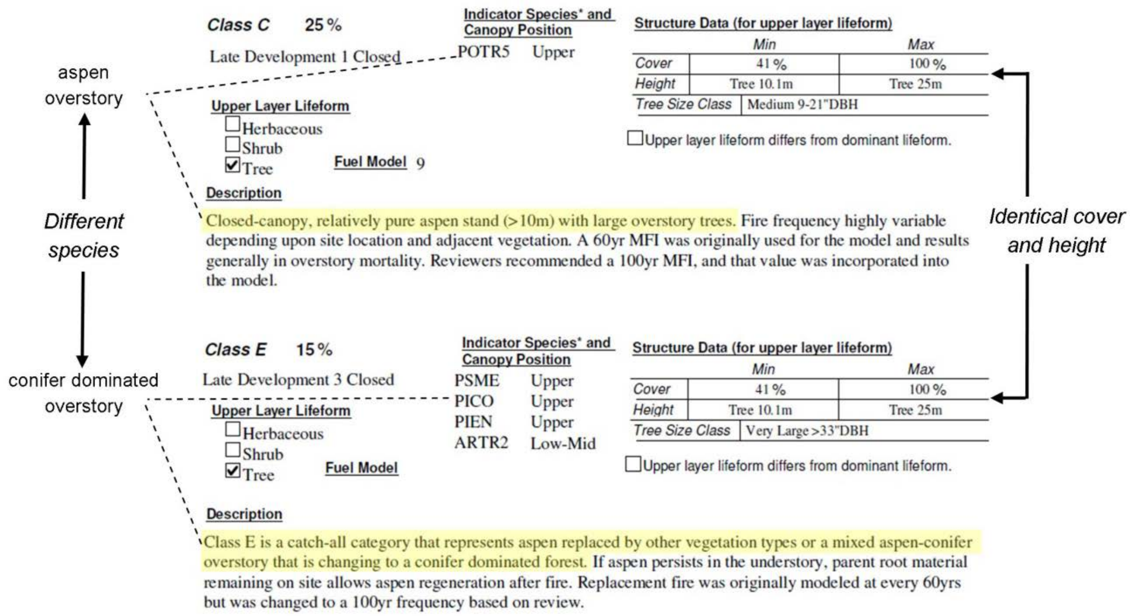


Figure 12. Example of a biophysical setting where species composition (not lifeform) is the primary variable for differentiating between succession classes. As of version 1.2.0 (LANDFIRE 2010) succession class E was not mapped for this biophysical setting in map zone 21 thus requiring GIS post-processing to map it.

Another consideration is that structure, as defined in the vegetation dynamics models, may be difficult to map using remote-sensing based techniques (as is done in mapping LANDFIRE existing vegetation). For example, although a rule may differentiate between succession classes based on whether herbaceous vegetation height is less than or greater than 0.5m, this level of precision is difficult to map accurately using the satellite-based predictive modeling approach described above (Riano et al. 2002). Conversely, the existing vegetation height classes in forested vegetation (Table 7) may be too coarse to accurately differentiate between succession classes (e.g., 10m to 25m and 25m to 50m) or a poor surrogate for vegetative development stage altogether. Chapter 6 contains additional considerations for using the LANDFIRE succession class data layer in vegetation departure analyses.

Chapter 5: Fuels

LANDFIRE produces geospatial data depicting surface and canopy fuel characteristics. For surface fuel data we will focus on the 40 Scott and Burgan fire behavior fuel models data layer (Scott and Burgan 2005), as it is the most commonly used LANDFIRE surface fuel data product. However, the concepts presented in the Considerations section of this chapter are applicable to the other LANDFIRE surface fuel products as well—13 Anderson fire behavior fuel models (Anderson 1982), Canadian forest fire danger rating system fuel types, fuel characteristic classification system fuelbeds, and fuel loading models.

In combination with forest canopy cover, forest canopy height, and topographic data (slope, aspect and elevation), LANDFIRE fire behavior fuel model and canopy fuel data (canopy base height and bulk density) can be used to create a “landscape” file (LCP) required by common geospatial fire behavior modeling systems used in the United States, such as FlamMap (Finney 2006), FARSITE (Finney 1998), and FSPro (USDAFS 2009). Although an LCP file may be downloaded directly from the LANDFIRE data distribution website, we do not discuss the critique or modification of these data in the LCP file format.

This chapter presents an overview of the LANDFIRE fuel mapping process and common considerations for critiquing LANDFIRE fuel data with examples relevant to local applications.

Fuel Mapping Process

Surface Fuels

Technically, a fire behavior fuel model—Anderson (1982) or Scott and Burgan (2005)—is a set of fuelbed inputs required by fire behavior modeling systems that use the Rothermel (1972) fire spread model. More practically speaking, a fire behavior fuel model represents a range of fuelbed conditions (e.g., load, depth, surface-area-to-volume ratios) in which fire behavior may be expected to respond similarly to changes in fuel moisture, slope, and wind speed (Figure 13). In this sense, a fire behavior fuel model is not so much a model of fuels as it is a model of fire behavior.

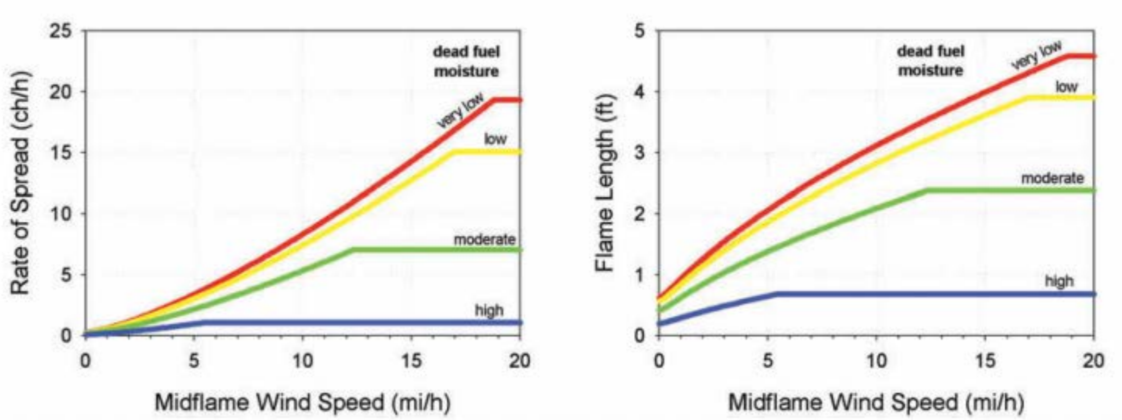


Figure 13. Differences in rate of spread and flame length by dead fuel moisture content and wind speed for fuel model Timber-Understory 1 (Low Load Dry Climate Timber-Grass-Shrub; Scott and Burgan 2005).

Like succession class (Chapter 4), all LANDFIRE surface fuel data products are mapped using an expert-opinion, rule-based approach, where mapping rules are based on unique combinations of: existing vegetation type, cover, and height; biophysical setting; and disturbance (Chapters 3 & 4). Fire behavior fuel model mapping rules were developed by fire and fuel specialists through a series of fuel calibration workshops held across the United States. The purpose of these workshops was to elicit regional expertise about fire behavior characteristics (i.e., how fire burns) in various vegetation types and structures. The calibration workshops were conducted at the extent of a LANDFIRE map zone or multiple adjacent zones. There are 80 LANDFIRE map zones across the continental U.S., Alaska, and Hawaii, ranging in size from 4 to 60 million acres (Figure 2).

The LANDFIRE total fuel change tool (formally known as ToFu Δ) (LFTFCT 2011) is a custom ArcGIS toolbar that links to the national fuel mapping rules through a Microsoft Access database (Figure 14). This tool, originally developed for use in the national calibration workshops, can now be downloaded from the [Wildland Fire Management Research, Development and Application] website and is highly useful in local LANDFIRE fuel data critiques.

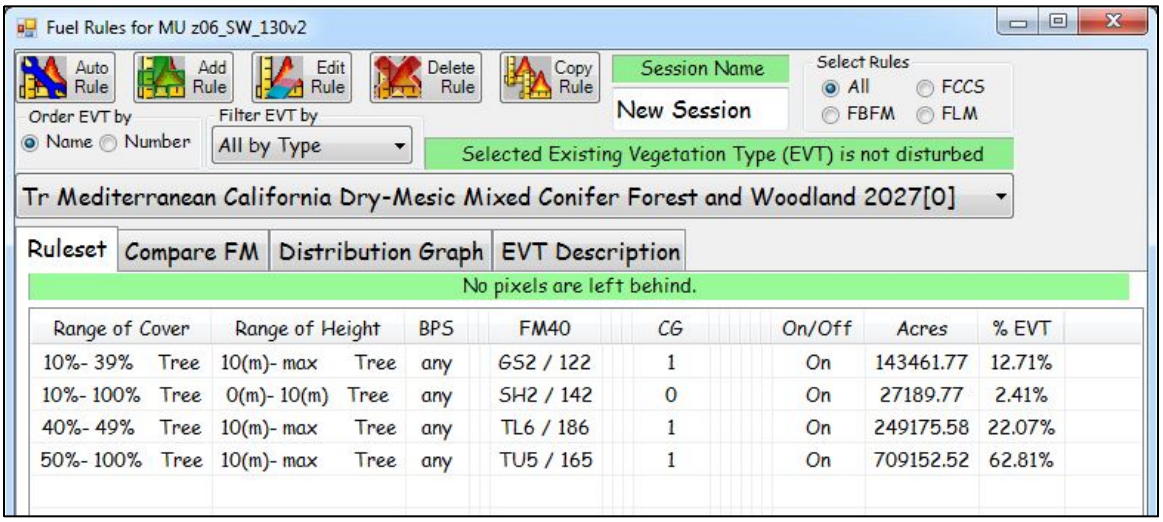


Figure 14. Example LANDFIRE Total Fuel Change Tool (LFTFCT) rule set. The LFTFCT is a custom ArcGIS toolbar that links to the LANDFIRE fuel mapping rules through a Microsoft Access database.

Canopy Fuels

The LANDFIRE canopy fuel data products include canopy base height, canopy bulk density, forest canopy cover, and forest canopy height. Forest canopy cover and canopy height values represent the midpoint of the existing vegetation cover and height data layer classes, respectively. These values are used in some fire behavior modeling systems as variables in predicting dead woody fuel moisture, wind reduction, and crown fire spotting potential. All four variables are required to model crown fire behavior using U.S. fire behavior modeling systems.

Canopy base height is defined as the lowest height above the ground at which there is sufficient available fuel (i.e., ≤ 0.25 inch diameter) to propagate fire vertically through the canopy. It is important to differentiate canopy base height—which is a property of the group of trees represented by the pixel—from *crown* base height, which is a property of an individual tree. Canopy base height is an important variable for fire behavior modeling, as it is used to predict whether crown fire initiation is possible under a given set of environmental conditions (Scott and Reinhardt 2001; Scott 2012). Prior to LANDFIRE 2012, canopy base height was mapped based on plot level averages. Various combinations of existing vegetation type, cover, and height values were crosswalked to an average canopy base height value of associated plots. For the LANDFIRE 2012 canopy base height data layer, a predictive modeling approach was implemented where field referenced plot data were used to develop regression equations based on statistical relationships between canopy base height and existing vegetation type, cover, and height (USGS 2010).

Canopy bulk density is the mass of available canopy fuel per unit canopy volume (Scott and Reinhardt 2001). Like canopy base height, canopy bulk density is a property of a group of trees—*crown* bulk density is a property of an individual tree. In fire behavior modeling, canopy bulk density is used to predict whether an active crown fire is possible under a given set of environmental conditions assuming that a crown fire has initiated (Scott and Reinhardt 2001; Scott 2012). LANDFIRE maps canopy bulk density using a predictive modeling approach based on forest canopy cover, forest canopy height, and membership to a pinyon-juniper existing vegetation type as input to a generalized linear model (Reeves et al. 2009).

In deciduous forest vegetation types—typically not considered prone to crown fire—LANDFIRE assigns canopy base height and canopy bulk density values that prevent fire behavior modeling systems from predicting crown fire. Forest canopy cover and forest canopy height values are still mapped to account for the canopy's effect on fuel moisture and wind reduction.

Fuel Updates

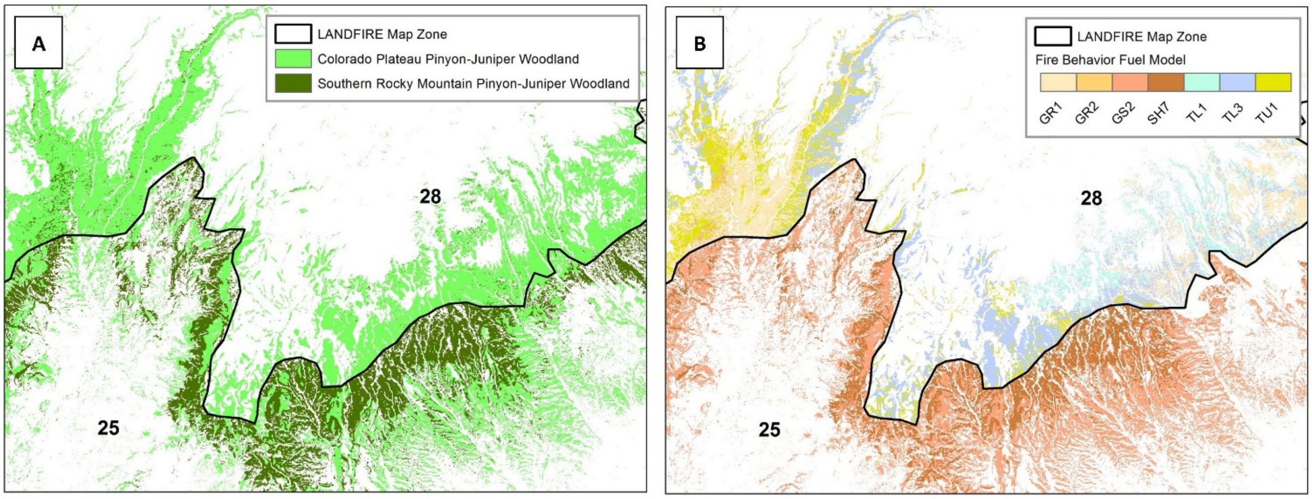
Because surface and canopy fuel mapping rules are tied to existing vegetation type, cover, and height, updates to existing vegetation data due to growth and vegetation succession automatically account for updates to fuels in non-disturbed areas. Whether an update to the fuel data layers occurs or not depends on the magnitude of the change and the threshold values in the mapping rules.

Rules for disturbed areas are independent of rules for non-disturbed areas so that the disturbance type, severity, and time-since-occurrence can be taken into account in combination with the post-disturbance vegetation characteristics, including unique lifeform and species specific disturbance response as discussed in the previous sections. The one-year time-since-disturbance category is used by LANDFIRE to update the immediate post-fire effects to canopy fuels but not used in the assignment of post-disturbance fire behavior fuel model. Fire behavior fuel model is the same for the one-year and two- to five- year time-since-disturbance categories, which are considered to represent the second growing season after the event (C. Martin, personal communication, July 10, 2015).

Considerations

Map Zone Boundaries

As mentioned earlier, fire behavior fuel model mapping rules are developed for individual map zones or groups of adjacent zones based on unique combinations of existing vegetation type, cover, and height; biophysical setting; and, disturbance. It is common to find differences in mapping rules between adjacent map zones that may lead to an "artificial edge" at the zone boundary (Figure 15). In situations where your analysis area overlaps more than one LANDFIRE map zone, a primary consideration is whether there are differences in mapping rules between the zones. If so, determine whether those differences are legitimate or if the rules from one zone more appropriately fit the analysis area as a whole. If working in an area with pinyon-juniper vegetation types, a specific mapping rule issue to watch for is whether or not there are differences between zones in the assignment of canopy fuels. In some cases, the rules for one map zone will consider the canopy fuels in pinyon-juniper vegetation types as part of the surface fuel model, while the rules for an adjacent map zone will not. This may lead to prediction of crown fire on one side of the zone boundary and surface fire on the other. The discrepancies are due to differences in mapping methodology rather than actual fire behavior potential. There may be valid reasons for each case but consistency should be strived for when an analysis area intersects multiple map zones, to ensure consistent interpretation of the results across the entire analysis area.



Existing Vegetation Type	Zone 28				Zone 25			
	Tree Cover (%)	Tree Height	FM	CG	Tree Cover (%)	Tree Height	FM	CG
Colorado Plateau Pinyon-Juniper Woodland	10-29	Any	GR1	1	10-19	Any	GR2	0
	30-39	Any	TU1	1	20-49	Any	GS2	0
	40-59	Any	TL3	1	50-100	Any	TL3	1
	60-100	Any	TL1	1	-	-	-	-
Southern Rocky Mountain Pinyon-Juniper Woodland	10-19	Any	GR2	1	10-29	Any	GS2	0
	20-59	Any	GS2	1	30-49	Any	SH7	0
	60-100	Any	TL3	1	50-100	Any	TL3	1

Figure 15. Example of variation in fire behavior fuel mapping rules by existing vegetation type and map zone. Panel A shows the existing vegetation type at the map zone boundary; panel B shows the fire behavior fuel model. FM refers to the standard Fire Behavior Fuel Model (Scott and Burgan 2005). CG refers to the canopy guide feature in the LANDFIRE Total Fuel Change Tool that controls how canopy fuels are mapped.

Multiple inconsistencies between map zones can be seen in Figure 15. The predominant pinyon-juniper existing vegetation type in map zone 28 is Colorado Plateau pinyon-juniper woodland; in map zone 25 it is southern Rocky Mountain pinyon-juniper woodland (Figure 15A). The fire behavior fuel model mapping rules for these two vegetation types vary both by type and by map zone, resulting in the obvious difference in fuel model seen at the boundary (Figure 15B). Furthermore, in map zone 28, the rules for both vegetation types include the assignment of canopy fuels (i.e., canopy guide of 1); in map zone 25 the rules do not assign canopy fuels to pixels with less than 50% canopy cover, indicating that the trees are part of the surface fuel stratum. This inconsistency forces a different interpretation of fire behavior modeling results for each map zone.

Application Scale and Location

As stated earlier, fire behavior fuel model mapping rules were developed at regional workshops for application to individual, or groups of adjacent, map zones. While these rules may be appropriate at this scale, they may need to be adjusted for application at finer scales. In other words, the "best fit" for an entire map zone may be a compromise between different parts of the zone. For finer-scale applications, fire behavior fuel model mapping rules should be locally critiqued whenever possible. We recommend doing this in a workshop setting, where local specialists with expertise in local fire behavior critique the national mapping rules and make adjustments as needed. Remember, the objective is to choose the fire behavior fuel model that most appropriately simulates the observed or expected fire behavior under a range of fire-environment conditions. It is therefore invaluable to have workshop participants who have seen fire burn under a range of conditions in the local vegetation types.

Another consideration common in more arid locations is whether the fuel models that are appropriate under a typical, or average, yearly weather scenario are appropriate in an atypical scenario. For example, in a typical year, fire behavior in many desert ecosystems may be best represented using a shrub fire behavior fuel model. However, in a year when an unusually wet winter is followed by an influx of annual grasses, the primary carrier of fire will be the herbaceous component and thus fire behavior would be better represented using a grass or grass-shrub fuel model. In this case, two separate versions of fuel data layers could be created to represent the different fuel scenarios.

Similarly, areas with a heavy deciduous tree component may experience very different fire behavior depending on the time of year. In fall, winter, and spring the leaves have fallen from deciduous trees, therefore adding to the load and structure of the surface fuels and associated surface fire behavior. As mentioned above, in deciduous forest vegetation types, LANDFIRE assigns pseudo canopy-fuel values that prohibit the simulation of crown fire in fire behavior modeling systems but retain the actual forest canopy cover and height values for modeling the influence of canopy cover on wind-reduction and fuel moisture. However, in mixed deciduous-conifer existing vegetation types LANDFIRE does not account for the proportion of deciduous-to-conifer cover; canopy bulk density is estimated from the total forest canopy cover. Depending on the proportion of conifer and deciduous trees, canopy bulk density may therefore be overestimated in these stands throughout the year, and wind-reduction and shading may be overestimated during the leaf-off times of the year.

Disturbance

Disturbances may affect both surface and canopy fuels depending on their type and severity. As with undisturbed fuels, the fuel mapping rules for disturbed areas should be critiqued by local fire specialists before application to finer-scale analyses.

In grass and shrub vegetation types the post-disturbance fire behavior fuel model is influenced by the affected species' response to disturbance. For example, wildfire in grass vegetation types is typically high-severity by nature—consuming all of the above-ground biomass. Most grasses, however, return to their pre-fire condition relatively quickly (i.e., one or two growing seasons) and in some cases will respond with increased biomass compared to the pre-fire condition due to an influx of nutrients and more favorable growing conditions. In shrub vegetation types, low-severity fire (less than 25% overstory mortality) may have little effect on the fuel load, fuelbed depth, and other components of a shrub-based fire behavior fuel model, whereas high-severity fire (greater than 75% overstory mortality) may result in immediate resprouting of shrubs or conversion to grass for some period of time, all dependent on the particular species' response to fire.

In tree-dominated vegetation types, low-severity fire will, generally speaking, consume litter (small dead branches and needles on the forest floor) and grass with minimal effect on understory shrubs and small trees. Moderate-severity fire may have a wide range of effects on litter and understory vegetation, but at the pixel level can generally be assumed to have consumed most of the litter and understory vegetation. By LANDFIRE severity definitions, moderate-severity fire in forested vegetation types results in 25% to 75% overstory tree mortality. High-severity fire results in greater than 75% mortality of the overstory trees. The mortality of overstory trees will influence the availability of light, water, and nutrients to understory vegetation, as well as contribute litter (through needle and branch fall) and large woody debris (as dead trees fall) as surface fuels over time.

These same principles apply to non-fire disturbance types. Ask yourself what is the response of the vegetation to the particular disturbance, what influences will this response have on post-disturbance fuel, how fire burns in the disturbed area, and what is the effect of time-since-disturbance.

As discussed in Chapter 3, the generalization of mechanical disturbance types to two categories—mechanical add and mechanical remove—may lead to a misrepresentation of effects. Critique of the LANDFIRE events polygon and individual year disturbance data by local experts can often confirm or provide additional information about the disturbance's effect on fuels.

The effect of time-since-disturbance varies by location and fuel type. Time-since-disturbance is split into three categories, the first of which is "one year". The need for the one year time-since-disturbance category can be evaluated based on your location, how you plan to apply the data, and how frequently you plan, or need, to update it.

Modeling

In-depth discussion of wildfire behavior modeling concepts is beyond the scope of this guidebook. Scott (2012) provides a comprehensive review of the topic in his *Introduction to Wildfire Behavior Modeling* guide. Nevertheless, a few considerations warrant discussion here. Wildfire behavior modeling requires an understanding of how the interaction among vegetation, fuels, and topography—as characterized in LANDFIRE data—influences modeling results. Wildfire analyst support may therefore be desired when critiquing and updating fuel data, depending on local wildfire behavior modeling expertise.

First, there is no direct, repeatable method for measuring canopy base height in the field, and multiple observers will often estimate significantly different values in the same stand. Methods exist to indirectly estimate canopy base height from plot data (Sando and Wick 1972; Cruz et al. 2003; Reinhardt and Crookston 2003; Scott and Reinhardt 2005), but canopy base height is challenging to map at a landscape scale because it is not well-related to characteristics that can be measured by remote sensing techniques. Canopy base height may include ladder fuels such as lichen, dead branches, needle drape, small trees, and shrubs. However, if shrubs and small trees are being considered as part of the fire behavior fuel model, they should not also be included in the canopy base height.

Next, understanding the interaction of fire behavior fuel model and canopy base height on modeling results is crucial in critiquing fuel data. The fire behavior fuel model predicts the surface fire intensity under a given set of environmental conditions (e.g., wind speed, slope steepness, fuel moisture). The lower the canopy base height, the milder these conditions can be in order to initiate crown fire. Given the difficulty of measuring and mapping canopy base height, working backwards— that is, adjusting canopy base height based on the conditions expected to initiate crown fire—is an effective way to critique canopy base height in relation to other variables. Tools such as NEXUS (Scott 1999) and BehavePlus (Andrews 2013) can provide information on the torching index—20' wind speed required for crown fire initiation—under various fuel and fire environments. The LFTFC tool also includes an option for calculating the critical canopy base height needed for crown fire initiation for different combinations of fire behavior fuel model, fuel moisture, and wind speed (Figure 16).

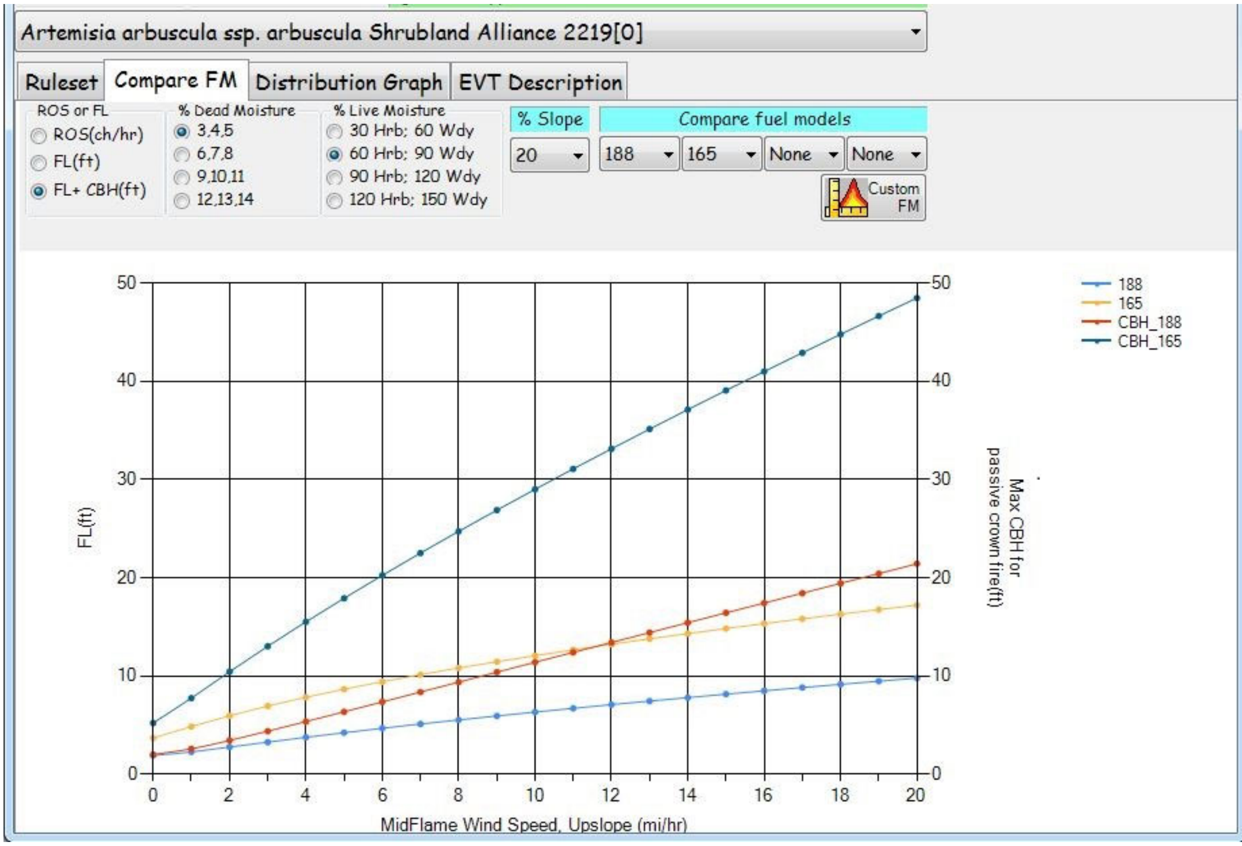


Figure 16. LANDFIRE Total Fuel Change Tool compare fuel model tab. This allows the user to calculate the critical canopy base height needed for crown fire initiation for different combinations of fire behavior fuel model, fuel moisture, and wind speed.

Lastly, in fire behavior modeling, canopy bulk density is a factor in determining whether an active crown fire can be sustained once initiated. Since the existing vegetation height classes used to predict canopy bulk density are rather coarse, they influence the resulting precision of the canopy bulk density values as well. Again, tools such as NEXUS and BehavePlus can be useful in determining if the data will predict the expected fire behavior under various conditions. Analysts are also encouraged to run geospatial fire behavior modeling systems to see if patterns in the results reveal any potential calibration issues that warrant a closer look. This is the *analyze* component of the data critique and modification framework discussed in Chapter 2.

Chapter 6: Fire Regime and Vegetation Departure

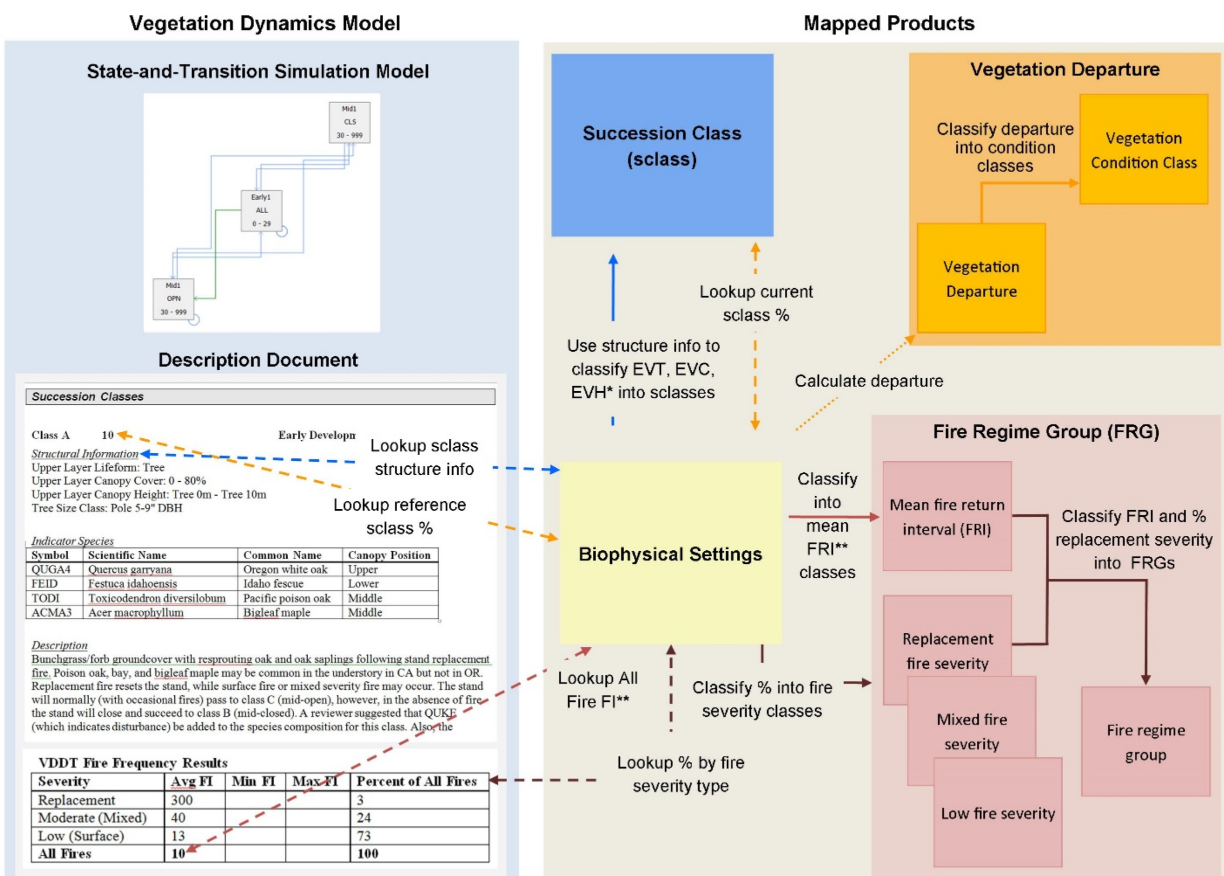
The fire regime and vegetation departure products are useful for understanding historical fire regimes and the current condition of vegetation on the landscape within the context of the historical disturbance regime. The fire regime products include fire regime group, mean fire return interval, percent low-severity fire, percent mixed-severity fire, and percent replacement-severity fire. The vegetation departure products include vegetation departure and vegetation condition class. The departure products were created for the LANDFIRE National, 2001, 2008 and 2012 versions but not for LANDFIRE 2010.

This chapter begins with an overview of the vegetation dynamics models, which form the basis of the fire regime and vegetation departure products, and then describes how those products are mapped by LANDFIRE. The chapter concludes by presenting common considerations for critiquing these data layers and provides examples of common pitfalls.

Fire Regime Mapping Process

Vegetation Dynamics Models

The foundation of the fire regime and vegetation departure products is a set of models that describe the vegetation dynamics and reference conditions of each biophysical setting mapped by LANDFIRE (Figure



*Existing vegetation type (EVT), cover (EVC) and height (EVH)

**Fire Interval (FI) and Fire Return Interval (FRI) refer to the average fire frequency modeled.

17). This section therefore begins with a brief overview of the models and how they relate to the fire regime and vegetation departure products. More information on the vegetation dynamics models can be found on the LANDFIRE program website.

Figure 17. The fire regime and vegetation departure products are created through crosswalks that link each BpS on the BpS data layer to the reference condition values modeled in the corresponding vegetation dynamics model.)

LANDFIRE collaborated with vegetation and fire ecology experts to create a vegetation dynamics model to estimate the reference (i.e., pre-Euro-American settlement) condition for each biophysical setting. The models were created in the Vegetation Dynamics Development Tool (VDDT, ESSA Technologies Ltd. 2007). A model represents a single biophysical setting and consists of five or fewer successional states, or classes, that compose the biophysical setting (Figure 18). Each state is equivalent to a succession class and each succession class is mapped in the succession class data layer (Chapter 4). A state has an age range that indicates how long it typically persists before it transitions to the next state. Disturbance pathways between states are used to represent the impact of important disturbances, and each pathway is defined by a probability that describes how often it occurs. The models were attributed based on scientific literature, available data, and the experience and judgment of the modelers (Rollins 2009).

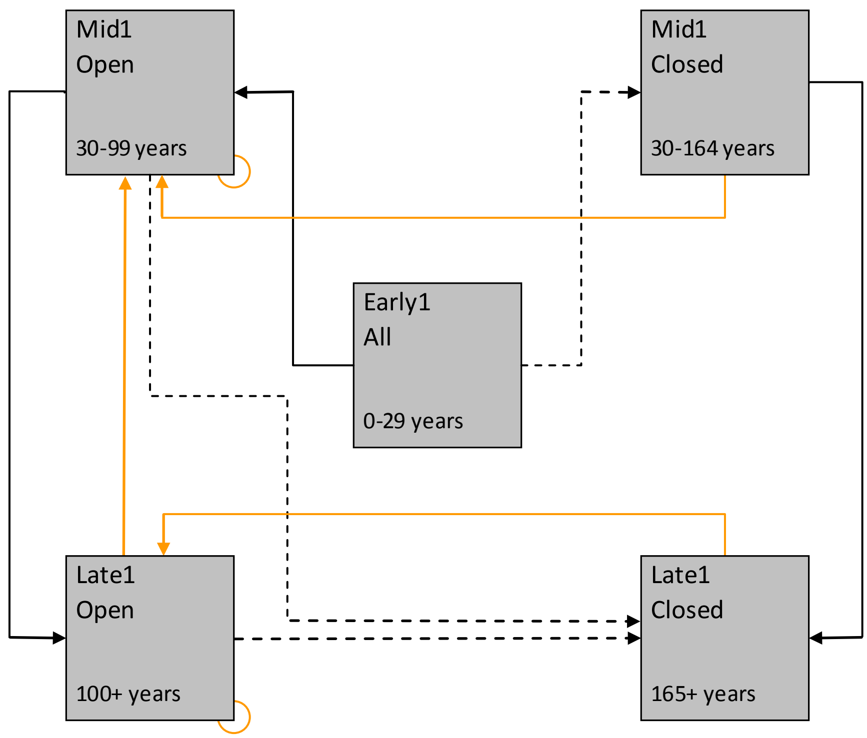


Figure 18. State-and-transition model of the Pacific Northwest Mixed Conifer BpS. This model is comprised of five successional states (boxes). Each state has an age range and is linked to other states through main successional pathways (solid lines), alternative succession pathways (dashed lines) and disturbance pathways (yellow line represent mixed fire transitions).

Once attributed, each model was run for ten 1,000-year simulations, in VDDT, and the results were averaged to estimate the biophysical setting reference conditions. The reference conditions include:

- the fire frequency expressed as a mean fire return interval,

- the fire severity expressed as the relative percent of low-, mixed-, and replacement-severity fire, and
- the relative amount represented by each succession class expressed as a percent.

The reference conditions are published in the LANDFIRE model descriptions along with the VDDT models available from the LANDFIRE website (Figure 19).

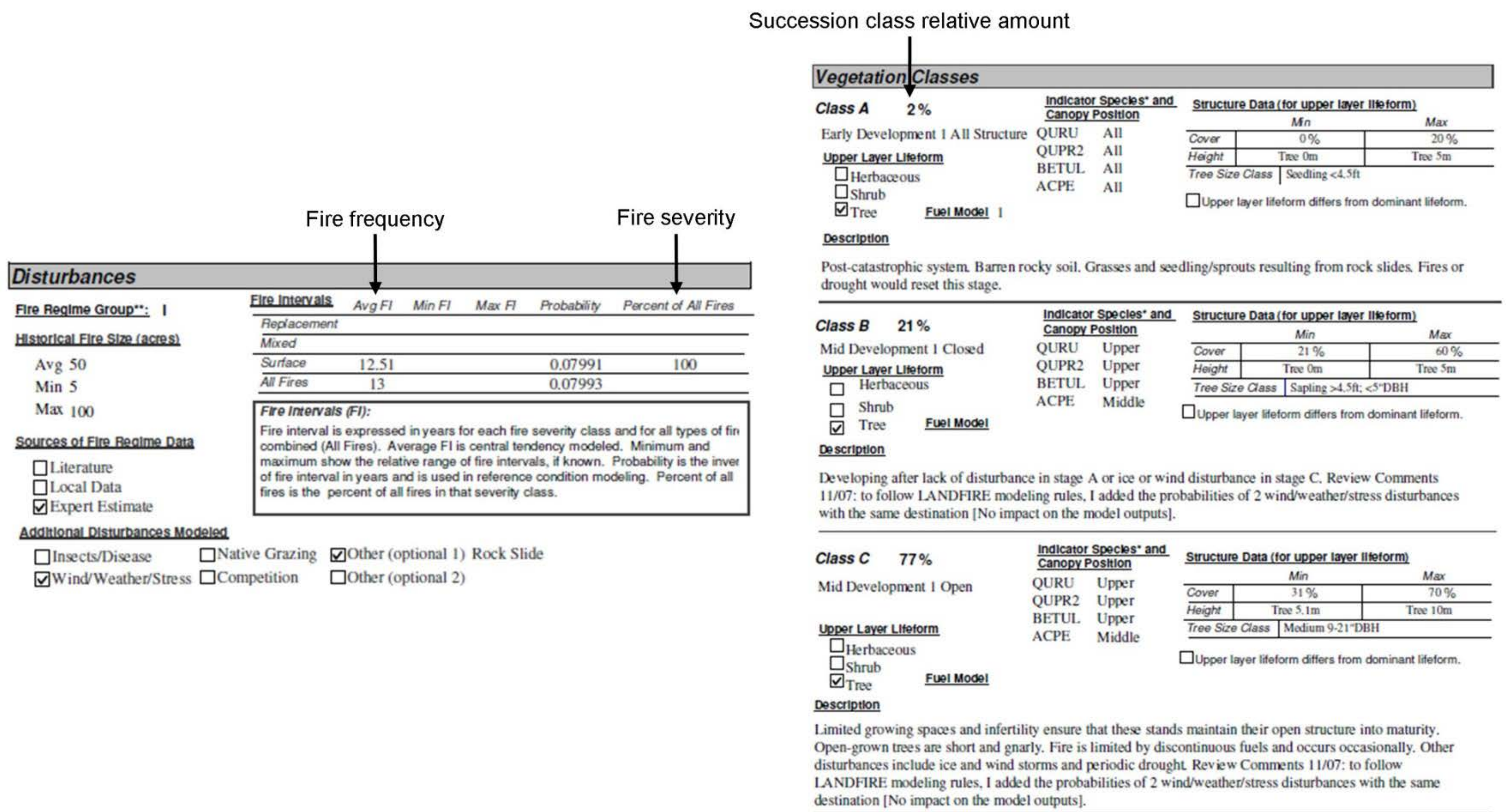
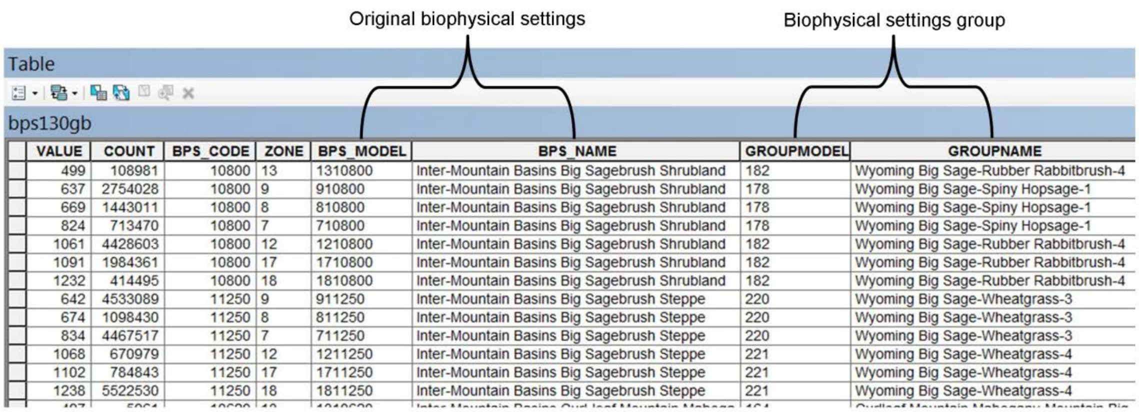


Figure 19. The biophysical setting model descriptions contain the reference conditions. The fire frequency and severity are found in the Disturbances section. The relative amount represented by each succession class is expressed as a percent and is found after the class letter name in the upper left of each vegetation class description.

Vegetation Dynamics Models and Biophysical Setting Map Units

The LANDFIRE biophysical setting data layer contains attributes for two nested map units: biophysical setting and biophysical setting groups. The biophysical setting attribute is the original biophysical setting classification based on NatureServe's Ecological Systems and described in the vegetation dynamics model description documents. LANDFIRE created biophysical setting groups to simplify the mapping of the fire regime products and to reduce the complexity of the vegetation dynamics model set for users. The original units were placed into groups based on similar vegetation and fire regime characteristics. Each biophysical setting group is represented by a single "exemplar" model chosen from the original model set. The fire regime products and the succession class data layer in LANDFIRE 2001 and 2008 are based on the biophysical setting groups and their associated exemplar models. All other LANDFIRE versions, including the most recent versions, use the original biophysical setting attribute. Although the biophysical setting and the biophysical setting groups are related, they can have different succession class definitions and different reference conditions, including different succession class proportions and fire frequency and severity values. Users need to pair the correct biophysical setting attribute in the biophysical setting data layer with the correct model based on the version of LANDFIRE data they are using. The relationship between biophysical setting and biophysical setting groups is described in the "BpS Groups Table" located on the LANDFIRE website (Figure 20).



VALUE	COUNT	BPS_CODE	ZONE	BPS_MODEL	BPS_NAME	GROUPMODEL	GROUPNAME
499	108981	10800	13	1310800	Inter-Mountain Basins Big Sagebrush Shrubland	182	Wyoming Big Sage-Rubber Rabbitbrush-4
637	2754028	10800	9	910800	Inter-Mountain Basins Big Sagebrush Shrubland	178	Wyoming Big Sage-Spiny Hopsage-1
669	1443011	10800	8	810800	Inter-Mountain Basins Big Sagebrush Shrubland	178	Wyoming Big Sage-Spiny Hopsage-1
824	713470	10800	7	710800	Inter-Mountain Basins Big Sagebrush Shrubland	178	Wyoming Big Sage-Spiny Hopsage-1
1061	4428603	10800	12	1210800	Inter-Mountain Basins Big Sagebrush Shrubland	182	Wyoming Big Sage-Rubber Rabbitbrush-4
1091	1984361	10800	17	1710800	Inter-Mountain Basins Big Sagebrush Shrubland	182	Wyoming Big Sage-Rubber Rabbitbrush-4
1232	414495	10800	18	1810800	Inter-Mountain Basins Big Sagebrush Shrubland	182	Wyoming Big Sage-Rubber Rabbitbrush-4
642	4533089	11250	9	911250	Inter-Mountain Basins Big Sagebrush Steppe	220	Wyoming Big Sage-Wheatgrass-3
674	1098430	11250	8	811250	Inter-Mountain Basins Big Sagebrush Steppe	220	Wyoming Big Sage-Wheatgrass-3
834	4467517	11250	7	711250	Inter-Mountain Basins Big Sagebrush Steppe	220	Wyoming Big Sage-Wheatgrass-3
1068	670979	11250	12	1211250	Inter-Mountain Basins Big Sagebrush Steppe	221	Wyoming Big Sage-Wheatgrass-4
1102	784843	11250	17	1711250	Inter-Mountain Basins Big Sagebrush Steppe	221	Wyoming Big Sage-Wheatgrass-4
1238	5522530	11250	18	1811250	Inter-Mountain Basins Big Sagebrush Steppe	221	Wyoming Big Sage-Wheatgrass-4

Figure 20. LANDFIRE biophysical settings were placed into groups based on similar vegetation and fire regime characteristics. Each biophysical setting group is represented by a single "exemplar" model chosen from the original model set. For example, in the table above notice that the seven original Inter-Mountain Basins Big Sagebrush Shrubland biophysical settings models were lumped into two groups: Wyoming Big Sage-Rubber Rabbitbrush-4 and Wyoming Big Sage-Spiny Hopsage-1.

Fire Regime: Frequency, Severity, and Fire Regime Group

The mean fire return interval data layer depicts the presumed historical fire frequency for each biophysical setting. The layer is created by linking the biophysical setting to the VDDT-modeled fire frequency results described in the vegetation dynamics model description document. The mean fire return interval is classified into 22 categories that vary in length to provide greater temporal resolution for frequently burned biophysical settings and less temporal resolution for biophysical settings that burn infrequently.

The fire severity data layers depict the relative percent of low-, mixed-, and replacement-severity fire under the presumed historical fire regime for each biophysical setting. Fire severity is defined as the percent mortality of the overstory vegetation: less than 25% mortality is classified as low-severity, 25-75% mortality is classified as mixed-severity, and greater than 75% mortality is classified as high-severity. The layer is created by linking the biophysical setting to the VDDT-modeled relative amount of each fire severity type as reported in the vegetation dynamics model description document. The results range from 0-100% and they are classified and mapped in 5% increments.

The fire regime group data layer characterizes the presumed historical fire frequency and percent replacement severity fire for each biophysical setting in five classes (Table 8). The fire regime group layer is created by linking the biophysical setting to the fire frequency and severity results described in the vegetation dynamics model description document.

Table 8: Fire regime group mapping rules. The fire regime group layer is created using a rule set that classifies combinations of fire frequency and relative percent replacement severity fire into one of five fire regime groups for each biophysical setting.

Fire Regime Group	All Fire Frequency (years)	Relative Percent Replacement Severity Fire
I	0-35	<66%
II	0-35	>=66%
III	36-100	<80%
	101-200	<66%
IV	36-100	>=80%
	101-200	>=66%
V	>=201	Any fire severity

All of the fire regime products include additional map units for water, snow/ice, barren, and sparsely vegetated systems which are mapped from the existing vegetation type data layer. The value "indeterminate fire regime characteristics" identifies a biophysical setting without fire disturbance in its associated vegetation dynamics model. These are typically biophysical settings that are either too wet or too dry to carry fire (e.g., Alaskan Pacific Maritime Sitka Spruce Forest biophysical setting).

Vegetation Departure

LANDFIRE provides geospatial data that characterize two metrics of vegetation departure: stratum vegetation departure and stratum vegetation condition class. Vegetation departure and vegetation condition class are calculated following the methodology described in the FRCC Guidebook (Barrett et al. 2010) and the FRCC Mapping Tool User's Guide (Jones and Ryan 2012). Both metrics describe the overall departure of the current vegetation conditions from the historical, or reference, vegetation conditions across all succession classes within a particular biophysical setting (i.e., stratum). The historical proportion, or relative amount, of each succession class in a biophysical setting is based on the average proportion modeled in the vegetation dynamics model and reported in the model description document (Figure 17). Current succession class proportions are calculated directly from the succession class data layer.

Stratum vegetation departure is calculated by determining the succession class "similarity" (the smaller of the reference, or the current proportion, for each succession class), summing the similarities, and then subtracting from 100. This provides the percent departure for a biophysical setting and that value is mapped in the vegetation departure data layer. To create the vegetation condition class data layer, the percent departure is classified into three classes: 0-33% departure in condition class 1, 34-66% departure in condition class 2, and 67-100% departure in condition class 3 (see sidebar).

Calculating Vegetation Departure

Stratum vegetation departure is calculated by comparing the reference distribution of succession classes (i.e., the proportion that each contributes to the whole expressed as a percent) to the current distribution of succession class for individual biophysical settings. In the table below, departure is calculated for a biophysical setting with three reference succession classes (A, B, and C), which are defined in its vegetation dynamics model. The Uncharacteristic succession class only includes a current value because by definition it does not occur under the reference condition. The uncharacteristic class proportion is the sum of the uncharacteristic native and uncharacteristic exotic proportions.

The first step in calculating stratum vegetation departure is to determine the succession class similarity (i.e., the lower of the reference or current percent) of each succession class. Next, stratum similarity is calculated by summing the succession class similarity values. Then, the current stratum vegetation departure is calculated by subtracting the stratum similarity value from 100. This is the value mapped in the LANDFIRE vegetation departure grid. Finally, the vegetation condition class is calculated by classifying the current stratum vegetation departure value into the three condition classes (1 = ≤ 33%, 2 = > 33% to ≤ 66%, 3 = > 66%). This is the value mapped in the LANDFIRE Vegetation Condition Class grid:

Succession Class (S-Class)	Reference Percent	Current Percent	S-Class Similarity
A-Early	15	3	3
B-Mid	40	25	25
C-Late	45	31	31
Uncharacteristic	0		
Stratum Similarity	**59**		
Current Departure	**41**		
Vegetation Condition Class	**2**		

Departure is calculated for a specific geographic area referred to as the landscape summary unit. For LANDFIRE National, departure was calculated for ecological subsections (Cleland et al. 2005) within a LANDFIRE map zone. In LANDFIRE 2001 and 2008 departure was calculated within nested hydrologic unit codes (HUCs; Seaber et al. 1987). Departure for biophysical settings in fire regime groups I and II was calculated at the sub-watershed level (HUC 12); biophysical settings in fire regime group III were calculated at the watershed level (HUC 10); and biophysical settings in fire regime groups IV and V were calculated at the sub-basin level (HUC 8). In LANDFIRE 2012 the landscape summary unit was defined as a biophysical setting with identical reference condition values regardless of map zone. To understand this, imagine that a biophysical setting is mapped in map zones 1, 2, and 3 and that zones 1 and 2 have identical reference conditions in their associated vegetation dynamics models but that zone 3 has a unique set of reference conditions. In this case, the departure would be calculated using the biophysical setting's extent in zones 1 and 2 as one summary unit and zone 3 as another summary unit.

Considerations

Understanding the Source Data

All of the fire regime and vegetation departure products are derived from other LANDFIRE products. Any assumptions, limitations, and issues associated with the source data are inherited by the fire regime and vegetation departure products. To understand and critique these products, the user must therefore understand the source data. The fire frequency, fire severity, and fire regime group values come from the vegetation dynamics models. Vegetation departure and vegetation condition class results are derived from the modeled reference conditions, the biophysical setting data layer, the succession class data layer, (which is itself derived from the biophysical setting, existing vegetation type, existing vegetation cover, and existing vegetation height data layers; see Chapter 4), and the landscape summary unit. The information from other chapters in this guide will help the user critique these geospatial data inputs. For more information on critiquing the vegetation dynamics models, refer to the *Reviewing and Modifying LANDFIRE Vegetation Dynamics Models* (The Nature Conservancy 2011a) user's guide.

Knowledge Uncertainty

The quality of the fire regime and vegetation departure products depends to a great extent on the quality and quantity of the information used to create the vegetation dynamics models. In general, there are more data to attribute models for economically valuable and heavily studied biophysical settings, such as forested ecosystems, than there are for biophysical settings with little economic value and those that are rare (Blankenship et al.2012). The quantity and quality of fire regime information also varies considerably based on the characteristics of the vegetation comprising the biophysical setting. Fire history from recent centuries tends to be most reliably documented in systems where the evidence of low- and moderate-severity fires is recorded and persists within the annual rings of long-lived tree species (Swetnam et al. 1999) such as longleaf pine and ponderosa pine, and/or where the time since the last stand-replacing fire can be determined from the stand age. In non-forested systems, little direct evidence persists for inferring the characteristics of historical fire regimes (Swetnam et al. 1999) although historical records, charcoal and pollen records, and dependence or sensitivity of long-persisting species provide clues to the fire frequency and severity. The vegetation dynamics model description documents often provide information about the sources and the quality of the information on which they are based and can provide users with valuable information for evaluating the fire regime products derived from them.

Map Zone Boundaries

The vegetation dynamics models were developed to apply at the level of a LANDFIRE map zone (Figure 2). Sometimes the same biophysical setting may have different succession class mapping rules, succession class reference proportions, and fire frequency and severity information in different map zones. This can lead to abrupt changes in the fire regime and vegetation departure products at map zone boundaries, even for the same biophysical setting. Users performing an independent departure analysis can address this issue (see Vegetation Departure Analysis below).

Changes in Departure Methods

The methods LANDFIRE used to create the departure products have changed between versions (Table 9). Users should be cautious when comparing the departure products (vegetation departure and vegetation condition class) between different LANDFIRE versions because changes in the biophysical setting map units and the landscape summary unit discussed above, as well as the source of the reference conditions, can change the departure score. Theoretically the LANDFIRE 2001 and 2008 departure data layers are comparable because they were calculated using the same method, but it may be too short a time period to see substantial change across broad areas. LANDFIRE 2001, 2008 and 2012 departure data layers are not comparable to LANDFIRE National because of the changes in the methods (USFS [n.d.] Fire Regime Data…).

Table 9: Comparison of the methods and input data used to create the LANDFIRE departure data products by data version.

Version	Departure Products Mapped	Departure BpS Unit[a]	Summary Unit	Reference Condition Source[b]
National	Yes	BpS	Ecological Subsections within Mapzones	VDDT & LANDSUM
2001	Yes	BpS Group	Nested Hydrologic Unit Codes	VDDT
2008	Yes	BpS Group	Nested Hydrologic Unit Codes	VDDT
2012	No	BpS	Unique Combination of BpS Code and BpS Model	VDDT

[a]Vegetation departure products were calculated for the biophysical setting (BpS) or the BpS groups depending on the version. In versions where departure products were not mapped, the Departure BpS Unit refers to the units used to map the fire regime and succession class layers.
[b]The reference conditions were derived from the Vegetation Dynamics Development Tool (VDDT) and the Landscape Succession Model (LANDSUM). For LANDFIRE versions 2001 and greater the reference condition source is as described in this guide. The reference conditions source for the National version is described in the document "Developing the LANDFIRE Fire Regime Data Products" on the LANDFIRE Program website.

Vegetation Departure Analysis

Rather than use the LANDFIRE vegetation departure products as-is, many users prefer to complete their own, local, departure analysis. Performing an independent departure analysis allows users to address the issues discussed above, critique and refine the succession class mapping rules, and integrate ancillary data (e.g., locally mapped invasive species distribution). The Fire Regime Condition Class Mapping Tool also allows for the calculation of additional vegetation departure metrics beyond stratum vegetation departure and stratum vegetation condition class, as well as fire *regime* departure analysis. In addition to the considerations listed above, there are some considerations specific to an independent departure analysis using LANDFIRE data.

Biophysical Setting Thematic Resolution

Users performing an independent departure analysis may want to consider the thematic resolution (Chapter 1) of the biophysical setting data layer in relation to their analysis objectives (Chapter 2), especially if there are concerns about the source data or knowledge uncertainty as discussed above. Using the biophysical setting group attribute is one way to "coarsen" the biophysical setting data layer to a more appropriate thematic resolution, but careful critique of the "exemplar" vegetation dynamics model associated with the biophysical setting group is critical. In some cases, the user may want to choose a different "exemplar" model that better represents the biophysical setting group for their analysis location.

Biophysical setting classes may also be grouped using local, ancillary information. For example, in a vegetation condition analysis of Southern Sierra National Forests, analysts grouped models based on similarity of vegetation characteristics and fire regimes following a crosswalk between LANDFIRE biophysical setting and presettlement fire regime groups presented in Van De Water and Safford (2011), thus reducing the number of biophysical settings from 25 to 15.

If biophysical settings are grouped to coarsen the thematic resolution of the biophysical setting data layer, the user will usually be required to manually map succession class due to differences in succession class definitions between the original and chosen vegetation dynamics models. The guide *How to Map Successional Stages Using LANDFIRE Products* (The Nature Conservancy 2013) provides step-by-step instructions on how to do this.

Biophysical Settings that Cross Map Zone Boundaries

In situations where the analysis area overlaps more than one LANDFIRE map zone, a primary consideration is whether there are differences in the vegetation dynamics models between zones, and if such differences reflect reality. If the map zone boundary reflects an ecological transition, then differences between models for the same biophysical setting may be acceptable and necessary. However, if the map zone boundary creates an artificial demarcation in the analysis area, users will want to choose the model that best fits the analysis area and make the appropriate modifications to the related geospatial data. If a new biophysical setting model is chosen, the succession class data layer will need to be adjusted so that it reflects the succession class mapping definitions of the new model (the guide *How to Map Successional Stages Using LANDFIRE Products* provides instructions for re-mapping succession classes) (The Nature Conservancy 2013).

Succession Class Mapping Rules

It is particularly important to critique the succession class mapping rules because the vegetation departure calculation is very sensitive to the amount of area mapped to each succession class. The LANDFIRE succession class data layer is created by applying rule sets to combinations of biophysical setting, existing vegetation cover, existing vegetation height, and to a lesser extent existing vegetation type (Chapter 4). Any problems in the input data layers will carry through to the succession class data layer. Three general concerns with the succession class mapping rules that can impact departure assessments are: 1) the mappability of the classes; 2) the completeness of the succession class rule set; and, 3) the classification of uncharacteristic types.

> *Mappability of Succession Classes*. Succession class is a concept that can be difficult to translate into mappable criteria. Height and cover, the primary variables LANDFIRE uses to map succession class, may not always be the best surrogate for vegetative development and can be difficult to map (Chapter 4). In particular, the height classes for shrub and herbaceous lifeforms are difficult to discern using LANDFIRE's two dimensional satellite imagery. For example, it may be difficult to distinguish 0.5m tall grass from 1.0m tall grass using Landsat data, but some succession classes are mapped based on this distinction. In forests, the height classes tend to be mapped more accurately (see Chapter 4 - Existing Vegetation), but they may be too coarse to adequately differentiate succession classes (e.g., 10 to 25m and 25 to 50m).

> *Completeness of the Rule Set.* Ideally, the succession class rule sets would cover all possible mapped combinations of existing vegetation type, existing vegetation cover, and existing vegetation height for every biophysical setting without gaps or overlaps. In other words, the rules should be mutually exclusive and exhaustive, but this is not the case for all LANDFIRE succession class rules.

Take, for example, a hypothetical shrub biophysical setting with two succession classes defined as follows:

A - shrubs 10-100% cover and height < 1m

B – shrubs 50-100% cover and height > 0.5m

In this case shrubs .5-1m tall with >50% cover could be classified in either succession class A or B; the rule set is not mutually exclusive.

Take another hypothetical example of a tree-dominated biophysical setting:

A – trees 0-100% cover and < 5m height; or herbs or shrubs 0-100% cover and "any" height

B – trees 0-100% cover and 5-10m height

C – trees 0-100% cover and 10-25m height

In this example, if trees are not established or trees are less than 5m in height, the pixel is mapped as succession class A. Trees that are 5-10m in height are mapped as succession class B and trees 10-25m in height are mapped to succession class C. What about trees greater than 25m in height? Did the model developers intend for this condition to be mapped as uncharacteristic? In many cases this is not the intent; rather, the rule was developed before the geospatial data were mapped and the modelers chose the most reasonable height class without knowledge of the possible mapped height range. When the rules are not exhaustive and/or mutually exclusive, pixels can be mapped into an inappropriate class.

Users also should watch for rules that overlap in structure (cover and height) but differ by species composition. Some vegetation dynamics model descriptions use existing vegetation type as criteria for distinguishing between succession classes, but it was not a primary variable used in mapping—although this varies by biophysical setting and data version. In these cases the succession class assigned by LANDFIRE may not be in agreement with the vegetation dynamics model description. For example, in LANDFIRE map zone 21, the Rocky Mountain Aspen Forest and Woodland vegetation dynamics model differentiates between succession classes C and E by species composition (Figure 12). Both classes have the same structural criteria but succession class C represents a "relatively pure aspen stand," whereas succession class E represents "aspen replaced by other vegetation types or a mixed aspen-conifer overstory that is changing to a conifer dominated forest." These classes should be differentiated by existing vegetation type, but as recent as LANDFIRE 2010 no pixels were mapped to succession class E because existing vegetation type was not used in the succession class mapping process.

If manually mapping succession class, the existing vegetation type data layer can be used to mitigate this issue. For instance, where the structural criteria are met, succession class C would be assigned to pixels classified as the Rocky Mountain Aspen Forest and Woodland existing vegetation type; succession class E would be assigned to pixels classified as an aspen-mixed conifer or a pure conifer existing vegetation type.

Classification of Uncharacteristic Types. LANDFIRE classifies uncharacteristic vegetation as either uncharacteristic native or uncharacteristic exotic (Chapter 4). The uncharacteristic native class indicates that the existing characteristics (i.e., cover, height, and composition) of native vegetation are outside the reference condition range. When conducting a local vegetation departure analysis users may want to critique the mapping rule thresholds for local relevance. For example, if the maximum canopy cover in the vegetation dynamics model is 40%, any cover greater than 40% will be mapped as uncharacteristic native. Does local research of reference conditions corroborate the 40% threshold? Another instance in which the succession class might be mapped as uncharacteristic native is when a *native* riparian existing vegetation type is mapped to a non-riparian biophysical setting. As discussed in Chapter 4 this situation may be due to differences in the mapping methodologies for biophysical setting and existing vegetation type (see Chapter 4 - Potential vs. Existing Vegetation Type Rectification). Users may wish to further critique the data in such situations.

The uncharacteristic exotic class indicates that an exotic species has become established in an area. Succession class is mapped as uncharacteristic exotic wherever an "introduced" existing vegetation type is mapped (e.g., introduced upland vegetation-perennial grassland and forbland). A consideration related to the presence of exotic species is that LANDFIRE classifies less than 10% vegetation cover as "sparsely vegetated." For some analysis objectives, it may be important to identify sparse cover of exotics, such as cheatgrass (*Bromus tectorum)*, and this may require ancillary data sources (Provencher et al. 2009).

Landscape Summary Unit

Independent vegetation departure analyses are not tied to the landscape summary units used by LANDFIRE. The key criterion for landscape delineation is that the summary unit needs to be large enough to encompass the full range of succession classes expected under the historical disturbance regime (Barrett et al. 2010). Careful consideration should be given to the choice of the landscape summary unit using the guidance in the Fire Regime Condition Class Guidebook and Fire Regime Condition Class Mapping Tool User's Guide, keeping in mind that departure scores may vary with changes in the summary unit. If the landscape summary unit is so small that it would not contain the full range of succession classes under the historical disturbance regime, misleading departure scores can result, and lead to errors in the subsequent planning process (Barrett et al. 2010). In contrast, summary units that are too large may make it difficult to discern changes in departure due to planned (e.g., restoration treatments) and unplanned disturbances (Barrett et al. 2010). This may be the case for some biophysical settings under the LANDFIRE 2012 methodology for mapping departure, in which the full extent of the biophysical setting in one or multiple map zones is used as the summary unit to calculate departure. However, it is the intent of the off-the-shelf LANDFIRE products to assess departure at a much broader scale than that of a typical local analysis.

Chapter 7: Interpreted Examples

In this chapter, we (the authors) illustrate the data critique and modification process in two example applications. The first example critiques LANDFIRE data for use in fire behavior analysis of the Rogue Basin located in southwest Oregon (Figure 21). The second example focuses on the critique of LANDFIRE data for use in vegetation departure analysis in the southern Sierra Nevada Mountains of California (Figure 21).

Figure 21. Project area boundaries for interpreted examples.

There are multiple approaches and tools available for critiquing and modifying geospatial data. In these examples we demonstrate the use of common approaches and tools that are available to most natural resource professionals. The following examples summarize the concepts and considerations for modifying LANDFIRE data discussed in previous chapters and therefore should be beneficial to all readers regardless of expertise in working with geospatial data. Details on geospatial analysis and data manipulation tasks, however, are beyond the scope of this document and are only outlined here.

Example 1: Critiquing LANDFIRE data for local fire behavior analysis

Define objectives

For this example we turned to the 3.3 million-acre Rogue Basin in southwest Oregon, where the Southern Oregon Forest Restoration Collaborative and its partners are undertaking the development and implementation of a cohesive forest restoration strategy. A key component in the development of this strategy was an understanding of the current wildfire hazard and associated risk to the Basin's natural resources and assets. Our objective was to conduct a wildfire hazard analysis using LANDFIRE data and geospatial wildfire behavior modeling software.

Identify data requirements

Eight geospatial data layers are required inputs for simulating the full range of wildfire behavior—surface through active crown—in the geospatial fire modeling systems used in this analysis. These layers characterize surface fuels (fire behavior fuel model), canopy fuels (canopy base height and canopy bulk density), forest canopy structure (canopy cover and canopy height), and topography (elevation, aspect, and slope). Each geospatial data layer is available from LANDFIRE.

Given our objective to geospatially analyze wildfire hazard, it was important that the geospatial data represent the fuels and wildfire potential as appropriately as possible for the scale of the analysis. To evaluate the LANDFIRE fuels data we would use the LANDFIRE Total Fuel Change Tool (LFTFCT 2011), which allows for the critique, modification, and analysis of fuel mapping rules and their effect on simulated fire behavior within the tool itself. Because LANDFIRE fuel data (Chapter 5) are derived from existing vegetation type, cover, and height (Chapter 4), biophysical setting (Chapter 4), and disturbance (Chapter 3), the tool requires these geospatial data layers as input, thus increasing our data requirements. We downloaded the additional data layers using the LANDFIRE Data Access Tool (Figure 22, LFDAT 2012).

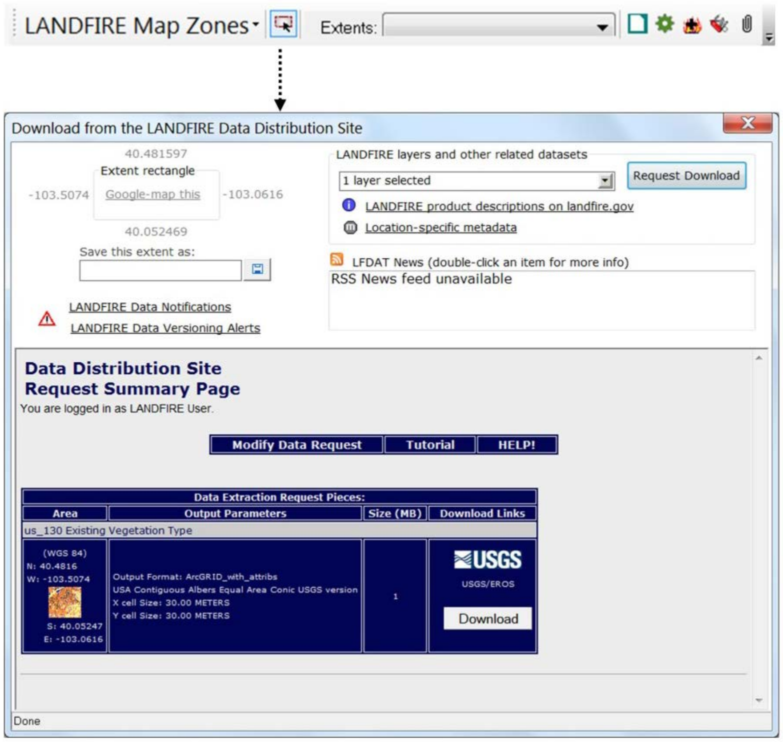


Figure 22. The LANDFIRE Data Access Tool (LFDAT). The LFDAT is a custom ArcGIS toolbar that links to the LANDFIRE Data Distribution Site.

Critique

The fundamental question of our critique was whether LANDFIRE data would be appropriate for simulating wildfire behavior at the analysis location and scale. The LANDFIRE Total Fuel Change Tool would be used to assess the fuel mapping rules in addressing this question; however, *data currency* and *map unit accuracy* (Chapter 1) are also important to accurately simulate the current wildfire hazard so we began our critique there.

The wildfire analysis component of this project began in January 2015, just after LANDFIRE version 1.3.0 (LANDFIRE 2012) data were released for the region. This meant the data were two years out-of-date at the time of acquisition. A critical first task was therefore to determine how much the landscape had changed in the preceding two years.

Approximately 200,000 acres of wildfire and 11,500 acres of mechanical disturbance had occurred over 2013 and 2014 within the wildfire simulation landscape. Given this information, it was clear that currency updates to the LANDFIRE vegetation and disturbance data inputs would be required prior to critiquing the fuel mapping rules with the LANDFIRE Total Fuel Change Tool.

The input data were also critiqued for map unit accuracy. Upon field review, local resource managers felt that oak woodland ecological systems were underrepresented in the LANDFIRE existing vegetation type data layer and that ancillary data would be required to address this issue. In critiquing the LANDFIRE disturbance data, local resource specialists also determined that certain disturbance type assignments were not correct for the local area. For example, the assignment of mechanical remove to all silvicultural treatments (i.e., clearcut, harvest, thinning) was not appropriate for the Rogue Basin because not all local harvesting methods are accompanied by activity-fuel treatments such as hand-pile burning or biomass extraction. Similarly, there were activities assigned to the "other mechanical" event type (a mechanical-add disturbance) that participants felt should be assigned to mechanical-remove. In addition, participants felt that although mastication event types add fuel to the surface fuelbed, they should be differentiated from the other mechanical add disturbances due to the effect of the structure and compactness of masticated fuel on fire behavior.

Finally, as discussed in Chapter 3, LANDFIRE does not currently use a cumulative effect approach to assign disturbance attributes in the composite fuel disturbance data layer. Rather, if multiple treatments occurred in the same location within the update period, the attributes of the most recent treatment are assigned (except where fire has occurred; see Chapter 3). This was also a potential cause of inaccurate map unit assignment.

To summarize, the following information was gathered from the data critique and used to modify the geospatial data inputs to the LANDFIRE Total Fuel Change Tool.

- Data is not current through 2014.
- Oak woodland ecological systems are underrepresented.
- Some disturbance type map unit assignments are inaccurate due to generalization of treatment types at the national scale and/or incorrect accounting of cumulative treatment effects.
- Grouping of mastication treatments with other mechanical add disturbances does not represent the unique fire behavior of masticated fuel.

Modify LANDFIRE Total Fuel Change Tool inputs

As discussed above, the LANDFIRE Total Fuel Change Tool requires geospatial data layers of: existing vegetation type, cover, and height; biophysical setting; and disturbance as inputs. The amount of updating required for these layers varies depending on analysis objectives. The following sections describe the modifications that were made, or why modification was determined to be unnecessary, for each of the required geospatial data layers based on our data critique.

Disturbance

Data Currency

Because the LANDFIRE 2012 composite fuel disturbance data layer only represents conditions through 2012, two data currency updates were required to create an up-to-date 2014 disturbance layer: 1) the time-since-disturbance attribute needed to be updated to reflect the two additional years that had passed, and 2) new disturbances—those that occurred in 2013 and 2014—would need to be added. The following methods were used to create the updated disturbance layer.

First, we determined the years for which the time-since-disturbance attribute would need to be updated (Table 6). Disturbances that occurred from 2005-2007 would remain in the 6-10 year time-since-disturbance class. Likewise, disturbances that occurred in 2010 and 2011 would remain in the 2-5 year time-since-disturbance class. However, disturbances that occurred in 2008 and 2009 would need to be updated to the 6-10 year class and disturbances that occurred in 2012 would need to be updated to the 2-5 year class.

The 2003 and 2004 disturbances would now be greater than ten years old. LANDFIRE removes disturbances greater than ten years old from the composite vegetation and fuel disturbance data layers (Chapter 3) and may also update existing vegetation layer map units to reflect a vegetation transition based on the ecology of the region. For example, a forested, existing vegetation type that experienced a high-severity wildfire, and was subsequently reassigned as an herbaceous or shrub existing vegetation type, may be reassigned to a forest vegetation type after ten years if reestablishment of trees is expected. More information on LANDFIRE vegetation transition rules is available on the program's website. Based on our analysis objectives we determined that we could leave the 2003 and 2004 disturbances in the 6-10 year time-since-disturbance class since we were only concerned with the fuel data layers required for wildfire hazard analysis and therefore not required to update existing vegetation layers.

Next, we downloaded the individual-year disturbance data layers for the years 2008-2012 using the LANDFIRE Data Access Tool. These layers were used to create two "geospatial masks" using the ArcGIS Spatial Analyst extension—one representing the 2008-2009 disturbances and one representing the 2010-2012 disturbances (Figure 23). Masks are used in geospatial analysis to constrain operations to certain pixels within a raster dataset. In our case, we used the masks to identify and update the time-since-disturbance of pixels where a disturbance had occurred in 2008 or 2009 without subsequent disturbances in 2010-2012. As in the LANDFIRE mapping process, if a fire disturbance occurred prior to 2008 we retained the time-since-disturbance of the fire (Chapter 3).

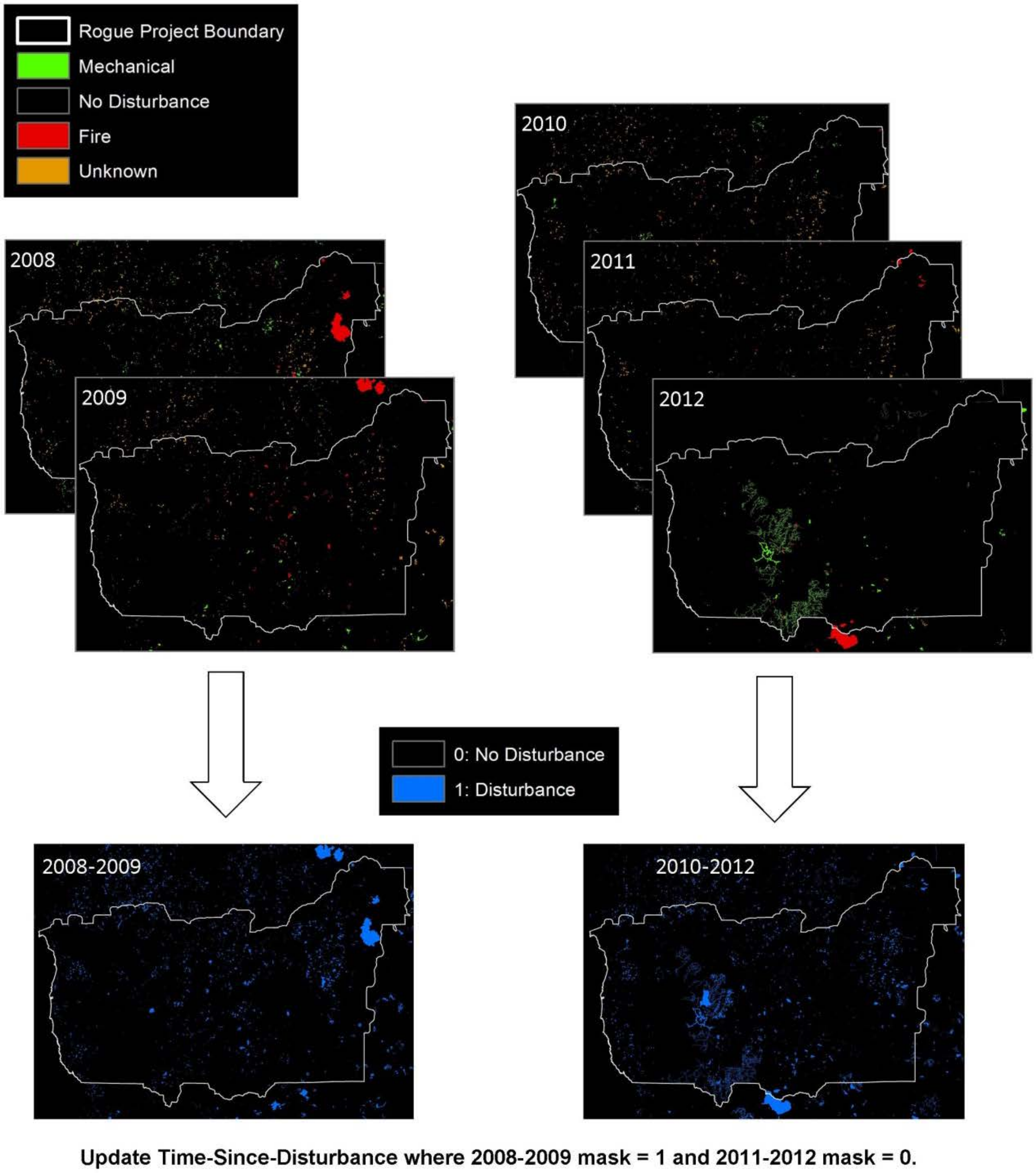


Figure 23. Updating time-since-disturbance. Two geospatial masks were created from the LANDFIRE individual year disturbance layers: one representing disturbances from 2008-2009 and the other representing disturbances from 2010-2012. Time-since-disturbance was updated from the 2-5 year class to the 5-10 year class only where disturbances occurred in 2008-2009 without subsequent disturbance in 2010-2012

With the time-since-disturbance updates complete, we next needed to incorporate 2013 and 2014 disturbances into our updated composite fuel disturbance layer. To reflect large wildfires (> 1,000 acres), we acquired wildfire severity data from the Forest Service Rapid Assessment of Vegetation Condition after Wildfire (RAVG) program website. Recall from Chapter 3 that the LANDFIRE disturbance severity classes represent the effect of disturbances on the vegetation cover of the dominant lifeform. The RAVG program produces a raster data layer representing canopy cover reduction, as a result of fire, through a process that correlates percent change in canopy cover to a remote sensing change detection protocol (Miller and Thode 2007, Miller et al. 2009). We used this data layer to further update the composite fuel disturbance layer based on the percent canopy cover reduction using the ArcGIS Spatial Analyst Extension *Reclassify* tool (Figure 24).

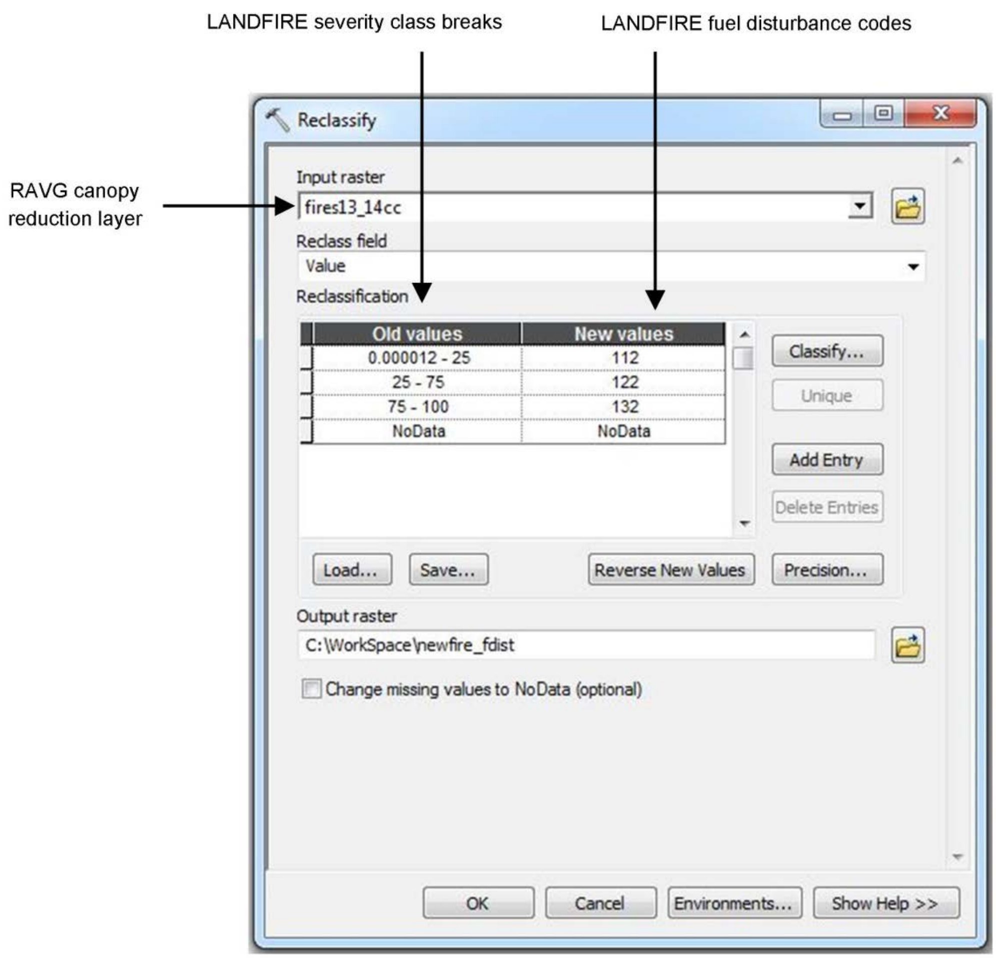


Figure 24. Reclassification of canopy cover reduction estimates from the Rapid Assessment of Vegetation of Condition after Fire (RAVG) program data to LANDFIRE fuel disturbance codes.

We followed a similar process for non-wildfire disturbances. First we acquired 2013 and 2014 Forest Service activities data from the agency's Forest Activities Tracking System (FACTS) and Bureau of Land Management activities from the National Fire Plan Operations and Reporting System (NFPORS). Forest Service and Bureau of Land Management personnel assigned the LANDFIRE disturbance type (mechanical add, mechanical remove, or prescribed fire), severity, and time-since-disturbance codes to each of the activity polygons. If subsequent activities occurred in the two-year time frame, the cumulative effect of those activities was used to determine the most appropriate disturbance attributes. We converted the polygon data to raster format and used ArcGIS Spatial Analyst Extension tools to further update the composite fuel disturbance layer.

Map Unit Accuracy

As mentioned above, our data critique identified two map unit accuracy issues in the disturbance data layer: 1) disturbance type map unit assignments were inaccurate due to generalization of treatment types at the national scale and/or incorrect accounting of cumulative treatment effects, and 2) the grouping of mastication treatments with other mechanical add disturbances does not represent the unique fire behavior of masticated fuel.

We used the ArcGIS Spatial Analyst *combine* function to combine the composite fuel disturbance layer with the individual disturbance layers from 2003-2012. The combine function creates a new raster where each unique combination of values from the input layers represents a single row in the attribute table. Using this table we were able to identify four unique situations and make adjustments based on local resource specialist input (Table 10).

Table 10: Adjustments made to mechanical disturbance type based on local input.

Criteria	Acres	Adjustment
Silvicultural treatments only	200,039	Disturbance type was changed from mechanical remove to mechanical add.
Mastication treatments only	9,188	Created a mask of mastication only pixels and changed the final fuel model values to a "post-mastication" fuel model within the mask during post-processing.
'Other mechanical' treatments only	75,936	Modified disturbance type only if local resource specialists felt the cumulative effect of the treatments was incorrectly assigned.
Combination of mechanical treatment types	289,248	Typically a combination of "other mechanical" and silvicultural treatment. Modified disturbance type only if local resource specialists felt the cumulative effect of the treatments was incorrectly assigned.

Biophysical Setting

Since the biophysical setting data layer represents potential vegetation based on the biophysical characteristics and historical disturbance regime of the site (Chapter 4), disturbances by definition do not have an effect on this layer[2]. Furthermore, because biophysical setting criteria are infrequently used in the LANDFIRE fuel mapping rules for the Northwest Geographic Area, we did not critique this layer for content accuracy.

Existing Vegetation Type

As mentioned above, our data critique identified that oak woodland ecological systems were underrepresented in the existing vegetation type layer. We therefore acquired ancillary geospatial vegetation data developed by the Landscape Ecology, Modeling, Mapping, and Analysis team (LEMMA). We extracted the oak woodland vegetation cover types from this data and augmented the LANDFIRE existing vegetation type data layer using ArcGIS Spatial Analyst tools.

Disturbances may result in a change to the existing vegetation type. For example, tree- or shrub-dominated vegetation may transition to herbaceous-dominated vegetation as a result of high-severity fire. If the existing vegetation type layer was to be used for purposes beyond the critique and development of fuel data, a separate data layer would need to be created to account for any post-disturbance effects to the existing vegetation type. However, since we were only concerned with post-disturbance effects on fuels, we were able to omit this step and rely on our updates to canopy structure and the *canopy guide* feature of the LANDFIRE Total Fuels Change Tool (see below) to correctly assign post-disturbance fuel attributes.

Existing Vegetation Cover

Two updates to the existing vegetation cover layer were required based on our data critique. First, because we used the LEMMA cover type data to augment our existing vegetation type data layer for oak woodland, we also updated the existing vegetation cover layer with LEMMA canopy cover values to ensure consistency across layers. That is, wherever existing vegetation type was updated with LEMMA data, we also updated existing vegetation cover with LEMMA data. Second, we needed to update existing vegetation cover to reflect the 2013 and 2014 disturbances added to the composite fuel disturbance layer.

The structural characteristics of existing vegetation are what the fire behavior fuel model mapping rules are keyed to (Figure 14). We were therefore required to adjust the existing vegetation cover for the new (i.e., 2013 and 2014) disturbances we added to the composite fuel disturbance layer. The post-disturbance canopy cover of forested vegetation types is also required for calculating post-disturbance canopy base height and canopy bulk density.

For the 2013 and 2014 large wildfire disturbances we used the RAVG canopy cover reduction data layer directly to adjust existing vegetation cover. For the non-wildfire disturbances we first assigned a canopy cover reduction value to each severity class midpoint (low severity: 12.5%, moderate severity: 50%, high severity: 87.5%). We did not allow values to be reduced below the lowest canopy cover class (10%-20%) because with few exceptions (e.g., clearcuts), even high-severity disturbances leave some cover. In the case of forested vegetation, leaving 15% forest canopy cover allows for simulating a slight effect of shading and wind reduction to surface fuel from the standing dead trees.

Existing Vegetation Height

[2] Although there are exceptions that could lead to a biophysical setting type conversion, such as those influenced by climate change, uncharacteristic disturbances, and/or exotic species, these occurrences are rare and even if present would have little effect on the assignment of fuel model in this analysis area—that is, biophysical setting criteria are infrequently used in the LANDFIRE fuel model mapping rules in the western states.

As with existing vegetation cover we first updated the existing vegetation height with the LEMMA data in the oak woodland vegetation type.

LANDFIRE existing vegetation height represents the basal-area weighted average of the dominant and co-dominant trees (Chapter 4). In forested vegetation types it is therefore typically not necessary to reduce forest canopy height due to disturbance, as most disturbances would not change the average height significantly enough to reduce existing vegetation height to a lower height class (Table 7). Certain silvicultural methods that target dominant trees, such as clearcuts or thinning from above, are exceptions. For high-severity wildfire, we retained the pre-disturbance canopy height. In combination with the low canopy cover value we assigned, retaining a canopy height value would allow us to simulate a slight effect of the standing dead trees on shading and wind reduction to surface fuel. We were able to prohibit crown fire from being predicted in the post high-severity fire pixels through use of the LANDFIRE Total Fuel Change Tool "canopy guide" function (see below).

Integration of steps with the LANDFIRE Total Fuel Change Tool

With the preliminary critique and updates to the required vegetation and disturbance data layers complete, we then critiqued the LANDFIRE fuel mapping rules using the LANDFIRE Total Fuel Change Tool. A user's guide, tutorial, and information on training for this tool are available on the Wildland Fire Management Research Development and Application – Fuels and Fire Ecology Program website. In this section we will highlight key features of the tool that were used to critique and update fuels for the Rogue Basin analysis.

The LANDFIRE Total Fuel Change Tool provides users the ability to critique and modify the LANDFIRE fire behavior fuel model mapping rules. Additionally, the tool will create canopy fuel data layers (canopy base height and canopy bulk density) using LANDFIRE's methodology, or allow users to "hardcode" base height and bulk density values to unique combinations of vegetation and disturbance attributes. This allows the user to "fine-tune" the interaction of fuel model, canopy base height, and canopy bulk density that is so critical to accurately simulating wildfire behavior.

Critique and Modification of Fire Behavior Fuel Model

The fuel critique was done in a workshop setting where local fire and vegetation specialists from the Forest Service, Bureau of Land Management, and The Nature Conservancy participated. This collaborative approach not only provides a wide range of local knowledge and expertise but also facilitates a sense of ownership and confidence in the end product.

We critiqued the fire behavior fuel model mapping rules for each of the major existing vegetation types in the analysis area. For each existing vegetation type, we first reviewed its description and where it was mapped. If photos were available they would be displayed to provide further context. Next, we discussed which factors—canopy cover; canopy height (a surrogate for stand age); biophysical setting; and disturbance type, severity, and time-since-occurrence—influenced the surface fuels and reviewed how the mapping rules used different combinations of these variables.

Adjustments to the fuel model mapping rules can be made in one of two ways, either to the fuel model assignment itself or to the combination of variables that define a rule (Figure 25). Adjustments to the fuel model assignment were made if workshop participants felt the specified fuel model didn't represent the expected surface fire behavior for the vegetation type and structure identified (that is, if the flame length was too high/low or the rate of spread was too fast/slow). The LFTFC tool provides an interface for comparing the flame length and rate of spread of different fuel models under varying combinations of fuel moisture, slope, and wind speed (Figure 26) as an aid to making modification decisions. Adjustments to the canopy cover and height thresholds, or addition of biophysical setting criteria will influence the spatial distribution and proportion of area assigned to each fuel model. We modified these criteria if participants felt the location or distribution of fuel models did not reflect on-the-ground conditions.

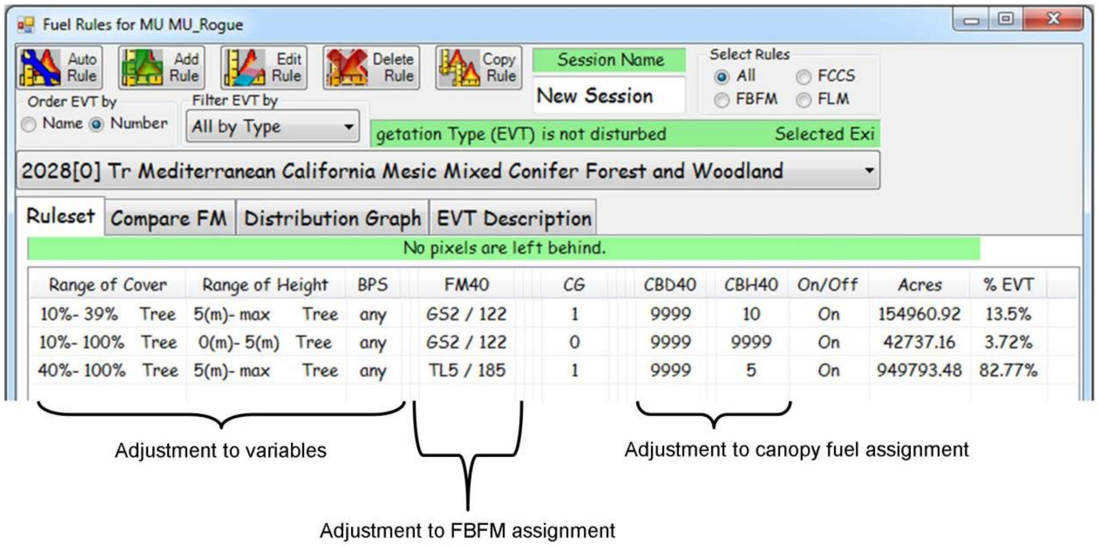


Figure 25. LANDFIRE Total Fuel Change Tool rulesets. Adjustments can be made to the range of variables, fire behavior fuel model (FBFM), and canopy fuel.

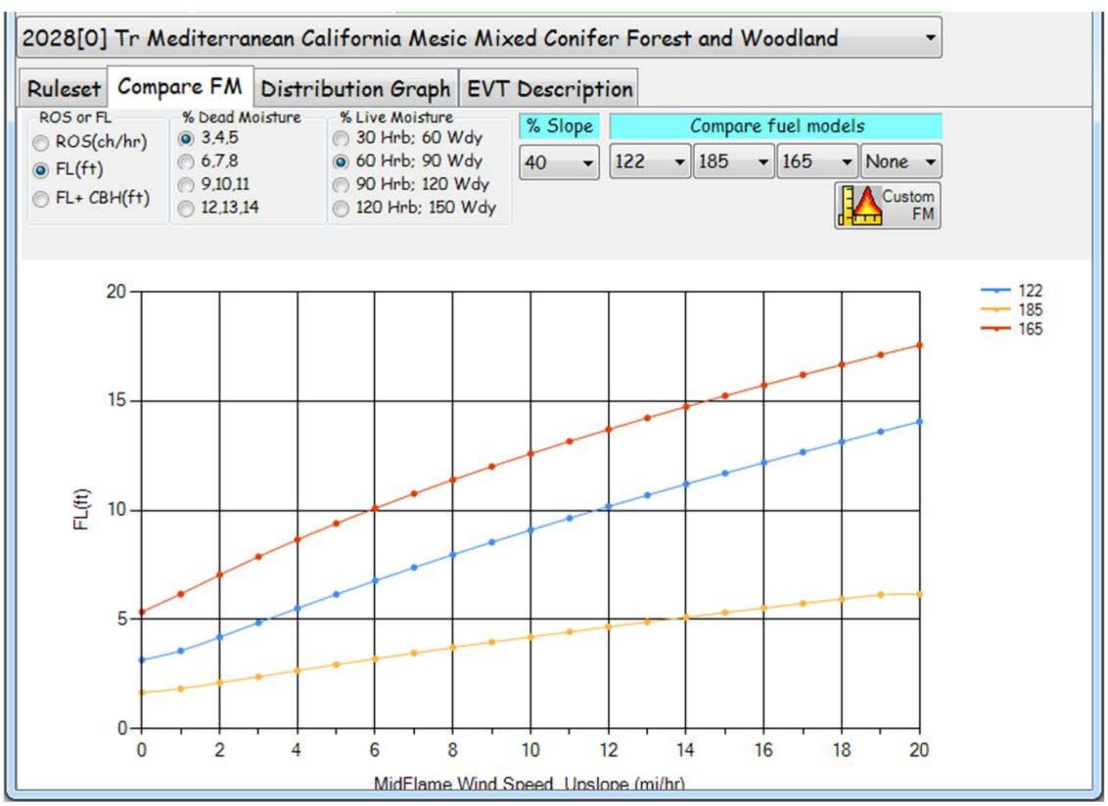


Figure 26. Comparing fuel models. The LANDFIRE Total Fuel Change Tool has built-in functionality to compare fire behavior between fuel models under a variety of fuel moisture and slope conditions.

Finally, for areas where a mastication treatment occurred we assigned the fire behavior fuel model outside of the LANDFIRE Total Fuel Change Tool using ArcGIS Spatial Analyst tools.

Critique and Modification of Canopy Fuels

There are two ways a user has control over how canopy fuels are mapped with the LANDFIRE Total Fuel Change Tool. The first is to use the tool's canopy guide feature; the second is to "hardcode" canopy fuel values. The canopy guide options are as follows:

- 0: No forest canopy structure characteristics (i.e., cover and height) or fuels are assigned. In forested existing vegetation types this may be used to represent a disturbance that removes the forested canopy (e.g., clearcut) or when the "forested" canopy is already considered in the fire behavior fuel model assignment (e.g., short trees).
- 1: The standard LANDFIRE methodologies (Chapter 5) are used to calculate canopy structure and canopy fuel values.
- 2: The canopy base height and canopy bulk density are artificially set to a point where crown fire—passive, active, or conditional (Scott and Reinhardt 2001)—will not be simulated (canopy base height of 10m and canopy bulk density of 0.012 kg/m-3). This value may be used in cases where canopy height and canopy cover values are still desired due to their influence on reducing wind speed and dead fuel moisture content through shading (Chapter 5) but where crown fire is unlikely (e.g., broadleaf forests).

We set the canopy guide value to 2 for all high-severity fire disturbances. As discussed previously, this technique allows for the standing dead trees to still have some, albeit minimal, influence on dead fuel moisture content and wind reduction, but eliminates crown fire and spotting from being modeled in fire behavior modeling systems. The use of a canopy guide value of 2 also served as an alternative to modifying the existing vegetation type due to high-severity fire. That is, by "turning off" crown fire and assigning the appropriate fire behavior fuel model for the expected change in the dominant vegetative lifeform, we accomplished the same goal.

For non-disturbed, and low- and moderate-severity fire disturbances, we assessed the effect of fire behavior fuel model and the LANDFIRE default canopy base height values on crown-fire initiation using the NEXUS (Scott 1999) fire modeling system (Figure 27). Canopy base height values were "hardcoded" (Figure 25) in the fuel rules if workshop participants felt that simulated crown-fire initiation didn't accurately represent expected crown-fire initiation. There are many factors to consider when assigning a canopy base height value. Knowledge of local wind patterns and/or analysis of the wind data that will be used in your analysis are paramount. We accepted the LANDFIRE default canopy base height assignments for all mechanical disturbances.

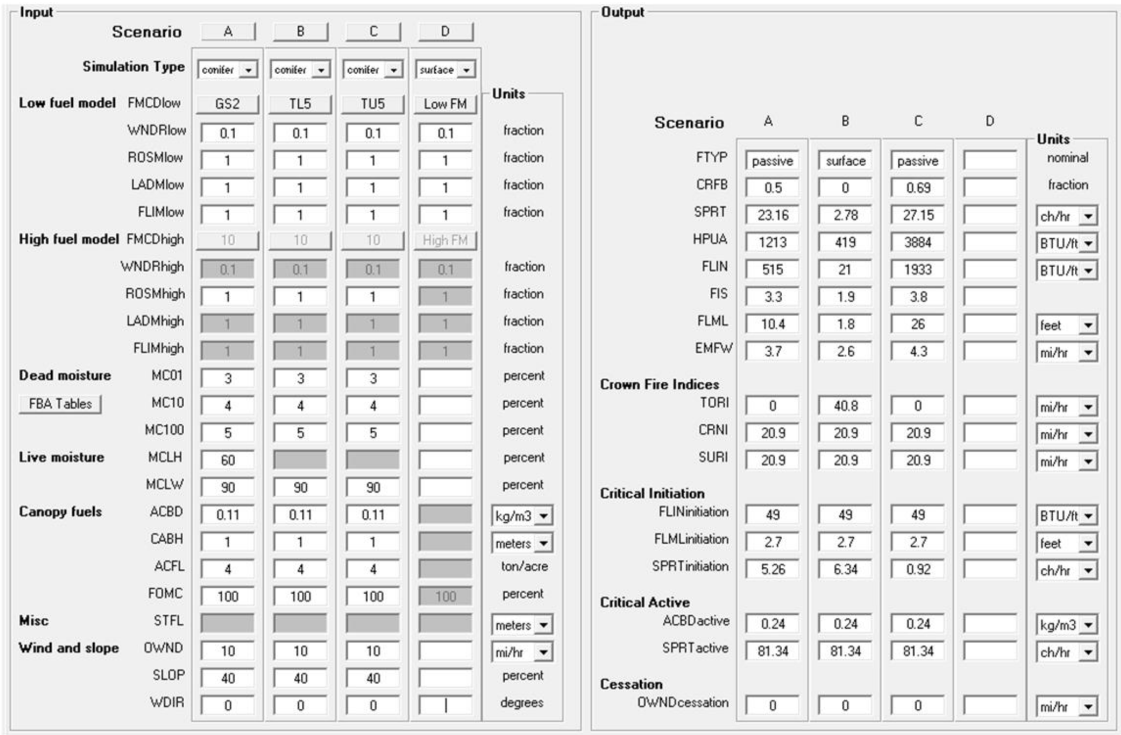


Figure 27. NEXUS fire modeling system. NEXUS facilitates in depth fire behavior critique and is particularly useful in assessing the environment conditions required to transition surface fire to crown fire based on fire behavior fuel model and canopy base height values.

Analysis

We created a new fire behavior modeling landscape (i.e., LCP file) based on our updated disturbance data layers and fuel model mapping rules. We then used this LCP to run basic fire behavior simulations as an additional critique. This final analysis step was used to highlight issues that were possibly overlooked or might have been hard to detect during the fuel calibration, thus necessitating further data modifications. After completion of this final step, the modified fuel data layers were used to analyze wildfire hazard in the Rogue Basin.

Example 2: Using LANDFIRE for local vegetation departure analysis

In this example we illustrate the data critique and update tasks conducted as part of an analysis of vegetation departure in the southern Sierra Nevada Mountains. The 12.5 million-acre planning area includes the Inyo, Sequoia, and Sierra National Forests; Sequoia and Kings Canyon National Parks; and portions of Yosemite and Death Valley National Parks (Figure 21). Because LANDFIRE provides wall-to-wall geospatial vegetation data, it was an obvious choice for vegetation departure analysis at such a broad spatial extent.

Define objectives

The objective of this project was to conduct a vegetation departure analysis using the FRCC Mapping Tool (Hutter et al. 2012) and LANDFIRE data. The results of this analysis would be further integrated into a wildfire hazard and risk assessment. The analysis was conducted in the fall of 2013.

Identify data requirements

Vegetation departure analysis requires data that characterize both the historical and current vegetation condition. LANDFIRE vegetation dynamics models (Chapter 6) would be used to describe the baseline historical conditions for each biophysical setting mapped to the analysis extent. LANDFIRE vegetation data would be used to characterize the current vegetation composition and structure. LANDFIRE 2008 vegetation data layers were acquired and updated for disturbance through 2012 by USDA Forest Service regional office geospatial analysts.

Critique and modification

We began our critique by listing biophysical settings by analysis area acreage from largest to smallest. A team of regional ecologists, vegetation specialists, and GIS and remote sensing specialists reviewed the data list to determine which biophysical settings to assess for departure. Biophysical setting classes comprising insignificant acreage, those that were difficult to accurately map (Chapter 6), and those determined not important to the analysis objectives were dropped. The review team further determined that the thematic resolution (Chapter 1) of the biophysical setting data layer was too fine, given local knowledge of historical vegetation dynamics and disturbance regimes (Chapter 6). Biophysical setting classes were therefore grouped (Table 11) based on recently developed presettlement fire regime groups that summarize presettlement fire frequency estimates for California ecosystems dominated by woody plants (Van de Water and Safford 2011).

Because the analysis area intersects multiple LANDFIRE map zones, we next reviewed the vegetation dynamics models for each of the biophysical settings for consistency across zones. It is common for the vegetation dynamics model to differ across zones for the same biophysical setting. If the map zone boundary reflects an ecological transition, then the differences between models may be appropriate (Chapter 6). However, if the map zone boundary creates an artificial demarcation in the analysis area, users will want to choose a single model that best fits the analysis area. The review team chose the most representative vegetation dynamics model for each biophysical setting or group of biophysical settings to be assessed. The LANDFIRE 2008 biophysical setting data layer was reclassified using the *reclassify* tool in the ArcGIS Spatial Analyst extension to the final 15 classes represented in Table 11.

Table 11: LANDFIRE biophysical setting (BpS) model groupings for the Southern Sierra vegetation departure analysis.

LANDFIRE Biophysical Setting Name	LANDFIRE BpS Code	Presettlement Fire Regime[a]	LANDFIRE Model Used in VCA[b]
Inter-Mountain Basins Big Sagebrush Shrubland	10800	Big Sagebrush	610800
Inter-Mountain Basins Big Sagebrush Steppe	11250		611260
Inter-Mountain Basins Montane Sagebrush Steppe	11260		
Great Basin Xeric Mixed Sagebrush Shrubland	10790	Black and Low Sagebrush	610790
California Mesic Chaparral	10970	Chaparral-Serotinous Conifers	611050
California Montane Woodland and Chaparral	10980		
Great Basin Semi-Desert Chaparral	11030		
Northern and Central California Dry-Mesic Chaparral	11050		
Sonora-Mojave Semi-Desert Chaparral	11080		
Mediterranean California Dry-Mesic Mixed Conifer Forest and Woodland	10270	Dry Mixed Conifer	610270
Sierra Nevada Subalpine Lodgepole Pine Forest and Woodland	10580	Lodgepole Pine	610581
Sierra Nevada Subalpine Lodgepole Pine Forest and Woodland - Wet	10581		
Mediterranean California Mesic Mixed Conifer Forest and Woodland	10280	Moist Mixed Conifer	610280
California Lower Montane Blue Oak-Foothill Pine Woodland and Savanna	11140	Oak Woodland	611140
Mediterranean California Mixed Oak Woodland[c]	10290	Mixed Evergreen	410140
Great Basin Pinyon-Juniper Woodland	10190	Pinyon-Juniper	610190
Mediterranean California Red Fir Forest - Cascades	10321	Red Fir	610321
Mediterranean California Red Fir Forest - Southern Sierra	10322		610322
Mediterranean California Subalpine Woodland	10330	Subalpine Forest	610330
Northern California Mesic Subalpine Woodland	10440		
Sierra Nevada Subalpine Lodgepole Pine Forest and Woodland - Dry	10582	Lodgepole Pine	
Inter-Mountain Basins Subalpine Limber-Bristlecone Pine Woodland	10200	Subalpine Forest	610200
Rocky Mountain Subalpine-Montane Limber-Bristlecone Pine Woodland	10570		
California Montane Jeffrey Pine(-Ponderosa Pine) Woodland	10310	Yellow Pine	610310
Mediterranean California Lower Montane Black Oak-Conifer Forest and Woodland	10300		

[a] Van de Water and Safford (2011) pre-settlement fire regime vegetation types shown for reference.
[b] Vegetation Condition Assessment.
[c] Based on local knowledge and ancillary vegetation data, workshop participants felt that areas mapped as a Mediterranean California Mixed Oak Woodland biophysical setting were incorrectly classified and should be classified as Central and Southern California Mixed Evergreen Woodland (BpS model 410140).

Next, the succession class mapping rules for each of the final vegetation dynamics models were assessed. Adjustments were made to ensure rules were *exhaustive* and *mutually exclusive*, and that uncharacteristic native conditions were appropriately represented for the local area (Chapter 6). Succession class was then remapped, accounting for the adjustments, using ArcGIS software (Figure 28). First, the existing vegetation type layer was reclassified to create an exotic vegetation mask, where exotic vegetation types were assigned a value of 1 and native vegetation types were assigned a value of 0. Next, the biophysical setting, existing vegetation cover, existing vegetation height, and exotic vegetation mask data layers were combined using the ArcGIS Spatial Analyst extension *combine* tool. A new field was then added to the output combine layer and populated with the new succession class values by first selecting combinations of the data layer attributes as defined in the mapping rules and using the *field calculator* function. Finally, after all combinations had been assigned a new succession class value, the ArcGIS Spatial Analyst extension *lookup* tool was used to create a new succession class data layer.

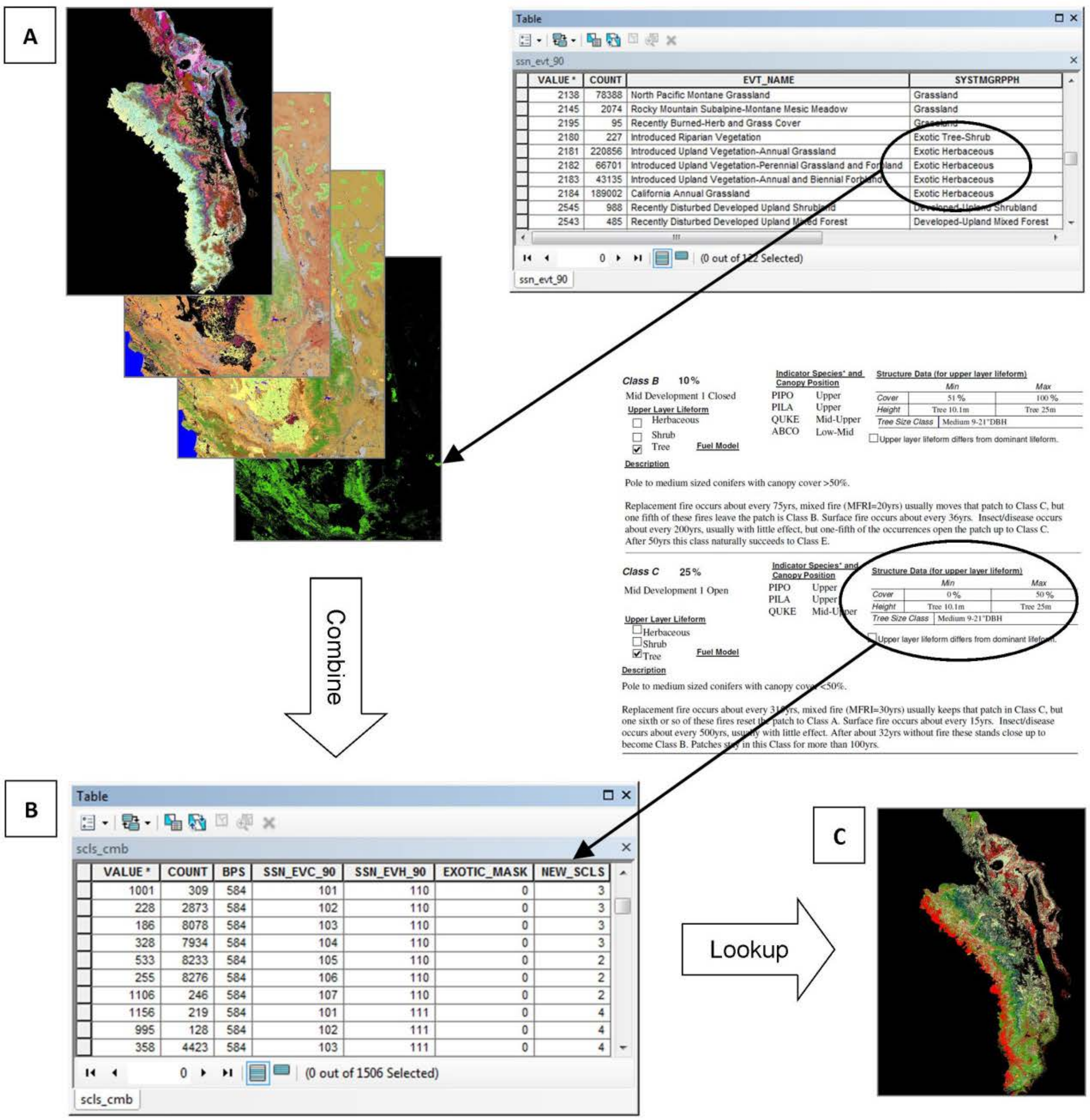


Figure 28. Succession class remapping process. (A) Biophysical setting, existing vegetation cover and height, and exotic vegetation data layers were combined using ArcGIS Spatial Analyst. (B) New succession class values were then assigned based on vegetation dynamics models and adjustments defined by local specialists. (C) Finally, the ArcGIS Spatial Analyst *lookup* tool was used to create a new succession class spatial data layer.

Analysis

We created a spatial landscape assessment unit data layer for conducting the vegetation departure analysis. Each biophysical setting was assigned to an assessment unit based on fire regime characteristics, including historical fire-size distribution (Barrett et al. 2010). Finally, we ran the Fire Regime Condition Class Mapping Tool and reviewed the results.

No issues were identified and the results informed managers where on the landscape specific vegetation development classes (i.e., succession class) were in either surplus or deficit in relation to their presettlement condition. As noted in Example 1 of this chapter, sometimes an analysis may highlight issues that were overlooked or hard to detect earlier in the data critique process that necessitate further data modifications. Analysis should be viewed as an iterative process.

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REVIEWS

"Biases" in Adaptive Natural Resource Management

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Abstract

Uncertainties about the consequences of natural resource management mean that managers are required to make difficult judgments. However, research in behavioral economics, psychology, and behavioral decision theory has shown that people, including managers, are subject to a range of biases in their perceptions and judgments. Based on an interpretative survey of these literatures, we identify particular biases that are likely to impinge on the operation and success of natural resource management. We discuss these in the particular context of adaptive management, an approach that emphasizes learning from practical experience to reduce uncertainties. The biases discussed include action bias, the planning fallacy, reliance on limited information, limited reliance on systematic learning, framing effects, and reference-point bias. Agencies should be aware of the influence of biases when adaptive management decisions are undertaken. We propose several ways to reduce these biases.

Introduction

Natural resource management is often a complex and uncertain process. The underlying environmental and physical processes are sometimes not well understood. Even when they are understood, there are likely to be uncertainties about the quantitative outcomes of management. The current actual status of the resource may be difficult to determine. Managers cannot always fully control which on-ground actions are undertaken due to lack of resources, legal powers, or capacities (Williams & Brown 2014).

These complexities and uncertainties mean that managers are required to make judgments. However, it has been shown that, in making judgments of these types, decision makers do not always undertake decisions "rationally." Simple rational decision-making models assume that agents always take decisions to maximize the achievement of their objectives, based on accurate knowledge of the outcomes, costs, and constraints. In reality, however, people have limited information, limited time, and limited cognitive capacity. As a consequence, they are restricted in formulating and solving complex problems, and they are susceptible to different types of biases (Arnott 2006; Tasic 2011)—beliefs that are inconsistent with reality (Chira *et al.* 2011) or behaviors that compromise the achievement of objectives. For example, Guthrie *et al.* (2000) found that some of the biases listed in Box 1 affect judges when they are making judicial decisions. Similarly, Hirshleifer (2008) found that financial regulators are subject to a different set of biases that influence their decisions, plans, and polices. The impacts of such biases can be substantial. For example, Kahneman (2012) reports on a 2005 study of rail projects worldwide undertaken between 1969 and 1998. Passenger usage of the rail system was overpredicted in 90% of cases. On average, planners overestimated passenger usage of new train lines by over 100%, reflecting a common bias known as the "planning fallacy."

Box 1: Selected behavioral biases with potential impact on adaptive management

- Action bias: Tendency to take actions even when it is better to delay action
- Framing effect: Tendency to respond differently to alternatively worded but objectively equivalent descriptions of the same item
- Reference-point bias: Tendency to overemphasize a predetermined benchmark for a variable when estimating the level of that variable
- Availability heuristic: Tendency to give more weights to events that can be recalled more easily
- Planning fallacy: Making judgments about a planned activity that are systematically over-optimistic, including underestimating project completion time, underestimating costs, or over-estimating benefits
- "Satisficing rule": Tendency to stop searching for a better decision once a decision that seems sufficiently good is identified
- Loss aversion: Tendency to value losses more highly than similar gains
- Reliance on limited information: Tendency to use a subset of information even when full set of information is available
- Limited reliance on systematic learning: Tendency to use information from past successful efforts rather than using information from both successful and failed efforts

For a general list of behavioral biases, see Arnott (2006) and Gino & Pisano (2008).

Managers of natural resources and the environment are likely to be just as susceptible to these biases as are other professionals who must make complex judgments, such as judges and financial regulators (Carlsson & Johansson-Stenman 2012). However, these issues have received little attention in the conservation literature. Our aim in this article is to draw from psychology, behavioral economics, and behavioral decision theory research literatures to identify key insights about biases that are relevant to conservation, and to understand their implications for managers responsible for management of environmental projects or programs.

In doing so, we focus to some extent on Adaptive Management (AM), since this is a process that has been promoted or used to manage complex and uncertain natural resource issues. AM is a process of "learning by doing" (Walters & Holling 1990) where learning from experience is combined with the need for immediate action (Westgate *et al.* 2013). Under AM, management policies are formulated as experiments that investigate ecosystems' responses to changes in people's behavior or management actions (Lee 1999). Conceptually, a set of potential models representing relationships between human actions and ecological outcomes are developed and tested. Viewing the learning process through a Bayesian lens, each model is assigned a probability of being the true model. In each time step, a management decision is made based on the current model probabilities, the current system state, and predicted future states. Model probabilities are updated after each time step based on each model's success in predicting outcomes (Conroy & Peterson 2012), and management may subsequently be modified.

Traditionally, AM has focused on learning from experimental trials or pilots of management approaches for biological and ecological systems (Wilhere 2002; McCarthy & Possingham 2007). It has been assumed that the decision makers will interpret the information collected and make their choices or decisions rationally and without bias. We will explore the extent to which research on human behavior and decision making casts doubt on this assumption. Broader implications for management of natural resources and the environment will also be discussed.

AM: Definition and Stages of Learning

AM has been defined by Williams *et al.* (2009) as "a systematic approach for improving resource management by learning from management outcomes" (p. 1). In active AM, the learning process is supported by purposefully collected information (Walters & Holling 1990), rather from observation of management actions chosen without regard to their ability to provide useful information for future decisions. In active AM, learning is often represented through single- and double-loop processes (Figure 1). Under a single-loop learning cycle, the key steps involved are: (1) define management goals with stakeholders involvement (step 1); (2) develop alternative management options, including an option to maintain the "status quo" (step 2); (3) develop models or statistical processes to trace system responses to management actions (step 2); and (4) implement management options (step 3; Westgate *et al.* 2013). Steps 4 and 5 involve monitoring and assessment of the outcomes, respectively. In a single-loop learning cycle, it is often assumed that project objectives, societal needs, and policy structures are fixed (Allen & Gunderson 2011).

In double-loop learning, on the other hand, it is assumed that policy objectives and structure could change. For example, in long-term projects, societal values and

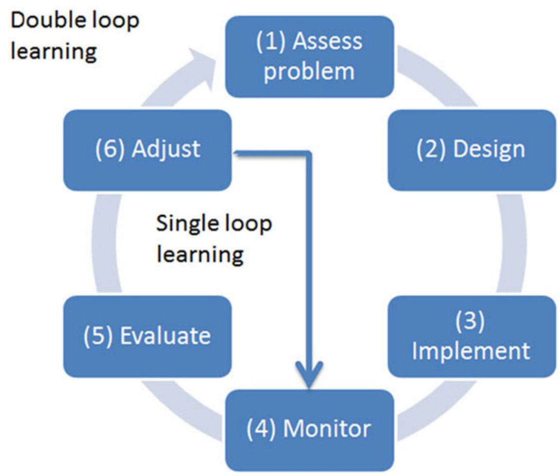

Figure 1 Different steps in active AM cycle with single- and double-loop learning (based on Williams & Brown 2014).

needs could change as time progresses and new management actions are introduced. The resource or the system under experimentation could also change to make the original project objectives unsuitable or unattainable. Therefore, the objectives, management options, or institutional arrangements might need to be changed. Under double-loop learning, original project objectives and management options are revisited after certain steps (step 6). New information from experimentation and model predictions are taken into account as well as changed policy and societal landscapes (Williams & Brown 2014).

In an AM regime, decision makers are responsible for defining management goals, identifying alternative management options, developing models, and implementing programs (Westgate *et al.* 2013). It is common to assume that in each step the resource managers would make "rational" decisions based on the information obtained from biological, physical, and social experiments. However, numerous studies inform us that people have cognitive limitation and bounded rationality, and are influenced by different types of biases. We expand on these issues in the following section.

Key Behavioral Biases

Both psychology and economics have rich literatures on the influences of different types of bias on behavior. Experimental economics serves three main purposes: testing theories, building new theories from observing experimental outcomes, and testing policy and management options. Behavioral economics also integrates insights from psychology to explain economic decision making. It studies the effect of psychological factors such as emotional, social, and cognitive factors on many decisions and economic processes (Camerer 1999). A related field is behavioral decision theory, which studies how people make decisions as well as how they should make decisions (Moore & Flynn 2008). The key biases identified in these research efforts that are relevant to AM are outlined below.

Action bias

"Action bias" occurs when the decision makers choose to take actions even when a "rational" decision maker would prefer to delay actions to allow further information collection, or to take no action. Possible reasons for action bias include that decision makers give higher weight to things that are readily observable and attributable (i.e., the management actions themselves), rather than to things that are delayed, indirect, or unobservable (i.e., potentially the outcomes from those actions; Patt & Zeckhauser 2000). For example, a study of elite soccer goalkeepers showed that they tend to jump to try to save goals even when the optimal strategy is to stay in place (Bar-Eli *et al.* 2007). In this case, taking action is valued in its own right, in addition to the value attributed to the outcome achieved. Similarly, environmental managers may feel that they will earn credit from their superiors, the general public, and the media if they take action even when it is not justified or should be of relatively low priority (Tasic 2011).

Action bias could be increased by uncertainty (Tan *et al.* 2012). In most environmental projects, knowledge of the effectiveness of interventions that will be taken on the ground is rather weak (Ferraro & Pattanayak 2006). As a result, taking action may be evaluated more positively than collecting additional information, partly because of a lack of evidence that actions would be ineffective.

The implication of "action bias" for AM is that it may be difficult to convince managers that an investment in information collection (i.e., AM) is worthwhile. They will tend to prefer to allocate the resources to additional on-ground management actions. Proponents of AM may enhance their persuasiveness by arguing that AM does not require actions to be delayed, and allows more effective or less costly actions to be taken in future. If AM is implemented, it should help to reduce action bias over time by providing additional information about whether the actions being undertaken are effective.

The planning fallacy

The "planning fallacy" is the tendency of project planners to be excessively optimistic about the performance of a project that they are developing (Kahneman & Tversky 1977; Kahneman & Lovallo 1993). For example, many investments in abatement of dryland salinity under Australia's National Action Plan for Salinity and Water Quality program were too small to make a notable difference to salinity outcomes (Auditor General 2008; Pannell & Roberts 2010). Apparently, managers choosing these investments greatly overestimated the effectiveness of the actions being funded, despite ample scientific evidence being available (Prosser *et al.* 2001; Dawes *et al.* 2002). The extent of bias due to the planning fallacy can be substantial. According to Griffin & Buehler (1999), only 1% of the U.S. military high-technology equipment purchases were delivered on time and on budget.

There are various factors that contribute to the planning fallacy. Buehler *et al.* (1994) observed that people estimate a project's expected completion time by constructing mental scenarios of how the project may develop. However, due to cognitive limitations, they generate a smaller range of scenarios than is realistically possible, overlooking many barriers and risks. The scenarios generated tend to reflect their hopes and preferences (Newby-Clark *et al.* 2000) and to neglect their own previous negative experiences with similar projects (Koole & van't Spijker 2000). To some extent, overoptimism is likely to reflect strategic biases adopted to increase the competitiveness of projects when funding is being allocated (Flyvbjerg 2007), but overoptimism is often present even when planners are attempting to be realistic (Kahneman 2012).

A strategy to reduce the planning fallacy is to ask managers to forecast the completion time, cost, or benefits for a range of comparable projects rather than a single project. This strategy, known as Reference Class Forecasting (Kahneman & Tversky 1977), has been effective in reducing time and cost overruns of large infrastructure projects (Buehler *et al.* 2010).

Where the planning fallacy is in evidence, AM may help to reduce its adverse consequences. AM, involving information collection and refinement of project design, helps in correcting decisions that were initially made on an excessively confident or optimistic basis. If necessary, targets can be modified or the project can be terminated following the collection of improved information (Dvir & Lechler 2004).

Reliance on limited information

Decision makers sometimes use only a subset of information even when the full-set information is available. In a series of experiments with common-pool resources, Apesteguia (2006) studied the impact of additional information on individual behavior and payoffs. The individual payoff depended on player's own investment as well as investments made by others. In one treatment, participants had complete information about the expected payoffs from their choices, while in another they had no relevant information. The experimenter observed that the aggregate outcomes (in terms of investment decisions and actual payoffs from the decisions made) were not significantly different between these two treatments (Apesteguia 2006). More-or-less similar observations have been made in other studies (Mookherjee & Sopher 1994; Oechssler & Schipper 2003; Van Huyck *et al.* 2007). One hypothesis to explain this phenomenon is that decision makers follow a "satisficing rule" to limit the cognitive costs of decision making (Hertwig & Pleskac 2010). Under such a rule, the decision maker stops searching for a better decision once he or she identifies a decision that seems sufficiently good.

Another version of this bias is "availability bias" in which people give more weights to certain types of events that can be recalled more easily (Tversky & Kahneman 1974). For example, a manager may assess the risk of bushfire higher than the risk of plant disease spread if bushfires have been more common or more salient in recent times. Underutilization of information is often observed in environmental planning. For example, it has been observed that many existing environmental planning systems fail to account for project costs (Mazor *et al.* 2013), for the effectiveness of management actions (Maron *et al.* 2013), or for behavior change (Pannell & Roberts 2010).

AM potentially provides a mechanism to counter this tendency of decision makers to ignore relevant information. It has been shown in many studies that use of systematic learning through use of data and models could outperform heuristic decision making and predictions by experts (Camerer 1981). It has also been shown that decision makers may employ information more comprehensively if they are asked to make a decision several times sequentially (with time delays) and to explain their decisions to third parties (Vul & Pashler 2008; Herzog & Hertwig 2014). By emphasizing the importance of using accurate information and encouraging use of a structured approach for doing so, AM may prompt a general strengthening of the evidence base for environmental decision making. There can also be a social aspect to AM, with different people contributing to decisions about how management should be adapted in response to new information. This socialization of the process may reduce the tendency of any individual to ignore information.

Limited reliance on systematic learning

Active AM involves systematic experimentation and learning from the outcomes. However, experimental studies on learning reveal that humans are not good at systematic learning. Instead, learning is often messy, noisy, and based on trial-and-error (Hertwig & Pleskac 2010). In practice, people hardly use systematic learning models where they compute and compare expected outcomes from every option before making a decision. Rather, they use heuristics and repeat their past successful choices without fully considering other potentially superior alternatives (Erev & Haruvy 2009).

One implication of limited reliance on systematic learning is that managers will try to learn only from their past "successful" project rather than learning from both "successful" and "failed" projects. In doing so, risk-averse managers are more likely to repeat their past successful choices instead of trying new management interventions (Denrell & March 2001). They are less likely (relative to risk-neutral managers) to invest resources to collect more information about the past unsuccessful strategy (Erev & Haruvy 2009). By contrast, a systematic AM approach would seek to learn from previous mistakes to avoid repeating them, and to enhance the resilience of the management system. AM encourages a systematic approach to learning, and to the use of new information for decision making. It makes explicit the importance of obtaining and using new information, at least partially countering tendencies not to do so.

An institutional barrier to systematic learning is staff turnover, which can be high in the environmental sector, sometimes due to the short duration of funding programs (Grafton 2005). Unless new staff commence before the departures of experienced staff, they must rely on written or verbal communication to learn about the existing or past project (Shogren & Taylor 2008). If the logic behind past decisions is not well-documented, new staff cannot integrate the successes or failures of past decision-making processes into their decision making. There are also differences in the way a new and an experienced manager would approach a problem. A new manager would use facts in a context-free manner whereas, for an experienced manager, problem recognition and action selection would be more intuitive (Hayes 2013).

One potential way to promote systematic learning is through the use of decision support systems (DSSs) that enable the storing of such information. There can be synergies between the use of DSSs and AM. Depending on the type of DSS, it may increase the transparency and evidence base of the initial decision to support a project. This transparent information can be updated as the AM process proceeds, allowing the DSS to inform decisions about modifications to the project (Dicks *et al.* 2014).

Framing effect and reference-point bias

The "framing effect" refers to a situation when people respond differently to statements that are worded differently but are objectively equivalent. Among the many ways of framing an environmental management issue, we mention three that are commonly discussed in the literature: (1) risky choice framing, where the expected outcomes of a risky option are described in different ways; (2) attribute framing, where some characteristics of an object or event are highlighted or focused on; and (3) goal framing, where different potential objectives of the program or activity are emphasized (Levin *et al.* 1998). In a risky choice, framing the outcomes from a lottery could be presented as a loss (say 50% chance of losing) or as a gain (50% chance of winning). In attribute framing, we might focus on only one or a few features of a project (say number of days required to complete a project) rather than all relevant features. For example, we could say that the project is successful if it is completed within a certain number of days (and ignore other features such as the achievement or nonachievement of environmental outcomes). In goal framing, we could focus on gain from undertaking a project (such as "Native animal population will increase if fox control bait is used") or loss from not undertaking the project (such as "Native animal population will continue to decline if fox control bait is not used"; Krishnamurthy *et al.* 2001).

Reference-point bias may cause managers to respond differently to a program or activity depending on the level of a predetermined reference point or benchmark. For example, the same level of environmental improvement could be seen as a success if it is well above a benchmark level of improvement or a failure if it is less than a benchmark, even if the benchmark is arbitrary (Kühberger 1998). It has been shown that people are more sensitive to losses relative to a benchmark than to gains (Camerer 1998). This may mean that managers are strongly motivated to prevent their program from being perceived to be a failure relative to the reference point, but less strongly motivated to seek to make a program perform above the reference point, even if a stronger performance would be feasible and worthwhile.

By regular monitoring and evaluation of project outcomes, AM may help to enhance flexibility in the setting of project goals and to reduce dependence on a fixed reference point. AM, in conjunction with a DSS could help in reducing the impacts of framing effect and reference-point bias by helping managers to assess potential strategies more comprehensively and objectively. Reasons why DSSs are not more commonly used by environmental managers include: lack of adequate training, no clear policy guideline to use the best possible information or DSS, and pressure to spend money within a deadline that is too short to allow time for using the DSS (Shtienberg 2013). To address the last of these issues, in particular, agencies should ideally plan and prepare for potential programs or the next phase of an existing program well before the existing program has concluded.

Discussion

Although many natural resource managers claim to use AM, rigorous and systematic applications are rare (McFadden *et al.* 2011; Westgate *et al.* 2013; Williams & Brown 2014). This is surprising given the theoretical attractiveness of AM in the face of risk and uncertainty (Stankey *et al.* 2005). There has been little research about the impact of psychological biases on decision making by managers of environmental or natural-resource programs (Westgate *et al.* 2013). Based on a survey of the economics and psychology literature, we have identified a set of biases that have implications for AM in particular and NRM in general. As a result of this review, there are grounds to expect that: (1) the managers are likely to take on-ground actions even when these are not worthwhile (Patt & Zeckhauser 2000); (2) they could suffer from the cognitive illusion of being more in control of the system than they actually are (Koole & van't Spijker 2000); (3) they could be overconfident about the expected outcome of their decisions (Flyvbjerg 2007); (4) they may be overly optimistic in terms of expected completion time of the project (Kahneman 2012); (5) they might rely on a partial set of information for decision making even when fuller information is available (Hertwig & Pleskac 2010); (6) they might rely on trial-and-error learning and repeat their past successful choices instead of collecting and comparing information about the full set of decision options (Erev & Haruvy 2009); and (7) managers could try to achieve predefined goals rather than the best possible outcomes from a project (Kühberger 1998; Table 1).

Different biases could influence various steps of the AM cycle differently. For example, action bias could influence the design phase of the AM cycle and lead the planners and managers to design projects with more emphasis on on-ground actions and less on the expected outcomes. Similarly, overconfidence and reliance on limited information would mean the managers would fail to consider all relevant information during the design and monitoring phases. Limited use of systematic learning process would mean failure to learn from previous mistakes during the evaluation phase. Lack of systematic learning would also make mangers susceptible to framing effect and reference-point bias (Klayman & Brown 1993). Agencies should be cautious about the impact of these biases and take remedial measures (Fischhoff 1982).

First, the agencies need to promote a culture of learning (e.g., García-Morales *et al.* 2012). It needs to be recognized that both successful and failed projects generate valuable information about the future state and expected impacts of the management interventions. This could be done by providing appropriate incentives (tangible and intangible) for the managers and decision makers to consider the full range of options before making any decision (Arnott 2006), requiring them to repeat the same decision several times before finalizing it (Vul & Pashler 2008; Herzog & Hertwig 2014), or asking mangers to justify their decisions to external parties (Gollwitzer & Sheeran 2006).

Second, adoption of a DSS could facilitate retention and storing of relevant information (e.g., Behrens & Ernst 2014). It may also make learning from past projects easier and help in systematic evidence-based decision making. Relevant staff should be adequately trained and properly incentivized to use DSSs (Dicks *et al.* 2014).

Third, conducting benefit-cost analyses of planned options would help to refine and prioritize the options during the design phase of the AM cycle (e.g., Pannell *et al.* 2012, 2013). Benefit-cost analysis provides a systematic and objective framework to include all relevant costs and benefits (both market and nonmarket goods and services) related to a project. In the process of identifying benefits and costs, it also helps in identifying if there is complementarity among them (to avoid double counting) and the time lag and uncertainty attached to realization of each benefits and costs. Thus, benefit-cost analysis could be used as a tool to comprehensively assess the expected benefit of a project (Sunstein 2000; Atkinson & Mourato 2008).

Fourth, involvement of external third-party reviewers may also help in designing more realistic and feasible projects (Chen & Volden 2013; Behrens & Ernst 2014). Finally, scenario analysis should be conducted as part of the assessment and design phase of AM cycle to anticipate the expected outcomes of different options (Lautenbach *et al.* 2009). The likely impact of different types of biases, their impact and the effectiveness of potential remedial measures should be systematically analyzed and studied before making any final recommendation for use in decision making for natural resources.

Table 1 Potential psychological biases, their impacts on different steps of the AM cycle, and potential remedial measures to overcome the impact of the biases

Biases	Potential impact on behavior	Potential impact on different steps of AM cycle	Potential remedial measures
Action bias	• Tendency to rely more on actions rather than on results	• During design phase (step 2) projects with visible actions will be prioritized which may lead to wastage of valuable resources (money and time)	• Emphasize the value of information and learning from the AM cycles during the evaluation (step 5), adjustment (step 6), and assessment (step 1) phases rather than on the actions undertaken on ground • Conduct a benefit-cost analysis during the design phase (step 2) of the cycle
Planning fallacy	• Overoptimistic or wrong judgments on the expected benefits, completion time, and costs of the project	• Failure to implement the project (step 3) in due time • During the monitoring phase (step 4), all relevant indicators may not be included, which lead to inadequate assessment during the evaluation phase (step 5)	• Conduct feasibility study as part of the assessment of the problem (step 1) and design of the options (step 2) • Involve external third parties during design phase (step 2) to review proposed actions and their underlying assumptions.
Reliance on limited information	• Make quick judgment • Lack of clearly specified project goals	• During assessment of the problem (step 1), full set of information will not be considered, which will lead to faulty prioritization of projects	• Develop DSSs which will automate incorporation of available information and facilitate consideration of full range of available information during assessment (step 1) and design (step 2) phases
Limited reliance on systematic learning	• Failure to consider the full range of the options • Repetition of the "safe" options • Failure to learn from previous mistakes	• Failure to consider learning from "failed" projects during the evaluation phase (step 5) may lead to missed opportunities to learn and realize the full potential of the situation	• During the evaluation (step 5) and adjustment (step 6) phases, consider learning from all projects (complete/incomplete, successful/failed, etc.) • Always conduct a scenario analysis with a range of options and expected future states during assessment (step 1) and design (step 2) phases
Framing effect and reference-point bias	• Failure to understand the real implications of an option • Success as well as failure is measured relative to a reference point • Follow a satisficing rule rather than a maximization rule while making decisions	• Use wrong measures to evaluate a project (step 5) • Managers may not give their full efforts if they think that they have performed better than others (or with respect to a predefined goal) already (step 3)	• Use DSSs and train managers on how best to use it • A scenario analysis could demonstrate the best possible outcomes from a given situation

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Global fire emissions buffered by the production of pyrogenic carbon

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Landscape fires burn 3–5 million km^2 of the Earth's surface annually. They emit 2.2 Pg of carbon per year to the atmosphere, but also convert a significant fraction of the burned vegetation biomass into pyrogenic carbon. Pyrogenic carbon can be stored in terrestrial and marine pools for centuries to millennia and therefore its production can be considered a mechanism for long-term carbon sequestration. Pyrogenic carbon stocks and dynamics are not considered in global carbon cycle models, which leads to systematic errors in carbon accounting. Here we present a comprehensive dataset of pyrogenic carbon production factors from field and experimental fires and merge this with the Global Fire Emissions Database to quantify the global pyrogenic carbon production flux. We found that 256 (uncertainty range: 196–340) Tg of biomass carbon was converted annually into pyrogenic carbon between 1997 and 2016. Our central estimate equates to 12% of the annual carbon emitted globally by landscape fires, which indicates that their emissions are buffered by pyrogenic carbon production. We further estimate that cumulative pyrogenic carbon production is 60 Pg since 1750, or 33–40% of the global biomass carbon lost through land use change in this period. Our results demonstrate that pyrogenic carbon production by landscape fires could be a significant, but overlooked, sink for atmospheric CO_2.

Globally, landscape fires, which include wildfires, deforestation fires and agricultural burns, emit approximately 2.2 Pg C yr^{-1} to the atmosphere (1997–2016)[1]. The majority of this total emission flux is contributed by non-deforestation and non-peatland fire emissions, which are approximately balanced by vegetation regrowth and thus have no net influence on atmospheric stocks of carbon on decadal timescales[2,3]; however, around ~0.4 Pg C yr^{-1} are emitted during tropical deforestation and peatland fires, which contribute to the net global emissions of carbon due to land use change (~1.1–1.5 Pg C yr^{-1} (Fig. 1))[4–6]. These global carbon budget (GCB) estimates are generated by models that represent the temporally distinct processes of immediate carbon emission from burned areas and decadal-scale sequestration through vegetation (re)growth in a spatially explicit manner[1,7,8]. However, such models routinely overlook the coincident flux of biomass carbon to recalcitrant by-products of fire, which can be stored in terrestrial and marine pools for centuries to millennia, and thus provide a long-term buffer against fire emissions (Fig. 1)[9,10–13]. Consequently, the legacy effects of fire that operate on the longest timescales are systematically excluded from models of the carbon cycle and from GCBs[12,14].

These legacy effects are due to the incomplete combustion of vegetation during landscape fires, which transforms part of the remaining organic carbon in the biomass to a continuum of thermally altered products that are collectively termed pyrogenic carbon (PyC)[10,12,15]. The majority of the PyC produced during landscape fires remains initially on the ground in charcoal particles of varying size and is subsequently transferred to its major global stores in soils[16–18], sediments[19,20] and water bodies[21,22]. A smaller fraction of fire-affected vegetation carbon is emitted as PyC in smoke[23,24]. PyC includes labile products of depolymerization reactions as well as aromatic molecules that result from condensation reactions, the latter of which are depleted in functional groups and thus chemically and biologically recalcitrant[25–27]. The enhanced resistance of PyC to biotic and abiotic decomposition leads to its preferential storage in environmental pools[15,20] and a residence time that is typically 1–3 orders of magnitude greater than that of its unburnt precursors[12]. This makes PyC one of the largest groups of chemically discernible compounds in the soil with a contribution to the soil organic carbon stocks of 14% globally[16]. A fraction of the PyC is also conserved across the land-to-ocean aquatic continuum and thus accounts for approximately 10% of riverine dissolved organic carbon[28], 16% of riverine particulate organic carbon[29] and 10–30% of the organic carbon in ocean sediments[13,19,30,31].

A series of reviews and data syntheses have recognized the potential of PyC production to invoke a drawdown (sink) of photosynthetically sequestered CO_2 to pools that are stable on timescales relevant to anthropogenic climate change and its mitigation[9,10,12,13,32–37]. Owing to the relative recalcitrance of PyC, the conversion of biomass carbon to PyC represents an extraction of carbon from a pool cycling on decadal timescales to a pool cycling on centennial or millennial timescales[13,19,20,25,38]. This storage potential contrasts with that of dead vegetation, which degrades on timescales of months to decades or enters soil pools with a shorter residence time than that of PyC[7,11,25,39,40]. Consequently, postfire PyC pools emit carbon to the atmosphere over a significantly longer time period than would be the case in the absence of PyC production and also provide a buffer that moderates atmospheric CO_2 stocks (Fig. 1)[9,12,13]. At present, the fire-enabled vegetation models that are used to make GCB calculations account for short-term fire emissions but routinely exclude fluxes of carbon from biomass to PyC or the delayed emission of carbon from legacy PyC stocks to the atmosphere (Fig. 1)[7,8,14,41,42]. This introduces systematic errors to GCBs through misrepresentation of the effects of modern and historical fires on the exchange of carbon between the atmosphere and terrestrial–marine pools[12–14].

Although PyC has been recognized as a major component of the postfire ecosystem carbon stocks for a number of decades[10,35],

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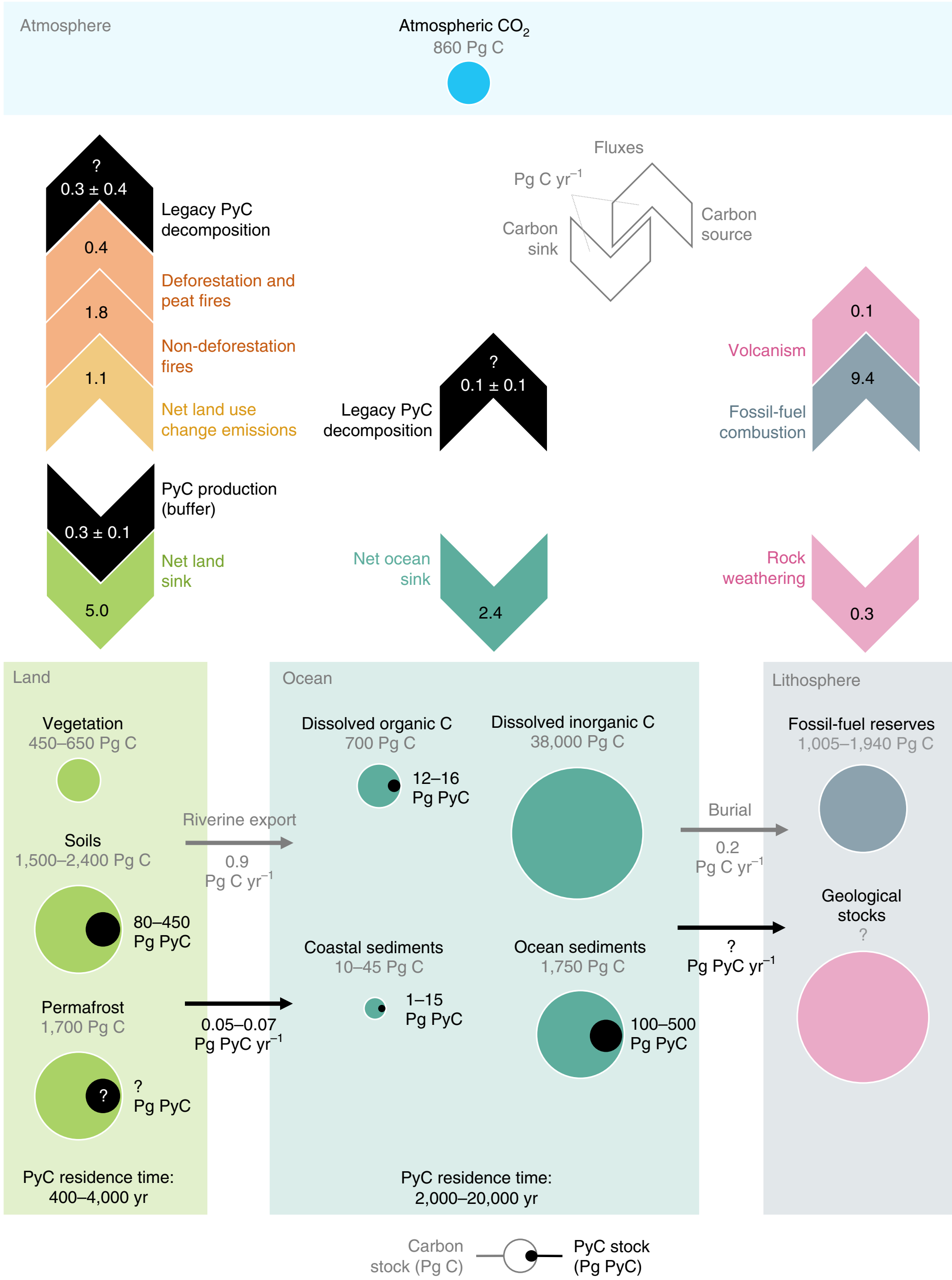

Fig. 1 | Schematic of the global carbon cycle including the buffer and legacy roles of PyC. Stocks (Pg C (1 Pg C = 1×10^{15} g of carbon)) and fluxes (Pg C yr^{-1}) of the global carbon cycle are represented by values from the GCB assessment of the decade 2008–2017[4] and the Intergovernmental Panel on Climate Change fifth assessment report of the decade 2000–2009[6]. Fluxes of carbon due to the net land sink are modified from the GCB to exclude non-deforestation fire emissions, whereas net land use change emissions are modified to exclude deforestation fire emissions. Carbon emissions from deforestation and peat fires and from non-deforestation fires were derived from GFED4s (ref. [1]) and relate to the period 1997–2016. PyC production fluxes due to deforestation and non-deforestation fires are based on estimates from GFED4s+PyC (this study). PyC stocks in soils, ocean dissolved organic carbon and ocean sediments are based on representative PyC/organic carbon ratios in the literature[13,16,68] applied to the estimates of organic carbon stocks and fluxes. PyC fluxes through rivers are the sum of global dissolved and particulate PyC export fluxes[28,29]. Residence times shown for soils derive from a meta-analysis of PyC decomposition in space-for-time substitution studies[69] and incubation experiment estimates extrapolated to field conditions[25]. Residence times for oceanic PyC pools are derived from the literature[19,70]. First-order estimates for legacy PyC decomposition fluxes and their uncertainties are calculated in quadrature for land and ocean pools as the product of the PyC stocks and the reciprocal of the residence times for PyC in these pools, assuming that the low- and high-end estimates for each term represent a consistent portion of normally distributed uncertainty.

quantification of its production rate at the global scale has been problematic and estimates vary by roughly an order of magnitude (50–379 Tg C yr^{-1}) (refs. [12,13,34,36]). A cause of the large range of production estimates is that calculations previously relied on incomplete information regarding the spatial distribution and type of fires, the allocation of carbon among the biomass fuel

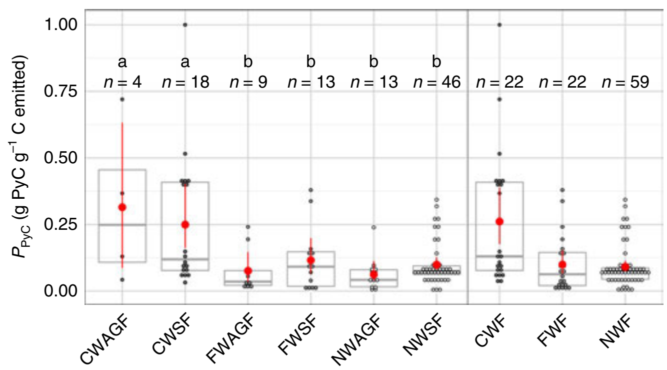

Fig. 2 | The box plots show the distributions of P_{PyC} values for each of the biomass component classes in the production factor dataset. Dots mark the distribution of P_{PyC} values across 1% intervals on the *y* axis. Red dots show the mean P_{PyC} values and red lines show the bootstrapped 95% confidence interval (Methods). Boxes illustrate the median and interquartile range of values. Letters a and b indicate biomass components with statistically similar P_{PyC} distributions at the 95% confidence level according to Tukey honest significant difference tests. The number of data entries (*n*) is also shown. CWF includes both CWSF and CWAGF. CWAGF, coarse woody aboveground fuels; CWSF, coarse woody surface fuels; FWAGF, fine woody aboveground fuels; FWSF, fine woody surface fuels; NWAGF, non-woody aboveground fuels; NWSF, non-woody surface fuels; FWF, fine woody fuels (includes both FWAGF and FWSF); NWF, non-woody fuels (includes both NWAGF and NWSF).

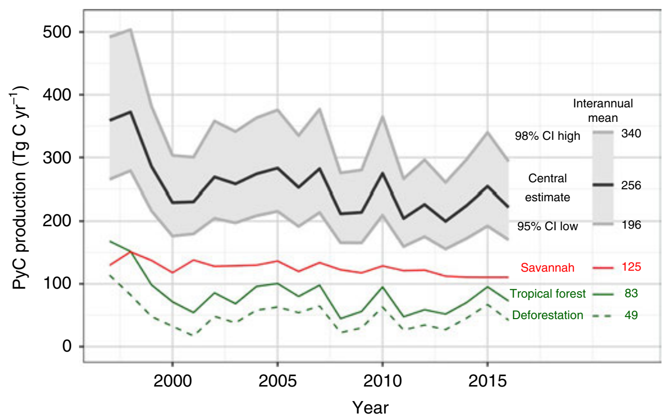

Fig. 3 | Annual global PyC production estimates from GFED4s+PyC for the period 1997–2016. The black line plots the modelled rate of production based on central P_{PyC} ratios (g PyC g^{-1} C emitted) from the global dataset. The shaded area indicates the uncertainty range of the modelled values based on the 95% confidence intervals (CIs) of P_{PyC} values (Fig. 2). The contributions of savannah burning (red line) and tropical forest burning (green solid line) to global PyC production totals are shown, the latter of which includes tropical deforestation fires (green dashed line).

components in burned areas and the specific PyC production factors for these distinct biomass fuel components. To alleviate these issues, we enhanced the Global Fire Emissions Database version 4 with small fires (GFED4s)[1], which is one of the principal process-based models used to make estimates of carbon emission from landscape fires[41,43,44]. Specifically, PyC production was incorporated by following a three-step approach that consisted of: (1) the assembly of the most comprehensive global database of PyC production factors (P_{PyC} (g PyC g^{-1} C emitted)) compiled to date, (2) the assignment of production factors for individual fuel classes stratified as coarse or fine and as woody or non-woody (Fig. 2) and (3) the application of P_{PyC} values to fuel-stratified carbon emissions (grams of C emitted) modelled by the native fuel consumption model in GFED4s. The output is the first global gridded dataset for monthly PyC production at a resolution of 0.25 × 0.25°, covering the years 1997–2016.

Global PyC production

Our central estimate for global PyC production in the period 1997–2016 was 256 Tg C yr^{-1} (Fig. 3), with an uncertainty range of 196–340 Tg C yr^{-1}, which includes variability in the measured P_{PyC} and interannual variability in global production, but excludes uncertainty in GFED4s emissions estimates (Methods). Interannual variability in global PyC production, expressed as the s.d. around the mean, was 47 Tg C yr^{-1} and was most strongly associated with variability in woody fuel combustion, which includes standing wood and coarse woody debris (CWD) (Supplementary Section 1 and Supplementary Fig. 1). Coarse woody fuels (CWF) produce PyC at a greater rate than finer fuels (Fig. 2) and consequently forest fires have disproportionate potential to influence global rates of PyC production (Supplementary Fig. 2).

The El Niño–Southern Oscillation (ENSO) is the primary driver of interannual variability in the burned area in the tropics[45] and previous analyses conducted with GFED showed that carbon emissions from tropical forest ecosystems more than doubled on average during the positive (El Niño) phases relative to the negative (La Niña) ENSO phases[46]. Correspondingly, we calculated that global rates of PyC production in tropical forests were 111% greater during the main fire season of the El Niño phases than during the La Niña phases (Supplementary Table 1). As rates of PyC production by non-forest fires were not sensitive to ENSO (Supplementary Table 1), the major driver of interannual variability in the total PyC production was variability in the tropical forest burned area (Fig. 3). The production of PyC was anomalously high in 1997–1998 (366 Tg C yr^{-1}), which aligns with a particularly strong positive El Niño phase that promoted extensive burning of (tropical) forests in South and Central America and in Southeast and Equatorial Asia[1,46].

Major production regions

The PyC production rates modelled by GFED4s+PyC conformed to a latitudinal pattern (Fig. 4) in which the tropical latitudes clearly dominated production at the global scale. Of the global production, 91% occurred in the tropics and subtropics (0–30° N and 0–30° S), whereas temperate (30–60° N and 30–60° S) and high-latitude (60–90° N) regions provided small contributions to the global total (8% and 1%, respectively).

The global distribution of PyC production also shows intricate regional patterns driven by variation in both the frequency at which fuel stocks were exposed to fire and the magnitude of the fuel stocks that were combusted during the fires that occurred (Supplementary Figs. 3 and 4). Fire frequency was ultimately the key determinant of PyC production rate, which explains why the tropics and subtropics were the dominant source regions. Although savannah fires affect low fuel stocks (Supplementary Section 2), these fires occur frequently and were spatially extensive (Supplementary Fig. 5 and Supplementary Table 2). They thus made the largest contribution to the global PyC production flux (125 Tg C yr^{-1}). Although tropical deforestation fires affected approximately 1% of the area of savannah fires, they affected large stocks of fuel (Supplementary Table 2) and were thus the second largest driver of global PyC production, at 49 Tg C yr^{-1}. The area affected by non-deforestation tropical forest fires was more than a factor of four larger than that of deforestation fires, but fuel consumption was relatively low (Supplementary Table 2). These fires provided the third major component of the global PyC production flux (34 Tg C yr^{-1}). Overall, 81% of the total global PyC production in the period 1997–2016 occurred in savannahs (49%) and tropical forests (32%).

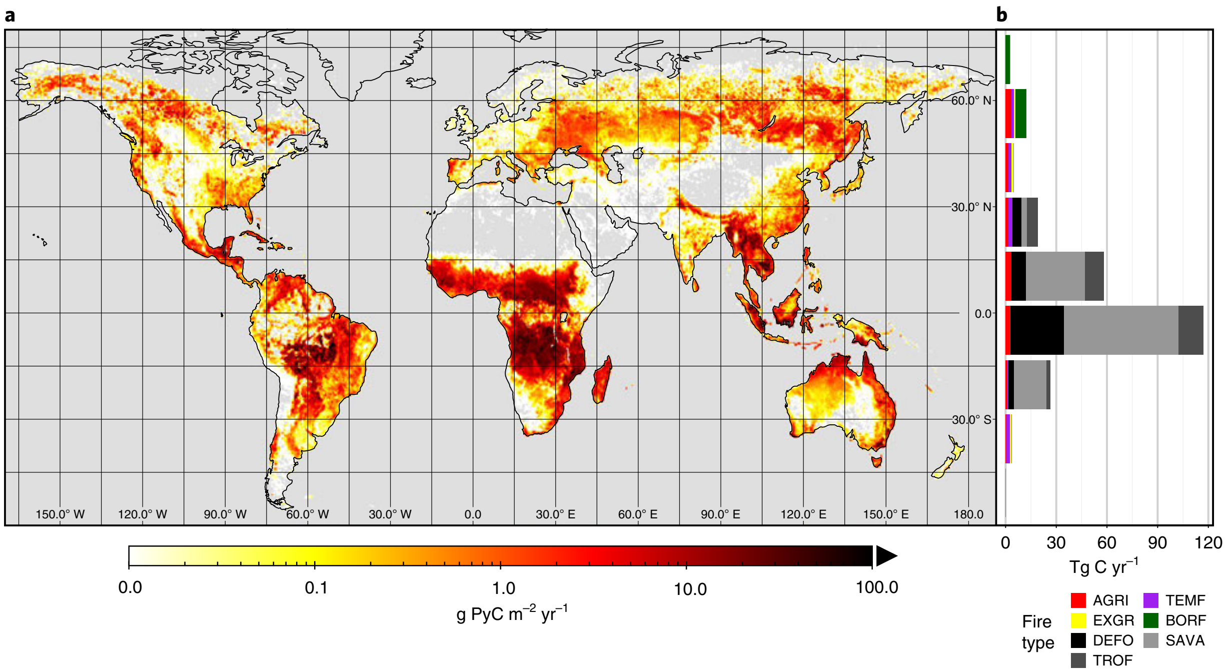

Fig. 4 | Annual average PyC production rates for the period 1997–2016 from GFED4s+PyC, based on central production factors (Fig. 2). a, The average global distribution of PyC production (g C m^{-2} yr^{-1}; note the log scale). **b**, The total production of PyC (Tg C yr^{-1}) in 15° latitudinal bands segregated according to the fire type, which includes savannah fires (SAVA), non-deforestation tropical forest fires (TROF), tropical deforestation fires (DEFO), agricultural fires (AGRI), temperate forest fires (TEMF), extratropical grassland fires (EXGR) and boreal forest fires (BORF).

Global carbon budget implications

Here we have quantified the global gross sink of atmospheric carbon caused by the transfer of photosynthetically sequestered biomass carbon to stocks of PyC during vegetation fires. Our central global PyC production flux estimate (256 Tg C yr^{-1}) is non-trivial within the context of the global carbon cycle (Fig. 1), as it equates to 12% of the global carbon emissions flux due to biomass burning and ~8% of the land sink for atmospheric CO_2 (~3.0–3.2 Pg C yr^{-1}) (refs. [4,6]). The global PyC production flux also equates to 75% of the carbon emitted from tropical deforestation and peat fires, which are the main categories of fire that cause a net loss of carbon to the atmosphere[1,9,47]. The PyC flux modelled here occurs in addition to the smaller global flux of 2 Tg C yr^{-1} caused by the emission of PyC in smoke from vegetation fires (according to equivalent estimates made using GFED4s in the years 1997–2016)[1].

The magnitude of our global estimate for PyC production indicates that the production of PyC during vegetation fires has the potential to significantly influence the atmospheric stock of carbon. A net sink of atmospheric carbon to stocks of PyC can be expected to develop if the flux associated with its production is unmatched by remineralization fluxes from legacy PyC stocks in terrestrial–marine pools (Fig. 1). Earth system models (ESMs) are the most sophisticated tools available to quantify the exchange of carbon between the atmosphere and these pools in time periods for which robust empirical data are sparse or unavailable. Despite previous attempts to highlight the importance of PyC production for carbon storage over timescales relevant to anthropogenic climate change and its mitigation[34,35,48], the absence of the PyC cycle from ESMs has restricted the scope to quantify its role in the carbon cycle[14]. The method introduced here allows for the routine integration of PyC production into fire-enabled vegetation models in a manner that systematically considers the spatial distribution of fire, the composition of the fuel stocks affected and the specific PyC production factors that apply to individual fuel components. This procedure is simple to implement in other fire-enabled vegetation models, which means that the major outstanding challenge to quantifying the net exchange of carbon between the atmosphere and PyC stocks with ESMs is to improve constraints over its storage and residence time in terrestrial and marine pools (Fig. 1)[13,14].

We also show that the PyC cycle must be integrated into ESMs if they are to represent accurately the role of fire in Earth's carbon cycle. The production flux of PyC represents the quantity of carbon that models would otherwise treat either as emitted or as unburned biomass with a residence time in terrestrial pools on the order of months to decades[7,11,25,39,40,49]. At present, the fate of 11% of the global biomass carbon stocks affected annually by fire is misrepresented in global models. As PyC dynamics are not represented in the ESMs used to make GCB calculations[4], this pool may represent a quantitatively significant missing sink or source of carbon to the atmosphere[14,50]. Recent estimates suggest that total carbon emissions from biomass burning in the period 1750–2015 amounted to ~500 Pg C (averaging 1.9 Pg C yr^{-1}) (ref. [41]). Under the assumption that the modern global PyC production flux maintained a constant ratio with the carbon emissions flux throughout this period, we estimate that since the beginning of the industrial revolution ~60 Pg C was transferred to the PyC stocks. This value is equivalent to 33–40% of the carbon lost from biomass pools due to land use change in the same time period (145–180 Pg C) (refs. [6,51]).

Our estimates for the modern and historical PyC production incorporate the best current understanding of PyC production through the combustion of vegetation biomass; however, the limitations of these estimates are worthy of mention. Notably, we do not include the production of PyC through the combustion of organic matter in soils, which may be an important process that drives the accumulation of PyC stocks in environments with deep organic layers, particularly peatlands[52]. We also do not account for the recombustion of PyC in locations that experience secondary burns, which can drive losses of the PyC that remains exposed at the surface[53]. PyC mass losses through recombustion have been reported as <8% in savannahs[54] and 17–84% in boreal forests[53,55]; however, the long fire return intervals in the latter biome typically allow sufficient time for PyC to be protected from recombustion through its burial in soils[17]. Our exclusion of recombustion is deliberate as we consider the process to be a component of the legacy PyC decomposition flux, which we do not quantify here (Fig. 1). Finally, our dataset of PyC production factors provides values for P_{PyC} that are modulated by fuel class (Fig. 2), but does not take into account fire characteristics (for example, temperature and duration) that are relevant to the formation of PyC[36,56,57]. The continued study of PyC production, with a particular focus on regions with high or rising fire incidence[58–60] and a range of fire intensities[61], will facilitate the application of more specific production factors in spatially explicit global models and thus result in reduced uncertainties in the global PyC production.

The production of PyC may become an increasingly important process for global carbon cycling in future centuries. Although the global burned area has declined in at least the past two decades, due predominantly to the conversion of savannah and grassland to agriculture[62,63], recent fire modelling studies generally agree that this decline is unlikely to continue past the year 2050[58–60]. It is also likely that a higher fraction of global burned area will be distributed in forests in which significant stocks of vegetation carbon are held[58,64,65]. As woody fuels generate more PyC per unit of biomass carbon than other fuels (Fig. 2), the spread of fire into forests can be expected to disproportionately enhance the global PyC production (Supplementary Fig. 2). Although it is less clear how fire prevalence will change in tropical and temperate forests owing to a stronger human control over burning in these regions[58,62], recent increases in fire extent caused by an increasing drought frequency in Amazonia already counteract reductions in the extent of deforestation fires[66]. Notwithstanding the significant uncertainty that exists in model predictions of future fire regimes, there are strong indications that PyC production rates will increase in some of the Earth's most carbon-dense regions in response to a changing climate[7,9,67]. This implies that the buffer for atmospheric CO_2 emissions that results from PyC production will grow in future centuries.

Online content

Any methods, additional references, Nature Research reporting summaries, source data, statements of code and data availability and associated accession codes are available at https://doi.org/10.1038/s41561-019-0403-x.

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Author contributions

M.W.J., C.S. and S.H.D. designed the study. S.H.D. led the Leverhulme Trust Research Project grant that funded the main body of the work. M.W.J. collated the PyC production factor dataset with support from C.S. C.S. and S.H.D. provided unpublished PyC production data. G.R.vdW. provided access to the GFED4s code. M.W.J. adapted the GFED4s code to include PyC production with the support of G.R.vdW. M.W.J. conducted the formal analysis of the production factor dataset and model outputs. All the authors contributed to the interpretation of the results. M.W.J. wrote the manuscript and produced all the figures. All the authors contributed to the refinement of the manuscript.

Competing interests

The authors declare no competing interests.

Additional information

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Methods

Global fuel consumption modelling in GFED4s. In GFED4s, carbon emissions to the atmosphere are quantified based on burned area and fuel consumption per unit of burned area. Burned area is derived from satellite data[71] and fires that are too small to be detected by regular burned area algorithms are derived statistically based on active fire detections and relations with, among others, vegetation indices[72]. Fuel consumption is modelled using a satellite-driven biogeochemical model[1] and tuned to match observations[73]. Most of the underlying satellite input datasets have a 500 × 500 m resolution but are aggregated to the model resolution of 0.25° × 0.25°. Total fuel consumption is based on the fuel consumption of several fuel components, which include leaves, grasses, litter, fine woody debris, CWD and standing wood. van der Werf et al.[1] give more information on the GFED4s modelling approach.

To calculate the PyC production within GFED4s we added the production factor P_{PyC}, which quantifies the production of PyC per unit carbon emitted. Until now, the principle obstacle to performing a global modelling exercise of this type was the lack of a sufficiently rich and standardized dataset with which to constrain representative values for P_{PyC}.

Our estimates of uncertainty in the annual PyC production relate only to variability in the PyC production factors and interannual variability in emissions and do not include uncertainties in carbon emission estimates propagate from GFED4s. Uncertainties in GFED4s emissions estimates are discussed at length in van der Werf et al.[1,74] and are predominantly the result of uncertainties in the satellite detection of small fires using thermal anomalies and burn scars. As carbon emissions and PyC production are codependent on the burned area, estimation errors that relate to fire detection introduce scalar uncertainties. Uncertainty in the fuel consumption is an additional component of the overall uncertainty in GFED4s emission estimates[1] and has been reduced from previous versions (for example, GFED3) through its incorporation of a global dataset of fuel consumption estimates[73]. As discussed in the primary literature that relates to the development of the GFED4s[1], a formal global-scale assessment of the uncertainties in fuel consumption cannot be completed due to a paucity of ground truth data for some input datasets. For the previous version of GFED (GFED3), Monte Carlo simulations that accounted for uncertainty in both burned area detection and fuel consumption were used to obtain first-order constraints on the uncertainty in carbon emissions, which were ±20–25% at global, annual scales as a 1 s.d. (1σ) value[74]. Developments of GFED4s included the incorporation of small-fire burned area detection, which led to important reductions in the negative bias in the emissions estimates[72]; however, small fires are also challenging to detect and a lack of validation data prevents the formal investigation of uncertainty in burned area for GFED4s[1,72]. Hence, the true uncertainty of GFED4s is not known precisely, but it is likely to be on the same order as that of GFED3 ($1\sigma = \pm 20$–25%). Nonetheless, uncertainty ranges are likely to be greater in regions where small fires are prevalent or where organic soils are affected (for example, Central America, Europe and Equatorial Asia)[1,72].

Regional-scale field studies of fire emissions have served to validate that the GFED modelling framework produces reliable estimates at large scales, for example, in Alaska[75] and the tropics[76]. Studies that involve atmospheric tracers have also provided vital diagnostics for the performance of GFED[1], and generally highlight its proficiency at large scales but reveal some weaknesses in specific regions or during isolated events[77–82]. Overall, GFED4s is highly suited to the investigation of the effects of fire in global-scale biogeochemical cycles and is thus regularly used in GCB assessments[4] and as a reference point for the fire modules of ESMs[7].

Collating a global dataset of PyC production factors. We compiled a new database of P_{PyC} factors (Supplementary Dataset) from a global collection of 22 published studies that reported on PyC production in 91 burn units, as well as two new datasets produced by the authors with 23 burn units reported for the first time here, and we standardized their reporting. All the studies used one of the following two broad approaches to quantify the impacts of fire on the biomass carbon stocks, either prefire and postfire stocks of biomass carbon and PyC are measured or space-for-time substitution is used to constrain burned and unburned stocks of biomass carbon and PyC, which are assumed to be equivalent to prefire and postfire stocks, respectively. Hereafter, the terms 'prefire' and 'postfire' are used to refer to both types of assessment. Here we focus only on PyC present in charcoal and ash[83] on the ground following fire and on charred vegetation. PyC emitted with smoke, transported in the atmosphere and deposited on a regional-scale area is not included as this process has been studied in separate dedicated studies conducted by atmospheric scientists[23] and represents a relatively small flux in comparison (see main text)[12,13].

The P_{PyC} values were calculated for each of the six classes of widely used biomass components: CWSF, which includes CWD or downed wood defined by typical diameter thresholds of >7.6 cm or >10 cm (refs. [84,85]); FWSF, which includes fine woody debris or any other woody debris with diameters below the thresholds for CWSF; CWAGF, which includes trees or branches with diameters greater than the thresholds for CWSF; FWAGF, which includes material described as shrubs, trees or branches with diameters below the thresholds for CWSF; NWSF, which includes litter, understory vegetation, grass, root mat and any other form of non-woody material directly in contact with the ground surface[85,86] and, finally, NWAGF, which includes foliage, leaves, needles, crown fuels and any other forms of non-woody material that attach to standing wood structures above the ground surface.

For each biomass component, P_{PyC} (PyC produced per C emitted) was calculated using the following equation:

$$P_{PyC} = \frac{C_{Py}}{C_{PRE} - C_{POST} - C_{Py}}$$

where C_{Py} is the mass of PyC created during the fire that was attributed to the component, C_{PRE} is the prefire stock of biomass carbon in the component and C_{POST} is the postfire stock of biomass carbon in the unburnt component. C_{Py}, C_{PRE} and C_{POST} are all expressed in the units g C km^{-2}.

Criteria were applied as filters to the dataset to ensure that P_{PyC} could be calculated in a consistent and representative manner. Specifically, P_{PyC} was calculated if the following conditions were met: first, both prefire and postfire biomass stocks were reported and the carbon content (%) was either measured or assumed based on representative values from the literature; second, postfire stocks of pyrogenic organic matter (charcoal, ash and the charred components of partially affected vegetation) were reported and their PyC content (%) was either measured or assumed based on representative values from the literature; third, the type of fire that occurred was representative of a widespread regional fire type (for example, wildfires, slash-and-burn deforestation and prescribed fire) and fourth, in experimental fires, the biomass carbon stock was designed to replicate the density and structure of biomass carbon stocks observed in the field and the burning efficiency was not optimized or adapted as a factor of the study design.

The set of criteria outlined above does not exclude studies that assess the PyC content of charcoal using one of the various chemical or thermochemical techniques available for the separation of PyC from bulk organic carbon[87,88]. Such techniques are frequently used for the detection of PyC in well-mixed soil, sediment and aquatic matrices. However, we note that none of the studies included in our dataset utilized a chemical or thermochemical approach to separate PyC from non-PyC; instead, these studies considered all the organic carbon in residual products of interest (charcoal, ash and the charred components of partially affected vegetation) to be PyC. Thus, we highlight that our estimates of P_{PyC} are free of the intermethod variability in PyC quantification that often confounds the comparison of PyC concentration in environmental matrices across studies and contributes to the notable uncertainty in the magnitude of Earth's major PyC stocks[12,13] (Fig. 1).

Like biomass carbon, total PyC stocks are distributed across several components, which include charcoal and ash on the ground, charcoal attached to CWD and charcoal attached to aboveground vegetation[12]. The majority of the studies included in the production factor dataset matched the studied PyC components to individual biomass carbon components from which they were known to derive. However, as some individual components of the PyC stocks can have a mixture of sources that are indistinguishable from their location or appearance alone, it was occasionally necessary to make assumptions about the biomass components that were sources of these components. This was done on a study-by-study basis. In cases where the source of each PyC component was not explicitly stated, the following procedural steps were adhered to. On a first basis, the PyC component was assigned to a biomass component according to the most probable source inferred, but not explicitly stated, in the primary literature. Second, where more than one biomass component was inferred to be a source of the PyC stock in the primary literature, the PyC stock was weighted proportionally to the prefire stock of carbon present in each of the implicated biomass components. Otherwise, if no sources of PyC were inferred in the primary literature it was necessary to make independent assumptions about the source of PyC in a manner that was consistent with the other studies included in the dataset and our collective experience of quantifying PyC production in the field.

Summary of the production factor values for use in GFED4s+PyC. Our global database suggested that CWSF and CWAGF produce significantly more PyC, relative to carbon emitted, than other fuel classes (their P_{PyC} averaged at 0.25 and 0.31 g PyC g^{-1} C emitted, respectively (Fig. 2)). In contrast, the mean P_{PyC} values for FWSF and FWAGF (0.12 and 0.076 g PyC g^{-1} C emitted, respectively) did not differ significantly from those of NWSF and NWAGF (0.099 and 0.062 g PyC g^{-1} C emitted, respectively). These results are consistent with previous studies, which suggest that large-diameter woody fuels burn less completely and produce PyC in greater proportions than finer fuels[34,35].

For each class, the mean PyC production factor was used as the central estimate for P_{PyC} and the confidence interval around the mean P_{PyC} was calculated through a bootstrapping procedure. Specifically, the available PyC production factors from the dataset were resampled 50,000 times, the mean P_{PyC} was calculated for each resample and the 95% confidence interval was calculated as the middle 95% of the observed 50,000 means (that is, those ranked 1,250th to 48,750th).

According to an analysis of variance with a Tukey honest significant difference post hoc test, no significant differences in mean P_{PyC} were observed between the distributions of P_{PyC} for coarse, fine and non-woody fuels positioned at the ground surface and those same fuels located above the ground surface. Therefore, the

P_{PyC} values applied in GFED4s+PyC are based on the distribution of values in three simplified fuel classes (Fig. 2): CWF (mean 0.26 g PyC g^{-1} C; 95% confidence interval, 0.18–0.39 g PyC g^{-1} C), FWF (mean 0.096 g PyC g^{-1} C; 95% confidence interval, 0.064–0.15 g PyC g^{-1} C) and NWF (mean 0.091 g PyC g^{-1} C; 95% confidence interval, 0.074–0.11 g PyC g^{-1} C).

Assigning PyC production factors in GFED4s+PyC. P_{PyC} values were assigned to each of the native fuel classes of GFED4s[1], which are leaves, grasses, surface fuels (which include litter and fine woody debris), CWD and standing wood (which includes trunks, stems and branches). Mean P_{PyC} values and bootstrapped confidence interval values for CWF, FWF and NWF from the global dataset were used to define representative P_{PyC} values for each of the GFED4s fuel classes (Fig. 2). Full details as to the assignment of P_{PyC} values to each GFED4s fuel class are provided in Supplementary Section 3 and Supplementary Table 3). Briefly, leaf, litter and grass were assigned the relevant P_{PyC} values of NWF, fine woody debris and CWD were assigned the values of FWF and CWF, respectively, and P_{PyC} values for standing wood were applied in a spatially explicit manner as weighted combinations of the P_{PyC} values for CWF (carbon in trunks) and FWF (carbon in branches). The weighted CWF:FWF ratio was assigned according to empirical relationships that defined biomass carbon apportionment to branches and trunks in the various forest types of the GFED4s land cover scheme (Supplementary Section 3 and Supplementary Table 4)[89].

Quantifying ENSO impacts on PyC production. To investigate the influence of pantropical climatic variability driven by the ENSO on the production of PyC, we replicated the analysis presented by Chen et al.[46] with a focus on PyC production rather than on carbon emissions. The pantropics were defined as consisting of Central America, Northern Hemisphere South America, Southern Hemisphere South America, Northern Hemisphere Africa, Southern Hemisphere Africa, Southeast Asia, Equatorial Asia and Australia (Supplementary Fig. 6). The PyC production in El Niño and La Niña phases was compared for the major fire season periods defined in each tropical region by Chen et al.[46]; their study gives a thorough explanation of the rationale for selecting these comparison periods. We summed PyC production in the major fire season period of each region and disaggregated this total to forest and non-forest fires according to the dominant land cover type in the GFED4s land cover scheme (based on the MODIS Land Cover Type Climate Modelling Grid product MCD12C1)[90].

Apportioning sources of PyC. After the GFED4s+PyC model runs, PyC production was assigned to specific sources following a method developed previously for use in GFED4s model runs[1,74]. Specifically, PyC production that occurs as a result of non-deforestation fires was disaggregated in each cell to tropical forest, savannah/grassland, boreal forest, temperate forest and agricultural fires using an existing algorithm that utilizes fractional tree cover, climate and fire-persistence variables. van der Werf et al.[74] give a full discussion of this algorithm. We added an additional latitudinal constraint (30° N to 30° S) to further disaggregate the savannah compartment, which thus separates tropical savannahs and grasslands from extratropical grasslands.

Data availability

The global dataset of the PyC production factors is available as a supplementary data file (GlobalPyC_supplementarydataset.xlsx). This dataset will also be uploaded to the GFED website (http://www.globalfiredata.org) and updated with new data as it becomes available. Supplementary Section 4 contains full references to the studies included in the production factor dataset. Burned area and fire emissions data are publicly available at the GFED website. Additional ancillary data are available from the corresponding author on request.

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Livestock Impacts on Riparian Ecosystems and Streamside Management Implications... A Review

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Historically, riparian vegetation has been defined as vegetation rooted at the water's edge (Campbell and Franklin 1979). Quite often, however, the stream influences vegetation in many ways and well beyond the water line. In lotic systems, the stream is not only responsible for increased water availability, but also for the soil deposition, unique microclimate, increased productivity, and the many consequential, self-perpetuating biotic factors associated with riparian zones. These factors all contribute in the formation of a unique assemblage of plant communities quite distinct from upland communities surrounding the riparian zone. Therefore, along streambanks, other lotic systems, and even ephemeral drainages, riparian ecosystems could best be defined as those assemblages of plant, animal, and aquatic communities whose presence can be either directly or indirectly attributed to factors that are stream-induced or related (Kauffman 1982).

Riparian zones can vary considerably in size and vegetation complexity because of the many combinations that can be created between water sources and physical characteristics of a site (Odum 1971, Platts 1979, Swanson et al. 1982). Such characteristics, include gradient, aspect, topography, soil type of streambottom, water quality, elevation, and plant community (Odum 1971). However, riparian zones, particularly those bordering streams or rivers, have several characteristics in common. They are ecotonal, with high edge to area ratios (Odum 1978). As functional ecosystems they are very open with large energy, nutrient, and biotic interchanges with aquatic systems on the inner margin (Cummins 1974, Odum 1978, Sedel et al. 1974) and upland terrestrial ecosystems on the other margin (Odum 1978).

Thomas et al. (1979) stated that all riparian zones within managed rangelands of the western United States have the following in common: (1) they create well-defined habitat zones within the much drier surrounding areas; (2) they make up a minor proportion of the overall area; (3) they are generally more productive in terms of biomass—plant and animal—than the remainder of the area; and (4) they are a critical source of diversity within rangelands. Both density and diversity of species tends to be higher at the land/water ecotones than in adjacent upland, especially where regional climates are characterized by dry periods (Odum 1978). Ganskopp (1978) described 44 vegetation communities in a 49-hectare riparian zone in the Blue Mountains of northeastern Oregon. Kauffman et al. (1984) stated that the several biotic, environmental and other abiotic factors interacting in a riparian zone in Oregon created a disproportionately greater number of niches compared to surrounding upland ecosystems. Two-hundred and fifty-eight stands of vegetation representing 60 discrete plant communities were identified within this study area. The higher diversity, productivity, and other unique factors associated with the riparian zone when compared to the surrounding uplands are the primary factors that create the importance of these areas as focal points for the management of the livestock, fishery, and wildlife resources.

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Importance of Riparian/Stream Ecosystems

Importance to Instream Ecosystems

Vegetation along small streams is an important component of the riparian/stream ecosystem (Campbell and Franklin 1979, Jahn 1978). Riparian vegetation produces the bulk of the detritus that provides up to 90% of the organic matter necessary to support headwater stream communities (Cummins and Spengler 1978). In these tributaries of forest ecosystems 99% of the stream energy input may be imported from bordering riparian vegetation (i.e., it is heterotrophic) and only 1% derived from stream photosynthesis by attached algae (periphyten) and mosses (Cummins 1974). Berner (in Kennedy 1977) found that even in large streams such as the Missouri River, 54% of the organic matter ingested by fish is of terrestrial origin. The riparian zone vegetation functions both in light attenuation and as the source of allochthonous inputs, including long-term structural and annual energy supplies (Cummins 1974).

Vegetation along streams exercises important controls over physical conditions in the stream environment. It acts as a roughness element that reduces the velocity and erosive energy of overbank flow during floods (Li and Shen 1973). The result is a higher flood peak than a channel without riparian vegetation but lower erosional factors acting on the floodplain and bank (Schumm and Meyer 1979). Healthy riparian vegetation tends to stabilize streambanks, determines bank morphology and may help reduce streambank damage from ice, log debris, and animal trampling (Platts 1979, Swanson et al. 1982).

Channel and floodplain obstructions such as branches, logs, and rocks enhance detention and concentration of organic matter, thereby facilitating its use locally rather than washing downstream (Everest and Meehan 1981, Jahn 1978, Swanson et al. 1982). In addition, wood debris in channel bottoms appears to play an important role in the dynamics of stream morphology. Large pieces of woody debris in streams dissipate stream energy, control routing of sediment and water through channel systems, and serve as substrates for biological activity by microbial and invertebrate organisms (DeBano 1977, Swanson et al. 1982).

Streamside vegetation strongly influences the quality of habitat for anadromous and resident coldwater fishes (Duff 1979, Everest and Meehan 1981, Marcuson 1977, Meehan et al. 1977). Riparian vegetation provides shade, preventing adverse water temperature fluctuations (Meehan et al. 1977). The roots of trees, shrubs, and herbaceous vegetation stabilize streambanks, providing cover in the form of overhanging banks (Marcuson 1977, Meehan et al. 1977). Streamside vegetation acts as a "filter" to prevent sediment and debris from man's activities from entering the stream (Meehan et al. 1977). Riparian vegetation also directly controls the food chain of the ecosystem by shading the stream and providing organic detritus and insects for the stream organisms (Cummins 1974, Meehan et al. 1977).

Importance to Wildlife

It is believed that, on land, the riparian/stream ecosystem is the single most productive type of wildlife habitat, benefiting the greatest number of species (Ames 1977, Hubbard 1977, Miller 1951, Patton 1977). The riparian zone provides an almost classic example of the ecological principles of edge effect (Odum 1978). Riparian habitat provides living conditions for a greater variety of wildlife than any other types of habitat found in California (Sands and Howe 1977), the Great Basin of southeast Oregon (Thomas et al. 1979), the Southwest (Hubbard 1977), the Great Plains (Tubbs 1980), and perhaps the entire North American continent (Johnson et al. 1977).

Examples of the wildlife values of riparian habitat are numerous (Carothers et al. 1974, Carothers and Johnson 1975, Henke and Stone 1978, Hubbard 1977, Thomas et al. 1979). Hubbard (1977) reported that 16–17% of the entire breeding avifauna of temperate North America occurs in 2 New Mexico river valleys over the course of a "few score" miles. Johnson et al. (1977) reported that 77% of the 166 nesting species of birds in the Southwest are in some manner dependent on water related (riparian) habitat and 50% are completely dependent on riparian habitats. In western Montana, 59% of the land bird species use riparian habitats for breeding purposes and 36% of those breed only in riparian areas (Mosconi and Hutto 1982). Thomas et al. (1979) stated that of the 363 terrestrial species known to occur in the Great Basin of southeastern Oregon, 299 are either directly dependent on riparian zones or utilize them more than any other habitats.

When riparian vegetation is eliminated, several wildlife species dependent on riparian ecosystems may be either severely reduced or may disappear altogether. Henke and Stone (1978) found 93% fewer bird numbers and 72% fewer avian species on 2 riprapped plots from which riparian vegetation had been removed, and 95% fewer birds and 32% fewer species on cultivated lands previously occupied by riparian forests.

The influence of riparian ecosystems on wildlife is not limited to those animal species that are restricted in distribution to the streamside vegetation. Population densities of birds in habitats adjacent to the riparian type are influenced by the presence of a riparian area (Carothers 1977). When a riparian habitat is removed or extensively manipulated, not only are the riparian species of the area adversely influenced, but wildlife productivity in the adjacent habitat is also depressed (Carothers 1977).

Riparian ecosystems are valuable to wildlife as a source of water, food, and cover (Stevens et al. 1977, Thomas et al. 1979). They also provide nesting and brooding habitat for avian species (Carothers et al. 1974, Johnson et al. 1977, Tubbs 1980). By furnishing abundant thermal cover and favorable micro-climates, especially when surrounded by nonforested ecosystems, they facilitate the maintenance of of homeostatis, particularly for big game (Thomas et al. 1979). Riparian ecosystems also serve as big game migration routes between summer and winter range (Thomas et al. 1979), and provide routes and nesting cover for migrating avian species (Stevens et al. 1977, Wauer 1977).

Importance to Livestock

Livestock grazing on rangelands is the most extensive form of land use in the interior Pacific Northwest (Skovlin et al. 1977). Cattle tend to congregate on meadows and utilize the vegetation much more intensively than the vegetation of adjacent ranges (Reid and Pickford 1946).

In northeast Oregon, Reid and Pickford (1946) stated that moist meadow soils in riparian ecosystems are generally so highly productive than an acre of mountain meadow has a potential grazing capacity equal to 10–15 acres of forested range. Although riparian meadows cover only about 1–2% of the summer range area of the Pacific Northwest, potentially they can produce 20% of the summer range forage (Reid and Pickford 1946, Roath and Krueger 1982). However, Roath and Krueger (1982) found that because of livestock concentrations, limits on livestock movements imposed by steep slopes, and erratic distribution of watering areas away from the creek, the riparian zone (covering about 2% of a Blue Mountain grazing allotment) accounted for 81% of the total herbaeous vegetation removed by cattle.

Cattle exhibit a strong preference for riparian zones for a number of the same reasons other animals prefer and use these areas. The main attributes believed to attract and hold cattle to riparian areas are the availability of water, shade, and thermal cover, and the quality and variety of forage (Ames 1977, Severson and Boldt 1978). In addition, sedges (Carex spp.) tend to retain relatively constant crude protein levels until the first killing frost. Several sedges common to riparian zones of the Pacific Northwest outrank key upland forage species in sustained protein and energy content (McLean et al. 1963, Paulsen 1969, Skovlin 1967).

Livestock Riparian Relationships

The impact of livestock on riparian zones in public grazing lands of the western states has received much attention recently. Several studies are presently underway examining the impact of livestock grazing on stream ecology, water quality, channel stabilization , salmonid fish habitat and physiology, terrestrial riparian wildlife populations, and riparian vegetation.

It is often difficult for one to interpret science from opinion in the literature. Many of the studies reported in this paper have not necessarily followed the generally accepted "scientific method" for research today. However, it is not the purpose of this paper to determine, even if possible, which published reports represent quality scientific results and which are little more than a forum to express one's opinion. Rather the purpose of this paper is to familiarize the reader with the accepted facts and management theories available today concerning livestock interactions in riparian zones with the other valid resources also dependent or utilizing this resource. Where possible, in this paper, results of properly conducted research are reported using terms such as "significant", referring to a statistically significant result and those of reports relying on observational data or "hearsay" will be reported as suggestions or observations.

General Considerations for Livestock-Riparian Management

The quality of the riparian habitat and its associated aquatic environment, both formed over geologic time, are fragile ecosystems which currently serve as focal points for management of livestock, recreation, and fisheries and timber resources. It has been reported that inappropriate livestock management results in overuse and subsequent degradation of the riparian/stream ecosystem (Behnke and Raleigh 1978, Oregon-Washington Interagency Wildlife Council 1978, Platts 1979). Davis (1982) suggested that one of the most destructive forces in riparian ecosystems is the long-term impact of overgrazing by cattle. Livestock grazing can affect 4 general components of an aquatic system-streamside vegetation, stream channel morphology, shape and quality of the water column and the structure of the soil portion of the streambank (Behnke and Raleigh 1978, Marcuson 1977, Platts 1979, Platts 1981). Improper livestock use of riparian ecosystems can affect the streamside environment by changing, reducing, or eliminating vegetation bordering the stream (Ames 1977, Behnke and Raleigh 1978, Platts 1979). The channel morphology can be changed by widening and shallowing of the streambed, gradual stream channel trenching, or braiding, depending on soils and substrate composition (Behnke and Raleigh 1978, Gunderson 1968, Marcuson 1977, Platts 1979). The water column can be altered by increasing water temperatures, nutrients, suspended sediments, bacterial counts and by altering the timing and volume of water flow (Behnke and Raleigh 1978, Johnsen et al. 1978, Rauzi and Hanson 1966, Platts 1979). Overgrazing can cause bank sloughoff creating false setback banks, accelerated sedimentation, and subsequent silt degradation of spawning and food producing areas (Behnke and Raleigh 1978, Everest and Meehan 1981, Platts 1979, Platts 1981). These impacts on the water column due to abusive livestock practices result in decreased fish biomass and in percent of salmonid fishes in the total fish composition (Behnke and Raleigh 1978, Bowers et al. 1979, Duff 1979, Gunderson 1968, Marcuson 1977).

Livestock abuse of riparian areas can severely impact terrestrial wildlife habitat causing a subsequent decrease in wildlife species and numbers (Ames 1977, Townsend and Smith 1977, Tubbs 1980, Wiens and Dyer 1975).

Improper grazing can have a considerable effect on vegetation, resulting in decreased vigor, biomass and an alteration of species composition and diversity (Ames 1977, Bryant et al. 1972, Evans and Krebs 1977, Knoph and Cannon 1982, Pond 1961).

While various other land management activities have caused serious losses or reductions in wildlife habitat productivity, livestock grazing has been suggested as the major factor identified in numerous studies throughout the 11 western states (Oregon-Washington Interagency Wildlife Council 1978). Conversely, Busby (1979) suggested that it was not reasonable to conclude that livestock grazing is the only, nor necessarily the major cause of impacts to riparian ecosystems.

Impacts of Livestock on the Instream Ecology

A healthy instream environment is vital for the aquatic life forms inhabiting the stream, as well as for various human needs that directly depend on water quality. High concentrations of suspended solids or other sediment loads, and fecal coliforms or fecal streptococci are usually associated with the degree of impact of man's activities, and can have a major impact in altering an existing stream ecosystem or even creating an entirely new ecosystem (Johnson et al. 1977, Johnson et al. 1978, McKee and Wolf 1963).

During the grazing season, Johnson et al. (1978) could not find any significant differences in physical and chemical properties of streamwater (suspended solids, total dissolved solids, and orthophosphates) between an area grazed at 1.2 ha/AUM and an ungrazed area. After the grazing season, however, there was a significant increase in total dissolved solids which indicated that some livestock waste products may have eventually reached and enriched the stream, probably from the action of rain showers. The presence of cattle significantly elevated the fecal coliform and fecal streptococci for about 9 days after cattle were removed.

Winegar (1977) found sediment loads were reduced 48–79% while flowing through 3.5 miles of a stream protected from grazing.

Rauzi and Hanson (1966) found a nearly linear relation between runoff and infiltration to the degree of grazing intensity. They found that runoff from a heavily grazed watershed (1.35 acre/AUM) was 1.4 times greater than from a moderately grazed watershed (2.42 acre/AUM) and 9 times greater than from a lightly grazed watershed (3.25 acre/AUM).

Changes in water temperature have been shown to have drastic effects on fisheries and aquatic insect populations (Johnson et al. 1977). Changes in average temperature or daily fluctuations can in effect create an entirely new aquatic ecosystem (Johnson et al. 1977).

Van Velson (1979) found average water temperatures dropped from 24°C to 22°C after 1 year of livestock exclusion on a creek in Nebraska. Claire and Storch (unpublished) compared stream temperatures between an area that had been grazed season long (June 1–October 15) and an area that had been rested for 4 years and, thereafter, grazed only after August 1. The maximum water temperatures outside and downstream from the exclosure averaged 7°C higher than those sampled within the exclosure. Daily fluctuations of water temperatures averaged 15°C outside the exclosure as compared to 7°C inside the exclosures. Winegar (pers. comm. 1982) observed similar results in an exclosure along Beaver Creek in central Oregon.

The effects of livestock grazing have been shown to vary greatly depending upon several factors, in particular, the nature of the stream studied. Duff (1979) stated that introduction of livestock for 6 weeks into a riparian area rested for 4 years resulted in elimination of overhanging banks and a fracturing of the streambank, causing it to erode into the stream. In contrast, after 6 weeks of mid-summer grazing by cattle, Roath (1980) gave a visual estimate of 90% bank stability with little indication that trampling was contributing to or causing erosion. He attributed nearly all erosion present to geologic erosion caused by the actions of streamflow.

Buckhouse et al. (1981) could find no particular relationship between streambank erosion and various grazing treatments (including nonuse) in northeastern Oregon. There appeared to be no significant patterns of accelerated streambank deterioration due to moderate livestock grazing (3.2 ha/AUM and 60–65% utilization of the riparian vegetation). Most bankcutting losses in this system were associated with over-winter periods where ice floes, high water, and channel physiognomy were critical factors involved in the erosional process.

Hayes (1978) found that stream channel movement did not occur more frequently in grazed riaprian meadows under a rest-rotation grazing scheme compared to ungrazed meadows after 1 year of study. Rather, streambank degradation appeared to occur more often and to a greater magnitude along ungrazed streams. However, Hayes stated that sloughoff increased as forage removal was above 60%. High forage removal, high amount of foraging time along banks, and high percentages of palatable sedges along the bank were shown to significantly increase the probability of sloughoff occuring during the grazing season.

Kauffman et al. (1983b) measured significantly greater streambank losses in grazed areas (1.3–1.7 ha/AUM) compared to ungrazed areas in northeastern Oregon. The grazed pastures had utilization levels greater than 35% and less than 85% on the different vegetation stands while utilization by native animals was less than 20% on every stand. During 2 late season grazing periods (late August–mid-September), a mean of 13.5 cm of streambank was lost in grazed areas and 3.0 cm was lost in ungrazed areas. Total annual streambank losses were 30 cm in grazed areas and 9 cm in ungrazed areas.

Marcuson (1977) found the average channel width to be 53 meters in an area grazed season long at 0.11 ha/AUM and an average channel width of only 18.6 meters in areas that were ungrazed. Marcuson (1977) also recorded 224 meters of undercut bank/km in the grazed area and 686 meters of undercut bank/km in the ungrazed area. Heavy grazing and trampling by cattle were suggested to cause the excessive erosion.

Duff (1979) found the stream channel width in a grazed area was 173% greater than the stream channel not grazed for 8 years inside an exclosure. Similar results have been reported (Behnke and Zarn 1976, Dahlem 1979, Gunderson 1968, Heede 1977) where overgrazing and excessive trampling caused a decrease in bank undercuts, increases in channel widths, and a general degradation of fish habitat.

Claire and Storch (unpublished) stated that the production of game fish in headwater streams can be used as a biological indicator of the quality of land management that is occurring within the watershed and/or streamside. Overgrazing, causing a reduction in vegetative cover and the caving in of overhanging banks, has been suggested as one of the principal factors contributing to the decline of native trout in the West (Behnke and Zarn 1976).

Bowers et al. (1979) reported an average increase in fish production of 184% for 5 independent studies where livestock use was light or eliminated by fencing. They concluded with a prediction that trout production in streams currently being heavily grazed could be increased about 200% if management decisions were made to optimize habitat conditions for trout.

Van Velson (1979) found rough fish made up 88% of a fish population before relief from grazing and only 1% of the population after 8 years' rest. Rainbow trout *(Salmo gairdneri)* made up 1% of the fish population before cessation of grazing and 97% of the population after relief from grazing. Marcuson (1977) found that an overgrazed section (.11 ha/AUM) of Rock Creek, Montana, supported only 71 kg of brown trout *(Salmo trutta)* per hectare; whereas an ungrazed section produced 238.8 kg of brown trout per hectare. Claire and Storch (unpublished) found in the Blue Mountains of Oregon that game fish were 24% of the total population in area grazed season long, contrasted to a 77% game fish composition within a livestock exclosure.

Chapman and Knudsen (1980) found 8 sections of streamside vegetation in western Washington, judged to be moderately to heavily affected by livestock, had significant reductions in total biomass for Coho salmon *(Oncorhychus kisutch)*, Cutthroat trout *(Salmo clarki)*, and all salmonids compared to those areas that had not been grazed. Similar relationships between livestock grazing and salmonid fish populations have been reported by Dahlem (1979), Duff (unpublished), Gunderson (1968), Keller et al. (1979), and Lorz (1974).

Impacts of Livestock on Terrestrial Wildlife

Riparian zones are the most critical wildlife habitats for many species in managed rangelands (Thomas et al. 1979). It is readily apparent that riparian ecosystems are of paramount importance in producing and maintaining a large degree of biotic diversity in North America (Hubbard 1977, Johnson et al. 1977).

Changes in plant vigor, growth form and species composition due to grazing have frequently been related to the increase or decline of various species of birds (Townsend and Smith 1977). Several studies have shown a negative impact on certain avian populations due to grazing (Dambach and Good 1940, Overmire 1963, Owens and Meyers 1973, Reynolds and Trost 1980, Smith 1940). The tendency for livestock to congregate and linger around ponds and streambanks may result in the elimination of food and cover plants and reduces nest sites and habitat diversity (Buttery and Shields 1975, Behnke and Raleigh 1978, Crouch 1978, Evans and Krebs 1977). However, grazing may improve habitat for some avian species (Burgess et al. 1965, Crouch 1982, Kirch and Higgins 1976). In areas of higher precipitation (or productivity), grazing may be highly desirable to open up "roughs" and provide more diversity and patchiness (Ryder 1980). Grazing effects on breeding avifaunas are not uniform nor easily defined, primarily because grazing varies so much in its local intensity and because of the general difficulties in unraveling cause-effect relationships in rangeland faunas (Wiens and Dyer 1975).

Several studies have shown wildlife numbers increased when a riparian area that was abused by improper grazing practices was fenced and allowed to recover (Crouch 1978, 1982, Duff 1979, Van Felson 1979, Winegar 1977). Duff (1979) reported a 350% increase in small mammal songbird and raptor use after 8 years' rest from grazing. Van Velson (1979) reported increased pheasant *(Phasianius colchicus)* production, increased deer populations, and that watefowl production occurred for the first time in the rested area. Crouch (1982) found more ducks (primarily mallards) *(Anas platyrhynchos)*, more upland game animals, and twice as many terrestrial birds in an ungrazed bottomland rested for 7 years compared to adjacent grazed bottomlands on the South Platte River in northeastern Colorado. The grazed areas, utilized at "varying intensities, provided habitat for significantly more aquatic species of birds.

Mosconi and Hutto (1982) found no significant differences in total bird densities between heavily grazed riparian communities (2.5 cow-calf units/ha) and lightly grazed riparian communities (0.3 cow-calf units/ha). However, significant differences were recorded in bird species composition and foraging guilds. The majority of the bird species significantly affected were of the flycatcher, ground-foraging thrust, or foliage-gleaning insectivore guilds.

Similar results were reported by Kauffman (1982) and Kauffman et al. (1982). No significant differences in total avian densities were noted between riparian communities grazed under a late-season grazing scheme (2.0–2.5 ha/AUM) and those totally excluded from grazing. However, forage removal causing a change in habitat physiognomy did appear to cause some differential use in species and foraging guilds. These differences were particularly evident immediately after forage removal and negligible during seasons when cover and plant growth were similar between treatments. The grazed riparian communities were preferred by birds of insect foraging guilds; ungrazed riparian communities were preferred by birds of herbivorous/granivorous foraging guilds.

Livestock grazing and the subsequent removal of forage in the riparian zone has been shown to cause significant short-term decreases in small mammal composition and densities (Kauffman et al. 1982). When mammal densities before and after the grazing season in 1979 (stocking rate of 2.0–2.5 ha/AUM) were compared, small mammal communities decreased from 800 to 83 mammals/ha in Douglas hawthorn *(Crataegus douglasii)*-dominated communities; from 450 to 60 mammals/ha in riparian meadow communities; and from 129 to 42 mammals/ha in black cottonwood *(Populus trichocarpa)*-mixed conifer communities. By late summer the following year (10 months after grazing) and just prior to the grazing season, small mammal densities were not significantly different between grazed and ungrazed areas.

When properly managed, the grazing of domestic livestock is generally compatible with wildlife, and may even increase the numbers of some species (Tubbs 1980). Nongame wildlife which depend on riparian ecosystems have intangible values which are very hard to evaluate (Peterson 1980). It has been demonstrated that livestock can graze streamsides without causing serious damage, and the capability to achieve positive on-site livestock control appears to be the limiting factor (Claire and Storch unpublished).

Impacts of Livestock on Riparian Vegetation

Recently there has been much published research and opinion on the effects of livestock in riparian ecosystems. Specifically, these reports have dealt with soil compaction and its relationship to root growth; plant succession and productivity; and species diversity and vegetation structural diversity. Opinions on the subject have varied from there being no evidence of heavy, season-long cattle grazing affecting the productivity of a riparian zone, or causing bank deteriorations by trampling (Roath 1980) to grazing only a few days seriously impairing a riparian zone's reproductive capability.

Impacts to riparian vegetation induced by livestock can basically be separated into: (a) compaction of soil, which increases runoff and decreased water availability to plants; (b) herbage removal, which allows soil temperatures to rise and increases evaporation to the soil surface; and (c) physical damage to vegetation by rubbing, trampling, and browsing (Severson and Boldt 1978).

Impacts of Trampling

The impact of livestock trampling on soil compaction bulk density and subsequent effects on forage growth have been documented. Alderfer and Robinson (1949), Bryant et al. (1972), Orr (1960), and Rauzi and Hanson (1966) all found soil compaction increased linearly with increases in grazing intensity.

Alderfer and Robinson (1949) found grazing and trampling Kentucky bluegrass *(Poa pratensis)* upland pastures to a 1-inch (2.5 cm) stubble height reduced vegetation cover, lowered yields, decreased noncapillary porosity, and increased the volume weight of the 0-1 inch (0-2.5 cm) layer of soil.

Rauzi and Hanson (1966) found water intake rates on silty clay and silty clay loam soils to be 2.5 times greater in an area grazed at 1.35 acres/AUM compared to an area grazed at 3.25 acres/AUM. After 22 years of grazing at this intensity, not only had species composition been altered but soil properties had been changed as well.

In a riparian zone continuously grazed season long, Orr (1960) found bulk density and macropore space to be significantly greater in grazed areas over exclosures. Differences in total pore space (both macro- and micro-pores) between grazed and exclosed areas were small because of a transformation of macropore spaces to micropore spaces by trampling. Macropore space is a more sensitive indicator of compaction or recovery from compaction than either micro or total pore space (Orr 1960).

Bryant et al. (1972) found increasing trampling pressure had an adverse effect on Kentucky bluegrass swards, particularly during the months of June and September. After one overwinter period, there was a significant difference in soil compaction between an area trampled by 120 cow trips over bluegrass plots and an area that was untrampled.

Impacts of Herbage Removal

Impacts of herbage removal can be divided into 2 categories according to vegetation structure: (1) utilization of herbaceous vegetation and subsequent impacts on species composition, species diversity, and biomass produced and (2) utilization of woody vegetation and subsequent impacts on foliage cover, structural height diversity and stand reproduction.

A major vegetation change that has taken place in mountain riparian systems of the Pacific Northwest is replacement of native bunchgrass with Kentucky bluegrass. It has successfully established itself as a dominant species in native bunchgrass meadows as a result of overgrazing by herbivores and subsequent site deterioration (Volland 1978).

Pond (1961), in Wyoming, found clipping native bunchgrass meadows every 2 weeks for 4 years caused a marked reduction in native sedges (*Carex* spp.), tufted hairgrass *(Deschampsia caespitosa)* and fostered the appearance of Kentucky bluegrass where it was not present before. Kauffman et al. (1983a) found that when grazing was halted in moist meadows, succession towards a more mesic/hydric plant community occurred. Exotic grasses such as meadow timothy *(Phleum pratense)* and forbs more attuned to drier environments were decreasing and were being replaced by native sedges and mesic forbs.

In central Oregon, Evenden and Kauffman (unpublished) compared plant communities on each side of a fence that was heavily grazed on one side and protected from grazing on the other. The grazed site was dominated by Kentucky bluegrass and Baltic rush *(Juncus balticus)*, while the ungrazed site was dominated by panicled bullrush *(Scirpus microcarpus)*. Twenty herbaceous species were recorded in the grazed area with 12 herbaceous species recorded in the ungrazed area. Dobson (1973) also found an increase in species numbers due to grazing in a riparian zone in New Zealand. He concluded the effect of grazing had been to open up the vegetation, creating more niches in which weeds could establish themselves. Hayes (1978) in central Idaho also observed that the abundance of forb species appeared to be higher in grazed areas than in pristine areas.

The impact of cattle on herbaceous productivity in riparian zones has been examined along several streamsides in the western United States. Duff (1979), Gunderson (1968), Kauffman et al. (1983a), Marcuson (1977), McLean et al. (1963), and Pond (1961) found either decreases in biomass due to herbage removal or increases in biomass due to cessation of grazing in riparian ecosystems.

Kauffman et al. (1983a) compared grazed and ungrazed responses on 10 riparian plant communities in northeastern Oregon from 1978 to 1980. Three of 10 communities displayed significant standing biomass differences. Production in ungrazed moist meadows dominated by Kentucky biomass, meadow timothy, and sedges was significantly less after 2 years of rest compared to grazed meadows but was not significantly different after 3 years of rest. Standing biomass in a Douglas hawthorn-dominated community and in a Kentucky bluegrass-dominated community was significantly greater in ungrazed stands compared to grazed stands after 3 years. Conversely, Volland (1978) could find no significant differences in biomass between a Kentucky bluegrass meadow grazed annually and one that had been rested for 11 years.

Effect of herbivory on shrub and tree production is a critical impact in riparian ecosystems, because of the importance of the woody vegetation to wildlife habitat and its dominant influence in altering the riparian microclimate. While mature vegetation approaches senescence, excessive grazing pressures have prevented the establishment of seedlings, thus producing an even-aged nonreproducing vegetative community (Carothers 1977, Glinski 1977).

The effects of excessive herbivore use on woody vegetation bordering streamsides can generally be termed as negative. Knopf and Cannon (1982) found that cattle significantly altered the size, shape, volume, and quantities of live and dead stems of willows. Cattle grazing was also found to influence the spacing of plants and the width of the riparian zone. Marcuson (1977) found shrub production to be 13 times greater in an ungrazed area than in a severely overgrazed area. Cover was 82% greater in the natural area. On a stream rested from continuous grazing for 10 years, Claire and Storch (unpublished) found alders (*Alnus* sp.) and willows (*Salix* spp.) provided 75% shade cover over areas that had been devoid of shrub canopy cover before exclosure. Similar herbivore-woody vegetation relations have been reported by Crouch (1978), Davis (1982), Duff (1979), Evenden and Kauffman (1980), Gunderson (1968), and Kauffman (1982).

Management of Riparian Ecosystems

Recognizing and understanding the impacts on the streamsides which resulted from all previous land use practices is a prerequisite to streamside planning (Claire and Storch unpublished). Because of their small extent, riparian zones in the past were considered "sacrifice areas" (Oregon-Washington Interagency Wildlife Council 1978, Skovlin et al. 1977). Riparian vegetation has been intensively used by livestock over several decades causing a reduction in the productivity of fish and wildlife habitats and degrading water quality as well as promoting increases in flow fluctuations (Oregon-Washington Interagency Council 1978).

Platts (1979) indicated that riparian ecosystems are the most critical zones for multiple-use planning and offer the most challenge for proper management; therefore, stream habitats should be identified as separate management units from the surrounding upland ecosystems. Even among riparian zones the need to identify and classify them adequately is important for proper stewardship of these systems (Claire and Storch unpublished, Platts 1978, 1979).

However, there have been few attempts to come up with a viable classification scheme of riparian vegetation that is feasible for land management activities (Cowarden 1978, Norton et al. 1981, Padgett 1982, Pase and Layser 1977, Tuhy and Jenson 1982). The major problem has been the lack of successional knowledge to formulate classification schemes based upon potential climax communities. Other problems have been the lack of continuity of terminology. For example, terms such as riparian dominance type (Padgett 1982), community type (Tuhy and Jenson 1982), and riparian type (USFS-R-4 file data) have all been used to define the basic unit of land which supports a riparian community.

Land management agencies responsible for managing livestock grazing have not adequately considered the influence of grazing on the other uses and users of riparian ecosystems (Platts 1979). Often what is good range or timber management (in short-term economic terms) is not good riparian or stream management (Platts 1979). On the other hand, it has been suggested that proper stream management practices that protect stream banks from damage also improve the potential for riparian zones to enhance fisheries, wildlife, and livestock uses (Gunderson 1968, Marcuson 1977).

Methods discussed for riparian zone rehabilitation include exclusion of livestock grazing, alternative grazing schemes, changes in the kind or class of animals, managing riparian zones as "special use pastures," in-stream structures and several basic range management practices (eg. salting, alternative water sources, fencing, range riders, etc.).

The use of instream structures as a method of riparian rehabilitation has met with some success where instream structures are combined with rest from livestock grazing (Duff unpublished, Heede 1977). Bowers et al. (1979) indicated that some instream structures (e.g., trash catchers, gabions, small rock dams, individual boulder placement, rock jetties, and silt log drops) could serve the dual purpose of increasing the water table in areas of former wet meadows as well as improving salmonid habitat.

Heede (1977), combining rest from grazing with construction of check dams, obtained vegetation cover improvements, a change from an ephemeral stream flow to a perennial flow and a stabilization of gully erosion.

After losing 23 out of 26 instream structures in a grazed area in Utah, Duff (unpublished) suggested that stream improvement structures cannot work effectively to restore pool quality and streambank stability as long as livestock grazing continued. Keller et al. (1979) in Idaho found that rest from grazing negated the need for artificial instream structures intended to enhance trout production for stream ecosystems. Kimball and Savage (in Swan 1979) found aquatic ecosystems can be restored through intensive livestock management at a lower cost than through installation of instream improvement structures.

Grazing systems have achieved some success in riparian rehabilitation and much success in riparian ecosystem maintenance. The damage caused by heavy season or yearlong grazing is well documented (Evans and Krebs 1977, Gunderson 1968, Marcuson 1977, Severson and Boldt 1978). It appears that rest-rotation grazing schemes and/or specialized grazing schemes in which riparian zones are treated as special use pastures have been the most successful.

Hayes (1978), in Idaho, stated that species composition appeared to be improved under a rest-rotation grazing system and bank sloughoff occurrences were not increased if utilization was under 60%. In other Idaho mountain grazing studies, Platts (1982) stated that when rest-rotation strategies call for livestock to utilize riparian vegetation at a rate of 65% or more, some riparian habitat alteration occurs. He also indicated that riparian alteration may be insigificant when utilization is equal to 25% or less.

Claire and Storch (unpublished) found a rest-rotation system to be favorable for achieving desired streamside management objectives if 1 year's rest out of 3 is included in the scheme.

Davis (1982), in Arizona, found that a four-pasture rest-rotation system was a cost-effective and successful method for rehabilitation of the riparian resource when each pasture received spring-summer rest for 2 years out of 3. On 2 grazing allotments, cottonwood and willows had a mean increase from 78 plants/ha to 2,616 plants/ha, 2 years after implementation of the system. A rest-rotation system also obtained a very favorable response for vegetation surrounding a livestock pond in South Dakota (Evans and Krebs 1977).

Criticism of rest-rotation systems includes reports that objectives for herbaceous vegetation were not being achieved within desired time limits (Storch 1979), and that rest-rotation systems may increase trailing and trampling damage, causing streambank erosion and instability (Meehan and Platts 1978).

Fencing and managing riparian zones separately from terrestrial upland sites as special use pastures has been shown to be an adequate multiple use system of riparian zone management (Kauffman 1982, Winegar 1977). Simulated grazing of a fenced riparian zone annually after August 1 had no measurable effect on production or species composition in riparian meadows, contrasted to decreased production and composition in a simulated season-long scheme in northcentral Wyoming (Pond 1961).

Kauffman (1982) suggested that positive characteristics of a late season grazing scheme on a riparian zone in Oregon included increased livestock production, good plant vigor and productivity, minimal soil disturbance, and minimal short-term disturbance to wildlife populations dependent on riparian ecosystems.

Another grazing system for fenced riparian zones includes winter grazing, where possible, to minimize damage (Severson and Boldt 1978). For riparian meadows dominated by Kentucky bluegrass, Volland (1978) recommended an initial year's rest, then late spring grazing alternated with late fall grazing to discourage flowering, increase tiller development, maintain plant vigor, and maximize productivity.

Changes in the kind or class of animal as well as selective culling and breeding may be another positive tool for riparian rehabilitation or maintenance. Roath (1980) found that cattle exhibited distinctive home range patterns in which certain groups of cattle preferred upland sites and groups preferred riparian sites. As forage became limiting on stream bottoms, some cattle actually decreased intake rather than move away from the riparian zone. Selective culling of these cattle and replacing them with those that prefer uplands may be beneficial for the livestock operator as well as for the riparian zone.

Platts (1982) stated that because sheep grazing on public lands is usually controlled by the use of herders, it may be possible to graze a watershed without exerting direct significant influence on riparian habitats. May and Davis (1982) suggested that sheep have been shown to exert a lesser influence on certain riparian and aquatic ecosystems and converions back to a sheep operation may be necessary to improve some riparian areas.

The most successful riparian management alternative on public lands to date has been intensive livestock management by permit holders (Storch 1979). Herding livestock on a somewhat daily basis has been successful in limiting the number of livestock that visit streambottoms and improving utilization of upland areas. Proper stewardship of riparian ecosystems is, in effect, money in the bank for the floodplain rancher (Marcuson 1977). Proper management of riparian zones means decreased streambank erosion and floodplain losses (Duff 1979, Gunderson 1968, Marcuson 1977), increased forage production (Evans and Krebs 1977, Pond 1961, Volland 1978), and an increased wildlife and fisheries resource (Buttery and Shields 1975, Duff 1979, Tubbs 1980, Van Velson 1979).

In conclusion, public grazing lands must be managed on a true multiple use basis that recognizes and evaluates the biological potential of each ecological zone in relation to the present and future needs of our society as a whole (Behnke et al. unpublished). Management strategies that recognize all resource values must be designed to maintain or restore the integrity of riparian communities (Behnke et al. unpublished).

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The Status of Our Scientific Understanding of Lodgepole Pine and Mountain Pine Beetles – A Focus on Forest Ecology and Fire Behavior

A synthesis of our current knowledge about the effects of the mountain pine beetle epidemic on lodgepole pine forests and fire behavior, with a geographic focus on Colorado and southern Wyoming.

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Introduction

Major lodgepole pine forest changes and how they affect us. Mountain pine beetle populations have reached outbreak levels in lodgepole pine forests throughout North America. The geographic focus of this report centers on the southern Rocky Mountains of Colorado and southern Wyoming. The epidemic extends much more widely, however, from the southern Rocky Mountains in Colorado in the United States to the northern Rocky Mountains in British Columbia and Alberta, Canada.

Worries about large-scale tree mortality in lodgepole pine forests have created public concerns across the West. The appearance of red trees during the last decade, a clear sign of recent beetle attack, has been followed by bare dead tree skeletons throughout this large area. Unquestionably, millions of dead trees foretell large forest changes in the near future, and more might be anticipated in areas where the mountain pine beetle has not yet reached epidemic levels.

People are concerned for many reasons. At a minimum, the loss of mature lodgepole pine trees will significantly change the present and future appearance of affected forests for half a century or more. Extensive areas of dead trees and snags are not as aesthetically appealing as live forests. Perhaps more seriously, dying and dead trees raise fears of increased fire danger. Some people worry that the dead needles and wood generated by the mountain pine beetle epidemic will lead, perhaps quickly, to severe wildfires that threaten lives, property, wildlife, and watersheds. Many are concerned that trees not yet attacked will succumb to the epidemic. Some people worry that the forest in and around our communities and recreation areas will become sparse or disappear forever, and that these forest changes will affect timber commodities, game habitat, and recreation resources.

Some contend that the current epidemic with synchronous outbreaks at many locations is unprecedented and a clear warning of global climate change impacts on ecosystems around the world. Scientists and others point to other changes occurring in our region – *Ips* beetle-caused mortality of piñon pine in the Southern Rocky Mountains, aspen decline, and large fires in Front Range ponderosa pine forests and elsewhere. It is difficult to prove cause and effect, but all of these changes began during the last 10-15 years, coinciding with recent warm climatic conditions, increasing numbers of large trees, and advancing age of many forests. Whether or not the current epidemic is unprecedented is a question to which there is currently no clear answer because of the lack of precise information on extent and severity of beetle outbreaks prior to the early 1900s. Nevertheless, many in the scientific community believe the probability of a similar event historically over at least the past few 100 years is low.

There are many insights and opinions about lodgepole pine being discussed by stakeholders of all kinds -- forest managers, agency administrators, researchers, policy-makers, politicians, the news media, industries, and the general public. Some concerns and fears are supported by scientific evidence. Others are probably justified given the current status of our scientific knowledge, but lack clear scientific support. Still others are myths with little or no basis in science. A further complication is that some of the information emerging from the science community has appeared on the surface to be somewhat contradictory.

The reason for this report. This document is written to report our current scientific understanding of the ecology and fire behavior of lodgepole pine, with a focus on the direct and indirect effects of the current mountain pine beetle epidemic that is so dominant in our minds. We recognize that important socio-economic implications stemming from the mountain pine beetle epidemic exist, and we hope that examining the status of science will aid in addressing these issues. While this document focuses on lodgepole pine and mountain pine beetles, there are also many other forest types and non-forested systems subject to extreme or at least unexpected impacts of climate, other insect and pathogen species, and other disturbances including fire and wind.

This report results from a meeting in January 2008 convened in Colorado by The Nature Conservancy, bringing together expertise of scientists who study lodgepole pine throughout its geographic range. We hope to provide as much scientific help to stakeholders as possible by sorting out what is known with a high degree of certainty, what we are confident about but with less certainty, and what is truly not understood and in need of more research. While our primary geographic focus during the workshop was Colorado and southern Wyoming, some of the findings may be appropriate for lodgepole pine throughout much of its natural range of distribution. *We urge caution, however, in applying our findings beyond our initial area of focus or to other forest types in the region.*

During the workshop and through subsequent email dialogue, the lodgepole pine team reached consensus on nine key points. As always, science is a work in progress, and uncertainties surfaced during discussion of some key points. For some points we provide what is known with adequate confidence rather than waiting for more definitive information, when this information is useful to interested stakeholders. This report provides the nine key points along with explanatory material intended to help the reader understand the degree of confidence we have from scientific study for these key points. To help the reader, we provide a list of suggested reading at the end of this report for more detailed information on many of the topics discussed. We begin with the obvious.

A. Lodgepole pine forests are being heavily impacted by the ongoing mountain pine beetle epidemic.

From British Columbia to Colorado, forests are experiencing high mortality of lodgepole pine trees from attack by mountain pine beetles. An insect epidemic with multiple outbreaks at this scale has not been observed during the last century of scientific study, though small outbreaks have occurred. This mortality is changing forest structure and composition, and modifying fuels in ways that will affect fire behavior for decades.

Many believe the mountain pine beetle epidemic, now nearly a decade in duration, might be unprecedented at least in recent centuries, stemming from a unique alignment of factors. These factors include extensive forests of trees at the right age, size, and density to support large numbers of mountain pine beetles, and a climate warm enough over the last decade to favor beetle reproduction and survival. But records are short. Modern records cover little more than a century, and for this period there is no account of a similar severe mountain pine beetle epidemic in lodgepole pine over such a large area.

For earlier periods, however, little scientific evidence exists to suggest that severe mountain pine beetle outbreaks either did or did not occur. Forest fires, another important natural disturbance, often scar living trees, which provides physical evidence indicating dates, locations, and severity of fires back through much of the last millennium. Fire-scarred wood is often resistant to rot and may persist for centuries, preserving a record of fire. But mountain pine beetle attacks that might have occurred more than a century ago leave little or no physical evidence helpful for determining dates or severity of such attacks. Wood from trees killed by beetles rots quickly, especially where wood moisture is high (e.g. fallen trees). Both stand-replacing fires and beetle epidemics that kill large numbers of trees allow stands of trees of the same age to establish in the wake of the disturbance. The ages of these trees can be used to estimate the time of the last stand-replacing disturbance, but it is often not possible to tell what kind of disturbance initiated the stand, and disturbances such as beetles, fire, and wind may act synergistically. Thus we cannot exclude the possibility that factors aligning so perfectly to result in the current epidemic could not have aligned equally in past centuries or millennia.

Regardless of whether or not the current mountain pine beetle epidemic and lodgepole pine mortality are within the historical range of variability at some time scale, the epidemic and associated tree mortality are large and are having immediate effects on forest structure and function over a vast area.

B. Not all lodgepole pine forests are the same.

Some forests are composed of nearly pure lodgepole pine established following large fires decades or centuries ago. Others are mixtures of lodgepole pine with subalpine species such as Engelmann spruce, subalpine fir, and aspen at higher elevations, or with mixed conifer species such as ponderosa pine, Douglas-fir, and aspen at lower elevations. Each type of forest has unique features of ecology and fire behavior. And lodgepole pine trees in all three types are vulnerable to attack by mountain pine beetles.

Lodgepole Pine Ecology 101. Lodgepole pine is found over a large area in western North America, from northwestern Canada in the northern Rocky Mountains; Washington, Oregon, and California in the Cascades and Sierra Nevada; Idaho, Montana, and Wyoming in the central Rockies; down to Colorado and even northern New Mexico in the southern Rockies. It comes as no surprise that across this large area and also locally, lodgepole pine trees are found in diverse forest conditions. In Colorado and southern Wyoming, pure stands of lodgepole pine occur. Even where pure stands occur, lodgepole pine forests may range from extremely dense to open and savanna-like. Elsewhere, lodgepole pine is mixed with other species. These differences in species composition of forests influence the way forests are affected by mountain pine beetles and fire, and how forests may change in the future.

Two key features of lodgepole pine are especially important in the way the species interacts with the environment and with other trees. Lodgepole trees are relatively intolerant of shade, and they are adapted to reproduce prolifically after fire. Unshaded lodgepole trees survive and grow more readily than trees overtopped either by larger lodgepole pines or by other species. Fire adaptation in trees occurs in two primary forms: the capacity to survive fire, or the ability to reproduce after fire even if killed. While species like ponderosa pine are adapted to survive fire, lodgepole pine is adapted to reproduce readily after fire.

Many lodgepole pine trees have serotinous cones that remain closed and store viable seeds in the crowns of trees for years, actually requiring the heat of a fire for seed release and dispersal. When crown fires kill trees, the resin sealing the cones melts, allowing the cones to open shortly after the fire. Huge numbers of seeds are released at once to the forest floor, falling on exposed soil that is an excellent seedbed for germination and seedling establishment. It is not uncommon to find 50,000 or more seedlings per acre several years after a stand-replacing fire. Competition then thins out trees naturally as these young forests grow to maturity. After a mountain pine beetle epidemic, lodgepole pine stands also generally regenerate, because serotinous cones on branches that have fallen near the ground heat adequately to release seeds, and seeds previously released from non-serotinous cones may exist in the forest litter. However, the role of serotinous and non-serotinous cones as seed sources, and the effect of cone serotiny on subsequent stand density, are not well understood.

The three most common natural agents influencing lodgepole pine in Colorado and southern Wyoming other than fire are mountain pine beetles, dwarf mistletoe, and wind. Of these, mountain pine beetles have the capacity like fire to change forests at large scales. Beetle populations can occasionally reach epidemic densities over large areas, though not usually as large as the current epidemic. The spatial extent of the current epidemic is probably related to large numbers of suitable host trees existing over much of the range of lodgepole pine in the West. Mountain pine beetles are a native insect that has evolved with lodgepole pine. They normally exist in endemic populations that kill a few trees but are regulated by weather. Endemic populations of beetles typically infest diseased or stressed trees. Because temperature regulates beetle development, prolonged warm periods may help trigger outbreaks. Natural enemies also help regulate endemic bark beetle populations but their role under epidemic populations is not as effective.

Dwarf mistletoe typically occurs in localized patches. While mistletoe slowly spreads, it often remains only locally significant, and trees may live for decades with mistletoe. This native parasite, which also evolved closely with lodgepole pine, is periodically reduced by fires that kill the infected trees. Major wind events may topple trees and create small to large openings. In many places lodgepole pines are shallowly rooted in rocky soils or on steep slopes. Typically even the largest blowdowns affect forests only locally, and while they contribute to the landscape mosaic of forest age and composition, they are unlikely to affect forests regionally unless they become centers of another disturbance agent (e.g. spruce beetle).

Three kinds of lodgepole pine forest. Lodgepole pine forests occur along gradients of elevation and latitude that control the length of growing season, available moisture, and frequency of natural disturbances. Fire and mountain pine beetles affect forest structure and composition differently in each ecosystem, just as environmental conditions regulate the occurrence and intensity of the disturbances. To understand this, it is useful to identify three specific types of forest in which lodgepole pine occurs. In Colorado and southern Wyoming, these are pure lodgepole pine, subalpine forest, and mixed conifer forest.

Pure lodgepole pine forests may occur where environmental conditions are poorly suited for other tree species, or where human impacts such as logging followed by burning eliminate other species. Lodgepole pines are tolerant of cold, dry conditions and poor, rocky soils. Individuals rarely live more than 400 years. Typically, pure lodgepole pine stands result after stand-replacing fires have killed all or most trees, leaving behind lodgepole seeds stored in serotinous cones as the only significant seed source. Alternatively, fire-killed stands without serotinous cones may still reproduce if lodgepole pine seeds are blown in from unburned trees nearby. Stand-replacing fires may occur in healthy, green forests under extreme weather conditions. Similar fires might occur under more moderate conditions when mountain pine beetle mortality or mistletoe infestation in stands creates additional dry fuels, though there is no firm evidence thus far confirming this. Pure lodgepole pine stands are often established within a few years after the fire and have one dominant age class or cohort for the life of the new stand, although some stands may develop continuously over longer periods of time and have multiple age classes. However, if aspen is present even in small amounts before large fires, its sprouting capability may lead to aspen patches which often give way over time to slower-growing lodgepole pine.

The spatial extent of pure lodgepole pine forests typically reflects the size of the fires that established them. As a general rule, pure lodgepole pine forests occur more commonly at upper elevations in the mixed conifer (upper montane) zone and the lower portion of the subalpine forest zone, between 9,000 and 10,000 feet elevation in Colorado and southern Wyoming. Less commonly, pure stands exist because sites are unsuitable for other tree species. Historically, past fires may have been tens to hundreds of thousands of acres in size, resulting in large lodgepole pine stands that dominate the landscape for several hundred years. However, even large intense fires do not burn uniformly, and within a fire perimeter, some patches of trees or individuals may survive intact. The 1988 Yellowstone fires are a good example of this. Alternatively, smaller crown fires may have created patches of pure lodgepole pine as small as an acre or less.

If not renewed by fire every few centuries, pure lodgepole pine stands often but not always experience ingrowth by other tree species, especially those tolerant of moderate shade. Ingrowth of other species depends strongly on site suitability for the other species, and availability of seeds. Eventually these species may replace lodgepole pine as the dominant trees in the stand. Lodgepole pine may persist in these mixed stands even if only a limited number of seedlings become established periodically, usually as a consequence of minor local disturbances such as very small fires, wind, insects, or disease.

Subalpine forests at higher elevations (usually above 10,000 feet elevation but as low as 9,000 feet) often include lodgepole pine as a component along with Engelmann spruce, subalpine fir, and aspen. Stand-replacing fires may occur in subalpine forests, but intervals between fires are usually several to many centuries (compared to one to several centuries for pure lodgepole pine forests). After stand-replacing fire, lodgepole pine seedlings grow faster than spruce or fir seedlings and may dominate stands during early developmental stages, even when spruce and fir seeds are available nearby. When aspen is present, however, creation of openings by fire or other disturbances may shift species dominance to aspen because of its sprouting ability.

Mixed conifer forests at lower elevations (usually between 7500 and 9000 feet elevation) often include lodgepole pine along with ponderosa pine, Douglas-fir, aspen, and perhaps small amounts of subalpine fir, Engelmann spruce, and limber pine. Large stand-replacing fires can occur in mixed conifer forests and may lead to pure lodgepole pine stands. More typically, however, mixed-severity fires create smaller openings providing opportunities for patches of lodgepole pine establishment and persistence within the complex landscape mosaic of mixed conifer. Once again, aspen may become temporarily dominant if it existed prior to the fire.

C. Forests are living systems subject to constant change.

It is normal and expected that many natural agents, including mountain pine beetles, fire, and wind, change forests over time. Some changes are so gradual that we barely notice them, while others are relatively sudden and extensive. The forests that are presently losing many trees to insect attack will not look the same in our lifetimes, but healthy and vigorous forests will eventually return in most locations.

We tend to think of forests as static over time because their change is slow relative to human time scales. Yet forests are non-equilibrium systems, and we should expect them to change. Our adult human experience is measured in years or decades at most, and we often fail to notice all but the more dramatic changes that occur in forests. Thus we may believe that the structure and composition of forests typically do not (and even should not) change, and, when they do, it means something alarming has happened. However, lodgepole pine and other tree species live several centuries or more and during their life cycles a number of very natural, and ecologically predictable, forest-changing events or processes often occur. The 1988 Yellowstone fires are often cited as an example of natural change in lodgepole pine ecosystems.

Taking this more comprehensive view, it is clear that combinations of fire and other natural disturbance agents, along with differences in ecological characteristics of the various tree species suited for the landscape, result in frequent changes in forest landscapes over time. The overall forest mosaic is in fact not static, but rather experiences significant shifts and adjustments, all a part of the natural ecology of forests. Thus at any location in a given landscape, the species composition, distributions of tree sizes and ages, and stand density all are subject to change, even if in our memory they do not appear to.

Understanding and predicting the consequences of natural disturbance effects on landscapes is difficult. All of the natural disturbance factors – fire, insects, pathogens, wind, drought, etc. – are capable of affecting forest landscapes at various scales and may act individually or in combination. In the current mountain pine beetle epidemic, interactions between fire and beetle effects are certain, because the insects are changing fuel characteristics of forests significantly.

D. Lodgepole pine will not disappear from the southern Rocky Mountains.

The make-up of our forests is already changing where mountain pine beetles cause high mortality of lodgepole pine. However, this event will not cause the extinction or disappearance of lodgepole pine, and forests dominated by or including lodgepole pine will persist in the southern Rockies, though they may look different from those of the past due to changing climate. Future forests will continue to provide valuable ecological services and aesthetic and recreational benefits.

When viewed from a distance, it may appear that many pure lodgepole pine forests in Colorado and southern Wyoming are being completely killed. It even appears that in some places all the lodgepole pine trees in subalpine or mixed conifer forests are being killed. Yet there is wide variability in the amount of tree mortality, and even where all the mature trees have died, understory saplings may be released and new lodgepole pine seedlings are likely to emerge. Thus it is untrue that lodgepole pine will disappear from our forests.

Scientific knowledge is not complete, however, and there is considerable uncertainty about the composition of future forests after the epidemic. Clearly, major changes in these forests are occurring, but multiple factors will affect what kind of new forest will result. A high proportion of larger lodgepole pine trees (diameters greater than six inches) are dying, and in many places many smaller trees are being killed as well. Mortality may approach 100% in pure lodgepole pine stands having few small trees.

Recovery of lodgepole pine forests following previous beetle outbreaks suggests, however, that in many places significant numbers of lodgepole seedlings and small saplings will survive. These may produce new pure stands of lodgepole pine if no other species are present, or help sustain a lodgepole component in stands of mixed species. Height growth of Engelmann spruce or subalpine fir seedlings is slow compared with that of lodgepole seedlings. Where small seedlings of spruce or fir existed beneath a pure lodgepole pine overstory, lodgepole pine may still predominate after the first decade because of their more rapid growth. However, if saplings of spruce and fir trees are left under the dead pines, they may grow quickly into the canopy and dominate the site. If aspen is present, sprouting and rapid early growth may result in an aspen forest, perhaps with the shade tolerant conifer species in the understory. However, aspen sprouting after mountain pine beetle mortality is not as well understood as it is for disturbances that more directly affect aspen trees or roots.

In pure lodgepole pine forests with few or no surviving trees, it is reasonable to expect a new lodgepole forest to regenerate on suitable sites, but difficult to predict with certainty. The existing seed bank (seeds stored in cones of dead trees and in the litter) or seeds produced by non-serotinous trees near the time of tree death may produce enough new seedlings to regenerate a new lodgepole pine forest. It is also possible that other species will colonize the sites, including other wind-dispersed trees such as spruce and fir, ponderosa pine and Douglas-fir, or bird-dispersed trees such as limber pine. Grasses, forbs, and shrubs may flourish in the new openings for periods of time, and tree establishment may be limited or slowed. Under such conditions, the landscape is likely to become more diverse than it was in the previously pure, single-aged lodgepole forests. This in itself may be beneficial for reducing the risk of a future large-scale mountain pine beetle epidemic or other monolithic disturbance.

E. Active vegetation management is unlikely to stop the spread of the current mountain pine beetle outbreak.

Mountain pine beetles are so numerous and spreading so rapidly into new areas that they will simply overwhelm any of our efforts where trees have not yet been attacked, and no management can mitigate the mortality already occurring. However, judicious vegetation management between outbreak cycles may help mitigate future bark beetle-caused tree mortality in local areas.

In the current epidemic, it is impractical to expect that silvicultural treatment of lodgepole pine forests will prevent or even impede the advance of the epidemic in Colorado and southern Wyoming. There are simply too many suitable host trees over too large an area, and unusually high insect populations. Unless climatic conditions become less favorable for beetle reproduction and spread, the most likely scenario is that the epidemic will be sustained until host trees are depleted.

Preventive spraying of high-value trees with insecticides is effective in protecting trees from bark beetle attack. Direct control measures such as removing infested trees may provide some mitigation on a small local scale but are not be effective at a landscape scale.

The current epidemic is so extensive and severe in part because large areas of lodgepole pine forest are suitable hosts for mountain pine beetles. As noted earlier, it is unclear if epidemics occurred at such a large scale historically, though smaller-scale or less severe epidemics most likely did occur and are expected in the future. Active vegetation management *between* periods when lodgepole pine forests are vulnerable to a mountain pine beetle epidemic may reduce the magnitude of future landscape-scale outbreaks, if that is chosen as a management objective. Creating diverse patch ages and sizes (including young patches) and perhaps more mixed-species forests across the landscape may or may not reduce the spread of future mountain pine beetle outbreaks, but it likely would reduce the amount of forest susceptible through time to a monolithic disturbance, including mountain pine beetle attack or fire. Thus while unproven, this increased landscape heterogeneity may be effective for limiting the scale and severity of future mountain pine beetle impacts. The effectiveness of such measures cannot be assured, nor are all the ecological consequences known, though even in the current epidemic, stands and patches of younger lodgepole pine trees appear to have survived the epidemic with no or only limited mortality.

F. Large intense fires with extreme fire behavior are characteristic of lodgepole pine forests, though they are infrequent.

Very dry and windy conditions can lead to large intense fires in lodgepole pine forests. Such fires are a natural way for lodgepole pine to be renewed and are largely responsible for extensive pure lodgepole pine forests.

Fire history studies based on fire scars and stand structure evidence extending over at least the past 500 years show that large, severe fires (often involving multiple ignitions) occurred in subalpine lodgepole pine forests of Colorado and southern Wyoming during periods of exceptionally warm and dry weather. These studies also show that long intervals (e.g. of 80 to 100 years) during which large fires were absent from study areas extending over 10,000 or more acres were common during the past five centuries in subalpine lodgepole pine forests. Climatic variation at annual and multi-decadal time-scales has been the major driver of fire occurrence in these forests and is the key explanation for the non-equilibrium behavior of these ecosystems. Large fires shaped the amounts and locations of extensive lodgepole pine forests on the landscape and this process is relatively well understood, but additional research would be helpful to characterize stand history in local areas, especially in relation to past climate.

Fire is complex, and its behavior varies with variations in weather, ignitions, fuel amounts and arrangement, and fuel moisture. Historically, most ignitions in lodgepole pine forests were caused by lightning. The role of Native American ignitions is unknown, but given that extensive fire occurs in these forests only under dry and windy conditions, their contribution was probably small. Young and mature stands of pure lodgepole pine are relatively unlikely to burn except under the most extreme weather conditions. Unless residual fuels remain from the effects of previous fire or insect epidemic, fuels commonly are sparse in the understory, and closed canopies help keep the forest floor cool and somewhat moist. The snow-free period above 9000 feet elevation is relatively short, leaving little time for fuels to dry. The term “asbestos forest” has been applied to these forests, attesting to their low probability of an intense crown fire except under extreme weather conditions.

As lodgepole pine forests mature they become increasingly vulnerable to natural disturbances such as mountain pine beetles and wind. Even with only partial overstory mortality, openings created in the forest canopy allow more air circulation beneath the canopy, and drying of surface fuels. In addition, fuel amounts may be increased by the localized tree mortality, including fuel ladders provided by fallen trees, young understory trees, and shrubs that may help fire reach the overstory. Such changes may increase the probability of fuel ignition from lightning and may alter fire behavior in several ways. Fire behavior in maturing stands is not fully understood, however, and more research would be beneficial.

These remarks about fire behavior apply especially to pure lodgepole pine forests. In subalpine mixed forests, the likelihood of dry fuels is even less as the snow-free period is shorter. In mixed conifer forests below 9000 feet, the complexity of the landscape, greater productivity and longer and more frequent fire season encourages mixed-severity fires which have both surface and stand-replacing components. Even in the mixed conifer forests, however, fire extent is highly variable due to climatic variation, and fire-scar studies show that years of widespread fires during past centuries were dependent on exceptional drought. Typically, lodgepole pine occurrence can be suppressed with shortened fire intervals because its long-term presence depends on seed germination after fire and trees growing to reproductive maturity before the next fire. In general, fire history and potential fire behavior are less well understood in mixed conifer forests than in pure lodgepole pine forests.

G. In forests killed by mountain pine beetles, future fires could be more likely than fires before the outbreak.

Large intense fires with extreme fire behavior are again possible.

There is considerable uncertainty about fire behavior following a mountain pine beetle epidemic on this scale. In pure lodgepole pine forests, crown fires are possible both before an epidemic and after while needles are still on trees. Intense surface fires are possible after most dead trees have fallen to the ground. The probabilities of such fires are uncertain, and more research is needed to learn in what ways and how long the fuels and fire environment are altered by the beetles. Nevertheless, protection of communities and other values at risk continues to be imperative.

More research is required to fully understand fire behavior over time following a mountain pine beetle attack. Nonetheless, the extensive epidemic now occurring is precipitating enormous changes in fuel structure over large areas in Colorado and southern Wyoming, through changes in the condition and arrangement of the forest biomass (which is fuel for forest fires). The mature lodgepole pine trees that provided abundant but moist living fuels are now dead, dry, and falling, and have the potential to contribute to extreme fire behavior in post-beetle forests similar to historical fires in lodgepole pine forests. However, the realization of that potentially extreme fire behavior will depend on a number of contingencies, particularly future climatic conditions.

Empirical data are very limited. One study of fire extent and severity of wildfires that burned in subalpine forests in Colorado in the extreme drought of 2002 did not find that fire extent or severity were greater in stands recently killed by mountain pine beetle. The authors cautioned, however, that the conclusions regarding the influence of the recent beetle outbreak on fire extent and severity are limited by spatial and temporal limitations associated with aerial detection of the outbreak. More importantly, any broader applications of this case study would need to be tested by additional studies considering different initial forest (fuel) conditions and especially weather conditions that drive fire behavior. Even though only limited scientific information is available to predict likely fire behavior during and in the decades following a mountain pine beetle epidemic and under varying climate conditions, we believe that both field observations of fire behavior and modeling provide some insights into what could be expected. We offer these insights as preliminary guidance for those concerned with management of beetle-killed forests, even as new research is being conducted to clarify our scientific understanding.

Pure lodgepole pine. In the initial phases of the epidemic when trees are being killed, needles die, turn red and dry out but persist on trees for two or three years. During this phase, needles and small branches provide dry fine fuel that could burn in a crown fire. The amount of fuel is relatively unchanged compared with the pre-epidemic forest. However, fuel moisture is lower, and some think it likely that a crown fire could ignite and spread under somewhat less extreme fire weather conditions than were required for initiating a crown fire in an equivalent forest of live trees.

The fuel structure of dead lodgepole pine stands changes significantly when needles fall to the ground. During this phase, little fine fuel remains in the forest canopy to support an active crown fire that spreads from tree to tree. Furthermore, the fallen needles lie close to the ground surface and, in the absence of other fuels near the ground, provide a relatively poor fuel bed for generating significant flame heights. Increased growth of grasses, low shrubs and forbs may create a moist fuel bed during the growing season but provide dry fine fuels near the end of the growing season. However, large amounts of biomass in the boles and branches of standing trees remain well above typical flame heights, and without needles these canopy fuels are relatively unlikely to burn. Thus surface fires in years following needle fall may not be intense and crown fires may be nearly impossible (assuming the forest is relatively pure lodgepole pine and most or all large trees are dead). In some areas, rapid development of a tall shrub community (which may precede tree regeneration) may provide shade and protection from drying of fuels on or near the ground. However, this is unlikely in most lodgepole pine forests in Colorado and southern Wyoming (the focus area of this report), because few tall shrub species occur in these relatively dry forests. Instead, low shrubs such as huckleberry and buffaloberry are more common.

Trees killed by mountain pine beetle may remain standing for a number of years, but as they progressively decay and fall to the ground (often aided by wind), the fuel structure changes once again. In this phase (typically 10-20 years or more after death), a large amount of biomass becomes available as fuel within flame heights that can be generated by the fine surface fuels. Some of the biomass is elevated above the ground where it dries out more easily and becomes available to support intense fire with a large release of heat. Such a fire is relatively hard to control and nearby structures may be hard to protect. Furthermore, fire intensities under these conditions could cause high mortality of young trees that survived or regenerated after the mountain pine beetle attack. If widespread fire mortality occurs before trees have matured to cone production age, rapid re-establishment of lodgepole pine on this site is less likely.

At the scale of a stand, none of the changes in fire behavior that we have described would be outside the historical range of variability for this ecosystem. Even in stands with tremendous wood accumulation on the ground, fire behavior may differ little from historical fires within blow-downs or areas recently burned by stand-replacing fires. However, we are uncertain about fire behavior at landscape or regional scales because we have not seen systems with such heavy fuel loads over such extensive areas; and we know little about the ecological consequences of such fires at these scales.

Lodgepole pine with other species. Similar transitions in fuel structure also will occur in the lodgepole pine-dominated component of subalpine and mixed conifer stands. But the mixture of dead lodgepole pine with live trees of other species creates a more complex fuel structure. An important effect of lodgepole pine mortality in mixed stands is a change in the environmental conditions and thus the fuel moisture near the forest floor. Prior to beetle mortality of the overstory, solar radiation is largely intercepted by the forest canopy, and air movement beneath the forest canopy is moderated by the overstory. The understory beneath the canopy remains relatively cool and moist.

When lodgepole pine trees die and needles fall from dead trees, radiation reaching the forest floor and air movement beneath the residual live tree canopy are increased, and both contribute to fuel drying. More open canopies also contribute to greater understory vegetation growth. The consequences of these changes on fire behavior are not fully understood, but such conditions may favor ignition and spread of fire more readily than in forests having few canopy gaps or fuels created by mountain pine beetle mortality, particularly later in the growing season when fuels near the ground become drier. Because several associated species, firs and spruces, typically have low crown bases due to poor self-pruning, higher surface fire intensity from added lodgepole pine fine fuels coupled with drier, warmer, windier surface conditions, could lead to an increase in potential for passive crown fire (torching). Furthermore, increased human activity in today's forests has increased fire ignitions compared with the historical period.

H. Mountain pine beetle outbreaks are not likely to cause increased erosion.

Soils are not disturbed and protective ground cover is not reduced when mountain pine beetles kill lodgepole pine trees. If anything, understory plants may grow more vigorously in the increased light and with the higher available soil moisture and nutrients. Where tree mortality is high, annual streamflow may increase and the timing of water delivery may be changed, because of reduced canopy interception of precipitation and reduced water uptake by the trees.

Interactions between forest structure and hydrology have been studied extensively, and there is little question that major changes in the structure of Colorado and southern Wyoming forests alter several key hydrologic characteristics of these forests. Forests are widely viewed as important for protecting sloping terrain in watersheds from extreme runoff and erosion. Wildfire severe enough to kill forests is viewed as a major threat to watersheds, because protective vegetation, litter, and duff are often consumed. In many cases, fire exposes soils directly to precipitation, and runoff during heavy precipitation events (often exacerbated when fire makes soils temporarily hydrophobic) can result in extreme erosion for several months following a fire.

Soil type, steepness of slope, precipitation intensity and duration, and timing of understory vegetation recovery all affect the severity of erosion after fire.

Death of forest trees during a mountain pine beetle epidemic affects the forest floor and soil much differently than fire. Tree mortality caused by beetles leaves behind protective layers of litter and duff, and often quickly results in more productive understory vegetation. Thus severe erosion events are not expected as a result of the mountain pine beetle epidemic. In fact, mulching and seeding after fire often are used in attempts to mimic the stabilizing effects of litter, duff, and understory vegetation found after overstory mortality by beetles.

The potential for erosion from wildfire still exists, however, if extensive fire occurs in the decades following the epidemic, when large amounts of fuel are on the ground. Thus while the mortality of trees does not increase erosion significantly, erosion remains a possibility if a post-beetle fire occurs with heavy fuel loading on the ground. We note, however, that erosion is a natural process, and concerns about extreme erosion may be more a human issue than an ecological one.

There may be other hydrologic effects of mountain pine beetle mortality. Paired watershed studies around the world support the conclusion that substantially decreasing forest density results in increased runoff, though many factors affect the degree to which this occurs. Subalpine forest studies in Colorado and elsewhere are among the best examples supporting these findings. While no empirical studies of runoff in relation to the current mountain pine beetle epidemic have been completed in Colorado and southern Wyoming, it is reasonable to expect that the total annual runoff will increase where pure stands of lodgepole pine are killed by mountain pine beetles. More research is needed to determine how the hydrologic features of watersheds change during and after such epidemic changes in forest structure.

I. Climate changes will most likely contribute to substantial forest changes in the decades ahead.

Given the climate changes in the last several decades and projected changes for coming decades, large fires and other natural disturbances and shifts in vegetation composition and distribution are anticipated in many ecosystems of Colorado and southern Wyoming. These large disturbances and other changes in growing conditions will likely contribute to restructuring many forest landscapes.

Many uncertainties about climate and vegetation exist for the years, decades, and centuries ahead. As noted in our introduction, we have seen a number of significant ecological events in the last decade, including the mountain pine beetle epidemic in lodgepole pine. All of them coincide with warmer climatic conditions than were typical for the past century or more for which we have records. Warming temperatures (especially winter minimum temperatures), longer growing seasons, and growing season drought may be playing major roles in the widespread bark beetle outbreaks in Colorado and southern Wyoming and elsewhere. Fuel quantity and arrangement and fire behavior may be influenced directly or indirectly by the same variables. Germination and establishment success of new seedlings may be affected. However, it is difficult to prove whether climate changes, consequences of past forest management practices on forest conditions, or both are the primary causes of these ecological changes.

Implications for future forests. Models for predicting future climates have progressed dramatically in recent years, but their accuracy is questionable for planning purposes, particularly at local levels. Nonetheless, model predictions suggest significant alterations in climate from past observed patterns. These predictions are supported by recent climate events that themselves had largely been predicted several years ago. Therefore, the potential for future changes justifies thinking about future ecosystem dynamics that are very different from what we have seen in the last few centuries, including vegetation responses involving natural disturbance agents, species distribution, habitat suitability, and conservation of biodiversity. Areas at the elevational and latitudinal edges (ecotones) of lodgepole pine distribution may be the most likely to experience notable changes following the beetle epidemic.

Our understanding of natural disturbance phenomena such as fire, drought, and insect epidemics under new climatic scenarios is inadequate for us to judge the likely consequences of future climatic conditions. We all observe and acknowledge that natural disturbances can be major change agents regardless of their cause. Climate warming may be contributing to substantial forest changes now, but there may be more subtle changes in the future as well. Through time forest species (including insect associates and other animal species, shrubs, grasses, and forbs as well as trees) may shift to other elevations and latitudes where habitats have become more suitable for them. Some species with rapid generation times, such as mountain pine beetle, may adapt to the changing climate. Alternatively, without adaptation local extinction in bark beetle populations could occur with increased warming due to a disruption of their tightly coupled developmental timing with local weather. Groups of species may migrate together or separately, perhaps leading to unanticipated new forest communities. We cannot make firm predictions about the makeup of future forests or the biodiversity associated with these forests. Regeneration and plant community restructuring in the landscape may follow novel pathways. Information is lacking, however, and extensive research (including use of monitoring data and reconstructions of past changes) is needed to relate potential future climate and the requirements and environmental amplitudes of species and communities.

Is re-establishment of lodgepole pine assured after the mountain pine beetle epidemic? Undoubtedly, but subtle or even large shifts in its location and plant associations are not out of the question. Nonetheless, most of the area experiencing a mountain pine beetle epidemic will likely remain forest. Even if future forests differ from those of today, such forests are likely to provide valuable (if different) benefits and opportunities, both ecologically and socially. Monitoring of changes in forests as they occur is important for enabling research on such changes, and to allow managers to adapt practices to achieve desired effects as conditions change and consequences of past actions are better understood.

J. Summary

The current mountain pine beetle epidemic affecting lodgepole pine forests is an important ecological event with significant socio-economic implications. What will be the consequences for the affected ecosystems? How do we protect our communities and other human values at risk in ways that are socially and economically (as well as ecologically) feasible? These are difficult questions. This report has focused specifically on the ecology and fire behavior issues associated with lodgepole pine and the mountain pine beetle epidemic. We recognize that the socio-economic aspects are as important as the ecological issues, but they are beyond the scope of this report.

Ecologically, much is known about lodgepole pine and mountain pine beetles. Even though the scale of the current epidemic is unprecedented over the past approximately 100 years of reliable observations, beetle-caused tree mortality at some scale has long been part of the dynamics of the lodgepole pine ecosystems. Similarly, fire behavior and its role in ecological processes and fuel management practices are relatively well understood. While we are confident about our general understanding, we have identified at least some scientific uncertainties about lodgepole pine, mountain pine beetle effects, and fire behavior that should be acknowledged and further researched.

We are most concerned about several wildcard issues that create some uncertainty in applying what we know from science. The scale of this epidemic is larger than any mountain pine beetle epidemic studied thus far. We do not fully understand if or how the magnitude of this ecological event will affect future forests in terms of regeneration of the present species or transitions to different vegetation types. Furthermore, there is the question – both tantalizing and troubling – about possible climate change (including its rate, direction and magnitude) and the degree to which scientific findings need to be qualified as they are applied.

If humans were not a part of the equation, forests would simply mature, die, and regenerate or be replaced by other vegetation types, following ecological trajectories over time driven by climate, environment, and species capabilities.

Because humans cause changes in forests by choosing to live there and deriving economic services from them, our communities are impacted by forest changes, whether they are natural or not. Thus both the scale of the mountain pine beetle epidemic and the uncertainties about future forests leave us with questions that are important to us but may not be answerable with the knowledge we have now.

Knowledge from scientific research about lodgepole pine and mountain pine beetles is valuable in two ways. It offers answers to some of the questions we have about forest ecology and provides valuable insight for management of these forests for ecological and community protection purposes. It also clarifies what we do not know. This is valuable not just to direct new research, but also to inform stakeholders of the degree of confidence they should have as land and natural resource management practices are considered.

As noted in the introduction, science is a work in progress. Many of the scientific uncertainties discussed in this report already are receiving attention in the research community. Even as research continues, however, the scientific knowledge already available is usable by a wide variety of stakeholders and in the collaborative and adaptive management process. Adaptive management is perhaps best described as managing while learning on the fly. In this report, the scientific community provides information to managers and other stakeholders, but the scientific community also will help advance the knowledge base through lessons learned as management practices are planned, implemented, monitored, and evaluated. We humans must decide how to manage forests based upon their intrinsic value and natural processes as well as some desired future condition contingent on human wants and needs. We must be realistic about the degree to which we as observers, managers and stewards of the forest can affect what is happening now and what will happen in the future. Whatever we do from here should be done together.

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Chapter 3

c0015
Using Bird Ecology to Learn About the Benefits of Severe Fire

Richard L. Hutto, Monica Bond and Dominick DellaSala Au1

s0010
3.1 INTRODUCTION

p0010 In this chapter we do not provide an encyclopedic review of the more than 450 published papers that describe some kind of effect of fire on birds. In other words, we are not systematically proceeding through a litany of fire effects on birds of southeast pine forests, California chaparral, Australian eucalypt forests, South African fynbos, and so forth. Instead, we have chosen to highlight underappreciated principles or lessons that emerge from selected studies of birds in ecosystems born of, and maintained by, mixed- to high-severity fire. Those lessons show how important and misunderstood basic fire ecology is when it comes to managing fire-dependent forest lands and shrublands, and the lessons apply to all fire-dependent ecosystems that have historically experienced severe fire—fires that are severe enough to stimulate an ecological succession of plant communities (as described in Chapter 1). We also focus our attention primarily on conifer forest ecosystems of the western United States because they undergo an amazing transformation following severe fire and because studies of these systems clearly reveal how birds evolved with, and now require, severe fire. Insight that emerges from the study of bird populations is overlooked in management circles worldwide. This is unfortunate because the insight one can gain by studying the ecology of individual bird species argues strongly that severe fire needs to be maintained in the landscape if we hope to maintain the integrity of most fire-dependent ecological systems.

p0015 Most studies of fire effects on birds are disappointingly "empty" because they are merely lists of birds that benefit from or are hurt by fire; they are not placed in the broader context of what a self-sustaining fire-dependent system looks like. To understand whether a particular change in abundance is "good" or "bad" requires insight into what ought to be, which requires an understanding of the patterns that occur under conditions that are as natural as possible for any given vegetation system. That, in turn, requires replicated study of what we can expect to find after "natural" fire in any given system.

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Thus, a study of the effects of, say, prescribed understory fire on birds is meaningless without knowing what a "natural" fire in that system would ordinarily produce. Many studies might show that bird species A increases after a prescribed fire, but is that a good thing? If bird species B increases after postfire salvage logging, is that a good thing? If bird diversity is higher in one fire treatment versus another, is that a good thing? For studies of fire effects to be useful, we need to address questions that inform management by tapping into a solid understanding of what constitutes a "natural" response to fire, and that requires knowing something about the fire regime under which a given system evolved. Only through distribution patterns and adaptations of individual species (not through effects on bird guilds or on diversity and similar composite metrics) can we begin to understand which kind of fire regime necessarily gave rise to specific patterns of habitat use and to adaptations that have evolved over millennia. Birds are excellent messengers; they carry all the information we need to reconstruct the historical conditions under which they evolved. All we have to do is listen.

s0015 3.2 INSIGHTS FROM BIRD STUDIES

s0020 Lesson 1: The Effects of Fire Are Context Dependent; Species Respond Differently to Different Fire Severities and Other Postfire Vegetation Conditions

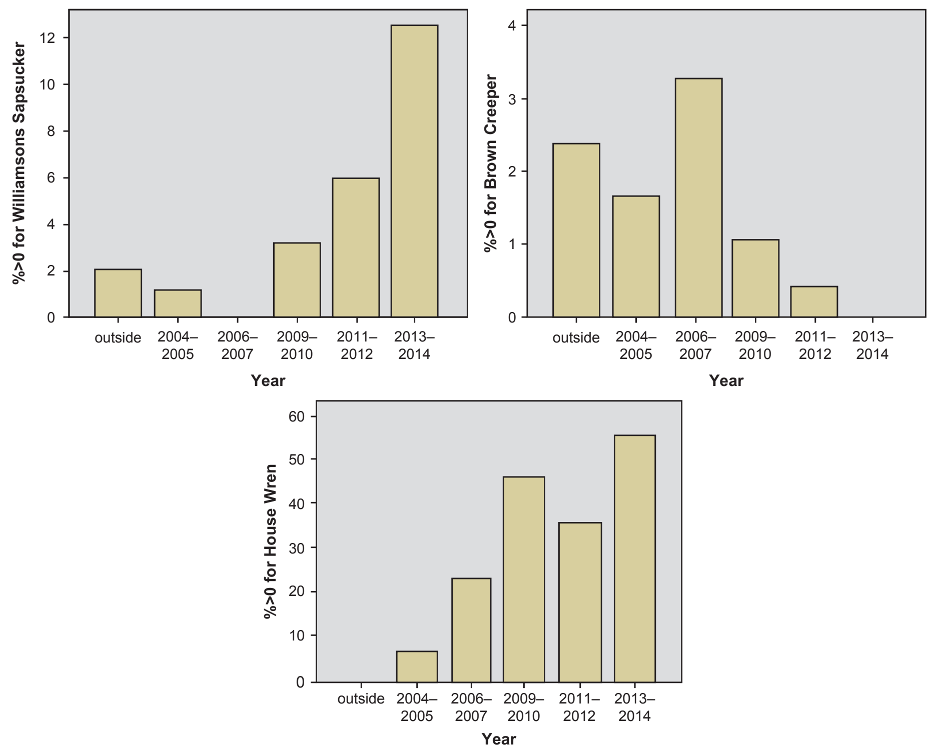
p0020 One extremely important lesson that has emerged from studies of the fire effects on birds is that a given effect depends entirely on the vegetation type, the kind of fire, and the time since the fire (Recher and Christensen, 1981; Woinarski and Recher, 1997). For years, individual bird species have been labeled as "positive responders" or "negative responders" or "mixed responders" when, in fact, any species can be all of the above. The actual response of a bird species (or of any species) to fire, then, is dependent on context. The earliest papers on fire effects rarely provided details about the nature of the fire being studied, so the first attempt to conduct a meta-analysis based on a compilation of published results of fire effects (Kotliar et al., 2002) necessarily generated a lot of "mixed" responses by birds because some papers said a species was positively affected and others said the same species was negatively affected by fire. The seeming disagreement among studies was, in most cases, a simple result of researchers looking at different postfire vegetation conditions and times since fire. It was not until Smucker et al. (2005) separated their data into categories of fire severity and time since the fire that responses began to look much more consistent among studies that share a particular vegetation type, fire type, and time since the fire. As soon as one accounts for these factors, it becomes clear that the responses of most bird species are quite consistent and that most bird species benefit from severe fire (as we discuss below).

s0025 ## *Time Since Fire*

p0025 Species that benefit from severe fire are not only those that flourish during the first year or two following the disturbance event. The same can be said for species that are restricted to years 2-4, years 5-10, or even years 50-100 following severe fire. In fact, *most* plant and animal species are present only during a limited time period following a disturbance. Therefore, *most* plant and animal species in disturbance-based systems depend on disturbance to periodically create the conditions they need. Many bird species that thrive after fire have been mislabeled as species hurt by fire because studies of bird response to fire typically involve only a brief period of time soon after the fire. For example, although Williamson's sapsucker (*Sphyrapicus thyroideus*) was labeled a "mixed responder" and brown creeper (*Certhia americana*) a "negative responder" in the meta-analysis by Kotliar et al. (2002), and the change in house wren (*Troglodytes aedon*) abundance was labeled "insignificant" in a recently published study by Seavy and Alexander (2014), each of these species typically reaches its peak abundance several years after a fire, as revealed in an 11-year postfire study conducted after the Black Mountain fire, which burned near Missoula, Montana, in 2003 (Figure 3.1). Thus, each species clearly benefits from severe fire when viewed in the proper (and perhaps very restricted) time frame after fire.

p0030 By extending the duration of a postfire study beyond the first few years after a fire, most bird species reveal a unimodal response to time since fire, and most benefit from fire; they reveal a greater probability of detection in the burned forest at some point during that postfire period than in the same forest before fire or in the surrounding unburned forest (Taylor and Barmore, 1980; Reilly, 1991a, 2000; Taylor et al., 1997; Hannon and Drapeau, 2005; Saab et al., 2007; Chalmandrier et al., 2013; Hutto, 2015). These results force one to appreciate that if for a period of time after a fire conditions remain better than they are in very old plant communities near the end of the late seral stage of succession, then disturbance is periodically necessary to create the conditions needed by that species. Thus a species being "hurt" in the short term by fire is not evidence that fire is somehow "bad" for that species and that it would have been better off without fire. In fact, once a system is beyond the ideal postdisturbance time period for a species, the only way to periodically "restore" conditions needed by that species is to disturb the system with another severe fire and then wait for the appropriate time period following disturbance again. The lesson is this: one cannot assess the effects of fire on any plant or animal species without examining whether the species is restricted to a period of time preceding the oldest possible vegetation condition.

p0035 A necessary consequence of different species occurring at different points in time following fire (in association with changes in vegetation type and structure) is that we must embrace natural severe disturbance processes because they create starting points for the development of the full range of vegetation-age



f0010 **FIGURE 3.1** The probabilities of occurrence of Williamson's sapsucker ~~(Sphyrapicus thyroideus)~~, brown creeper ~~(Certhia americana)~~, and house wren ~~(Troglodytes aedon)~~ were significantly greater several years after the 2003 Black Mountain fire than they were either before the fire (as determined from survey data "outside" the burn perimeter in unburned, mixed-conifer forest of the same type) or during the first 2 years following the fire (R.L. Hutto, unpublished data; sample sizes exceed 150 point counts for each time period; $P < 0.05$, log linear analyses). Therefore, the benefit of severe fire for some species cannot be detected without restricting data collection to within a specific time period after the fire event.

categories, which, in turn, are needed for the maintenance of biological diversity (in particular beta diversity, the turnover in species number across gradients). Moreover, mixed-severity fires (which can result only from high-severity fire events) help provide a variety of kinds of starting points, which, in turn, also help maintain biological diversity (Smucker et al., 2005; Haney et al., 2008; Rush et al., 2012; Sitters et al., 2014; see also Chapters 4-6).

s0030 Old Growth

p0040 As already emphasized, most bird species clearly depend on severe fire to reset the clock, which stimulates development of the particular postdisturbance "age" to which they are best adapted. Still, many bird species are restricted in their habitat distribution to an end-of-the-line successional stage—they are dependent on old growth. There are also ecosystems (e.g., eucalyptus forests, chaparral) where severe fire is natural but where there are few, if any, early

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f0015 **FIGURE 3.2** Resprouting eucalyptus trees following a severe fire that burned through the area only months earlier. *(Photograph by Richard Hutto, taken in November 1999 near −34.284030°S, 150.725373°E in the tablelands above Wollongong, New South Wales, Australia.)*

fire-dependent bird species because many of the dominant plant species resprout, yielding a plant community structure and composition that "recovers" rapidly after fire (Figure 3.2). In these instances most bird species are associated with "mature" forms of those plant communities and would appear to do well if there were no fire at all (e.g., Taylor et al., 2012).

p0045 In all vegetation types that undergo plant succession following mixed- to high-severity fire, there will always be some bird species that depend on long-unburned vegetation. Therefore, discovering that those species are absent in the short term or "hurt" by fire is not unexpected, nor is it a necessarily a problem that needs to be addressed. The fact that fire temporarily removes large parts of a landscape from the pool of suitable conditions for those species is not a problem because the loss of suitable conditions is temporary, and there are usually nearby "refuges" of suitable conditions in places that have not burned for a long time (Bain et al., 2008; Leonard et al., 2014; Robinson et al., 2014; Winchell and Doherty, 2014). Natural systems exist as an ever-changing mosaic of different postfire ages—all vegetation ages are present at some point in space all the time. A significant problem emerges only when humans remove or degrade so much of the older vegetation through timber harvesting or land conversion that there is now a perceived risk of fire to those species that depend on older vegetation stands that are too few and far between. Understand clearly, however, that the absence of late-succession forest refuges is a problem that stems from excessive logging or development, not from the presence of fire per se.

p0050 Now that we are down to the last remaining old-growth forest remnants in California and Oregon ~~landscapes that have been subjected to excessive logging~~, some feel that we should thin the forests around those remnants to protect them from fire. The effect of altering mature forest surrounding the last remaining old-growth remnants on the remnants themselves is, however, unknown.

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Moreover, as has been discussed in reference to eucalyptus forest systems, many old-growth forest patches are old precisely because they are situated in places that are relatively immune to severe fire (Bowman, 2000); the same is undoubtedly true of many old-growth mixed-conifer forest patches. Unburned forest patches surrounding unburned, old-growth forest patches also have been suggested to be important as dispersal corridors across which old-growth species may recolonize recently burned areas as succession proceeds toward later stages (Pyke et al., 1995; Robinson et al., 2014; Seidl et al., 2014). Therefore, proposals to thin the forest around remaining old-growth stands may be well intentioned but reflect a lack of appreciation for the resilience associated with plant communities born of, and maintained by, natural disturbance processes (a case in point is the spotted owl [*Strix occidentalis*]; see Box 3.1).

s0035 *Postfire Vegetation Conditions*

p0070 One must account not only for time since fire but also for fire severity and other forest conditions (e.g., vegetation composition and tree density) to adequately assess fire effects on animal species. Smucker et al. (2005) accounted for both time since fire and fire severity in an analysis of bird occurrence patterns following the Bitterroot fires of 2000 in Montana, and the results were profound.

b0010 BOX 3.1 Old-Growth Species and Severe Disturbance Events

p0055 There are a number of old-growth-dependent species in North American conifer forests, but severe fire may not pose anywhere near the threat to those species that one might suppose. Consider the spotted owl ~~(*Strix occidentalis*)~~, one of the most iconic old-growth-dependent bird species in the Pacific Northwest, California, and Southwest (extending into northern Mexico). This federally listed threatened raptor typically nests, roosts, and forages in dense conifer and mixed-conifer-oak forests dominated by large (>50-cm diameter at breast height), older trees and peppered with big decadent snags and fallen logs. High levels of canopy cover (generally >60%) from overhead foliage is an important component of nesting and roosting stands; thus, spotted owls were long presumed to be seriously harmed where severe fire burned the forest canopy. Indeed, over the past several decades, most forest management efforts in the range of the spotted owl (a Forest Service management indicator species) has been driven by logging to prevent or reduce fire to "save" the owl, including the latest U.S. Fish & Wildlife Service recovery plans for the northern and Mexican spotted owls. Yet, the forests where the owl dwells have experienced mixed- and high-severity fire for millennia. So how do these birds actually respond when severe fire affects habitat within their home ranges?

p0060 Several studies have demonstrated that all three subspecies of spotted owl can survive and thrive (i.e., successfully reproduce) within territories that have experienced moderate- and high-severity fire (Bond et al., 2002; Jenness et al., 2004;

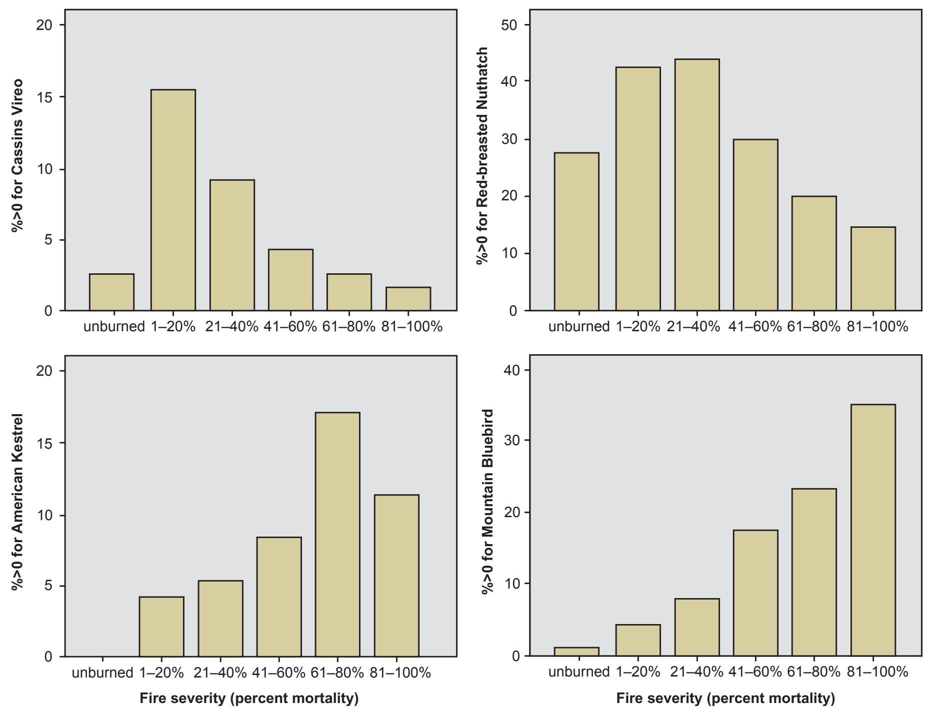
Continued

b0010

BOX 3.1 Old-Growth Species and Severe Disturbance Events—Cont'd

Roberts et al., 2011; Lee et al., 2012, 2013). Exceptionally high levels of severe fire in a nest stand can cause spotted owls to abandon that territory (Lee et al., 2013), but only a small fraction of sites ever exceed that threshold in any given fire. Moreover, a higher probability of abandonment after fire was documented only in a geographical region where prefire forest patches were limited or isolated; this did not occur in areas where prefire forest cover was more abundant (Lee et al., 2012, 2013) or in areas that were ~~salvage~~ logged after fire (Lee et al., 2013; Clark et al., 2013). For example, the year after the 2013 Rim Fire—one of the largest fires to occur in California within the past century—at least six pairs of California spotted owls were detected in sites where >70% of the "suitable habitat" around their nest stands burned at high severity. (At one occupied site severe fire burned 96% of the habitat!) Why do they stick around in burned territory? One study found California spotted owls selectively hunted (mostly for woodrats and gophers) in stands recently burned by severe fire when those burned forests were available to them and relatively near the nest or roost stand (Bond et al., 2009, 2013). Another study showed that during winter, Mexican spotted owls moved up to 14 km into burned forests where prey biomass was 2-6 times greater than in their breeding-season nesting areas (Ganey et al., 2014). Spotted owls are perch-and-pounce predators, so it is not surprising that they avoided foraging in areas that were logged after fire, as there were no longer any perch trees (Bond et al., 2009), nor is it surprising that postfire logging reduced site occupancy and survival rates (Clark et al., 2013; Lee et al., 2013). In these studies, spotted owls still preferred to nest and roost in green forests, underscoring the importance of unburned/low-severity refuges within the larger landscape mosaic of mixed-severity fire. Still, the point is that where severe fire is natural, even old-growth species can partake of its bounty. The spotted owl, too, is sending a message here: A natural fire regime provides a bedroom, nursery, and kitchen for even old-growth-dependent species, as long as the burned forest is left standing.

p0065 Despite this evidence, the U.S. Fish & Wildlife Service is now calling for aggressive, large-scale thinning in northern spotted owl habitat in dry forests as a means of reducing fire intensity (U.S. Fish and Wildlife Service, 2011). This "recovery" objective for the owl was developed over objections raised by scientists (Hanson et al., 2009, 2010) and professional societies such as The Wildlife Society and Society for Conservation Biology. Notably, Odion et al. (2014b) simulated changes in owl habitat over a four-decade period following fire and the kind of thinning proposed by federal land managers. The simulation study showed that thinning over large landscapes would remove 3.4-6.0 times more of their dense, late-successional habitat in the Klamath and dry Cascades, respectively, than forest fires would, even given a future increase in the amount of high-severity fire. Further, Baker (2015) documented that before extensive Euro-American settlement, mixed- and high-severity fires shaped dry forests in the Eastern Cascades of Oregon and provided important habitat for northern spotted owls there. These studies challenge the paradigm that severe fire is a serious threat to spotted owls, which evolved in landscapes shaped by such fire, and that extensive logging is needed to ameliorate this widely believed but overstated threat.



f0020 **FIGURE 3.3** Example plots of the percentage occurrence of four mixed-conifer bird species in relation to fire severity in the first few years after fire. Data were drawn from 7043 survey points distributed across 110 different fires that burned since 1988 in western Montana. Sample sizes exceed 700 point counts per severity category. All patterns are significant ($P < 0.05$, log linear analyses). Note that each species [illegible] abundant in burned than in unburned forest, and each is relatively abundant at a level o[illegible]everity (percentage of tree mortality) that differs from that occupied by the other species.

Once they accounted for fire severity alone, it became abundantly clear that many of the same bird species that had been labeled as "mixed responders" to fire by others (e.g., Kotliar et al., 2002) were not at all mixed in their response to fire. The importance of fire severity is strikingly apparent in even the simplest graphs of percentage occurrence across severity categories (Figure 3.3).

s0040 Lesson 2: Given the Appropriate Temporal and Vegetation Conditions, Most Bird Species Apparently Benefit from Severe Fire

p0075 After we combine information on the time since fire, fire severity, and perhaps one or two additional vegetation variables, most bird species apparently benefit from severe fire. For each species there is a particular combination of burned forest variables that creates ideal conditions for that species, as evidenced by an abundance that exceeds that in a long-unburned patch of the same vegetation type. Indeed, when Hutto and Patterson (2015) considered just two fire-context variables (time since fire and fire severity), they found 46 of 50 species to be

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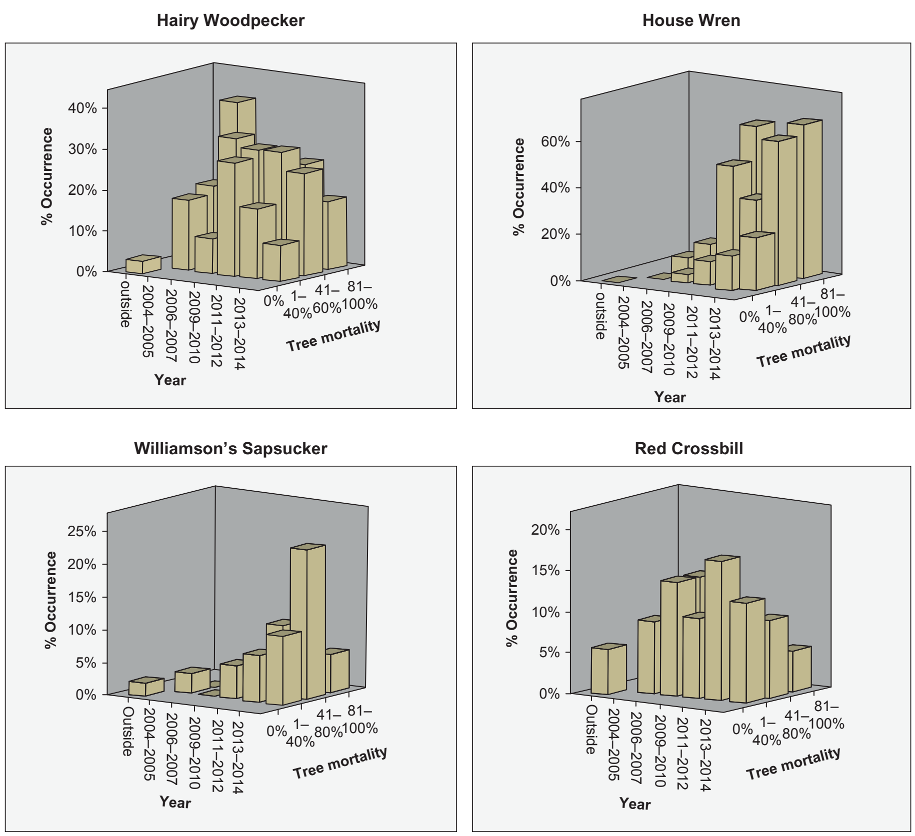
more abundant in some combination of those two variables than in long-unburned stands (Figure 3.4). Thus, not only are most species relatively abundant in one burned forest condition or another, but the average point in space and time occupied by each species is also species specific (Figure 3.5).

p0080 As an introduction to some of the fascinating biology surrounding severely burned forests, consider the following bird species. The black-backed woodpecker (*Picoides arcticus*), American three-toed woodpecker (*Picoides dorsalis*), hairy woodpecker (*Picoides villosus*), northern flicker (*Colaptes auratus*), and Lewis's woodpecker are all more abundant in severely burned than unburned mixed-conifer forest (see patterns of habitat occurrence for four of the five species in Figures 3.11 and 3.12) because of an abundance of food (beetle larvae and ants) and potential nest sites associated with standing dead trees.



f0025 **FIGURE 3.4** Example plots of percentage occurrence for various mixed-conifer bird species in relation to both time since fire and fire severity after the 2003 Black Mountain fire near Missoula, Montana (R.L. Hutto, unpublished; sample sizes exceed 35 point counts for each time-by-severity category; all patterns are significantly nonrandom as determined by log linear analyses [$P < 0.05$]). The examples were selected to illustrate that each species is more abundant in burned than in unburned forest (the occurrence rate in unburned forest shown in the first time period), and each is most abundant in a different combination of time since fire and burn severity (percentage of tree mortality).

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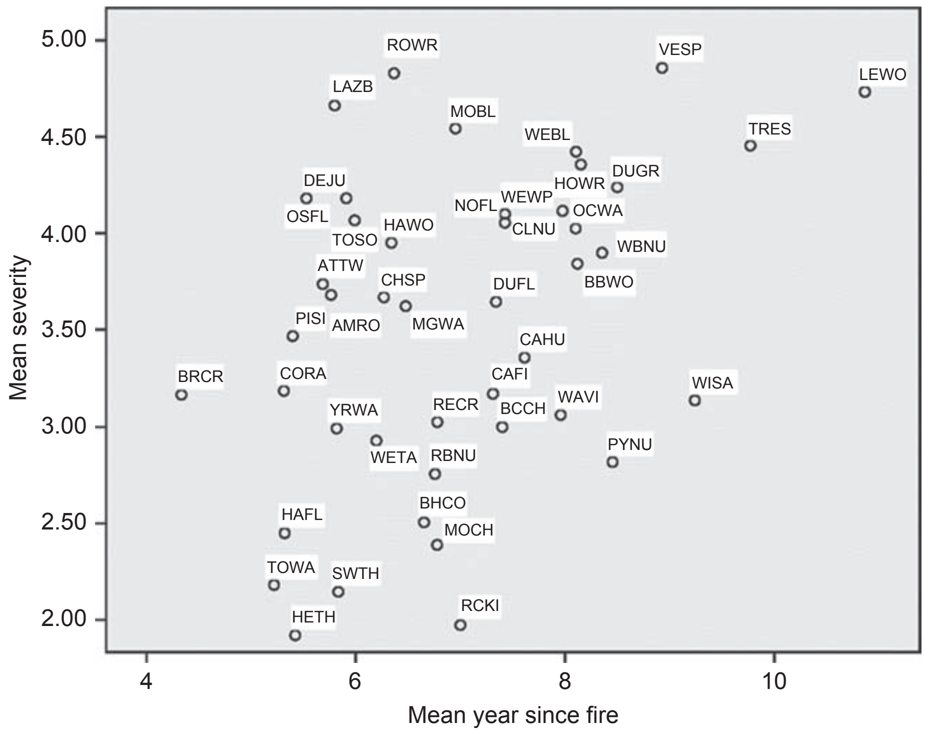

f0030 **FIGURE 3.5** In combination, the mean time since fire and mean fire severity at points of occurrence for each of 46 (mnemonically coded) species differs from that of every other species. Mean values were calculated from the kind of data presented in Figure 3.4.

The Williamson's sapsucker and olive-sided flycatcher (*Contopus cooperi*) find the abrupt edges between severely burned and unburned forest to be ideal nest locations (Figure 3.6). A host of secondary cavity-nesting and snag-nesting species (e.g., northern hawk owl [*Surnia ulula*], great gray owl [*Strix nebulosa*], mountain bluebird [*Sialia currucoides*], western bluebird [*Sialia mexicana*], house wren, and tree swallow [*Tachycineta bicolor*]) benefit from new forest openings, where they find a mature-forest legacy of already existing broken-top snags (Figure 3.7), where a disproportionately large number of nest sites are located (Hutto, 1995). These species depend on the kinds of snags that become common only after a forest reaches the mature- to old-growth stage and then burns in a severe fire. A variety of species (e.g., flammulated owl [*Psiloscops flammeolus*], mountain bluebird, Townsend's solitaire [*Myadestes townsendi*], and dark-eyed junco [*Junco hyemalis*]) make use of the cavities created by burned-out root wads or uprooted trees that happen to blow down in the first few years after severe fire (Figure 3.8). Many species (e.g., Clark's nutcracker [*Nucifraga columbiana*], Cassin's finch [*Haemorhous cassinii*], red crossbill (*Loxia curvirostra*], and pine siskin [*Spinus pinus*]) take advantage of seeds that are released or made available in cones that open after severe fire

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f0035 **FIGURE 3.6** Williamson's sapsucker ~~(*Sphyrapicus thyroideus*~~; left) and olive-sided flycatcher (~~*Contopus cooperi*;~~ right) are known to nest disproportionately often near the abrupt edges between severely burned and unburned forest. *(Photographs by Richard Hutto (left) and Bruce Robertson (right).)*

f0040 **FIGURE 3.7** Compared with burned trees with intact tops, broken-top snags that were already snags before the fire burned are used disproportionately more often as nest sites by cavity-nesting bird species. The black-backed woodpecker also roosts almost entirely in burned-out hollows, forked trunks, or other relatively unusual structures that create crevices in "deformed" snags that existed before the forest burned (Siegel et al., 2014). Pictured (left to right) are a young hairy woodpecker ~~(*Picoides villosus*)~~ in its nest cavity, an American robin (*Turdus migratorius*) nest, and a northern flicker ~~(*Colaptes auratus*)~~ nest. The implications are profound—old-growth elements (snags) are really important to birds that depend on burned forest conditions, so burned, old-growth forests are as valuable to wildlife as unburned old-growth forests. *(Photographs by Richard Hutto.)*

f0045 **FIGURE 3.8** The architecture of a burned forest becomes modified after trees begin to blow down in the first few years after a fire, and a number of bird species make use of the root wads as nest sites. A Townsend's solitaire ~~(*Myadestes townsendi*~~) nest is highlighted here. *(Photograph by Richard Hutto.)*

f0050 **FIGURE 3.9** Few people seem to realize how important Clark's nutcrackers ~~(*Nucifraga columbiana*~~) are as seed dispersers after severe fire in ponderosa pine forests. Pictured here are examples of a nutcracker extracting seeds from a ponderosa pine cone that opened after fire (left) and a nutcracker with a throat pouch full of seeds in the scorched ground beneath a ponderosa pine canopy. *(Photographs by Richard Hutto.)*

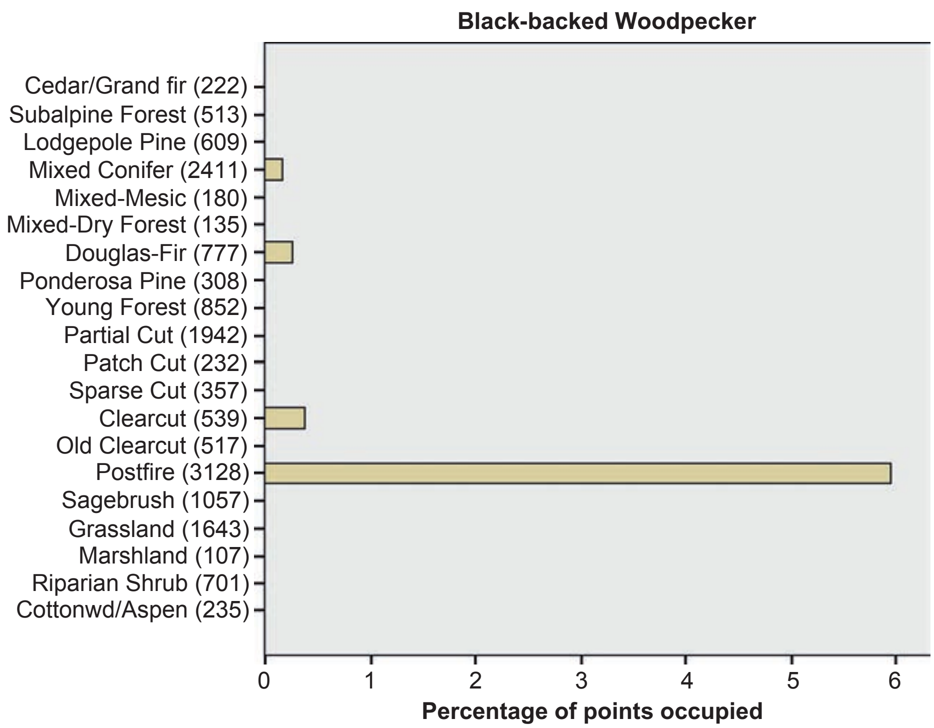
(Figure 3.9). Still more bird species (e.g., calliope hummingbird [*Selasphorus calliope*], lazuli bunting [*Passerina amoena*], and MacGillivray's warbler [*Geothlypis tolmiei*]) use the shrub-dominated early seral stage for feeding and nesting and as display sites (Hutto, 2014).

s0045 Lesson 3: Not only Do Most Bird Species Benefit from Severe Fire, but Some also Appear to *Require* Severe Fire to Persist

p0085 The black-backed woodpecker has become an iconic indicator of severely burned forests because its distribution is nearly restricted to such conditions. Bent (1939) provided the first description of the unusual association between this woodpecker species and burned forests when he noted that Manly Hardy wrote to Major Bendire in 1895 about finding the woodpecker to be "... so abundant in fire-killed timber areas that I once shot the heads off six in a few minutes when short of material for a stew." This anecdote, reflecting the importance of severe fire, went largely unnoticed until the 1970s, when Dale Taylor undertook a study of birds in relation to time since fire in the Yellowstone and Grand Teton National Parks. His more systematic study uncovered the same remarkable pattern. Taylor was the first person to evaluate data drawn from a series of burned conifer forest stands of differing ages, and he found the appearance of the black-backed woodpecker to be restricted to the first few years after fire (Taylor and Barmore, 1980). A subsequent before-and-after fire study by Apfelbaum and Haney (1981) and studies of burned versus adjacent unburned forest by Niemi (1978), Pfister (1980), and Harris (1982) provided additional evidence that this bird species is strongly associated with burned forest conditions. Following the Rocky Mountain fires of 1988, Hutto (1995) conducted a more comprehensive study of the distribution of black-backed woodpeckers across a broad range of vegetation types. That study served to reinforce the notion that this species is an ideal indicator of severely burned mixed-conifer forest. More specifically, Hutto provided a meta-analysis of his own and already published bird survey data collected from burned forests and from more than a dozen unburned vegetation types; those data showed the black-backed woodpecker to be relatively restricted to burned forests. To address the potential problem of putting too much faith in distribution patterns derived from bird occurrence rates that were based on a variety of study durations and methods, Hutto subsequently coordinated the collection of standardized bird survey data from more than 18,000 points distributed across every major vegetation type in the U.S. Forest Service Northern Region. The results (Hutto, 2008) were strikingly similar to what earlier studies showed: one is hard pressed to find a black-backed woodpecker anywhere but in a recently burned forest (Figure 3.10).

p0090 Numerous studies (most published just in the past decade) provide additional detail that can help us better understand this remarkable association between the black-backed woodpecker and severely burned forests. Here we list some of the insights we have gained:

o0010 **1.** The magical appearance of woodpeckers within weeks of a fire (Blackford, 1955; Uxley, 2014) suggests that either smoke, or perhaps the fire or burned landscape itself, provides a stimulus for birds to colonize newly burned forests.



f0055 **FIGURE 3.10** Histogram bars indicate the percentage of points (sample sizes in parentheses) at which the black-backed woodpecker was detected in each of 21 distinct vegetation types within northern Idaho and western Montana. The distribution is nonrandom ($X^2 = 559.43$; $df = 19$; $P < 0.0001$) and reveals that the black-backed woodpecker is highly specialized in its use of burned conifer forest. *(Data from Hutto (2008).)*

o0015 **2.** Breeding and nest densities increase more rapidly than expected on the basis of recruitment alone (Yunick, 1985; Youngman and Gayk, 2011), which suggests that the process of immigration after fire is significant.

o0020 **3.** Woodpecker diet, which is based mainly on wood-boring beetle larvae that feed almost exclusively on recently burned and killed trees (Murphy and Lehnhausen, 1998; Powell et al., 2002; Fayt et al., 2005), reflects the broad postfire change in animal community composition that accompanies severe fire.

o0025 **4.** The woodpecker's nonrandom use of forest patches containing dense, larger-diameter trees (Saab and Dudley, 1998; Saab et al., 2002, 2009; Nappi and Drapeau, 2011; Dudley et al., 2012; Seavy et al., 2012) that have burned at high rather than low severity (Schmiegelow et al., 2006; Koivula and Schmiegelow, 2007; Hanson and North, 2008; Hutto, 2008; Nappi and Drapeau, 2011; Youngman and Gayk, 2011; Siegel et al., 2013) is striking and consistent among studies.

o0030 **5.** The window of opportunity for occupancy by this species is not only soon after fire, but generally lasts only about a half-dozen years before the birds (and the abundant native beetle populations) disappear (Taylor and Barmore, 1980; Apfelbaum and Haney, 1981; Murphy and Lehnhausen, 1998; Hoyt and Hannon, 2002; Saab et al., 2007; Nappi and Drapeau, 2009; Saracco et al., 2011).

o0035 **6.** The size of the home ranges of black-backed woodpeckers within burned forests are significantly smaller (indicating better quality habitat) than those outside burned forests (Rota et al., 2014b; Tingley et al., 2014). Even more telling is that nest success is significantly higher inside than outside burned forests (Nappi and Drapeau, 2009; Rota et al., 2014a).

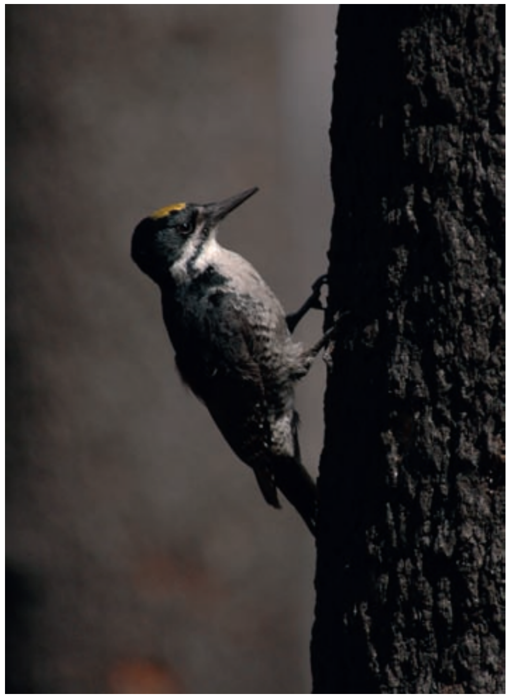
o0040 **7.** Estimated population growth rates are insufficient to maintain a growing population outside burned forests (Rota et al., 2014a). Thus, although one could argue that low woodpecker densities in green-tree forests multiplied by a much larger unburned forest area might yield even more woodpeckers in green forests (Fogg et al., 2014), a sink area alone (no matter how large) can never yield a viable population of woodpeckers (Odion and Hanson, 2013).

o0045 **8.** The importance of severely burned forests as foraging locations for wintering black-backed woodpeckers is virtually unknown; the only detailed work so far (Kreisel and Stein, 1999) revealed densities that were an order of magnitude greater in burned than in unburned forests.

p0135 The biology surrounding this single bird species clearly reflects not only the ecological importance but also the necessity of severely burned forests, but major environmental organizations have yet to focus conservation efforts on burned forests (Schmiegelow et al., 2006), and management guidelines developed by state agencies to designate important wildlife habitats (e.g., https://www.dfg.ca.gov/biogeodata/cwhr/) do not even have burned conifer forests on their radar.

p0140 The distributional stronghold of the black-backed woodpecker might be considered to lie within the boreal forests of Canada, which nobody doubts are among the most severe-fire-dependent ecosystems in the world, but the bird's distribution south into the California Sierras and Rocky Mountains of the Intermountain West confirms that severe fires in those areas have been historically important as well. A North American forest bird species that is more narrowly restricted to a single forest condition does not exist; the black-backed woodpecker is the definition of a specialist. Everything about this bird species, including its distribution, territory size, breeding success, and even coloration pattern (which matches blackened trees), all indicate that this species needs expansive patches of severely burned forest to persist (Figure 3.11).

p0145 We have taken the liberty to provide extensive detail on this particular species because its ecological story carries significant management implications. Because public land managers have a responsibility to manage for the maintenance of all vertebrate species, finding even a single species that depends on severe fire should be enough to raise their awareness that severely burned



f0060 **FIGURE 3.11** Black-backed woodpecker—a species that is relatively restricted in its distribution to severely burned forests. *(Photograph by Richard Hutto.)*

mixed-conifer forests provide necessary habitat as well. Thus the black-backed woodpecker is an ideal focal species for bringing attention to the fact that burned forest conditions are important to maintain in the landscape (DellaSala et al., 2014). The evolutionary history that has led to a strong association between burned forests and the woodpecker also raises questions about whether (as many assume) severe fires in mixed-conifer forests are really beyond the historical natural range of variation, whether we need to be thinning forests outside the wildland-urban interface to reduce fire severity, whether we need to be suppressing fire outside the wildland-urban interface, and whether we should "salvage" logging trees (including important legacy trees; see Chapter 11) after fire. Yes, the story surrounding this focal species is important.

s0050 *Bird Species in Other Regions That Seem to Require Severe Fire*

p0150 Do any other bird species seem not only to benefit from but also to require severe fire to persist? The presence of a species in a specific environment and its absence elsewhere would be a clear indication that it depends on that particular environment. For species that occur across a range of environmental conditions, the places where they are relatively abundant are also likely to represent places that are required for population persistence because they persist in source areas and they are generally less abundant in, and their abundance is

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more variable through time in, more marginal areas (Pulliam, 1988; Sergio and Newton, 2003). Although the same level of biological detail that has been amassed for the black-backed woodpecker has not been collected for most other fire-associated bird species, the habitat distribution patterns of numerous bird species reveal that they are nowhere more abundant than in recently burned forests. For example, Hutto (1995) listed 15 species that were more abundant in recently burned forests than in any of 14 other vegetation types. Graphs generated from surveys conducted across an even broader range of vegetation types show just how striking these habitat distribution patterns can be: numerous species are nowhere more abundant than they are in severely burned forests (Hutto and Young, 1999) (Figure 3.12).

p0155 Many mixed-conifer bird species (e.g., black-backed woodpecker, American three-toed woodpecker, hairy woodpecker, northern flicker, olive-sided flycatcher, western wood-pewee [*Contopus sordidulus*], dusky flycatcher [*Empidonax oberholseri*], mountain bluebird, Townsend's solitaire, house wren, tree swallow, lazuli bunting, Clark's nutcracker, red crossbill) fall consistently into a short-term "benefit" category, as revealed either by some measure of abundance or nest success in studies of burned versus unburned or before versus after fire (Bock and Lynch, 1970; Bock et al., 1978; Taylor and Barmore, 1980; Apfelbaum and Haney, 1981; Raphael et al., 1987; Hutto, 1995; Kotliar et al., 2002; Hannah and Hoyt, 2004; Smucker et al., 2005; Mendelsohn et al., 2008; Seavy and Alexander, 2014). Even severely burned patches within conifer forests that we have come to associate with low-severity fire can provide critically important habitat for species like the buff-breasted flycatcher (Kirkpatrick et al., 2006; Conway and Kirkpatrick, 2007; Hutto et al., 2008).

p0160 One of the most celebrated examples of a fire specialist involves the federally endangered Kirtland's warbler (*Setophaga kirtlandii*). It occurs almost exclusively in young (5- to 23-year-old) jack pine (*Pinus banksiana*) forest historically created by severe fire (Walkinshaw, 1983). In addition, pairing success is significantly higher in burned than in unburned forests (98% vs. 58% success; Probst and Hayes, 1987). The need for severe fire is obvious not only because, historically, it must have taken severe fires to stimulate forest succession but also because of how its critically endangered population increased dramatically after a fire accidentally escaped within its breeding range (James and McCulloch, 1995). Managers have had difficulty trying to recreate conditions that mimic natural postfire conditions through the use of logging techniques (Probst and Donnerwright, 2003; Spaulding and Rothstein, 2009), and efforts to use these artificial means to maintain warbler populations miss the point. Conservation efforts should be directed toward maintaining severely burned forests, not toward finding a way around the natural fire disturbance process.

p0165 In Australia, where few species are thought to be restricted to recently burned shrubland or forest conditions, early colonists are viewed as generalists, and management concerns are focused on postfire decreases in late-succession specialists (Serong and Lill, 2012). Nevertheless, recent data from Lindenmayer

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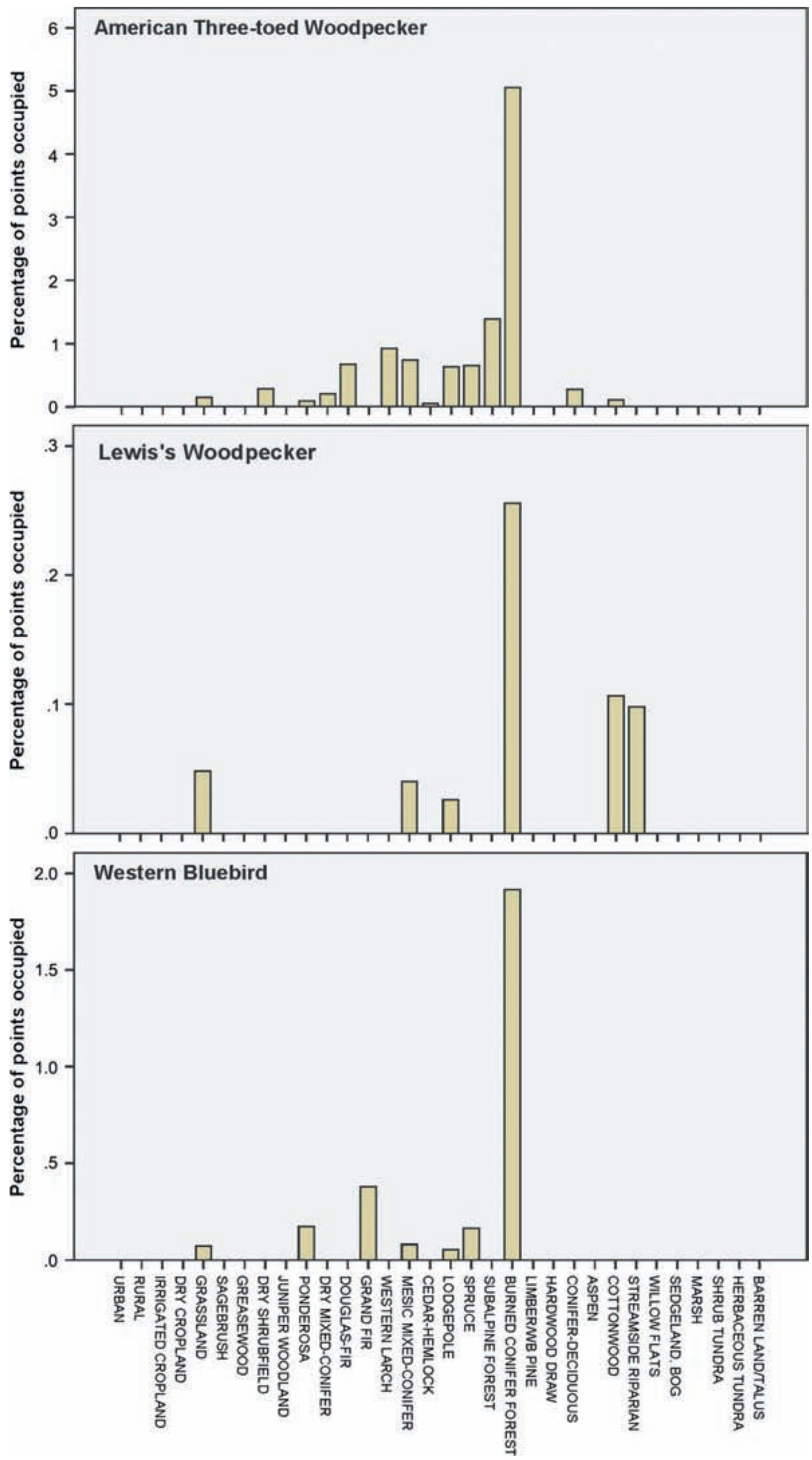

f0065 **FIGURE 3.12** Several graphs depicting species that seem to be more abundant in burned forests than in any other vegetation type in the northern Rocky Mountains. Data were drawn from a subset of the Northern Region Landbird Monitoring Program database consisting of 20,000 survey points distributed across northern Idaho and western Montana.

et al. (2014) show that a number of bird species decline in abundance 1-2 years after moderate to severe fire but then return to levels comparable to, or *higher* than, those in unburned forests within 3 years following fire. Indeed, upon further inspection, we found that the superb fairywren (*Malurus cyaneus*), gray fantail (*Rhipidura albiscapa*), yellow-faced honeyeater (*Lichenostomus chrysops*), white-fronted honeyeater (*Purnella albifrons*), dusky robin (*Melanodryas vittata*), flame robin (*Petroica phoenicea*), willie wagtail (*Rhipidura leucophrys*), gray shrike-thrush (*Colluricincla harmonica*), varied sittella (*Daphoenositta chrysoptera*), apostlebird (*Struthidea cinerea*), white-browed scrubwren (*Sericornis frontalis*), brown thornbill (*Acanthiza pusilla*), spotted pardalote (*Pardalotus punctatus*), welcome swallow (*Hirundo neoxena*), dusky woodswallow (*Artamus cyanopterus*), black-faced woodswallow (*Artamus cinereus*), and silver-eye (*Zosterops lateralis*) each have been shown by one or more authors to be more abundant in severely burned than in long unburned, dry sclerophyll forests (Christensen and Kimber, 1975; McFarland, 1988; Reilly, 1991a,b, 2000; Turner, 1992; Taylor et al., 1997; Fisher, 2001; Leavesley et al., 2010; Recher and Davis, 2013; Lindenmayer et al., 2014). Thus many eucalyptus forest species also seem to require severe fire to create the early successional forest conditions within which they are most abundant, but most of those species are not restricted to conditions that occur during the first year or two after fire. In comparison with the dramatic change in bird species composition following severe fire in mixed-conifer forests, there is, in fact, a notable lack of turnover in bird species composition following severe fire in eucalyptus forests (compare before-and-after fire data from Australia and the western United States in Figure 3.13). This difference in response to fire is presumably because eucalyptus trees resprout rapidly from epicormic shoots (Figure 3.2). Lindenmayer et al. (2014) also note that in montane ash forests, "... very rapid vegetation regeneration and canopy closure on severely burned sites ... may limit the influx of open-country birds and preclude the evolutionary development of early successional species" (p. 474). Nevertheless, the bird species listed above suggest that many may depend on slightly later stages of succession before the development of a fully mature forest and that a slightly different perspective might be needed to expose the ecological importance of severe fire to birds of Australian eucalypt forests.

p0170 Taken together, we hope we have provided enough ecological information derived from birds to solidify the notion that severe fire in most severe-fire-dependent shrublands and forests is both natural and necessary for maintenance of the ecological integrity of such systems.

s0055 Postfire Management Implications

p0175 Severe fire is natural and necessary in most—not relatively few—conifer forest types and in many other vegetation types worldwide as well (see Chapters 1 and 2). Current management practices designed to prevent fire,

Australian eucalyptus forest			Western North American mixed-conifer forest		
Species	unburned (n=39)	burned (n=35)	Species	unburned (n=1143)	burned (n=638)
New Holland Honeyeater	0.161	0	Townsend's Warbler	0.4	0.03
Little Wattlebird	0.095	0	Solitary Vireo	0.238	0.021
Scarlet Robin	0.019	0	Golden-crowned Kinglet	0.235	0.021
Yellow-faced Honeyeater	0.040	0.231	Gray Jay	0.084	0.009
Painted Button-Quail	0	0.035	Pileated Woodpecker	0.052	0.006
Grey Shrike-thrush	0	0.058	Swainson's Thrush	0.43	0.062
Olive-backed Oriole	0	0.131	Varied Thrush	0.103	0.015
			White-breasted Nuthatch	0.017	0.003
			Black-capped Chickadee	0.053	0.012
			Red-breasted Nuthatch	0.591	0.145
			Ruby-crowned Kinglet	0.316	0.086
			Hammond's Flycatcher	0.091	0.027
			Hermit Thrush	0.048	0.015
			Orange-crowned Warbler	0.098	0.036
			Western Tanager	0.398	0.163
			Mountain Chickadee	0.219	0.092
			MacGillivray's Warbler	0.201	0.095
			Yellow-rumped Warbler	0.521	0.249
			Warbling Vireo	0.145	0.098
			Clark's Nutcracker	0.022	0.047
			Pine Siskin	0.111	0.257
			Rufous Hummingbird	0.014	0.038
			Northern Flicker	0.076	0.21
			Calliope Hummingbird	0.01	0.03
			Song Sparrow	0.004	0.015
			Olive-sided flycatcher	0.025	0.107
			Rufous-sided Towhee	0.01	0.044
			Cassin's Finch	0.029	0.13
			American Kestrel	0.003	0.015
			Mourning Dove	0.004	0.021
			Hairy Woodpecker	0.021	0.124
			Three-toed Woodpecker	0.007	0.056
			Northern Waterthrush	0.003	0.033
			Green-tailed Towhee	0.001	0.012
			White-crowned Sparrow	0.002	0.027
			Lazuli Bunting	0.01	0.148
			House Wren	0.004	0.086
			Western Wood-pewee	0.003	0.104
			Mountain Bluebird	0.004	0.281
			American Robin	0.185	0.441
			Lincoln's Sparrow	0	0.015
			Tree Swallow	0	0.089
			Rock Wren	0	0.044
			Black-backed Woodpecker	0	0.05

f0070 **FIGURE 3.13** Probabilities of the occurrence of bird species in burned and unburned Australian eucalypt forests in the tablelands above Wollongong, New South Wales, and in burned and unburned mixed-conifer forests in western Montana (R.L. Hutto, unpublished data). Numbers of survey points are given in parentheses. Birds are ordered by the unburned-to-burned ratio of abundance, and species that are completely absent from or are significantly (Mann-Whitney *U* tests) less abundant in the opposite condition are highlighted in yellow. In both locations are bird species restricted to either early or later successional stages, but the amount of species turnover (degree of replacement of late with early succession specialists) is less pronounced after severe fire in Australia than after severe fire in the western United States.

suppress fire, mitigate fire severity, "restore" or "rehabilitate" burned forests after fire, and mimic the effects of severe fire are incompatible with the maintenance of ecosystem integrity (Chapter 13). Below we use results from bird research as evidence to support this statement, and we offer positive suggestions about what land managers could be doing differently.

s0060 *Fire Prevention Should Be Focused on Human Population Centers*

p0180 The dependence of so many bird (and many other plant and animal) species on conditions created by severe fire is clear. It necessarily follows that we cannot prevent fire and still retain anything close to a natural world. The obvious

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alternative is to focus prevention efforts toward population centers that are most at risk from severe fire so that fire can be left to periodically restore forest conditions elsewhere. Smokey Bear needs to refine his message so that it reflects a desire to save human lives and property, *not* a desire to save trees from fire in our wildlands (see Chapter 13).

s0065 *Fire Suppression Should Be Focused on the Wildland-Urban Interface (or Fireshed)*

p0185 Because many species depend on severe fire, it also necessarily follows that we should focus suppression efforts on areas immediately adjacent to human settlements (see Chapter 13). Wildland firefighters should serve primarily as support for firefighters who defend homes and human lives. Efforts to suppress fire beyond settled areas should be viewed as little more than efforts to save the forest from itself—forests need fire in the same way that they need sunlight and rain.

s0070 *High-Severity Fires Beget Mixed-Severity Results*

p0190 In contrast with high-severity fire, low-severity understory fires cannot create as broad a range of postfire conditions as severe fires can, nor can they stimulate the postfire process of ecological succession like a severe fire can. Therefore, managing for the maintenance of biodiversity requires more conscientious management for the maintenance of severe fires and the mixed-severity landscape effects that result from such fires (Nappi et al., 2010; Taylor et al., 2012).

s0075 *Mitigate Fire Severity Through Thinning only Where such Fuel Reduction Is Appropriate*

p0195 Because many species depend on severe fire, it necessarily follows that we should focus forest-thinning efforts in the wildland-urban interface and perhaps beyond that in what are basically artificial tree plantations that have resulted from past timber harvesting (see Odion et al., 2014a for review of this topic). The distributions of black-backed woodpeckers and many other fire-dependent plant and animal species make it abundantly clear that a reduction in fire severity is ecologically justified in only a very small proportion of vegetation types (Odion et al., 2014a; Sherriff et al., 2014). The presence of numerous fire-dependent species in most conifer forests throughout the American West (as illustrated by the abundance of bird research results considered in this chapter) is the strongest possible indication that the same forests have burned severely for millennia and are well within the historical range of natural variation.

p0200 The distribution of birds like the black-backed woodpecker and other fire-dependent plant and animal species, which blanket most of the forested land in the American West, are clearly at odds with claims (e.g., Haugo et al., 2015) that as much as 40% of public forested lands in parts of the United States are in need of restoration to prevent or mitigate the effects of severe fire. Lower-severity fires do not produce the mixed- and high-severity conditions needed by the most

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fire-dependent bird species, so efforts to mitigate fire severity in most places is incompatible with maintenance of the ecological integrity of most conifer forest systems (Odion et al., 2014a). So, what should we be doing differently? We could realize that modeled estimates indicating that our forests are in conditions that lie beyond the historical natural range of variation are just that—modeled estimates that rest strongly on many untested assumptions. We should always compare modeled results with insight gained by ecologists who can also draw strong inferences about historical conditions and, more specifically, about the kind of environments that necessarily led to adaptations of plants and animals—adaptations that reflect the distant past much more accurately than other methods commonly used to reconstruct natural fire regimes.

s0080 *Postfire "Salvage" Logging in the Name of Restoration or Rehabilitation Is Always Inappropriate*

p0205 Postfire "salvage" logging, seeding, planting, and shrub removal have overwhelmingly negative effects on natural systems (Lindenmayer et al., 2004; Lindenmayer and Noss, 2006; McIver and Starr, 2006; Swanson et al., 2011; DellaSala et al., 2014; Hanson, 2014), and birds have been instrumental in uncovering that fact. There is nothing as obvious to a birdwatcher as the negative effect of postfire salvage logging on the most fire-dependent birds (Uxley, 2014), and these anecdotal impressions are backed up by the strongest and most consistent scientific results ever published on any wildlife management issue (Hutto, 1995, 2006; Morissette et al., 2002; Nappi et al., 2004; Hutto and Gallo, 2006; Koivula and Schmiegelow, 2007; Hanson and North, 2008; Cahall and Hayes, 2009; Saab et al., 2009; Rost et al., 2013). One look at (Figure 3.14), or one walk through, a salvage-logged forest after knowing

f0075 **FIGURE 3.14** A vivid view of what can only be described as an ecological disaster following this postfire salvage logging operation, which took place after the 1988 Combination fire in Montana. *(Photograph by Richard Hutto.)*

something about the biological wonder associated with a severely burned forest should be enough to convince any thinking person that there is no justification for this kind of land management activity.

p0210 It is bad enough that forests logged after fire are made unsuitable for black-backed woodpeckers and other early postfire specialists, but much worse is that postfire logging and shrub removal through mechanical or chemical means may also act as an "ecological trap" (Robertson and Hutto, 2006). This can occur when birds are attracted to burned areas that seem to be suitable and then those areas are suddenly transformed by logging or shrub removal into unsuitable habitat in an unnaturally rapid period of time. This is the most reasonable explanation for why black-backed woodpeckers are more abundant in dense, burned forests that are logged after fire than they are in burned forests that are logged before fire—birds are not attracted to the latter, where tree densities are too low and sizes are too small to provide suitable habitat, but they are attracted to the former before the trees are unexpectedly removed (Hutto, 2008). Similarly, the disproportionate use of recently logged, unburned, old-growth forests in Canada (Tremblay et al., 2009) suggests that black-backed woodpeckers sometimes make the best of a marginal situation, not that they "prefer" recently logged forests.

p0215 Although the ecological responses of birds to postfire salvage logging may differ among globally different ecosystems (Rost et al., 2012), there is absolutely no ecological justification for this kind of logging in the mixed-conifer forests of the western United States, nor is there an economic justification to salvage log after fire, because there are always better places to harvest timber without anywhere near the negative ecological consequences associated with postfire salvage logging. This is a matter of setting priorities for timber harvest, and burned forests should be at the bottom of the list. Burned forests not only provide unique ecological value, they also set the stage for the development of a variety of future forest conditions—conditions that are much more varied than those associated with development after artificial disturbance from logging. Forests have their own rules and timetables associated with the natural process of ecological succession, and we should embrace that variety and complexity. What could be done differently? Postfire rehabilitation should focus on roads, culverts, and other infrastructure issues, and nothing else. We need to recognize that new forest conditions get created after fire, and a disturbance-dependent forest does not need to be "fixed" after disturbance takes place.

s0085 *We Can Do more Harm Than Good Trying to "Mimic" Nature*

p0220 Prescribed burning, forest thinning, and the use of other forms of artificial disturbance in an effort to mimic nature are often poor substitutes for natural disturbance processes. Prescribed burning is usually done out of season, too frequently, and in a manner that is far too mild to have the necessary effects in most systems that evolved with fire (England, 1995; Tucker and

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Robinson, 2003; Penman and Towerton, 2008; Peters and Sala, 2008; Arkle and Pilliod, 2010; Rota et al., 2014a). Thinning forests in a manner thought to mimic disturbance effects is also likely to be problematic because natural disturbance (the process of fire itself) produces effects that cannot be emulated through artificial means (Schieck and Song, 2006; Reidy et al., 2014). Moreover, a thinned forest that subsequently burns in a natural fire event will not be suitable as postfire habitat for early postfire specialists because of the reduction in tree densities and sizes (Hutto, 2008). Finally, the use of forest thinning in the name of forest restoration is inappropriately applied to relatively mesic mixed-conifer forests that are unlikely to be in need of restoration, as indicated by a lack of posttreatment change in bird communities toward what one would expect if the forests were actually outside the historical range of natural variation (Hutto et al., 2014).

p0225 Except in the case of an endangered species, the worst management approach is one that focuses narrowly on creating *artificial* conditions needed by a single species. This is "single-species management," which is not the same thing as using a "management indicator approach." Management indicators are not meant to be tools that enable land managers to artificially modify land conditions to benefit a single species. Instead, a management indicator species should be used as an indication of a particular kind of "natural" condition that needs to be maintained on the landscape and as a check that the land condition is indeed acceptable to a species that requires such conditions. Even for an endangered species, we should always be thinking about maintaining the "natural" conditions that historically maintained its population. Thus, although artificial tree plantations may provide conditions used by Kirtland's warbler (Spaulding and Rothstein, 2009), the bird historically nested beneath the canopy of young trees born of fire. Therefore we should create conditions safe enough to allow natural severe fire events to unfold throughout most of its historical range. As clearly stated in the Endangered Species Act (ESA, Section 2), "the purposes of this act are to provide a means whereby the *ecosystems upon which endangered species and threatened species depend* may be conserved . . ." (our italics). Conservation should be about the larger system (e.g., maintaining a fire disturbance-based jack pine forest system), not about ~~a~~ finding a way to maintain a species through artificial means. Thus the black-backed woodpecker is an "indicator" or "focal species" that should be used to inform us about a critically important "natural" disturbance process and vegetation condition we need to maintain—severely burned forests and all the associated organisms that thrive within them.

p0230 What could we be doing differently? We need to trust that disturbance-dependent systems need severe disturbance (yes, that means a lot of tree death) to stimulate ecological succession in a manner that is indeed natural. We also need to appreciate that modeled *means and standard deviations* associated with measures of forest structure are not the same things as historical *ranges of variation* associated with the same measures. While some places have tree densities that exceed some estimated historical average value, it does not mean they fall outside the historical range of natural variation. Land managers need to relax in response to severe fire. As long as we can reduce the frequency of human-caused fires and remain safe during naturally ignited fire events, a management option that lets nature take its course will work just fine (Gill, 2001; Bradstock, 2008). In this context, noting that safety is best achieved through mechanical treatments in small areas immediately adjacent to structures (Cohen, 2000; Cohen and Stratton, 2008; Winter et al., 2009; Stockmann et al., 2010; Gibbons et al., 2012; Syphard et al., 2014), and not through mechanical treatments in more remote wildlands, is important. Given this fact, why treatments in relatively remote, publicly owned wildlands have become the tactic most commonly used to reduce wildfire risk is puzzling (Schoennagel et al., 2009).

s0090 *Concluding Remarks*

p0235 The most important ecological lessons we can take away from the bird research described in this chapter are that (1) many species have evolved to the point where they now require severe fire to create the conditions they need, and (2) even though some ecological systems may have departed significantly from what are believed to be historical conditions (e.g., tree plantations in the Pacific Northwest), birds are telling us (through their behavior and distribution patterns) that the vast majority of fire-dependent ecosystems are still well within the historical range of natural variation, are plenty “resilient,” and are fully capable of proceeding quite naturally through the process of succession following a severe-fire event. Therefore, thinning forests in the name of restoration is largely unnecessary. If this were not true, the world would be full of places that experienced a severe fire disturbance and then underwent an unnatural transformation or “type conversion” following the disturbance event, never to return to what was there before disturbance. It is most telling that those kinds of places are rare indeed.

p0240 For those who would like to read, view, or hear more about the relationship between birds and severe fire, there are excellent children’s books (e.g., Peluso, 2007; Collard, 2015); several informative videos, including a field trip that illustrated many of the patterns discussed here (listed in the Preface); and a Fire Ecology Lab Facebook page (https://www.facebook.com/FireEcologyLab) devoted to building an appreciation for the role of severe fire in our forests.

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Non-Print Items

Abstract

Important lessons emerge from studies of birds in ecosystems born of, and maintained by, mixed- to high-severity fire. Specifically, (1) the effect of fire on any one species is context dependent. It depends on the time since the fire, the fire severity, and vegetation type and condition. (2) Bird species respond differently to any given postfire condition and, given an appropriate time since the fire and postfire vegetation conditions, most benefit from severe fire. (3) Some bird species (the black-backed woodpecker being iconic) seem to *depend* on conditions created by severe fire, as evidenced by their distribution patterns, territory sizes, nest success, and other adaptations. (4) Given these facts, current management practices designed to prevent fire, suppress fire, mitigate fire severity, “restore” or “rehabilitate” burned forests after fire, and mimic the effects of severe fire are incompatible with the maintenance of bird populations and, therefore, ecosystem integrity.

Keywords: Adaptation; Bird; Disturbance; Fire severity; Forest restoration; Salvage logging; Severe fire.

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SUPPLEMENTARY MATERIALS

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Materials and Methods
Figs. S1 to S5
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References (*22*–*29*)

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CONSERVATION

A global map of roadless areas and their conservation status

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Roads fragment landscapes and trigger human colonization and degradation of ecosystems, to the detriment of biodiversity and ecosystem functions. The planet's remaining large and ecologically important tracts of roadless areas sustain key refugia for biodiversity and provide globally relevant ecosystem services. Applying a 1-kilometer buffer to all roads, we present a global map of roadless areas and an assessment of their status, quality, and extent of coverage by protected areas. About 80% of Earth's terrestrial surface remains roadless, but this area is fragmented into ~600,000 patches, more than half of which are <1 square kilometer and only 7% of which are larger than 100 square kilometers. Global protection of ecologically valuable roadless areas is inadequate. International recognition and protection of roadless areas is urgently needed to halt their continued loss.

The impact of roads on the surrounding landscape extends far beyond the roads themselves. Direct and indirect environmental impacts include deforestation and fragmentation, chemical pollution, noise disturbance, increased wildlife mortality due to car collisions, changes in population gene flow, and facilitation of biological invasions (*1*–*4*). In addition, roads facilitate "contagious development," in that they provide access to previously remote areas, thus opening them up for more roads, land-use changes, associated resource extraction, and human-caused disturbances of biodiversity (*3*, *4*). With the length of roads projected to increase by >60% globally from 2010 to 2050 (*5*), there is an urgent need for the development of a comprehensive global strategy for road development if continued biodiversity loss is to be abated (*6*). To help mitigate the detrimental effects of roads, their construction should be concentrated as much as possible in areas of relatively low "environmental values" (*7*). Likewise, prioritizing the protection of remaining roadless areas that are regarded as important for biodiversity and ecosystem functionality requires an assessment of their extent, distribution, and ecological quality.

Such global assessments have been constrained by deficient spatial data on global road networks. Importantly, recent publicly available and rapidly improving data sets have been generated by crowd-sourcing and citizen science. We demonstrate their potential through OpenStreetMap, a project with an open-access, grassroots approach to mapping and updating free global geographic data, with a focus on roads. The available global road data sets, OpenStreetMap and gROADS, vary in length, location, and type of roads; the former is the data set with the largest length of roads (36 million km in 2013) that is not restricted to specific road types (table S1). OpenStreetMap is more complete than gROADS, which has been used for other global assessments (*7*), but in certain regions, it contains fewer roads than sub-global or local road data sets [see the example of Center for International Forestry Research data for Sabah, Malaysia (*8*); table S1]. Given the pace of road construction and data limitations, our results overestimate the actual extent of global roadless areas.

The spatial extent of road impacts is specific to the impact in question and to each particular road and its traffic volume, as well as to taxa, habitat, landscape, and terrain features. Moreover, for a given road impact, its area of ecological influence is asymmetrical along the road and can vary among seasons, between night and day, according to weather conditions, and over longer time periods. We conducted a comprehensive literature review of 282 publications dealing with "road-effects zones" or including the distance to roads as a covariate, of which 58 assessed the spatial influence of the road (table S2). All investigated road impacts were documented within a distance of

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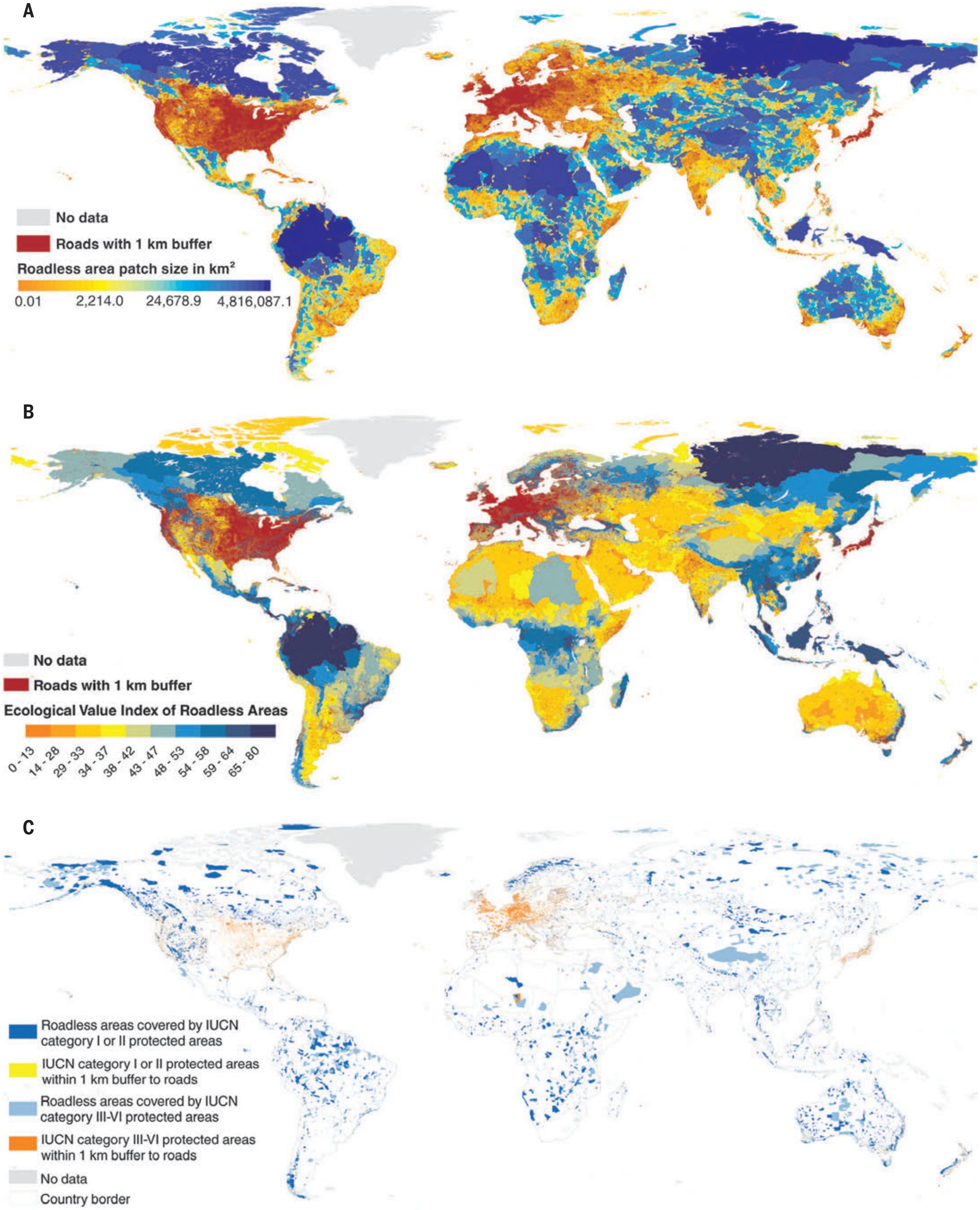


Fig. 1. The global distribution of roadless areas, based on a 1-km buffer around all roads. The distribution is depicted according to (**A**) size classes, (**B**) the ecological value index of roadless areas (EVIRA; based on patch size, connectivity, and ecosystem functionality), and (**C**) representation in protected areas (*8*).

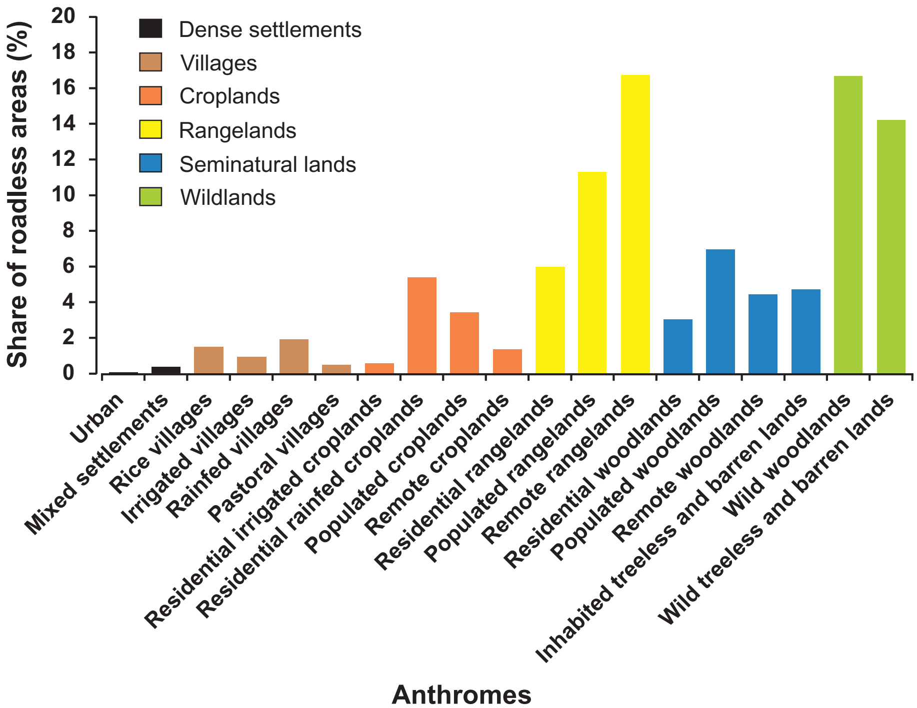


Fig. 2. Extent of roadless areas (1-km buffer) across anthromes. The majority of the world's roadless areas are in remote and unmodified landscapes, but they also occur in anthropogenically modified landscapes. The so-called anthromes were mapped according to (*10*).

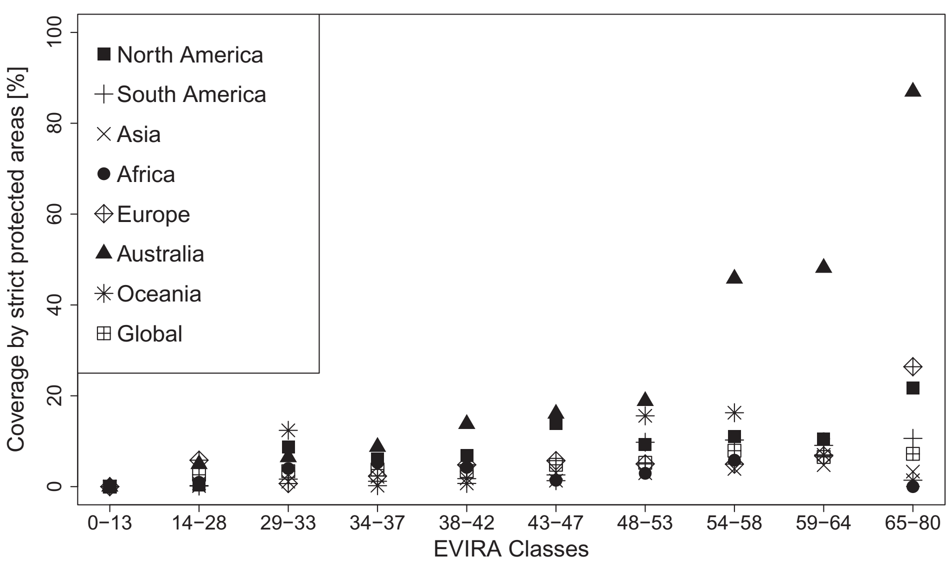


Fig. 3. Coverage of roadless areas by strictly protected areas (IUCN categories I and II) compared with global and continental EVIRA values. If priority were given to protecting roadless areas with high ecological functionality, we should see a positive correlation, with higher coverage associated with higher EVIRA values.

1 km from the road, 39% reached out to 2 km from the road, and only 14% extended out to 5 km from the road (fig. S1). Because the 1-km buffer along each side of the road represents the zone with the highest level and variety of road impacts, we defined roadless areas as those land units that are at least 1 km away from all roads and, therefore, less influenced by road effects. We compared results from using this criterion with the outcomes from using an alternative 5-km buffer (see fig. S2 and table S3). We excluded all large water bodies, as well as Greenland and Antarctica, which are mostly covered by ice, from the analyses.

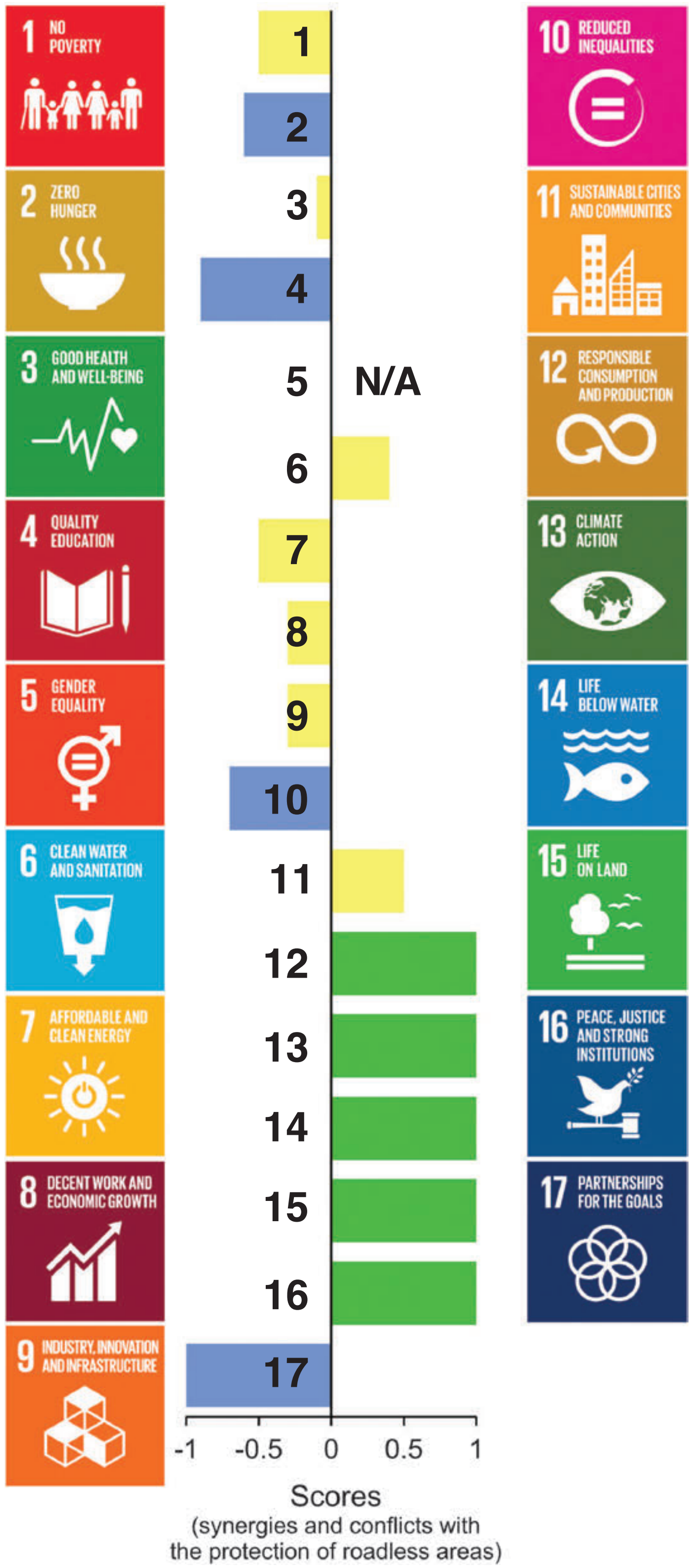
Roadless areas with a 1-km buffer to the nearest road cover about 80% of Earth's terrestrial surface (~105 million km^2). However, these roadless areas are dissected into almost 600,000 patches. More than half of the patches are <1 km^2; 80% are <5 km^2; and only 7% are >100 km^2 (table S4 and fig. S3). If the buffer is extended to 5 km, there is a substantial reduction in roadless areas to about 57% of the world's terrestrial surface (~75 million km^2), dissected into 50,000 patches (fig. S2 and table S3). The occurrence, distribution, and size of roadless areas differ considerably among continents (Fig. 1A and fig. S4). For instance, the mean size of roadless patches (1-km buffer) is 48 km^2 in Europe, compared with >500 km^2 in Africa. Because of comparatively large gaps in available spatial data on roads in many segments of the tropics, the number and size of roadless areas are overestimated and should be treated with caution (e.g., Borneo; table S1).

All identified roadless areas were assessed for a set of ecological properties that were selected to reflect their relative importance to biodiversity, ecological functions, and ecosystem resilience: patch size, connectivity, and ecosystem functionality (*9*) (table S5). We normalized these three indicators to between 0 and 100 to calculate an additive and unitless index of the ecological value of each roadless area identified (termed the ecological value index of roadless areas, or EVIRA) [Fig. 1B and fig. S5; the specific rationale and technicalities of the chosen indicators are described in table S5 (*8*)]. The EVIRA values range from 0 to 80. A sensitivity analysis shows that ecosystem functionality and patch size are the best single indicators for the final index values (table S6 and figs. S6 to S8). Areas with relatively high index values tend to have a lower coefficient of variation (fig. S9).

We used the International Union for Conservation of Nature (IUCN) and UN Environment Programme–World Conservation Monitoring Centre data set of global protected areas to determine the extent of roadless areas that are protected (*8*) (Fig. 1C). The roadless areas distribution across human-dominated landscapes was determined following the classification of so-called anthromes, defined as biomes shaped by human land use and infrastructure (*10*) (Fig. 2 and table S7).

When examining the density of roads within different biomes, large discrepancies in distribution are apparent. The tundra and rock and ice-covered biomes are nearly entirely roadless, whereas temperate broadleaf and mixed forests have the lowest share of roadless areas (41%; figs. S9 and S10). Boreal forests of North America and Eurasia still retain large tracts of roadless areas (figs. S10 and S11). In the tropics, large roadless landscapes (>1000 km^2) remain in Africa, South America, and Southeast Asia, with the Amazon having the single largest roadless segment. In relation to the anthromes (*10*), about two-thirds of the world's roadless areas can be described as remote and unmodified landscapes [26% uninhabited or sparsely inhabited treeless and barren lands; 21% natural and remote seminatural woodlands, with 17% wild woodlands therein (*8*); Fig. 2 and table S7]. The remaining one-third consists of rangelands, indicating that roadless areas can also occur in anthropogenically modified landscapes.

Fig. 4. Synergies and conflicts between conservation of roadless areas and the United Nations' Sustainable Development Goals. Scores <–0.5 (blue bars) indicate that conflicts with the goal prevail; scores between –0.5 and 0.5 (yellow) indicate a mixture of synergies and conflicts with the goal; and scores >0.5 (green) indicate prevailing synergies with the goal [for details, see table S11 (*8*)]. The scores reflect substantial imminent conflicts between various Sustainable Development Goals and conservation of roadless areas (table S11).



About one-third of the world's roadless areas have low EVIRA values. Patches with relatively low EVIRA values (ranging from 0 to 37; namely, <50% of the maximum value) account for 35% of the overall roadless area distribution, because most are small, fragmented, isolated, or otherwise heavily disturbed by humans. Some large tracts of roadless areas, such as arid lands in northern Africa or central Asia, occur in areas of sparse vegetation and low biodiversity and, thus, have low index values for ecosystem functionality (*9*) (Fig. 1B). High EVIRA values occur both in tropical and boreal forests. The relative conservation value of roadless areas is context-dependent. Comparatively small or moderately disturbed roadless areas have higher conservation importance in heavily roaded environments, such as most of Europe, the conterminous United States, and southern Canada.

Although the world's protected areas cover 14.2% of the terrestrial surface, only 9.3% of the overall expanse of roadless areas is within protected areas (all IUCN categories; Fig. 1C and table S8). There is no major difference in the coverage of roadless areas by strictly protected areas (IUCN categories I and II) versus the coverage of the overall landscape by strictly protected areas (3.8% roadless versus 4.2% overall). Only in North America, Australia, and Oceania are more than 6% of roadless areas under strict protection (table S8). If conservation efforts were to prioritize functional, ecologically important roadless areas, we would find a positive relation between strict protection coverage and EVIRA values of roadless areas. However, with the exception of Australia, this is not the case (Fig. 3 and table S9). Asia and Africa have particularly low protection coverage for roadless areas with high EVIRA values. For instance, we found gaps in the Asian tropical southeast, as well as in boreal biomes.

The recent Global Biodiversity Outlook (*11*) gives a bleak account of the progress made toward reaching the United Nations' biodiversity agenda as specified in the 20 Aichi Targets of the Convention on Biological Diversity (*12*). Governments have failed on several accounts to keep their use of natural resources well within safe ecological limits (target 4); to halt or at least halve the rate of habitat loss and substantially reduce the degradation and fragmentation of natural habitats (target 5); and to appropriately protect areas of particular importance for biodiversity and ecosystem services (target 11). To achieve global biodiversity targets, policies must explicitly acknowledge the factors underlying prior failures (*13*). Despite increasing scientific evidence for the negative impacts of roads on ecosystems, the current global conservation policy framework has largely ignored road impacts and road expansion. Furthermore, key policies on road infrastructure and development, such as the Cohesion Policy of the European Union, fail to take into account biodiversity.

In the much wider context of the United Nations' Sustainable Development Goals, conflicting interests can be seen between goals intended to safeguard biodiversity and those promoting economic development (*14*). We analyzed how roadless areas relate to the global conservation and sustainability agendas. As a transparent synthesis, we calculated simple scores of conflicts versus synergies of Sustainable Development Goals and Aichi Targets with the conservation of roadless areas (tables S10 and S11). Roads are explicitly mentioned in the Sustainable Development Goals only for their contribution to economic growth (goal 8), promoting further expansion into remote rural areas, and consideration is given neither to the environmental nor the social costs of road development. The resulting scores reflect substantial imminent conflicts (Fig. 4 and table S10); only in five Sustainable Development Goals do synergies with conservation of roadless

areas prevail, and four Sustainable Development Goals are predominantly in conflict with conservation of roadless areas. Maybe even more surprisingly, several of the Aichi Targets are ambivalent with respect to conserving roadless areas, rather than being in synergy entirely [six conflicting versus 11 synergistic targets (*8*); table S11].

There is an urgent need for a global strategy for the effective conservation, restoration, and monitoring of roadless areas and the ecosystems that they encompass. Governments should be encouraged to incorporate the protection of extensive roadless areas into relevant policies and other legal mechanisms, reexamine where road development conflicts with the protection of roadless areas, and avoid unnecessary and ecologically disastrous roads entirely. In addition, governments should consider road closure where doing so can promote the restoration of wildlife habitats and ecosystem functionality (*4*). Our global map of roadless areas represents a first step in this direction. During planning and evaluation of road projects, financial institutions, transport agencies, environmental nongovernmental organizations, and the engaged public should consider the identified roadless areas.

The conservation of roadless areas can be a key element in accomplishing the United Nations' Sustainable Development Goals. The extent and protection status of valuable roadless areas can serve as effective indicators to address several Sustainable Development Goals, particularly goal 15 ("Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss") and goal 9 ("Build resilient infrastructure, promote inclusive and sustainable industrialization and foster innovation"). Enshrined in the protection of roadless areas should be the objective to seek and develop alternative socioeconomic models that do not rely so heavily on road infrastructure. Similarly, governments should consider how roadless areas can support the Aichi Targets (see tables S10 and S11). For instance, the target of expanding protected areas to cover 17% of the world's terrestrial surface could include a representative proportion of roadless areas.

Although we acknowledge that access to transportation is a fundamental element of human well-being, impacts of road infrastructure require a fully integrated environmental and social cost-benefits approach (*15*). Still, under current conditions and policies, limiting road expansion into roadless areas may prove to be the most cost-effective and straightforward way of achieving strategically important global biodiversity and sustainability goals.

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SUPPLEMENTARY MATERIALS

www.sciencemag.org/content/354/6318/1423/suppl/DC1
Materials and Methods
Figs. S1 to S11
Tables S1 to S11
Data Sources
References (*16–180*)

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PLANT PATHOLOGY

Regulation of sugar transporter activity for antibacterial defense in *Arabidopsis*

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Microbial pathogens strategically acquire metabolites from their hosts during infection. Here we show that the host can intervene to prevent such metabolite loss to pathogens. Phosphorylation-dependent regulation of sugar transport protein 13 (STP13) is required for antibacterial defense in the plant *Arabidopsis thaliana*. STP13 physically associates with the flagellin receptor flagellin-sensitive 2 (FLS2) and its co-receptor BRASSINOSTEROID INSENSITIVE 1–associated receptor kinase 1 (BAK1). BAK1 phosphorylates STP13 at threonine 485, which enhances its monosaccharide uptake activity to compete with bacteria for extracellular sugars. Limiting the availability of extracellular sugar deprives bacteria of an energy source and restricts virulence factor delivery. Our results reveal that control of sugar uptake, managed by regulation of a host sugar transporter, is a defense strategy deployed against microbial infection. Competition for sugar thus shapes host-pathogen interactions.

Plants assimilate carbon into sugar by photosynthesis, and a broad spectrum of plant-interacting microbes exploit these host sugars (*1*, *2*). In *Arabidopsis*, pathogenic bacterial infection causes the leakage of sugars to the extracellular spaces (the apoplast) (*3*), a major site of colonization by plant-infecting bacteria. Although leakage may be a consequence of membrane disintegration during pathogen infection, some bacterial pathogens promote sugar efflux to the apoplast by manipulating host plant sugar transporters (*4*, *5*). Interference with sugar absorption by bacterial and fungal pathogens reduces their virulence, highlighting a general

Science
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A global map of roadless areas and their conservation status

Pierre L. Ibisch, Monika T. Hoffmann, Stefan Kreft, Guy Pe'er, Vassiliki Kati, Lisa Biber-Freudenberger, Dominick A. DellaSala, Mariana M. Vale, Peter R. Hobson and Nuria Selva (December 15, 2016)

Science **354** (6318), 1423-1427. [doi: 10.1126/science.aaf7166]

Editor's Summary

Too many roads

Roads have done much to help humanity spread across the planet and maintain global movement and trade. However, roads also damage wild areas and rapidly contribute to habitat degradation and species loss. Ibisch *et al*. cataloged the world's roads. Though most of the world is not covered by roads, it is fragmented by them, with only 7% of land patches created by roads being greater than 100 km^2. Furthermore, environmental protection of roadless areas is insufficient, which could lead to further degradation of the world's remaining wildernesses.

Science, this issue p. 1423

esa ECOSPHERE

Toward a more ecologically informed view of severe forest fires

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Abstract. We use the historical presence of high-severity fire patches in mixed-conifer forests of the western United States to make several points that we hope will encourage development of a more ecologically informed view of severe wildland fire effects. First, many plant and animal species use, and have sometimes evolved to depend on, severely burned forest conditions for their persistence. Second, evidence from fire history studies also suggests that a complex mosaic of severely burned conifer patches was common historically in the West. Third, to maintain ecological integrity in forests born of mixed-severity fire, land managers will have to accept some severe fire and maintain the integrity of its aftermath. Lastly, public education messages surrounding fire could be modified so that people better understand and support management designed to maintain ecologically appropriate sizes and distributions of severe fire and the complex early-seral forest conditions it creates.

Key words: early succession; ecological integrity; ecological system; fire management; fire regime; forest resilience; forest restoration; severe fire; wildfire.

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Introduction

The spatiotemporal expression of fire events over time in any landscape produces a "fire regime" that influences ecosystem dynamics in that area (Heinselman 1981, Kilgore 1981). Even though the various characteristics of a fire regime (Table 1) are continuous in nature, the traditional approach in representing this variation has been to create a small number of discontinuous categories. Fire regimes in western North America, for example, are often classified into as few as three categories: (1) low-severity, (2) mixed-severity, and (3) high-severity or stand-replacement (Agee 1998, Brown 2000). Our attempt to categorize fire regimes is ". . . an oversimplification…for the convenience of humans" (Sugihara et al. 2006; p. 62), and has had the unfortunate consequence of minimizing rather than emphasizing variation in fire behavior and fire outcomes among vegetation types and across spatial scales (Morgan et al. 2014). In reality, relatively few forest types fit entirely within either of the two extremes—the low-severity (e.g., some interior ponderosa pine) or the stand-replacement (e.g., Rocky Mountain lodgepole pine) categories. Instead, as a simple analysis

Table 1. Characteristics or descriptors often used to describe disturbance regimes (from Keane 2013).

Disturbance Characteristic	Description	Example
Agent	Factor causing the disturbance	Fire is an agent that can kill trees
Source, Cause	Origin of the agent	Lightning is a source for wildland fire
Frequency	How often the disturbance occurs or its return time	Years since last fire (scale dependent)
Intensity	A description of the magnitude of the disturbance agent	Wildland fire heat output
Severity	The level of impact of the disturbance on the environment	Fuel consumption in wildland fires; change in biomass
Size	Spatial extent of the disturbance	Tree kill can occur in small patches or across entire landscapes
Pattern	Patch size distribution of disturbance effects; spatial heterogeneity of disturbance effects	Fire can burn large regions but weather and fuels can influence fire intensity and therefore the patchwork of tree mortality
Seasonality	Time of year of that disturbance occurs	Spring burn vs. fall burn
Duration	Length of time of that disturbances occur	Fires can burn for a day or for an entire summer
Interactions	Disturbance types may interact with each other, or with climate, vegetation and other landscape characteristics	Mountain pine beetles may create fuel complexes that facilitate or exclude wildland fire
Variability	The spatial and temporal variability of the above factors	Each of the above characteristics has variation associated with it

using LANDFIRE data (Rollins 2009, <http://www.landfire.gov>) reveals, roughly 85% of all forested lands within the western US fit within the mixed-severity category, which includes proportions of low-, moderate-, and high-severity (lethal to more than 70% of all trees) fire that vary widely across vegetation types and biophysical settings.

Agee (1993) captured the essence of this important idea in a graph depicting the proportion of low-, moderate-, and high-severity fire across the range of fire regimes (Fig. 1). Note that change from one fire regime to the next (movement along the x-axis) is accompanied not by the sudden appearance of a different fire severity, but by continuous changes in the proportions of each fire severity category. Thus, fire regimes blend imperceptibly into one another. More importantly, except for the two end points on the graph where the proportion of high-severity fire would be either 0% or 100%, most fire regimes consist of a mix of fire severities so, technically speaking, they fit best within a mixed-severity regime (Fig. 2). It is not the presence of a particular fire severity, but the proportion (and, presumably, the distribution and patch sizes) of each severity component that distinguishes regimes. Indeed, empirical

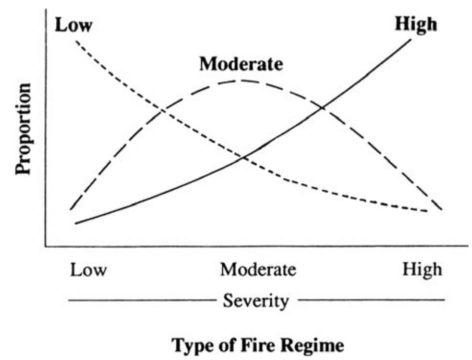

Fig. 1. This graph (from Agee 1993) illustrates that fire regimes are not characterized by the presence of only one kind of fire. Rather, it is the relative frequency of low-, moderate-, and high-severity fire in an average burn that varies among fire regimes.

data drawn from recent fires across the western United States between 1984 and 2008 (Fig. 3) reveal this continuous variation in proportions of different fire severities among fires. Thus, a more continuous view of fire regimes might be a better way to appreciate the infinite variability in fire behavior among forest types and geographic locations, and it might also promote a greater appreciation of severe fire as an integral

Fig. 2. Mixed-severity fires (fires that leave recognizable patches of low-severity, medium-severity, and high-severity effects) typify the majority of mixed-conifer forest systems in the western United States. The brown-needled and blackened areas harbor unique sets of plant and animal species found in no other forest conditions. This photograph of the North Fork of the Blackfoot River was taken 10 months after the 1988 Canyon Creek fire in Montana. Many fire-dependent plant and animal species were present in the more severely burned areas until they were helicopter logged, suggesting that unburned forests might be a better alternative for timber harvest.

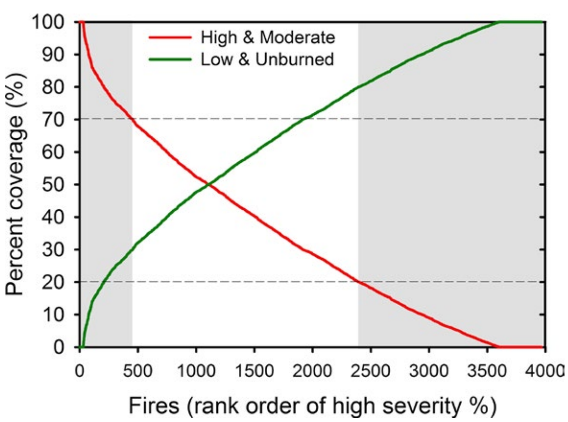

Fig. 3. The percent area within a fire perimeter that burned at low (green line) and at moderate to high (red line) severity is shown for a series of 3696 fires that burned in the western United States between 1984 and 2008 (after Belote 2015). The figure shows that the proportions of each severity category are continuously variable and that high-severity fire is a natural part of most forest fires in the West.

part of mixed- and high-severity conifer forest fire regimes.

Accordingly, we highlight the need for better information on the historical patterns and abundances of high-severity patches in different forest types. This is an important discussion because, even though our National Cohesive Wildland Fire Management Strategy (Wildland Fire Executive Council 2014) acknowledges that many fire regimes exist and that management needs to accommodate that variation and the variety of habitat such variation produces, contemporary fire management is focused heavily on the exclusion (prevention and suppression, collectively) or mitigation of severe fire. When either of those fails, management efforts seem to shift toward speeding the "recovery" of the forest after severe fire. With respect to the latter, there are repeated attempts to introduce legislation designed to expedite logging after fire (salvage logging). Although the removal of dead trees is justified near roads and structures for safety reasons, and although postfire logging can capture economic value of wood that would otherwise be lost, such logging has been shown to carry significant ecological costs (Hutto 2006, Lindenmayer and Noss 2006, Swanson et al. 2011, Lindenmayer and Cunningham 2013, DellaSala et al. 2015). The ecological benefits and necessity of severe fire (and its aftermath) has widespread implications for the flora and fauna that depend on the presence of burned forest conditions. Ecologically sound fire management includes land management designed to ensure the maintenance of ecologically appropriate mixes of fire severities within the forested landscapes of western North America while protecting homes and lives at the same time (Perry et al. 2011). An ecologically informed view of severe fire requires recognition that it is a natural component of many western conifer forests (Heinselman 1981, Arno 2000). Moreover, the severe-fire component must have been large enough and frequent enough to have favored the evolution of specialization by various plant and animal species to conditions that occur in the aftermath of severe fire. We offer the following points in an effort to better recognize and include severe fire as an integral part of fire management in mixed-conifer forest systems:

Severely burned forests create biologically unique conditions that cannot be created by other kinds of disturbances or through artificial means

Patterns in the habitat associations of plant and animal species can provide definitive evidence that severe fire plays an essential role in the ecology of mixed-conifer forests (Hutto et al. 2008). Specifically, if a plant or animal species occurs only in burned forest conditions created by severe fire events, then it cannot be using burned forest conditions merely opportunistically. Instead, the species must have evolved to depend on such conditions because it occurs rarely, if ever, in unburned habitat (Swanson et al. 2011, DellaSala et al. 2014). For example, some moss and lichen species are relatively restricted to severely burned forest conditions (Ahlgren and Ahlgren 1960), as are the fire morel mushroom (*Morchella elata*) and Bicknell's geranium (*Geranium bicknellii*) in forests throughout the West (Heinselman 1981, Pilz et al. 2004). The black-backed woodpecker (*Picoides arcticus*) is emblematic of a species that is relatively restricted to early successional conditions created by high-severity fire (Hutto 1995, Dixon and Saab 2000, Hoyt and Hannon 2002). Black-backed woodpeckers are attracted to postwildfire conditions because of the abundance of larvae of a number of wood-boring beetle species that are attracted to the fire-killed trees (Murphy and Lehnhausen 1998, Rota et al. 2015). Several of these beetle species are themselves relatively restricted to recently burned forests (Saint-Germain et al. 2004*a,b*, Boucher et al. 2012). Importantly, black-backed woodpeckers are significantly more likely to occur in the more severely burned portions of a mixed-severity fire (Hutto 2008, Latif et al. 2013). Although black-backed woodpeckers are known to occur outside severely burned forests on rare occasions, detailed study of survival and reproductive success shows that they exhibit growing populations only in forests recently burned by summer wildfires (Rota et al. 2014). The adaptations of thick bark, branch shedding, and serotiny in *Pinus* are thought to have evolved in response to a period of more intense crown fires in the mid-Cretaceous (He et al. 2012), and those adaptations also reflect the severe-fire backdrop against which pine, Douglas-fir, and larch are thought to thrive.

Many additional animal species, while not as narrowly restricted to burned forest conditions, clearly benefit from the burned forest conditions created by severe fires in mixed-conifer forests throughout the West (Hutto et al. 2015). For example, nest survival of white-headed woodpeckers is significantly higher in burned (wildfire) compared to unburned forest (Hollenbeck et al. 2011, Lorenz et al. 2015). In aquatic systems, severe fire events can rejuvenate stream habitats by causing large amounts of gravel, cobble, woody debris, and nutrients to be imported, resulting in increased production and aquatic insect emergence rates (Benda et al. 2003, Burton 2005, Malison and Baxter 2010, Ryan et al. 2011, Jackson et al. 2015). These changes can, in turn, affect food web dynamics in a way that results in higher growth rates in young trout, including young coastal cutthroat trout (*Oncorhynchus clarkii clarkii*) (Heck 2007) and rainbow trout (*Oncorhynchus mykiss*) (Rosenberger et al. 2011). Indeed, nonnative fish populations declined and native trout densities increased 3 yr after a severe fire in the Bitterroot River watershed, Montana, indicating that severe fire may help ensure ecological integrity of some western streams (Sestrich et al. 2011). In addition, native amphibians such as boreal toads (*Bufo boreas*) thrive in areas that burn severely (Dunham et al. 2007, Hossack and Corn 2007) and use severely burned areas more than expected due to chance (Hossack and Corn 2007, Guscio et al. 2008), as do some bat species (Buchalski et al. 2013).

These strong associations between organisms and severely burned forest patches suggests that many plant and animal species have evolved to rely on recurring severe wildfire events, and further indicates that severe fire events are a natural and important part of the fire regimes associated with many western mixed-conifer forest types. In other words, if one or more species occupy severely burned forests to the exclusion of other forest types (and if they do not tend to occupy forests disturbed through artificial means), then a severely burned forest would have to be considered natural, and would necessarily lie within the historical range of variation (Hutto et al. 2008). Moreover, a more intimate understanding of the biology of those plants and animals (e.g., knowledge of dispersal processes and patterns, foraging ecology, home-range sizes) can provide insight into the historical spatial scales at which severe fire operated across the broader landscape.

Fire history studies suggest that severe fire is an integral component of most fire regimes

In addition to the definitive evidence provided above, a growing body of fire history information points to the same conclusion—severe fire was historically, and is currently, an important component of many western conifer forest systems. At one end of the fire regime spectrum, conifer forests in the warmer, drier geographic areas in western North America are commonly characterized by frequent, low-severity fires that killed primarily juvenile trees historically, resulting in the maintenance of open pine forests with low densities of mature trees (Covington and Moore 1994*a,b*). Nevertheless, mixed and stand-replacement fires were possible even in these forest types after long inter-fire intervals, such as after an especially cold, wet period similar to what occurred during the Little Ice Age (Brown et al. 1999, Sherriff and Veblen 2007, Williams and Baker 2012, Odion et al. 2014, Hanson et al. 2015). At the other end of the fire regime spectrum, cooler, moister forest types, such as lodgepole pine forests, support fire regimes dominated by severe fire events (Brown and Smith 2000), although mixed- and low-severity fires are known to occur in these types as well (Barrett et al. 1991).

Between these two extremes lie the vast majority of mixed-conifer forest types in western North America. These include everything from the xeric, low-elevation, mixed ponderosa pine and Douglas-fir forest types to mesic, high-elevation, spruce-fir forest types. Unlike the forest types that are dominated by either the absence or presence of severe fire, mixed-conifer forests are best characterized by fire regimes of variable, or mixed severity (see Baker 2009: fig. 7.1), which means that the presence of sizable proportions of the three classes of fire severity characterize the fires that burn in those forest systems (Sherriff and Veblen 2006, 2007, Baker et al. 2007, Hessburg et al. 2007, Klenner et al. 2008, Perry et al. 2011, Schoennagel et al. 2011). Importantly, extreme weather (e.g., high temperature, low humidity, high wind speed) rather than quantity of woody fuels often exerts the greatest influence on fire severity and extent across that broad range of mixed-conifer forest types (Johnson et al. 2003, Schoennagel et al. 2004, Lydersen et al. 2014, Williams et al. 2015). This means that, in contrast with the situation in low-elevation or xeric-type ponderosa pine forests in some areas of the southwestern United States (Keane et al. 2008), the amount of high-severity fire in other mixed-conifer forest types is less likely to have departed significantly from historical ranges of variability, even though those forests may have experienced measurable twentieth century changes in fuels due to fire exclusion, timber harvest, and cattle grazing (e.g., Baker et al. 2007, Dillon et al. 2011, Marlon et al. 2012, Miller et al. 2012, Odion et al. 2014, Sherriff et al. 2014). We recognize the lack of relevant historical information on landscape-level distributions and spatial scales of different classes of fire severity for many forest types and regions, but severely burned forest patches have probably always occurred naturally, even in pure ponderosa pine forests of the Southwest, as Cooper (1961) and Weaver (1943) described long ago. We also know that, at least throughout the northern half of the western United States, the extent of severe-fire patches must have been both substantial enough in area and frequent enough to support those plant (e.g., lodgepole pine) and animal (e.g., wood-boring beetle and woodpecker) species that evolved to depend on severe fire itself or on the resulting severely burned forest conditions.

Maintaining ecological integrity means accommodating a broad spectrum of fire severities, including severe fire and its aftermath, in most mixed-conifer forests

We have now established two important facts: severe fire (moderate-to-high burn severity) is a natural agent of disturbance in many mixed-conifer forest types, and such fire is thought to be ecologically necessary for the presence or success of many plant and animal species. These two facts make it clear that management to maintain the ecological integrity of any ecosystem that harbors species that depend on severe fire as a disturbance agent will have to integrate severe fire and its effects into management goals. Moreover, if we better considered distribution patterns, home range sizes, movement patterns, and other animal adaptations that reflect the environment within which they evolved (e.g., Hutto et al. 2008), we could gain considerable insight into historical spatial scales under which severe fire operated as well. We are not questioning or attempting to discredit the evidence that some forest systems were historically dominated by low-severity fire; rather, we are encouraging land managers to also pay close attention to maintaining amounts and distributions of higher severity fire consistent with ecological integrity in our western mixed-conifer forests. The current science, management, and policy challenge for ecosystem managers is to estimate and incorporate amounts of low-, moderate-, and high-severity fire in a manner that maintains ecological integrity (Hessburg et al. 2007, Perry et al. 2011, Baker 2015).

While many fire ecologists understand the importance of more severe fire in forest ecosystems, politicians and the public at large have yet to reach the same understanding. Recent increases in the amount of forested area burned by wildfire over the past three decades in western North American forests (Westerling et al. 2006, Dennison et al. 2014) signaling what many believe to be the emergence of a new age of megafires (Attiwill and Binkley 2013), has created increased movement toward pre and postfire land management activities designed to reduce fire severity, mimic fire effects without the use of fire, or speed the recovery of a forest after fire. These activities may provide some societal benefits, but they can have real costs in terms of the way they negatively affect the ecological integrity of mixed-conifer forests born of mixed-severity fire. Removed from locations that pose a clear and immediate threat to human lives and property, the ecological costs associated with forest thinning may outweigh stated benefits by large margins. We highlight two types of land management (beyond fire suppression itself) that can have significant negative effects on fire-dependent species and, therefore, can interfere with our ability to maintain the ecological integrity of fire-dependent conifer forests: prefire fuel treatments and postfire salvage logging.

Prefire harvest treatments

We know a great deal about the effects of fuel treatments and restoration harvests on forest structure and vegetation recovery, but we know little about the ecological effects of such treatments on the prefire responses of most plant and animal species, and virtually nothing about postfire responses of the most fire-dependent plant and animal species after a treatment subsequently burns in a wildfire. This is because such treatments are rarely accompanied by "ecological effects monitoring," which, in contrast with implementation monitoring (evaluating whether a management activity was implemented) and effectiveness monitoring (evaluating whether the management activity achieved the stated goal), is specifically designed to address whether there are unforeseen negative ecological consequences of a management treatment (Hutto and Belote 2013).

Fuel treatments designed to restore fire-prone ecosystems should do so in the proper fire regime context; more specifically, they should produce appropriate postfire plant and animal responses when fire returns to the forest. Thus, treatments appropriate for dry forests that were historically maintained by a low-severity fire regime may be inappropriate for forests maintained by a mixed-severity fire regime. One serious negative consequence of canopy fuel reduction in forests that evolved with mixed-severity fire could be that fire-dependent species requiring high densities of large standing-dead trees created by the severe-fire component may not recruit after a subsequent fire. For example, the fire-dependent black-backed woodpecker was found to be even less abundant in mixed-conifer forests that were thinned before fire than in the same forest types logged after fire, even though the two pathways support similar standing dead tree densities. This is probably because birds rarely colonize thinned forests that burn, but they still make the best of a bad situation when trees are removed after they have already colonized a densely stocked, severely burned forest (Hutto 2008). Recent research on postfire soil conditions shows that soil C and N response following wildfire also depends on whether there have been fuel treatments, so the assessment of fuel treatment effects needs to include postfire response and not simply postharvest response (Homann et al. 2015). It has been suggested (e.g., Franklin and Johnson 2014) that variable-retention harvests could be designed to emulate early-seral conditions following natural disturbance events in forests born of mixed-severity fire, thereby avoiding the negative consequences associated with other tree harvesting methods. Unfortunately, that strategy is unlikely to satisfy the needs of those fire-dependent animal species that require high densities of fire-killed trees immediately following severe fire (Schieck and Song 2006, Hutto 2008, Reidy et al. 2014).

Postfire salvage logging

Salvage logging after fire is intended to recover economic value of timber that would otherwise be lost, to ensure human safety, and to reduce the risk of future fires. Unfortunately, salvage harvesting activities undermine the ecosystem benefits associated with fire (Lindenmayer et al. 2004, Lindenmayer and Noss 2006, Swanson et al. 2011). For example, postfire salvage logging removes dead, dying, or weakened trees, but those are precisely the resources that provide nest sites and an abundance of food in the form of beetle larvae and bark surface insects (Hutto and Gallo 2006, Koivula and Schmiegelow 2007, Saab et al. 2007, 2009, Cahall and Hayes 2009). No fire-dependent bird species has ever been shown to benefit from salvage logging (Hutto 2006, Hanson and North 2008). The ecological effects of salvage logging on aquatic ecosystems are also largely negative (Karr et al. 2004). In fact, the demonstrated negative ecological effects associated with postfire salvage logging are probably the most consistent and dramatic of any wildlife management effects ever documented for any kind of forest management activity (Hutto 2006). Therefore, because the National Forest Management Act and other legal mandates require public land managers to maintain the integrity of the larger ecological system, burned forests should perhaps be given special consideration compared with green-tree forests. Specifically, they could receive a low priority ranking when it comes to timber harvest decisions (with the obvious exception of small harvests associated with roads and other areas where safety or infrastructure are legitimate concerns). Timber can be harvested from many green-tree forests in a manner that imposes relatively little ecological cost in comparison with the costs associated with logging in burned forest (Lindenmayer and Cunningham 2013).

How do we move toward a more ecologically informed view of forest fires?

The ecological costs associated with some of the more commonly employed pre and postfire management activities in the western United States probably increase substantially as one moves from the low-elevation or xeric ponderosa pine or woodland forest types, where trees were widely spaced and severe fire historically played a spatially restricted role, to the broad array of more densely stocked mixed-conifer forest types, where severe fire historically played a major role. Therefore, a thorough understanding of the historical fire regime associated with any particular vegetation type or land area (as determined from multiple lines of evidence concerning regionally specific fire history) is critically important for land managers who concern themselves with the issues of wildfire risk, ecological restoration, or maintenance of the diversity of native species (Schoennagel and Nelson 2011). More specifically, quantification of appropriate fire rotations and proportions of low-, moderate-, and high-severity fire for any given forest landscape is critical for enlightened land management. For example, in some xeric ponderosa pine forest types, ecosystem restoration activities designed to decrease the severity of wildfire may be ecologically appropriate. The same management activities are not likely to be ecologically appropriate in many mixed-conifer forests, however, because key indicator species evolved to depend on significant amounts of severe fire in those forest types (Schoennagel et al. 2004, Hutto 2008, Klenner et al. 2008, Baker 2012, 2015, Williams and Baker 2012, Odion et al. 2014).

Land and fire managers are now facing future fires that many hypothesize will become larger and contain larger proportions of more severely burned patches under warming climate conditions (Rocca et al. 2014). Problems associated with climate change, however, must be solved through efforts directed toward the causes of climate change and not toward the symptoms of climate change. Any perceived problem with future changes in fire behavior cannot be solved by redoubling our effort to treat this particular climate change symptom by installing widespread fuel treatments that do nothing to stop the warming trend, and do little to reduce the extent or severity of weather-driven fires (Gedalof et al. 2005). Therefore, fuel management efforts to reduce undesirable effects of wildfires outside the xeric ponderosa pine forest types could be more strategically directed toward creating fire-safe communities (Calkin et al. 2014, Kennedy and Johnson 2014). A management emphasis directed toward altering conditions in and immediately adjacent to human communities is very different from an emphasis directed toward treating massive amounts of fuel on more remote public lands. Fuel treatment efforts more distant from human communities may carry the negative ecological consequences we outlined earlier and do little to stop or mitigate the effects of fires that are increasingly weather driven (Rhodes and Baker 2008, Franklin et al. 2014, Moritz et al. 2014, Odion et al. 2014).

Public land managers face significant challenges balancing the threats posed by severe fire with legal mandates to conserve wildlife habitat for plant and animal species that are positively associated with recently burned forests. Nevertheless, land managers who wish to maintain biodiversity must find a way to embrace a fire-use plan that allows for the presence of all fire severities in places where a historical mixed-severity fire regime creates conditions needed by native species while protecting homes and lives at the same time. This balancing act can be best performed by managing fire along a continuum that spans from aggressive prevention and suppression near designated human settlement areas to active "ecological fire management" (Ingalsbee 2015) in places farther removed from such areas. This could not only save considerable dollars in fire-fighting by restricting such activity to near settlements (Ingalsbee and Raja 2015), but it would serve to retain (in the absence of salvage logging, of course) the ecologically important disturbance process over most of our public land while at the same time reducing the potential for firefighter fatalities (Moritz et al. 2014). Severe fire is not ecologically appropriate everywhere, of course, but the potential ecological costs associated with prefire fuels reduction, fire suppression, and postfire harvest activity in forests born of mixed-severity fire need to considered much more seriously if we want to maintain those species and processes that occur only where dense, mature forests are periodically allowed to burn severely, as they have for millennia.

Another integral part of moving toward an ecologically informed perspective of forest fire involves getting the public, politicians, and policy-makers to better recognize and appreciate the critical role that severe fire plays in many forest systems. This has been difficult, and this difficulty has been exacerbated by public messages about severe fire that are uniformly negative. Progress toward allowing fires to burn is difficult unless the public begins to receive a message that differs markedly from the message that Smokey the Bear is sending them now. Fires in our wildlands are fundamentally natural and beneficial, so we must learn to live in a way that allows naturally occurring fires, including severe fires, to burn while minimizing risk to human property and lives (Calkin et al. 2014). That is a vastly different message from one that says severe fires are fundamentally bad and that we have to do everything in our power to prevent and suppress them, or from one that says severely burned forests are places where we should expedite efforts to capture residual economic value through "salvage" logging. We challenge ecologists and managers to pay greater attention to the degree of variation in fire regimes within mixed-conifer forests and to recognize that prefire thinning and postfire "restoration" activities may not always be compatible with maintenance of the ecological integrity of conifer forests that depend on complex mixed-severity fire disturbance.

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Research Article

A Case Study Comparison of LANDFIRE Fuel Loading and Emissions Generation on a Mixed Conifer Forest in Northern Idaho, USA

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ABSTRACT

The use of fire as a land management tool is well recognized for its ecological benefits in many natural systems. To continue to use fire while complying with air quality regulations, land managers are often tasked with modeling emissions from fire during the planning process. To populate such models, the Landscape Fire and Resource Management Planning Tools (LANDFIRE) program has developed raster layers representing vegetation and fuels throughout the United States; however, there are limited studies available comparing LANDFIRE spatially distributed fuel loading data with measured fuel loading data. This study helps address that knowledge gap by evaluating two LANDFIRE fuel loading raster options—Fuels Characteristic Classification System (LANDFIRE-FCCS) and Fuel Loading Model (LANDFIRE-FLM) layers—with measured fuel loadings for a 20 000 ha mixed

RESUMEN

El uso del fuego como herramienta de manejo de tierras es bien reconocido por sus beneficios ecológicos en varios ecosistemas naturales. Para continuar con el uso del fuego y a su vez cumplir con las regulaciones referidas a la calidad del aire, los gestores de tierras deben frecuentemente cumplir con tareas de modelado de emisiones durante el proceso de planificación de las quemas. Para alimentar tales modelos, el programa denominado *Landscape Fire and Resource Management Planning Tools* (LANDFIRE) ha desarrollado capas raster, que representan vegetación y combustibles a lo largo de todos los EEUU; desde luego, son limitados los estudios disponibles que puedan comparar los datos de carga de combustibles espacialmente distribuidos derivados del LANDFIRE, con datos similares producto de mediciones de carga de combustible en el terreno. Este estudio ayuda a dilucidar este vacío en el conocimiento mediante la evaluación de carga de combustible usando dos opciones del programa LANDFIRE—el *Fuels Characteristic Classification System* (LANDFIRE-FCCS) y el *Fuel Loading Model* (LANDFIRE-FLM) *layers*—

conifer study area in northern Idaho, USA. Fuel loadings are compared, and then placed into two emissions models—the First Order Fire Effects Model (FOFEM) and Consume—for a subsequent comparison of consumption and emissions results. The LANDFIRE-FCCS layer showed 200%* higher duff loadings relative to measured loadings. These led to 23% higher total mean total fuel consumption and emissions when modeled in FOFEM. The LANDFIRE-FLM layer showed lower loadings for total surface fuels relative to measured data, especially in the case of coarse woody debris, which in turn led to 51% lower mean total consumption and emissions when modeled in FOFEM. When the comparison was repeated using Consume model outputs, LANDFIRE-FLM consumption was 59% lower relative to that on the measured plots, with 58% lower modeled emissions. Although both LANDFIRE and measured fuel loadings fell within the ranges observed by other researchers in US mixed conifer ecosystems, variation within the fuel loadings for all sources was high, and the differences in fuel loadings led to significant differences in consumption and emissions depending upon the data and model chosen. The results of this case study are consistent with those of other researchers, and indicate that supplementing LANDFIRE-represented data with locally measured data, especially for duff and coarse woody debris, will produce more accurate emissions results relative to using unaltered LANDFIRE-FCCS or LANDFIRE-FLM fuel loadings. Accurate emissions models will aid

comparados con la medición de la carga para 20 000 ha de un área de bosques mixtos de coníferas en el norte de Idaho, EEUU. Las cargas de combustibles fueron comparadas, y luego ubicadas dentro de dos modelos—el *First Order Fire Effects Model* (FOFEM) y el *Consume*—para su subsecuente comparación de los resultados del consumo de combustibles y sus emisiones. El LANDFIRE-FCCS mostró una estimación 200%* superior en la carga del mantillo comparado con la carga medida a campo. Esto llevó a un valor 23% más alto en la media total de consumo y emisiones del combustible cuando fue modelado mediante el modelo FOFEM. El modelo LANDFIRE-FLM layer mostró menores cargas para combustibles de superficie relativo a datos medidos a campo, especialmente en el caso de restos de combustible leñoso grueso (*coarse woody debris*), que a su vez llevó a un 51% menos en el consumo y emisiones promedio cuando fueron modeladas por el modelo FOFEM. Cuando la comparación fue repetida usado el *Consume model outputs*, el consumo estimado por el LANDFIRE-FLM fue un 59% menor en relación a lo determinado en las parcelas medidas, con un 58% menos que las emisiones modeladas. Aunque ambos modelos de LANDFIRE y las cargas efectivamente medidas se ubican dentro de los rangos observados por otros investigadores en los ecosistemas mixtos de coníferas de los EEUU, la variación dentro de las cargas de combustible determinadas por las distintas fuentes fue alta, y las diferencias en carga de combustible llevan a diferencias significativas en consumo y emisiones, dependiendo éstos del modelo elegido. Los resultados de este estudio de caso son consistentes con aquellos obtenidos por otros investigadores, e indican que suplementando datos de LANDFIRE con datos locales obtenidos de mediciones a campo, especialmente para el mantillo y restos de combustible leñoso grueso, producirá resultados de consumo y emisiones más precisos que aquellos que usan solamente datos de carga provistos por LANDFIRE-FCCS o LANDFIRE-FLM. Los modelos de emisiones preci-

*Originally reported as 300%; corrected to 200% on 28 March 2018.

in representing emissions and complying with air quality regulations, thus ensuring the continued use of fire in wildland management.

sos ayudarán a representar emisiones y a cumplir con las regulaciones sobre la calidad del aire, de manera de asegurar el uso continuado del fuego en el manejo de áreas naturales.

Keywords: coarse woody debris, duff, fire effects, fuel loading models, Fuels Characterization Classification System, LANDFIRE

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INTRODUCTION

The use of fire as a land management tool is widely recognized for its ecological benefits, and as a historic disturbance that has driven succession across many ecosystems (Agee 1996, Hardy and Arno 1996, Rothman 2005). While fire science and policy has advanced in the last 50 years to better allow for the use of fire in managing wildlands (van Wagtendonk 2007), increasingly stringent air quality regulations (US EPA 1990, Hardy *et al.* 2001, US EPA 2015,) and an increased awareness of the health impacts from smoke (Liu *et al.* 2015) can make the use of fire as a management tool difficult. In a recent United States survey, prescribed fire practitioners expressed that smoke and air quality issues are the third greatest impediment to prescribed burning, following low work capacity and unfavorable weather conditions (Melvin 2012). To continue using fire as a management tool, land managers must plan to meet management objectives, while also limiting the impact of smoke on public health and keeping smoke levels within regulatory thresholds (NWCG 2014). Such planning may often require the use of models to determine the quantity of emissions generated by fire; these models require many pieces of information, including expected fire size, fuel loading characteristics, and fuel consumption. Of these, fuel loading has been identified as the most critical step in obtaining accurate smoke predictions (Drury *et al.* 2014). Unfortunately, in many areas there may be little or no measured data on fuel loading; this creates a major difficulty in estimating fuel consumed and emissions produced.

To address the lack of fuel loading information in planning, geospatial Fire Effects Fuel Model (FEFM) layers developed by the Landscape Fire and Resource Management Planning Tools (LANDFIRE) program are often used. LANDFIRE data layers were developed for the contiguous United States, Alaska, and Hawaii to provide consistent geospatial data describing the vegetation type, structure, fuel loading, and disturbances, regardless of land ownership boundaries (Rollins 2009). LANDFIRE is principally intended to inform management and planning decisions made by land management agencies in the United States. It is also the only resource available that provides the geospatial information outlined above across as wide an area as the continental US. To populate models for smoke production, LANDFIRE FEFMs describe fuel loading for duff, litter, woody fuels from timelag size classes ranging from one hour (≤0.6 cm) to 1000 hours (≥7.62 cm), and live herb and shrub loading. Currently, there are two FEFM choices available through LANDFIRE: one represents fuel loading based on the Fuel Loading Model (FLM) categories developed by Lutes *et al.* (2009), and the other based on Fuels Characteristics Classification System (FCCS) categories developed by Ottmar *et al.* (2007). Both methods are derived from extensive measured datasets; however, FCCS is stratified to represent fuel loading by vegetation type (Ottmar *et al.* 2007), while FLM is stratified to represent fuel loadings by their potential fire effects (Lutes *et al.* 2009). The two LANDFIRE FEFMs are different not only in how they stratify fuels, but also in their reported fuel loadings.

There have been few studies that detail the differences between these two LANDFIRE FEFMs. One study evaluated their mapping performance across the western United States (Keane *et al.* 2013), and another compared their loadings and resulting emissions as part of a broader comparison of factors affecting smoke predictions in Washington, USA (Drury *et al.* 2014). When Keane *et al.* (2013) compared fuel loading and mapping accuracy of FCCS and FLM LANDFIRE layers throughout the western United States to data from the Forest Inventory and Analysis (FIA) program, they found poor correlations between FIA and LANDFIRE represented loadings, mainly due to the high variability in fuel loadings. Drury *et al.* (2014) compared FLM and FCCS FEFM data with other local datasets and found the landscape fuel loadings to range from 2.7 million Mg to 8.8 million Mg for their research area in Washington, USA, depending on which fuel loading dataset they used.

Studies such as these are extremely valuable for documenting the complexity and variation within fuel loading data, and identifying the importance and challenges of applying FEFM fuels data to model emissions. Our study builds on the few evaluations of LANDFIRE FEFMs to date by comparing FEFM surface fuel loading with measured fuel loadings, and using these loadings in two popular consumption and emissions models—the First Order Fire Effects Model (FOFEM) and Consume—to compare the resulting differences in fuel consumption and emissions production, while holding the site and environmental conditions constant. This provides insight into the degree of fuel loading differences possible at smaller scales relative to the national or sub-regional scales that LANDFIRE was developed to represent. Yet this 20 000 ha area is large enough to fall within the range of fire management units that land managers are tasked to manage (USDI NPS 2005, USDA FS 2008). We compared duff, litter, herb, shrub, and woody fuel loadings measured in forest inventory plots to those shown on both LANDFIRE Fuel Loading Models (LANDFIRE-FLM) and LANDFIRE Fuels Characterization Classification System (LANDFIRE-FCCS) maps. Subsequent differences in modeled consumption and emissions using FOFEM and Consume are reported.

METHODS

Study Area

To evaluate potential differences in predicted fuel loadings and fire effects, we selected a 20 000 ha study area centered on Moscow Mountain in Latah County, Idaho, USA (Figure 1). The mountain lies in the Palouse Range of northern Idaho, with elevations ranging from 770 m to 1516 m. Moscow Mountain is dominated by mixed conifer forest tree species including ponderosa pine (*Pinus ponderosa* C. Lawson var. *scopulorum* Engelm.), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco var. *glauca* [Beissn.] Franco), grand fir (*Abies grandis* [Douglas ex D. Don] Lindl.), western red cedar (*Thuja plicata* Donn ex D. Don), western hemlock (*Tsuga heterophylla* [Raf.] Sarg.), and western larch (*Larix occidentalis* Nutt.). Ponderosa pine and Douglas-fir habitat types occur on the xeric southern and western aspects, grand fir and cedar-hemlock habitat types occur on the mesic northern and eastern aspects (Cooper *et al.* 1991). The majority of the land is owned by private timber companies, private non-commercial landowners, and public land holdings. Recent disturbances recorded between 2003 and 2009 were predominantly the result of forest man-

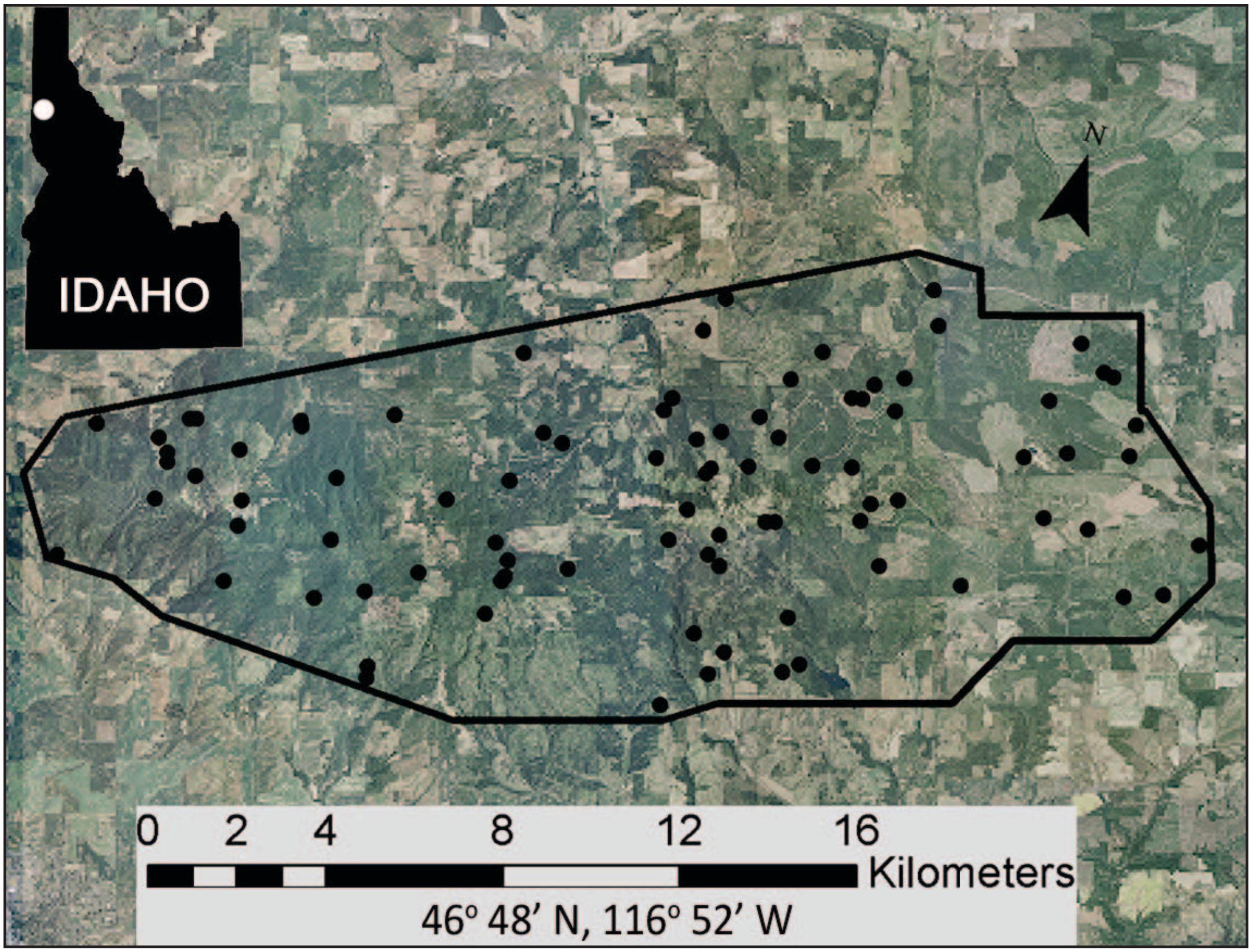

Figure 1. 2009 orthoimagery of the Moscow Mountain, Idaho, USA, study area (outlined) from the United States Geological Survey. The white dot in the inset is the study area. Plot locations are indicated with black dots.

agement practices including thinning, timber harvesting, and prescribed burning (Hudak *et al.* 2012). These activities have resulted in a forest with varying stand ages and structures that occur over a variety of biophysical settings (Falkowski *et al.* 2009, Martinuzzi *et al.* 2009, Hudak *et al.* 2012).

Plot Fuel Loadings

Plot data used in this study were collected in 2009, with information on plot placement and methodologies described in detail in Hudak *et al.* (2012). Following a stratified random sampling design of the study area, 0.04 ha fixed-radius field plots were placed randomly within strata based on elevation, slope, aspect, and percent forest cover. Plots that randomly fell within agricultural areas were subsequently excluded, leaving 87 forested plots for this analysis. Within each plot, duff; litter; coarse woody debris (CWD) in the ≥1000 hour (≥7.62 cm) size class; and fine woody debris in one hour (<0.635 cm), ten hour (0.635 cm to 2.54 cm), and 100 hour (2.54 cm to 7.62 cm) size classes were measured and loading was determined as described by Hudak *et al.* (2009), briefly summarized as follows: fuel loading was determined using two parallel 15 m Brown's transects (Brown 1974) centered 2.5 m upslope and downslope from plot center. On each transect, one hour and ten hour fuels were tallied over a 1.8 m segment, 100 h fuels over a 4.6 m segment, and 1000 h fuels over the entire length of both transects. Shrub and herbaceous cover were estimated ocularly and translated to loadings using equations from Brown (1981) and Smith and Brand (1983). Duff and litter depths were measured once at a set distance along each transect (Brown 1981), and loading was derived from relationships presented in Brown *et al.* (1982) with bulk densities from Woodall and Monleon (2008).

LANDFIRE Fire Effects Fuel Model Loadings

LANDFIRE FEFM map layers are available for both FCCS and FLM fuel classification systems. The FCCS system is composed of fuel loading data organized by vegetation type; each vegetation type is represented by loadings derived from field data collected from that vegetation type (Ottmar *et al.* 2007). FLM fuel loadings are the result of several field-collected datasets, which are grouped into statistically distinct groups based on fuel loading and modeled fire effects (i.e., emissions and soil heating; Lutes *et al.* 2009). In-depth comparisons of these approaches have been addressed by Keane (2013).

For this study, we compared LANDFIRE Refresh 2008 FEFMs to measured fuel loadings. LANDFIRE-FCCS and LANDFIRE-FLM layers were generated using different methodologies. LANDFIRE-FCCS layers were derived by matching FCCS fuelbeds to LANDFIRE vegetation communities (Comer *et al.* 2003) and vegetation type (McKenzie *et al.* 2012, LANDFIRE Team 2014*a*). LANDFIRE-FLMs were derived by a series of database queries that matched LANDFIRE data to the appropriate FLMs (Hann *et al.* 2012). More specifically, Forest Inventory and Analysis (FIA) data (Woudenberg *et al.* 2010) were keyed to FLMs (Lutes *et al.* 2009) and these FLMs were systematically matched to LANDFIRE vegetation types and cover. We should note that the scope of our study focuses on the surface fuel loadings represented in LANDFIRE map layers, not the FCCS and FLM fuel classification systems that the layers are intended to represent.

Generating Emissions within FOFEM and Consume

Consumption and emissions were generated using two common fire effects models: Consume version 4.2 (FERA Team 2014) and the Fire Order Fire Effects Model (FOFEM) version 6.0 (Lutes 2012). Consume calculates consumption and emissions based on empirical algorithms from many studies (Prichard *et al.* 2005). The FOFEM model generates consumption based on equations from the BURNUP model (Albini and Reinhardt 1997) and emission factors from Ward *et al.* (1993). Evaluating results in both models is important as FOFEM and Consume are both commonly used in fire management and are integrated into planning tools such as the Interagency Fuels Treatment Decision Support System (IFTDSS 2015). Consume is also integrated into the BlueSky Framework that is used for emissions calculations (AirFire 2015). For this study, we included the major compounds emitted by wildland fire that could be of concern for reasons of human health effects, regulatory impacts, or greenhouse gas emissions: carbon dioxide (CO_2), carbon monoxide (CO), methane (CH_4), and particulate matter 2.5 μm and 10 μm ($PM_{2.5}$ and PM_{10}). Nitrogen oxides (NO_x) and sulfur dioxide (SO_2) were also modeled using only FOFEM, and non-methane hydrocarbons (NMHC) were modeled using only Consume as these options are specific to each model. To parameterize these models we used the values in Table 1 to simulate summer fire conditions under which past fires in the region have ignited (McDonough 2003).

Table 1. Environmental parameters used to populate FOFEM and Consume under default 'Low' moisture conditions to simulate an early summer fire.

Parameter	Input
Moistures	
Duff	40 %
10 hour	10 %
CWD	15 %
Soil	10 %
Fuel type	Natural
Region	Interior West
Season	Summer

Statistical Comparison of Fuel Loadings

All analyses were conducted using R Statistical Software (R-Project 2013). We initially tested fuel loading differences using Bartlett's test for equal variance (Bartlett 1937). This indicated that the data did not meet the assumption of homoscedasticity required for parametric regression analysis. Therefore, we used non-parametric statistical methods. Analysis of variance was chosen and performed using the Anova test from the "car" package (Fox *et al.* 2014) as this version implemented the test using heteroscedasticity-corrected coefficient covariance matrices. If a significant difference was detected, further analysis was conducted with the Dunnett-Tukey-Kramer pairwise multiple comparison test adjusted for unequal variances and unequal sample sizes (Dunnet 1980) using the DTK package (Lau 2013) at the alpha = 0.05 significance level. This method was used to compare fuel loadings, consumption, and emissions. To examine the influence of different fuels on the total emissions produced, we used Hoffman and Gardner's Importance Index, a ratio of variances between total emissions generated and each individual fuel component (Hoffman and Gardner 1983 , Hamby 1994). Values close to one indicate higher significance than values closer to zero.

RESULTS

Fuel Loadings

In comparing LANDFIRE fuel loadings with measured fuel loadings, all fuel components differed at the alpha = 0.05 significance level with the exception of shrubs (Table 2, Figure 2). LANDFIRE-FCCS loadings over-represented duff and herbs; under-represented litter, 10 h, and 100 h fuels; and did not differ for 1 h fuels or CWD. LANDFIRE-FLM under-represented duff, litter, fine (1 h, 10 h, and 100 h), and CWD fuel loadings; over-represented herb loadings; but duff loading did not

Table 2. Mean fuel loads (Mg ha^{-1} and SD in parentheses) on measured plots and as modeled by LANDFIRE-FCCS and LANDFIRE-FLM. Asterisks indicate statistically significant difference relative to measured loading data at the $P < 0.05$ significance level.

	Mean plot loading		
Fuel	**Measured**	**LANDFIRE-FCCS**	**LANDFIRE-FLM**
Duff	10.55 (10.20)	31.89 (17.80)*	7.76 (12.19)
Litter	5.86 (4.13)	4.199 (1.37)*	3.66 (3.40)*
1 h	0.65 (0.47)	0.81 (0.46)	0.50 (0.32)*
10 h	2.57 (2.19)	1.85 (1.11)*	1.65 (1.13)*
100 h	4.98 (5.20)	2.47 (3.41)*	1.94 (1.66)*
CWD	20.087 (23.33)	18.45 (16.38)	2.75 (4.04)*
Herb	0.46 (0.28)	0.68 (0.76)*	0.73 (0.76)*
Shrub	1.179 (3.08)	1.36 (1.51)	3.65 (10.60)
Total fuel	46.26 (32.49)	61.63 (34.81)*	22.64 (21.16)*

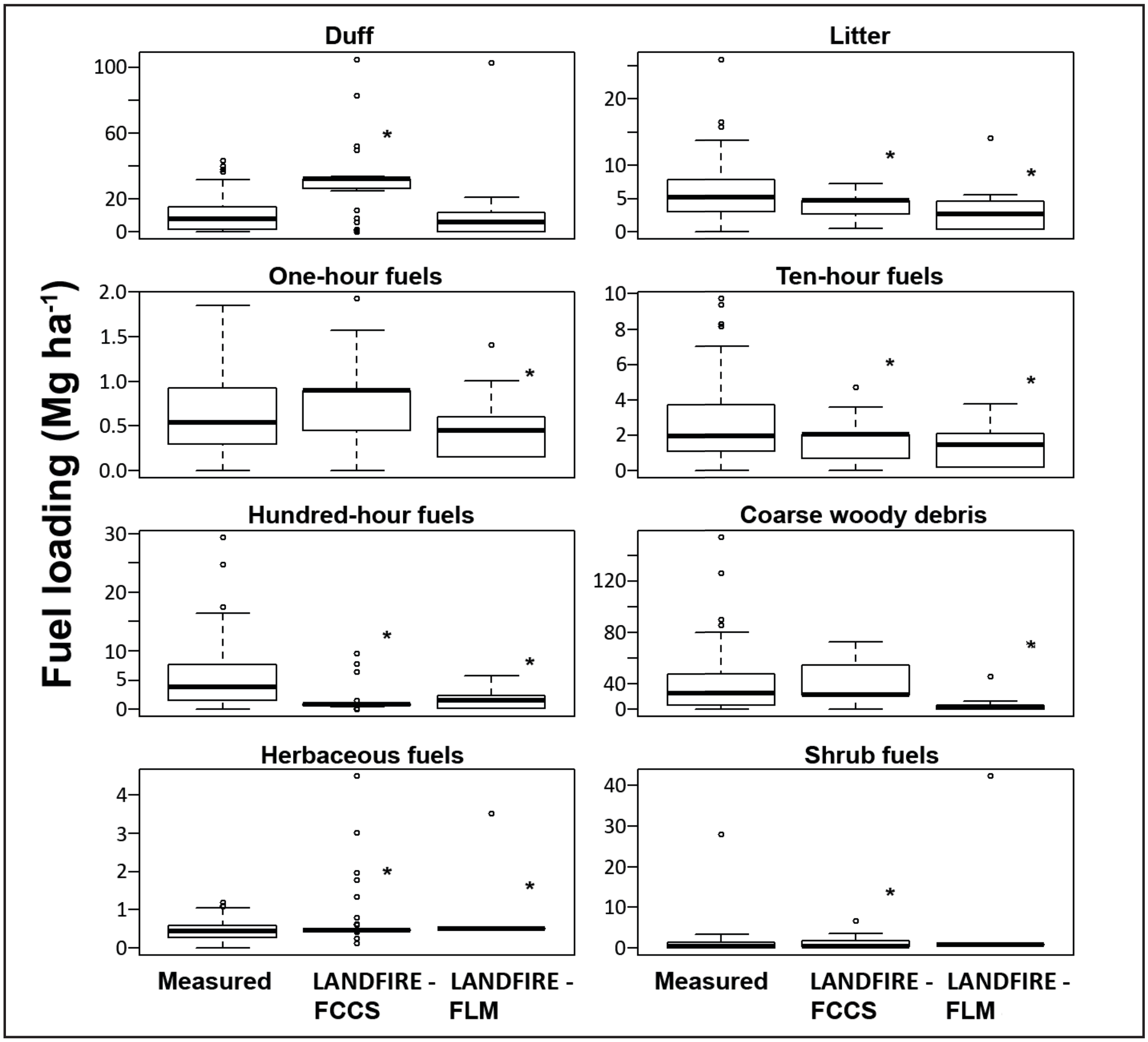

Figure 2. Differences in fuel loading for measured plots, LANDFIRE-FLM, and LANDFIRE-FCCS products. Bold horizontal lines indicate median values, asterisks represent significant differences relative to measured loadings. Circles indicate outliers, and whiskers indicate the region between the first and third quartiles.

differ. Duff and CWD fuel components showed the most pronounced difference in loadings, with LANDFIRE-FCCS duff loadings 200 % higher than measured loadings, and 300 % higher than LANDFIRE-FLM loadings. LANDFIRE-FLM CWD loading was 9 times lower than measured or LANDFIRE-FCCS loadings. When comparing LANDFIRE FEFMs to each other, only duff, CWD, and 1 h fuel loadings differed, with LANDFIRE-FCCS having the greater loadings.

Modeled Consumption and Emissions in FOFEM

The statistical relationships for fuel consumption mirrored those for fuel loading (Figures 2 and 3, Tables 2 and 3). Relative to measured consumption, the mean total surface consumption from LANDFIRE-FCCS was 23 % higher, and LANDFIRE-FLM was 51 % lower. It is apparent that the high LANDFIRE-FCCS duff loading led to the higher

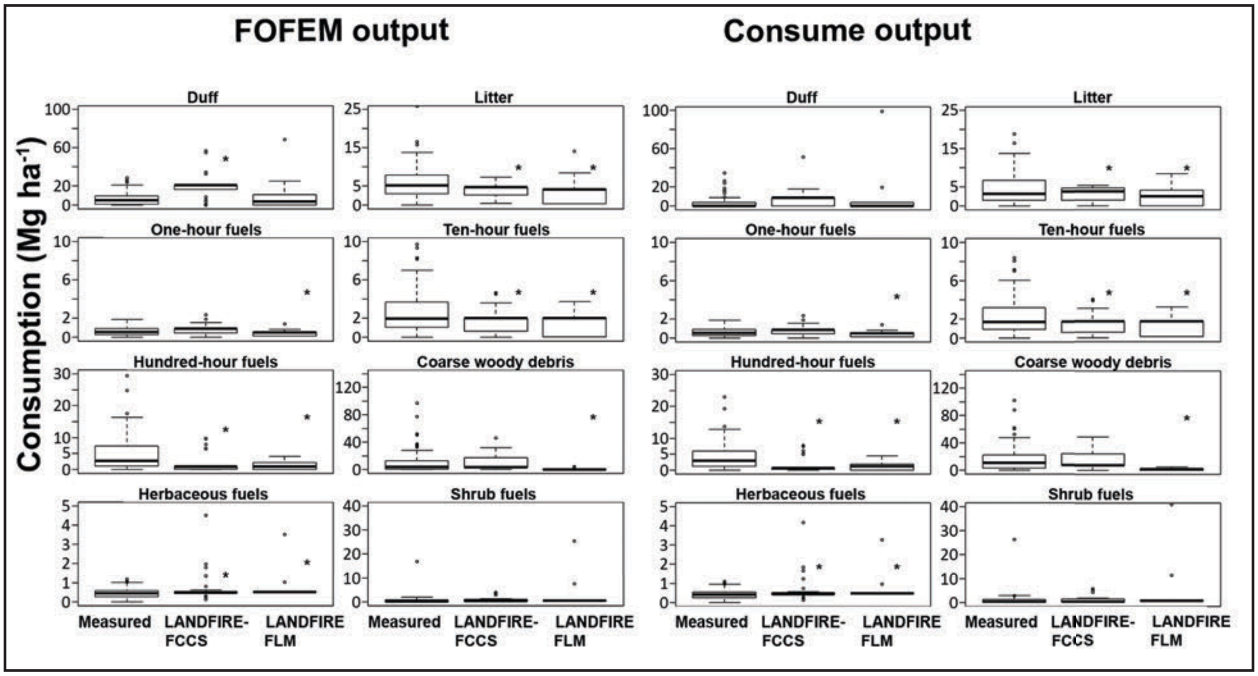

Figure 3. Differences in modeled consumption for measured, LANDFIRE-FLM, and LANDFIRE-FCCS fuel loadings. Bold horizontal lines indicate median values, asterisks represent significant differences relative to results derived from measured loadings.

Table 3. Mean fuel consumption (Mg ha^{-1} with SD in parentheses) under fixed environmental conditions or measured plots, and as modeled by LANDFIRE-FCCS and LANDFIRE-FLM, using FOFEM and Consume. Asterisks indicate statistically significant difference relative to estimates based on measured loading at the $P < 0.05$ significance level.

	Mean plot consumption in FOFEM			**Mean plot consumption in Consume**		
Fuel	**Measured**	**LANDFIRE-FCCS**	**LANDFIRE-FLM**	**Measured**	**LANDFIRE-FCCS**	**LANDFIRE-FLM**
Duff	6.98 (6.84)	20.64 (11.66)*	5.35 (8.38)	3.36 (6.48)	5.67 (8.48)	2.31(10.83)
Litter	5.83 (4.16)	4.14 (1.21)*	3.68 (3.23)*	4.45 (3.98)	3.11 (1.66)*	2.32 (1.71)*
1 h	0.65 (0.49)	0.76 (0.44)	0.49 (0.29)*	0.65 (0.49)	0.75 (0.44)	0.49 (0.29)*
10 h	2.58 (2.19)	1.77 (1.17)*	1.62 (1.19)*	2.24 (1.89)	1.53 (1.00)*	1.44 (0.98)*
100 h	4.56 (5.29)	2.32 (3.46)*	1.47 (1.25)*	3.93 (4.06)	1.84 (2.70)*	1.55 (1.30)*
CWD	10.59 (16.88)	8.60 (10.84)	0.53 (0.67)*	17.06 (19.41)	14.20 (14.17)	1.84 (1.56)*
Herb	0.45 (0.28)	0.66 (0.61)*	0.70 (0.70)*	0.42 (0.26)	0.61 (0.57)*	0.64 (0.65)*
Shrub	0.70 (1.85)	0.99 (1.20)	1.99 (5.87)	1.02 (2.88)	1.41 (1.74)	3.10 (9.48)
Total fuel	32.35 (25.46)	39.89 (23.40)	15.83 (14.30)	33.12 (25.90)	29.13 (18.61)	13.69 (16.08)*

overall consumption, and that the low CWD loading in the LANDFIRE-FLM contributed to less consumption. This in turn had a direct effect on the emissions modeled. All modeled emissions, with the exception of NO_x, were significantly higher when modeled using LANDFIRE-FCCS loadings, and lower when using LANDFIRE-FLM loadings, while emissions derived from measured fuel loadings fell in between (Table 4, Figure 4).

The relative importance of CWD and duff to total emissions was reaffirmed and quantified using the importance index (Table 5). Duff and CWD stood out as the primary con-

Table 4. Mean modeled emissions (Mg ha^{-1} with SD in parentheses) calculated using FOFEM and Consume for measured plots, LANDFIRE-FCCS, and LANDFIRE-FLM. Asterisks indicate statistically significant difference relative to estimates based on measured loading at the P < 0.05 significance level.

	Plot-level values FOFEM			Plot-level values Consume		
Effect	Measured	LANDFIRE-FCCS	LANDFIRE-FLM	Measured	LANDFIRE-FCCS	LANDFIRE-FLM
CH_4	0.32 (0.30)	0.46 (0.30)*	0.13 (0.14)*	0.19 (0.18)	0.19 (0.14)	0.06 (0.11)*
CO	6.83 (6.56)	10.00 (6.64)*	2.67 (3.00)*	3.67 (3.38)	3.65 (2.65)	1.20 (2.02)*
CO_2	45.20 (34.09)	52.81 (29.57)	23.39 (21.90)*	51.90 (39.72)	44.82 (28.11)	22.08 (25.22)*
$PM_{2.5}$	0.53 (0.50)	0.76 (0.50)*	0.22 (0.23)*	0.29 (0.25)	0.27 (0.19)	0.11 (0.15)*
PM_{10}	0.63 (0.59)	0.90 (0.59)*	0.25 (0.27)*	0.33 (0.28)	0.30 (0.21)	0.12 (0.16)*
SO_2	0.03 (0.03)	0.04 (0.02)	0.02 (0.01)*			
NO_x	0.03 (0.03)	0.02 (0.01)*	0.02 (0.03)			
NMHC				0.16 (0.14)	0.15 (0.11)	0.05 (0.08)*

tributors to total emissions in all cases, with the exception of LANDFIRE-FLM data, in which duff and shrub loadings were the primary contributors. Although shrub loadings did not statistically different in our study, shrub loadings tended to be higher in LANDFIRE-FLMs compared to other sources.

Modeled Consumption and Emissions in Consume

With the exception of duff, the relationships between fuel loading and modeled consumption when using Consume remained the same as with FOFEM; modeled duff consumption was much lower when using Consume (Table 3). Duff consumption using LANDFIRE-FCCS loadings did not significantly differ from consumption generated from measured loadings. Because of this, the overall modeled fuel consumption from LANDFIRE-FCCS did not significantly differ from the fuel consumption generated by measured loadings. However, the modeled consumption from LANDFIRE-FLM was significantly lower than consumption from measured loadings, with mean total surface fuel consumption 59% less than that derived from measured fuel loadings.

The importance index for the consumption and total emissions in Consume was similar to the FOFEM emissions importance index (Table 5). Duff consumption was still an important component with regard to emissions production, even though it did not statistically differ between measured and modeled fuel datasets when modeled with Consume. When emissions were evaluated, the LANDFIRE-FLM generated emissions were significantly lower than those generated using measured fuel loadings. Emissions generated using LANDFIRE-FCCS and measured fuel loadings did not differ from each other (Table 4).

DISCUSSION

Measured Versus Modeled Fuel Loading

Duff and CWD led to the most significant differences in modeled consumption and emissions. LANDFIRE-FLMs contained higher shrub loadings, although this number did not result in a statistically significant difference, nor was it great enough to influence the total surface fuel loading when consumption and emissions were modeled. While the cause for these LANDFIRE-FLM shrub values to be so much higher is not known, the FLM system itself was developed with very little available shrub data (Lutes *et al.* 2009). This likely influenced which FLMs were available to assign

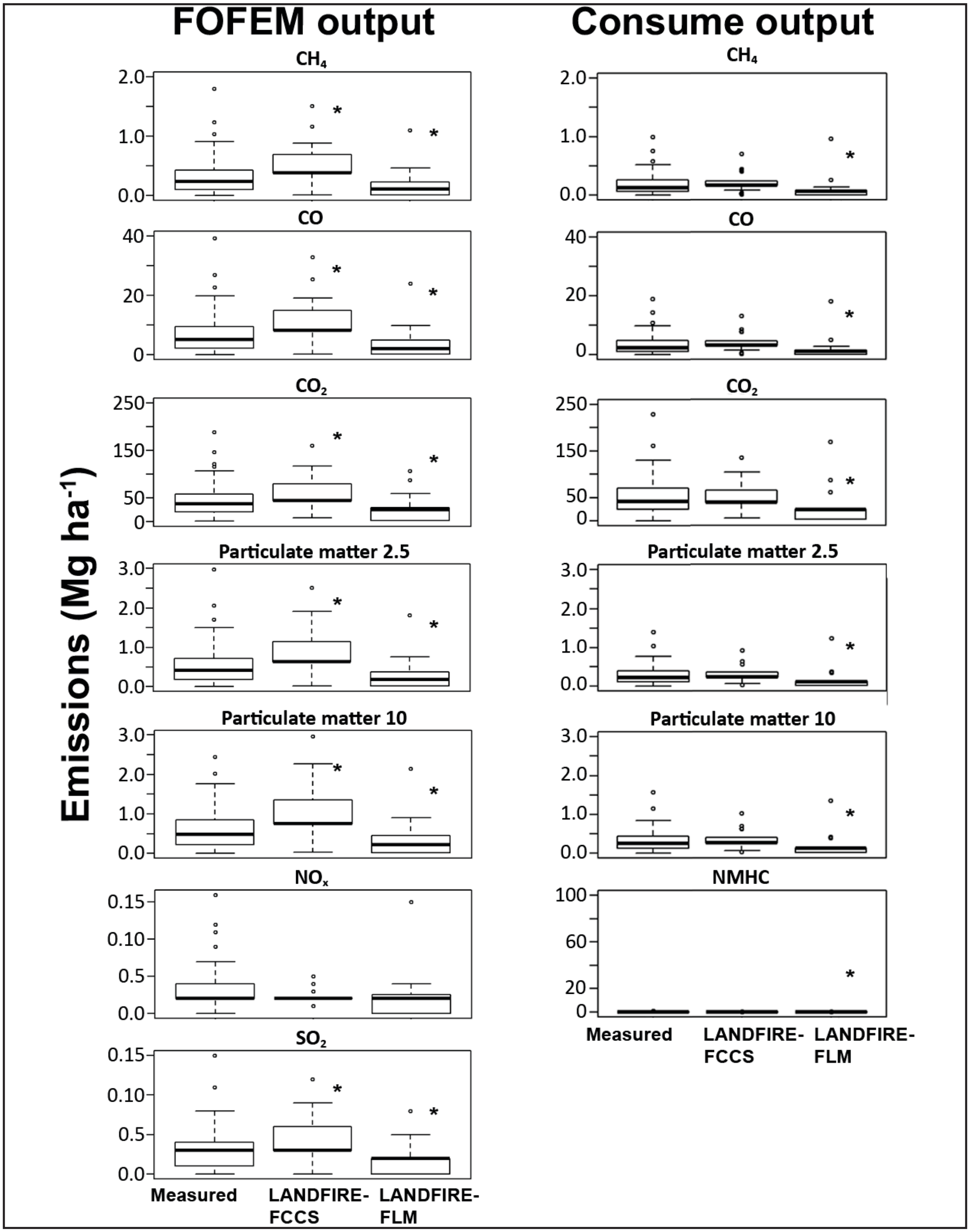

Figure 4. Differences in modeled emissions for measured, LANDFIRE-FLM, and LANDFIRE-FCCS fuel loadings. Bold horizontal lines indicate median values, asterisks represent significant differences relative to results derived from measured loadings.

Table 5. Hoffman and Gardner Importance Index for each FEFM and each fuel type shows that the fuel of relative importance to the total emissions produced varied depending by FEFM. Emissions from measured data and FCCS fuelbeds were most influenced by CWD and duff, and FLM by duff and shrubs, respectively. Highest values are indicated in bold.

	Importance Index FOFEM			**Importance Index Consume**		
Fuel	**Measured**	**LANDFIRE-FCCS**	**LANDFIRE-FLM**	**Measured**	**LANDFIRE-FCCS**	**LANDFIRE-FLM**
Duff	**0.012**	**0.043**	**0.053**	**0.022**	**0.073**	**0.0156**
Litter	0.002	0.000	0.004	0.008	0.003	0.004
1 h	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
10 h	0.001	<0.001	<0.001	0.002	0.001	0.001
100 h	0.003	0.002	0.001	0.009	0.007	0.002
CWD	**0.063**	**0.048**	0.001	**0.196**	**0.204**	0.003
Herb	<0.001	<0.001	<0.001	<0.001	<0.001	0.001
Shrub	0.001	0.001	**0.032**	0.004	0.003	**0.119**

to LANDFIRE maps when the LANDFIRE-FLM was created. Because the scope of this study focused on a mixed conifer ecosystem, our shrub data were somewhat limited and probably provided little insight in shrub-dominated ecosystems where shrubs are a large fuel component. Further investigation of these LANDFIRE layers in shrub-dominated systems and further fuel loading data from shrub ecosystems would be beneficial to further refining FLMs and the resulting LANDFIRE-FLM data for shrub ecosystems.

When comparing each fuel component for measured and LANDFIRE-represented loadings with those of other mixed conifer systems, all three of our fuel loading sets fell within the ranges observed by other researchers (Table 6). Focusing on duff and coarse woody debris, we found LANDFIRE-FCCS mean duff loading exceeded our measured values, but more closely resembled the ranges found in other mixed conifer forests. Thus, it is possible that our study area may have had less duff loading than other mixed conifer forests. When evaluating mean CWD loadings, we found the wide range noted in other studies, from 0.5 Mg ha^{-1} to 37 Mg ha^{-1}; LANDFIRE-FLM mean CWD loadings were at the low end of this range averaging 0.53 Mg ha^{-1}, while our measured data and LANDFIRE-FCCS were 10.6 Mg ha^{-1} and 8.6 Mg ha^{-1}, respectively.

Our results support a broader evaluation of the importance of various steps in the emissions modeling process in which Drury *et al.* (2014) compared LANDFIRE-represented loadings to a custom loading map based on measured data. Like our results, their duff loading was higher for LANDFIRE-FCCS relative to loadings represented using measured data, while in our study the LANDFIRE-FCCS total loadings were greater. Drury *et al.* found a wide range in possible fuel loadings depending upon the method chosen, as did we, and concluded that custom fuel loading layers derived from measured data produced the most reliable emissions estimates. Of the two LANDFIRE fuel layers, Drury *et al.* found the LANDFIRE-FCCS layer produced results closer to the custom loading layers. We found this to be true in our study when modeling emissions with Consume, but still found LANDFIRE-FCCS to produce higher emissions values when modeled using FOFEM.

In another study that compared classification, mapping accuracy, and fuel loadings of LANDFIRE-FCCS and LANDFIRE-FLM to Forest Inventory and Analysis (FIA) plot data across the western US, Keane *et al.* (2013) found poor performance in both LAND-

Table 6. Fuel loading for other mixed conifer forests in the western United States compared with mean fuel loading from this study (in Mg ha^{-1}). Standard deviations, when present, are indicated in parentheses. Values from this study are indicated in bold in the last three rows.

Source	Duff	Litter	1 hour	10 hour	100 hour	1000 h sound	1000 h rotten	Herb	Location	Elevation (m)
Hille and Stephens 2005	17.8 (3.6)	17.8 (3.6)	2.0 (0.2)	6.3 (0.7)	5.8 (1.6)	6.0 (3.3)	15.8 (4.3)	-	North-central Sierra Nevada, California	1200 to 1500
Sikkink and Keane 2008			0.019	1.649	0.513	0.683 (sound and rotten)		0.545	NW Rockies*	730 to 2130
Sikkink and Keane 2008			0.012	1.297	0.671	0.549 (sound and rotten)		0.659	NW Rockies	730 to 2130
Sikkink and Keane 2008			0.107	0.709	1.105	0.937 (sound and rotten)		0.581	NW Rockies	730 to 2130
Sikkink and Keane 2008			1.155	4.390	5.682	0.600 (sound and rotten)		0.615	NW Rockies	730 to 2130
Sikkink and Keane 2008			2.586	5.567	7.849	0.863 (sound and rotten)		0.636	NW Rockies	730 to 2130
Youngblood *et al*. 2008	22.27 (7.52)	5.9 (0.97)	0.94 (0.2)	1.56 (0.33)	4.16 (0.59)	9.63 (3.46)	7.31 (2.5)		Blue Mountain Region, Oregon	1040 to 1480
Youngblood *et al*. 2008	25.48 (7.03)	3.74 (0.44)	0.37 (0.12)	0.64 (0.21)	3.04 (0.67)	8.88 (4.26)	7.97 (0.61)		Blue Mountain Region, Oregon	1040 to 1480
Raymond and Peterson 2005			1.2	4.1	4.8			1.2	Oregon Coast Range	670 to 850
Raymond and Peterson 2005			4.4	6.8	8.7			1.2	Oregon Coast Range	670 to 850
Kobziar *et al.* 2006			1.25 (0.87)	4.53 (3.23)	9.93 (8.18)	7.52 (16.82)	14.18 (23.31)		North-central Sierra Nevada, California	1100 to 1410
Kobziar *et al.* 2006			1.13 (1.04)	5.53 (4.97)	6.17 (7.15)	7.91 (17.04)	29.02 (40.86)		North-central Sierra Nevada, California	1100 to 1410
Kobziar *et al.* 2006			0.9 (0.71)	2.9 (2.3)	4.25 (4.12)	2.57 (5.36)	26.62 (65.62)		North-central Sierra Nevada, California	1100 to 1410
Reinhard *et al.*1991	52 (1.3)		Other values are logging slash, not natural fuels						NW Rockies	900 to 1200
Reinhard *et al*. 1991	48.4 (1.6)		Other values are logging slash, not natural fuels						NW Rockies	900 to 1200
Measured	**10.55 (10.20)**	**5.86 (4.13)**	**0.65 (0.47)**	**2.57 (2.19)**	**4.98 (5.20)**	**20.09 (23.33)** (sound and rotten)		**0.46 (0.28)**	**NW Rockies**	**770 to 1516**
LANDFIRE-FLM	**7.76 (12.19)**	**3.66 (3.40)**	**0.50 (0.32)**	**1.65 (1.13)**	**2.47 (3.41)**	**2.75 (4.04)** (sound and rotten)		**0.73 (0.76)**	**NW Rockies**	**770 to 1516**
LANDFIRE-FCCS	**31.89 (17.8)**	**4.19 (1.37)**	**0.81 (0.46)**	**1.85 (1.11)**	**1.94 (1.66)**	**18.45 (16.38)** (sound and rotten)		**0.68 (0.76)**	**NW Rockies**	**770 to 1516**

* NW Rockies includes parts of Idaho and Montana, USA.

FIRE-represented FEFMs. LANDFIRE-FLM tended to under-predict loadings while LANDFIRE-FCCS tended toward over prediction. However, LANDFIRE-FLM loadings had lower root mean squared errors (Keane *et al.* 2013). Our findings here support the work of Keane *et al.* (2013) and Drury *et al.* (2014) in describing the tendency of LANDFIRE-FCCS to have higher loadings relative to LANDFIRE-FLMs.

Modeled Consumption and Emissions Using FOFEM

Relative differences in consumption values when modeled with FOFEM mirrored those of the loading values. High LANDFIRE-FCCS duff and low LANDFIRE-FLM CWD loading and consumption contributed to the total modeled emissions being highest when using LANDFIRE-FCCS inputs, and lowest when using LANDFIRE-FLM inputs. In examining the fuel loading data (Table 2), there is high variance in all fuel loading categories. This supports the work by Keane *et al.* (2013), who noted the high variance inherent in all categories of fuel loading, and the difficulties caused by spatial variation when trying to represent fuel loadings across large landscapes. Consumption followed the pattern of the total fuel loading values for the landscape, with LANDFIRE-FCCS being the highest, FLM being the lowest, and measured values in the middle. This in turn produced higher emissions from LANDFIRE-FCCS and lower emissions from LANDFIRE-FLM, highlighting the differences in emissions outcomes depending upon the choices made to represent fuel loadings.

Modeled Consumption and Emissions Using Consume

In comparing consumption and emissions from Consume, the choice of model has an effect on emissions generated. In this study, there were similar trends in modeled consumption with both fire effects models, but the lower duff consumption in Consume, relative to FOFEM, led to emissions outputs in which LANDFIRE-FCCS derived emissions did not differ from those derived from measured loadings. This difference in duff consumption is due to the fact that Consume and FOFEM calculate the consumption of duff using different equations, derived from different data sets (Reinhardt 2003, Prichard *et al*. 2005).

Research Implications

In modeling emissions, fuel loadings have been identified as the most crucial variables (Drury *et al.* 2014), yet they represent one of the greatest uncertainties in modeling emissions (French *et al.* 2011). In a detailed discussion on the topic, Keane *et al.* (2013) identified several factors creating difficulties in quantifying fuel loadings. These include lack of data to develop thorough loading maps; the use of classification systems that were developed from discrete plot locations but then applied to large, national-scale areas; and the inherent difficulty of classifying fuels into categories such as hourly size classes and duff, when each of these size classes may have different degrees of variation at different spatial scales (Keane *et al.* 2012, 2013). If existing fuel loading classification maps are to be improved, more data are necessary. The results of our study indicate that data on CWD and duff should be priorities, due to the relative importance of these fuels to overall emissions in mixed conifer forests (Table 5). While consumption didn't statistically differ for the specific case of shrubs, shrub loading accounted for a great deal of variability in emissions from LANDFIRE-FLMs (Table 5), a classification that was developed with little available shrub data (Lutes *et al.* 2009). For the case of LANDFIRE-FLMs, having additional data on shrub loadings would be beneficial.

Despite being represented at a 30 m resolution, LANDFIRE data layers are intended to be used at larger, sub-regional to national scales (LANDFIRE Team 2014*b*). Data from fuel loading maps may work for finer scales; however, there will likely be greater need to supplement that data with local knowledge. Based on our findings in a 20 000 ha area, using measured data, especially for duff and CWD loadings, is preferable relative to unaltered LANDFIRE layers. However, we understand that measured data are often unavailable, may be incomplete, or limited in availability.

Management Implications

If monitoring resources are available, emission estimates will be improved by having more information on duff loading, as differences in duff loading lead to the greatest differences in emissions, followed by CWD. For coarse woody debris, the planar intercept sampling methods have been most commonly used in forests such as in this study, although the Photoload (Keane and Dickinson 2007) method has also performed well (Sikkink and Keane 2008). Duff sampling is often performed via sampling points along a planar intercept to gather both loading and depth (Brown 1974). The fuel photoseries guides available for many ecosystems provide estimates of duff loading (Ottmar *et al.* 2003), but there are few studies comparing their performance relative to the traditional method. If measured data are not available, one could model with both the LANDFIRE-FLM and LANDFIRE-FCCS derived fuel loadings, and then average the two sets of results.

The use of systems such as the Wildland Fire Decision Support System (WFDSS) and the Interagency Fuels Treatment Decision Support System (IFTDSS) also hold potential for obtaining measured fuel loading information (IFTDSS 2015, WFDSS 2015). These systems provide online access to several models to represent fire behavior and effects (including emissions), but they also provide an easy platform from which data can be shared from user to user. In the future, it would be ideal to see a searchable database of user-provided fuel loadings within these decision support systems, similar to the searchable data available through the Fire Research and Management Exchange System (FRAMES) Resource Catalog (FRAMES 2015).

This study has characterized the potential differences in LANDFIRE-represented fuel loadings in a mixed conifer case study area, and their impact on the emissions modeling. While using measured data provides the most reliable outcome, either by itself or to help supplement the LANDFIRE data, this is not always possible. Web-based systems can aid in finding and sharing data; however, a search for the keywords "duff" and "coarse woody debris" in FRAMES returned 34 and 3 results, respectively. While online systems can be powerful sources of information, there is clearly a need for additional data with which tools such as the LANDFIRE map layers could be strengthened. In the interim, information on the relative differences in fuel loadings from LANDFIRE-represented data may be useful to managers who are tasked with quantifying emissions for fire management planning. Using all of these resources will aid in generating more accurate emissions estimates in a climate where regulatory pressure and the need to accurately represent potential emissions from fire are increasing.

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SUPPLEMENTARY MATERIALS

www.sciencemag.org/content/354/6318/1419/suppl/DC1
Materials and Methods
Figs. S1 to S5
Tables S1 and S2
References (*22*–*29*)

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CONSERVATION

A global map of roadless areas and their conservation status

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Roads fragment landscapes and trigger human colonization and degradation of ecosystems, to the detriment of biodiversity and ecosystem functions. The planet's remaining large and ecologically important tracts of roadless areas sustain key refugia for biodiversity and provide globally relevant ecosystem services. Applying a 1-kilometer buffer to all roads, we present a global map of roadless areas and an assessment of their status, quality, and extent of coverage by protected areas. About 80% of Earth's terrestrial surface remains roadless, but this area is fragmented into ~600,000 patches, more than half of which are <1 square kilometer and only 7% of which are larger than 100 square kilometers. Global protection of ecologically valuable roadless areas is inadequate. International recognition and protection of roadless areas is urgently needed to halt their continued loss.

The impact of roads on the surrounding landscape extends far beyond the roads themselves. Direct and indirect environmental impacts include deforestation and fragmentation, chemical pollution, noise disturbance, increased wildlife mortality due to car collisions, changes in population gene flow, and facilitation of biological invasions (*1*–*4*). In addition, roads facilitate "contagious development," in that they provide access to previously remote areas, thus opening them up for more roads, land-use changes, associated resource extraction, and human-caused disturbances of biodiversity (*3*, *4*). With the length of roads projected to increase by >60% globally from 2010 to 2050 (*5*), there is an urgent need for the development of a comprehensive global strategy for road development if continued biodiversity loss is to be abated (*6*). To help mitigate the detrimental effects of roads, their construction should be concentrated as much as possible in areas of relatively low "environmental values" (*7*). Likewise, prioritizing the protection of remaining roadless areas that are regarded as important for biodiversity and ecosystem functionality requires an assessment of their extent, distribution, and ecological quality.

Such global assessments have been constrained by deficient spatial data on global road networks. Importantly, recent publicly available and rapidly improving data sets have been generated by crowd-sourcing and citizen science. We demonstrate their potential through OpenStreetMap, a project with an open-access, grassroots approach to mapping and updating free global geographic data, with a focus on roads. The available global road data sets, OpenStreetMap and gROADS, vary in length, location, and type of roads; the former is the data set with the largest length of roads (36 million km in 2013) that is not restricted to specific road types (table S1). OpenStreetMap is more complete than gROADS, which has been used for other global assessments (*7*), but in certain regions, it contains fewer roads than sub-global or local road data sets [see the example of Center for International Forestry Research data for Sabah, Malaysia (*8*); table S1]. Given the pace of road construction and data limitations, our results overestimate the actual extent of global roadless areas.

The spatial extent of road impacts is specific to the impact in question and to each particular road and its traffic volume, as well as to taxa, habitat, landscape, and terrain features. Moreover, for a given road impact, its area of ecological influence is asymmetrical along the road and can vary among seasons, between night and day, according to weather conditions, and over longer time periods. We conducted a comprehensive literature review of 282 publications dealing with "road-effects zones" or including the distance to roads as a covariate, of which 58 assessed the spatial influence of the road (table S2). All investigated road impacts were documented within a distance of

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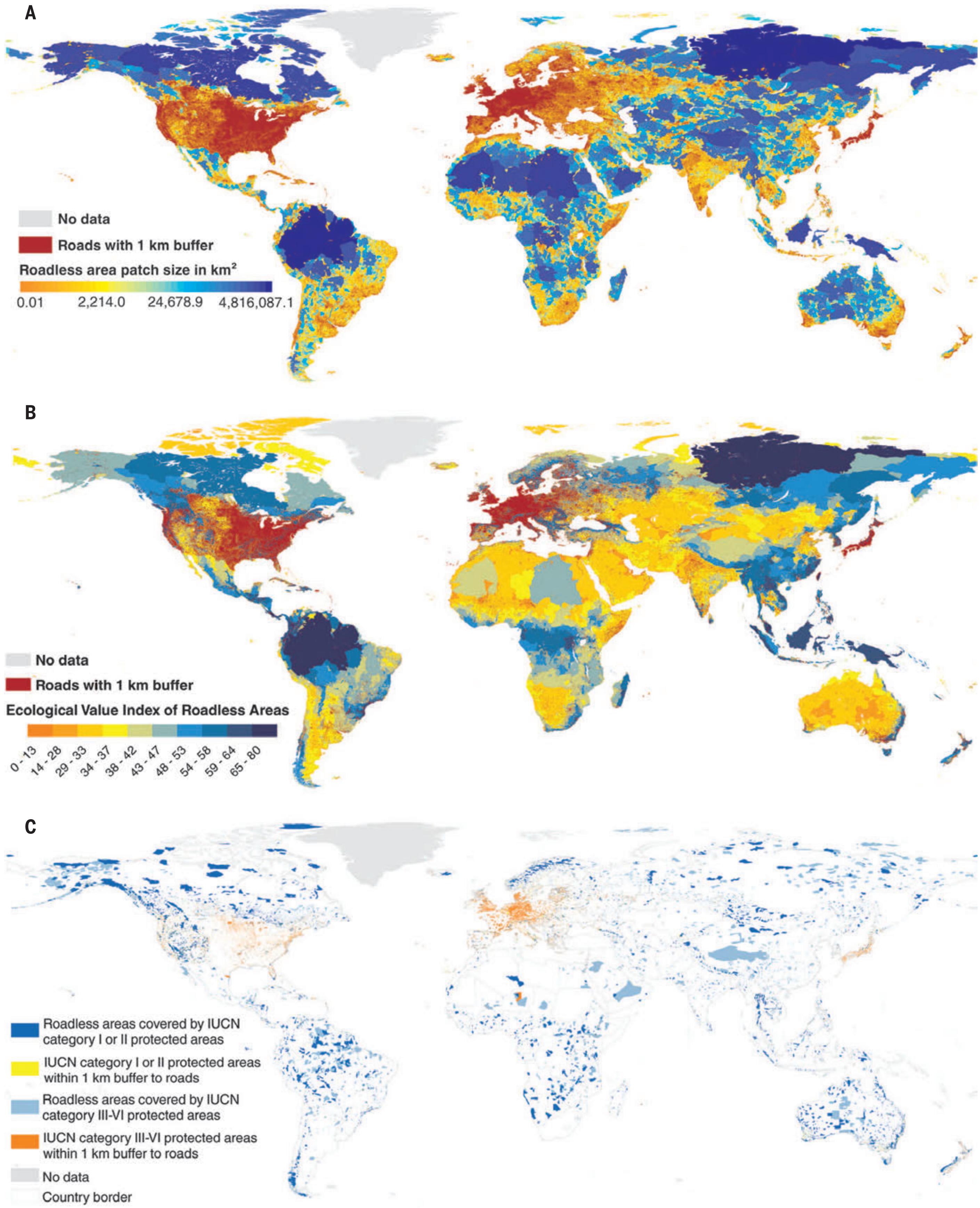

Fig. 1. The global distribution of roadless areas, based on a 1-km buffer around all roads. The distribution is depicted according to (**A**) size classes, (**B**) the ecological value index of roadless areas (EVIRA; based on patch size, connectivity, and ecosystem functionality), and (**C**) representation in protected areas (*8*).

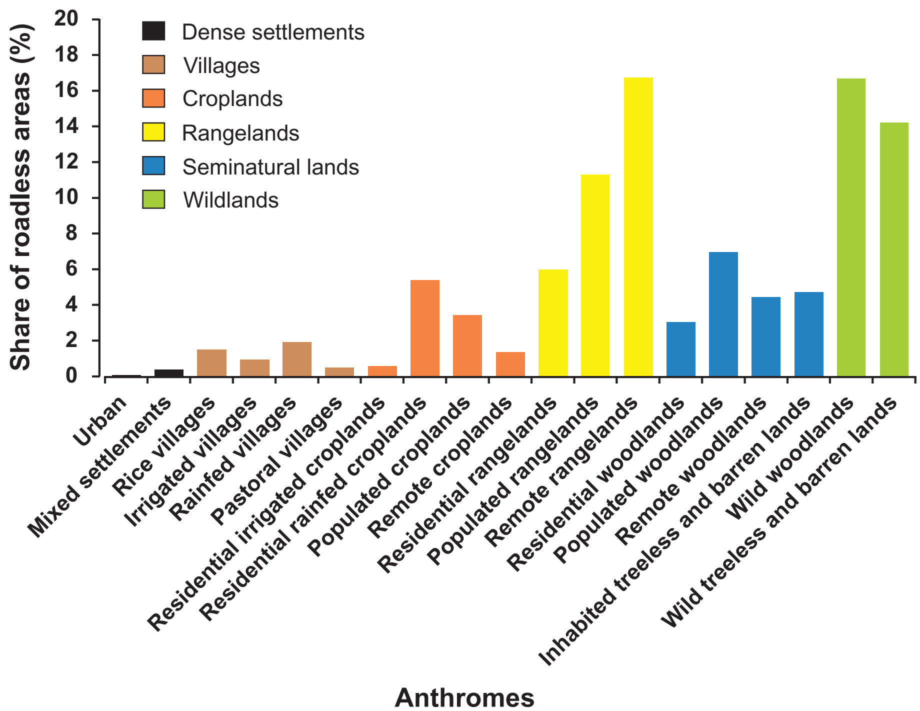

Fig. 2. Extent of roadless areas (1-km buffer) across anthromes. The majority of the world's roadless areas are in remote and unmodified landscapes, but they also occur in anthropogenically modified landscapes. The so-called anthromes were mapped according to (*10*).

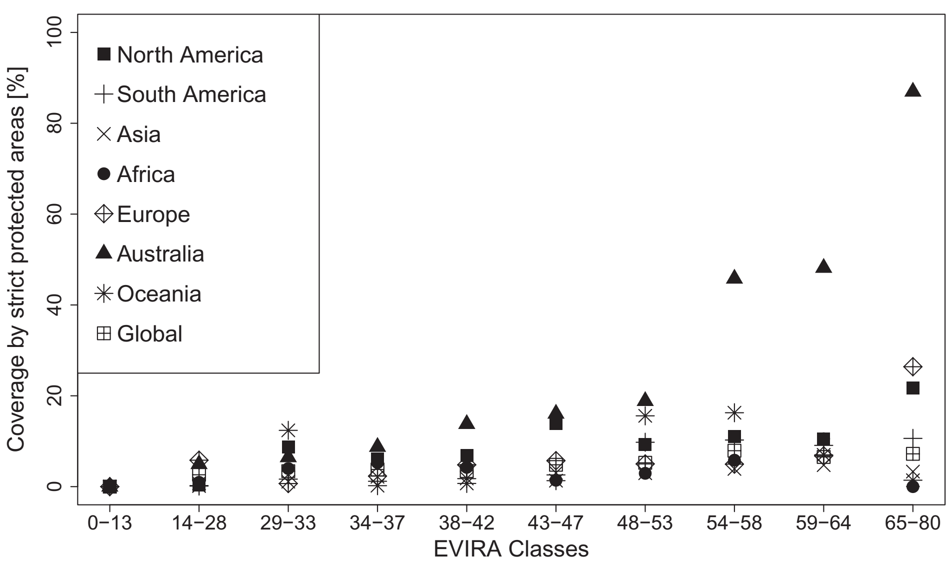

Fig. 3. Coverage of roadless areas by strictly protected areas (IUCN categories I and II) compared with global and continental EVIRA values. If priority were given to protecting roadless areas with high ecological functionality, we should see a positive correlation, with higher coverage associated with higher EVIRA values.

1 km from the road, 39% reached out to 2 km from the road, and only 14% extended out to 5 km from the road (fig. S1). Because the 1-km buffer along each side of the road represents the zone with the highest level and variety of road impacts, we defined roadless areas as those land units that are at least 1 km away from all roads and, therefore, less influenced by road effects. We compared results from using this criterion with the outcomes from using an alternative 5-km buffer (see fig. S2 and table S3). We excluded all large water bodies, as well as Greenland and Antarctica, which are mostly covered by ice, from the analyses.

Roadless areas with a 1-km buffer to the nearest road cover about 80% of Earth's terrestrial surface (~105 million km^2). However, these roadless areas are dissected into almost 600,000 patches. More than half of the patches are <1 km^2; 80% are <5 km^2; and only 7% are >100 km^2 (table S4 and fig. S3). If the buffer is extended to 5 km, there is a substantial reduction in roadless areas to about 57% of the world's terrestrial surface (~75 million km^2), dissected into 50,000 patches (fig. S2 and table S3). The occurrence, distribution, and size of roadless areas differ considerably among continents (Fig. 1A and fig. S4). For instance, the mean size of roadless patches (1-km buffer) is 48 km^2 in Europe, compared with >500 km^2 in Africa. Because of comparatively large gaps in available spatial data on roads in many segments of the tropics, the number and size of roadless areas are overestimated and should be treated with caution (e.g., Borneo; table S1).

All identified roadless areas were assessed for a set of ecological properties that were selected to reflect their relative importance to biodiversity, ecological functions, and ecosystem resilience: patch size, connectivity, and ecosystem functionality (*9*) (table S5). We normalized these three indicators to between 0 and 100 to calculate an additive and unitless index of the ecological value of each roadless area identified (termed the ecological value index of roadless areas, or EVIRA) [Fig. 1B and fig. S5; the specific rationale and technicalities of the chosen indicators are described in table S5 (*8*)]. The EVIRA values range from 0 to 80. A sensitivity analysis shows that ecosystem functionality and patch size are the best single indicators for the final index values (table S6 and figs. S6 to S8). Areas with relatively high index values tend to have a lower coefficient of variation (fig. S9).

We used the International Union for Conservation of Nature (IUCN) and UN Environment Programme–World Conservation Monitoring Centre data set of global protected areas to determine the extent of roadless areas that are protected (*8*) (Fig. 1C). The roadless areas distribution across human-dominated landscapes was determined following the classification of so-called anthromes, defined as biomes shaped by human land use and infrastructure (*10*) (Fig. 2 and table S7).

When examining the density of roads within different biomes, large discrepancies in distribution are apparent. The tundra and rock and ice-covered biomes are nearly entirely roadless, whereas temperate broadleaf and mixed forests have the lowest share of roadless areas (41%; figs. S9 and S10). Boreal forests of North America and Eurasia still retain large tracts of roadless areas (figs. S10 and S11). In the tropics, large roadless landscapes (>1000 km^2) remain in Africa, South America, and Southeast Asia, with the Amazon having the single largest roadless segment. In relation to the anthromes (*10*), about two-thirds of the world's roadless areas can be described as remote and unmodified landscapes [26% uninhabited or sparsely inhabited treeless and barren lands; 21% natural and remote seminatural woodlands, with 17% wild woodlands therein (*8*); Fig. 2 and table S7]. The remaining one-third consists of rangelands, indicating that roadless areas can also occur in anthropogenically modified landscapes.

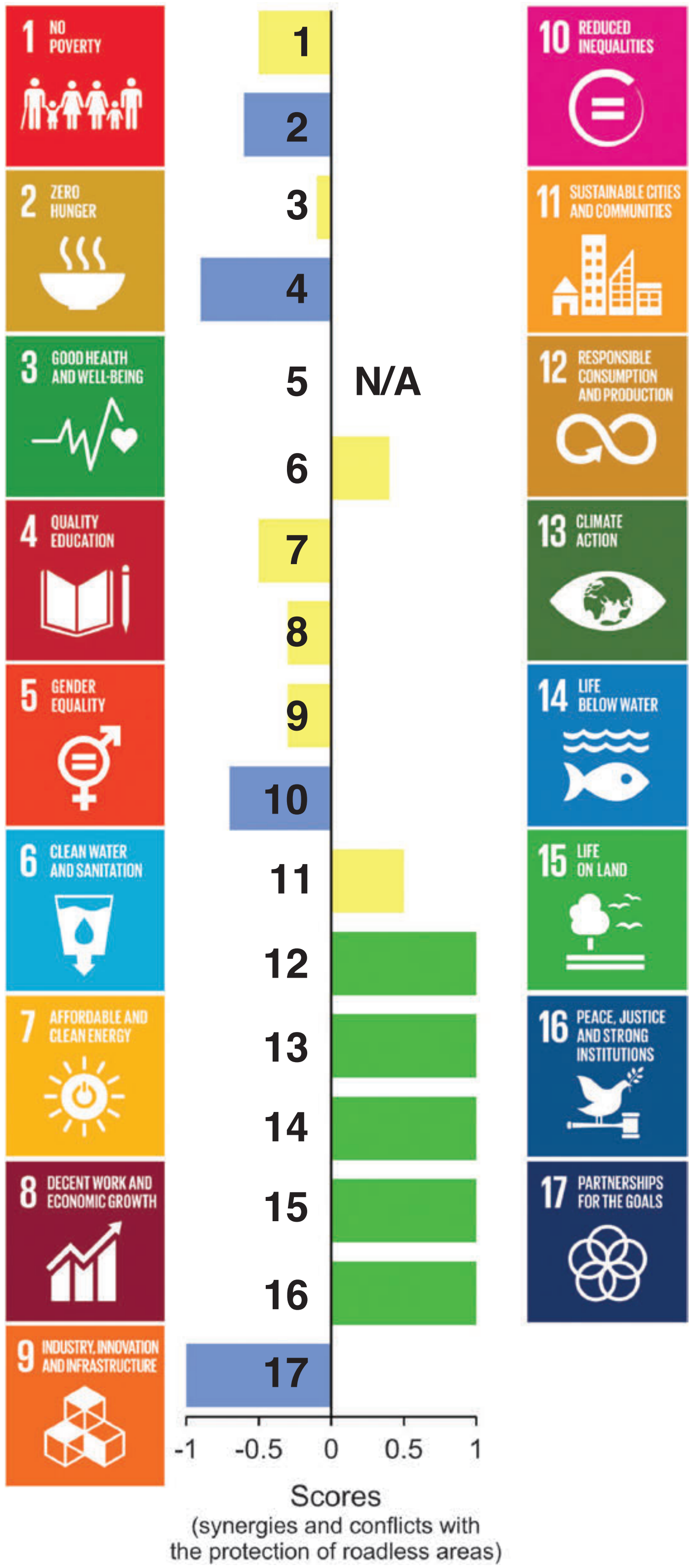

Fig. 4. Synergies and conflicts between conservation of roadless areas and the United Nations' Sustainable Development Goals. Scores <−0.5 (blue bars) indicate that conflicts with the goal prevail; scores between −0.5 and 0.5 (yellow) indicate a mixture of synergies and conflicts with the goal; and scores >0.5 (green) indicate prevailing synergies with the goal [for details, see table S11 (*8*)]. The scores reflect substantial imminent conflicts between various Sustainable Development Goals and conservation of roadless areas (table S11).

About one-third of the world's roadless areas have low EVIRA values. Patches with relatively low EVIRA values (ranging from 0 to 37; namely, <50% of the maximum value) account for 35% of the overall roadless area distribution, because most are small, fragmented, isolated, or otherwise heavily disturbed by humans. Some large tracts of roadless areas, such as arid lands in northern Africa or central Asia, occur in areas of sparse vegetation and low biodiversity and, thus, have low index values for ecosystem functionality (*9*) (Fig. 1B). High EVIRA values occur both in tropical and boreal forests. The relative conservation value of roadless areas is context-dependent. Comparatively small or moderately disturbed roadless areas have higher conservation importance in heavily roaded environments, such as most of Europe, the conterminous United States, and southern Canada.

Although the world's protected areas cover 14.2% of the terrestrial surface, only 9.3% of the overall expanse of roadless areas is within protected areas (all IUCN categories; Fig. 1C and table S8). There is no major difference in the coverage of roadless areas by strictly protected areas (IUCN categories I and II) versus the coverage of the overall landscape by strictly protected areas (3.8% roadless versus 4.2% overall). Only in North America, Australia, and Oceania are more than 6% of roadless areas under strict protection (table S8). If conservation efforts were to prioritize functional, ecologically important roadless areas, we would find a positive relation between strict protection coverage and EVIRA values of roadless areas. However, with the exception of Australia, this is not the case (Fig. 3 and table S9). Asia and Africa have particularly low protection coverage for roadless areas with high EVIRA values. For instance, we found gaps in the Asian tropical southeast, as well as in boreal biomes.

The recent Global Biodiversity Outlook (*11*) gives a bleak account of the progress made toward reaching the United Nations' biodiversity agenda as specified in the 20 Aichi Targets of the Convention on Biological Diversity (*12*). Governments have failed on several accounts to keep their use of natural resources well within safe ecological limits (target 4); to halt or at least halve the rate of habitat loss and substantially reduce the degradation and fragmentation of natural habitats (target 5); and to appropriately protect areas of particular importance for biodiversity and ecosystem services (target 11). To achieve global biodiversity targets, policies must explicitly acknowledge the factors underlying prior failures (*13*). Despite increasing scientific evidence for the negative impacts of roads on ecosystems, the current global conservation policy framework has largely ignored road impacts and road expansion. Furthermore, key policies on road infrastructure and development, such as the Cohesion Policy of the European Union, fail to take into account biodiversity.

In the much wider context of the United Nations' Sustainable Development Goals, conflicting interests can be seen between goals intended to safeguard biodiversity and those promoting economic development (*14*). We analyzed how roadless areas relate to the global conservation and sustainability agendas. As a transparent synthesis, we calculated simple scores of conflicts versus synergies of Sustainable Development Goals and Aichi Targets with the conservation of roadless areas (tables S10 and S11). Roads are explicitly mentioned in the Sustainable Development Goals only for their contribution to economic growth (goal 8), promoting further expansion into remote rural areas, and consideration is given neither to the environmental nor the social costs of road development. The resulting scores reflect substantial imminent conflicts (Fig. 4 and table S10); only in five Sustainable Development Goals do synergies with conservation of roadless

areas prevail, and four Sustainable Development Goals are predominantly in conflict with conservation of roadless areas. Maybe even more surprisingly, several of the Aichi Targets are ambivalent with respect to conserving roadless areas, rather than being in synergy entirely [six conflicting versus 11 synergistic targets (*8*); table S11].

There is an urgent need for a global strategy for the effective conservation, restoration, and monitoring of roadless areas and the ecosystems that they encompass. Governments should be encouraged to incorporate the protection of extensive roadless areas into relevant policies and other legal mechanisms, reexamine where road development conflicts with the protection of roadless areas, and avoid unnecessary and ecologically disastrous roads entirely. In addition, governments should consider road closure where doing so can promote the restoration of wildlife habitats and ecosystem functionality (*4*). Our global map of roadless areas represents a first step in this direction. During planning and evaluation of road projects, financial institutions, transport agencies, environmental nongovernmental organizations, and the engaged public should consider the identified roadless areas.

The conservation of roadless areas can be a key element in accomplishing the United Nations' Sustainable Development Goals. The extent and protection status of valuable roadless areas can serve as effective indicators to address several Sustainable Development Goals, particularly goal 15 ("Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss") and goal 9 ("Build resilient infrastructure, promote inclusive and sustainable industrialization and foster innovation"). Enshrined in the protection of roadless areas should be the objective to seek and develop alternative socioeconomic models that do not rely so heavily on road infrastructure. Similarly, governments should consider how roadless areas can support the Aichi Targets (see tables S10 and S11). For instance, the target of expanding protected areas to cover 17% of the world's terrestrial surface could include a representative proportion of roadless areas.

Although we acknowledge that access to transportation is a fundamental element of human well-being, impacts of road infrastructure require a fully integrated environmental and social cost-benefits approach (*15*). Still, under current conditions and policies, limiting road expansion into roadless areas may prove to be the most cost-effective and straightforward way of achieving strategically important global biodiversity and sustainability goals.

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SUPPLEMENTARY MATERIALS

www.sciencemag.org/content/354/6318/1423/suppl/DC1
Materials and Methods
Figs. S1 to S11
Tables S1 to S11
Data Sources
References (*16*–*180*)

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PLANT PATHOLOGY

Regulation of sugar transporter activity for antibacterial defense in *Arabidopsis*

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Microbial pathogens strategically acquire metabolites from their hosts during infection. Here we show that the host can intervene to prevent such metabolite loss to pathogens. Phosphorylation-dependent regulation of sugar transport protein 13 (STP13) is required for antibacterial defense in the plant *Arabidopsis thaliana*. STP13 physically associates with the flagellin receptor flagellin-sensitive 2 (FLS2) and its co-receptor BRASSINOSTEROID INSENSITIVE 1–associated receptor kinase 1 (BAK1). BAK1 phosphorylates STP13 at threonine 485, which enhances its monosaccharide uptake activity to compete with bacteria for extracellular sugars. Limiting the availability of extracellular sugar deprives bacteria of an energy source and restricts virulence factor delivery. Our results reveal that control of sugar uptake, managed by regulation of a host sugar transporter, is a defense strategy deployed against microbial infection. Competition for sugar thus shapes host-pathogen interactions.

Plants assimilate carbon into sugar by photosynthesis, and a broad spectrum of plant-interacting microbes exploit these host sugars (*1*, *2*). In *Arabidopsis*, pathogenic bacterial infection causes the leakage of sugars to the extracellular spaces (the apoplast) (*3*), a major site of colonization by plant-infecting bacteria. Although leakage may be a consequence of membrane disintegration during pathogen infection, some bacterial pathogens promote sugar efflux to the apoplast by manipulating host plant sugar transporters (*4*, *5*). Interference with sugar absorption by bacterial and fungal pathogens reduces their virulence, highlighting a general

Science AAAS

A global map of roadless areas and their conservation status

Pierre L. Ibisch, Monika T. Hoffmann, Stefan Kreft, Guy Pe'er, Vassiliki Kati, Lisa Biber-Freudenberger, Dominick A. DellaSala, Mariana M. Vale, Peter R. Hobson and Nuria Selva (December 15, 2016)

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Editor's Summary

Too many roads

Roads have done much to help humanity spread across the planet and maintain global movement and trade. However, roads also damage wild areas and rapidly contribute to habitat degradation and species loss. Ibisch *et al*. cataloged the world's roads. Though most of the world is not covered by roads, it is fragmented by them, with only 7% of land patches created by roads being greater than 100 km ^{2}. Furthermore, environmental protection of roadless areas is insufficient, which could lead to further degradation of the world's remaining wildernesses.

Science, this issue p. 1423

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Supplementary Materials for

A global map of roadless areas and their conservation status

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This PDF file includes:

MATERIALS AND METHODS

A. Definition of roadless areas
B. Data set and data accuracy
C. Data processing - Mapping of roadless areas and general processing
D. Data processing - Ecological Value Index of Roadless Areas (EVIRA)
E. Sensitivity analysis for the Ecological Value Index of Roadless Areas (EVIRA)
F. Policy analyses: synergies and conflicts between conservation of roadless areas and conservation and sustainability agendas

A. Definition of roadless areas

We reviewed 282 scientific papers, out of which 58 publications provided information on the spatial influence of various road impacts and/or on the road-effect zone (Table S2). All studied impacts were documented within a distance of 1 km from the road, 39% were observed in the 1-2 km zone, and only 14% extended out to 5 km. Road effects that go beyond 50 km and to even 100 km are rarely documented; they refer to deforestation in relation to distance to main roads, not including other minor roads and paths that are necessary for forest clearings (Table S2). The 1-km buffer would therefore rather underestimate than overestimate the extension of areas impacted by roads. Still it represents a reasonable approach to excluding with high certainty those areas that are significantly affected by roads. We consider 1 km as the minimum value for road-effect zones at a global scale, taking into account landscape heterogeneity, as well as the wide range of road impacts across biomes and road categories. Consequently, we defined roadless areas as terrestrial areas not dissected by roads and low impacted by road effects (which are at least 1 km away from the nearest road).

B. Dataset and data accuracy

We used a data set of OpenStreetMap (11/2013) to create a global map of roadless areas. This data set is updated on a daily basis and can be freely downloaded. We purchased a pre-processed data set provided by Geofabrik (http://www.geofabrik.de/de/). The pre-processing did not change the road data, but instead provided a filtered data set that contained only road layers in shapefile format. OpenStreetMap is a volunteered geographic information project founded in Britain in 2004 *(16)*. It is one of the most cited, analyzed and commonly used platforms of this type and became one of the best alternative sources for geodata *(17, 18)*. The aim of OpenStreetMap is to produce and distribute free global geographic data *(19)*. The OpenStreetMap data set used in this research provides six main road categories. Examples of 'major roads' can be motorways and freeways (*category one*); 'minor roads' are categorized as small local roads, residential roads, etc. (*category two*). *Category three* is represented by 'highway links' (sliproads/ramps) that connect roads with each other. Service roads or roads for agricultural use are considered as 'very small roads' under *category four*. *Category five* is called 'path' and mainly used for horse riding and cycling, but also for small or off-road vehicles. *Category six* roads are 'unknown' types of roads. As all road categories have ecological impacts (Table S2), we included all of them in the analyses.

The CIA World Factbook estimated the road length to be 64-million km in 2013 *(20)*. The OpenStreetMap data set (2013) used in this research consists of 36-million km of roads. In contrast, the Global Roads Open Access Data Set (gROADS), published in 2013, contains 9.1-million km of roads (CIESIN 2013). The gROADS data set has been used in global studies on road impacts, in spite of containing less data than OpenStreetMap (e.g. *(7)*).

OpenStreetMap relies on the willingness of volunteers, both to contribute entries and to edit them for errors *(21)*. Therefore, the data are a crowd-sourced product with unknown data quality standards. However, a quality assessment of the OpenStreetMap data, including spatial data quality, evolution of street network, polygon geometry, comparison of user activity, development, positional accuracy, and completeness is available for different regions *(17, 22-28)*. Gröchenig et al. (2014) conducted a global evaluation of the mapping progress of OpenStreetMap history between 2006 and 2013 *(29)*. Their results state that external and internal factors significantly influence the mapping progress. Some of these factors are regional activity of the mapping community, data imports, and environmental disasters or other unforeseen events *(29)*. Demographic characteristics affect the mapping progress, and the quality of the data can vary significantly among countries (*17, 29)*.

A high number of road assessments were conducted in Europe *(30-34)*. Often, commercial or administrative data sets are used to compare and evaluate OpenStreetMap *(17)*. A study published in 2010 assessed the quality of OpenStreetMap for Germany *(32)*. Among its findings, the total length of roads was calculated as 1,204,213.69 km, whereas the road length data made available by TeleAtlas (an enterprise that provides digital maps, user content navigation, and location-based services) was 1,272,681.77 km. TeleAtlas focuses more on roads suitable for cars, whereas OpenStreetMap includes all road types *(32)*. In the case of the Brazilian Amazon it has been found that the road data from the Brazilian Institute for Geography and Statistics (IBGE) are more complete, including ca. 157,000 km of roads in contrast to ca. 114,000 km in our OpenStreetMap data set.

In areas of the tropics where land conversion is advanced, the road network may not be well reflected by OpenStreetMap. An extreme example of missing roads in the OpenStreetMap data set is Borneo. We carried out a comparative analysis of roads in the Sabah region, Malaysia, in northern Borneo. In areas considered to be roadless, closer inspection on the ground (in 2015) revealed extensive networks of vehicle tracks, for instance, throughout oil palm plantations. A similar result was found in forested areas impacted by logging roads. Indeed, cumulative data (1970-2010) compiled by the Center for International Forestry Research (CIFOR) indicate that there would be 37,498 km of logging roads in the region of Sabah alone. The 2013 OpenStreetMap data set (for Sabah created since 2009) used in this study comprises just 4,880 km, which is still more than the 2,937 km included in the road data set gROADS (1980-2010) that was the basis for other global road assessments (CIESIN 2013, *7*). Applying a 1-km buffer to each of the three road data sets for Sabah demonstrates that roadless areas are underestimated by the OpenStreetMap and the gROADS data set (Table S1). According to the gROADS data set (CIESIN 2013), 92% of Sabah is roadless. The OpenStreetMap data set shows that 91% of Sabah is roadless. In contrast, buffering the logging roads (CIFOR) reveals that only 40% of Sabah remains roadless. However, on the other hand, the CIFOR data set seems to overestimate existing logging roads. The CIFOR logging roads were mapped in four time intervals (1970, 1990, 2000 and 2010) by visual interpretation of satellite imagery. Analyzing the CIFOR logging roads with current Google Earth satellite images suggests that numerous roads have been overgrown by forest. The amount of logging roads that were either non-existent in 2010 or were <10 m wide (therefore not included in the CIFOR analysis) is high *(35)*. This simple exercise highlights the methodological problems to be overcome in future mapping. The three data sets can only be compared to a limited extent, since the roads have been mapped in different ways, time intervals and for different purposes. The gROADS data set (CIESIN 2013) focuses on roads between settlements. For Malaysia, gROADS is based on the Vector Smart Map Level 0 data. The CIFOR road data set does not include any other road category besides logging roads. In general, the three different road data sets (OpenStreetMap, gROADS, CIFOR) vary in length, location and type of roads, with OpenStreetMap being the data set with the largest length of roads at a global scale, and not limited to one type of roads (Table S1).

C. Data processing - Mapping of roadless areas and general processing

The global road data set was analyzed and processed for each continent, except for Antarctica and Greenland. All roads were buffered on both sides with a geodesic buffer of 1 km. Due to a very high number of vertices, all buffered roads were generalized with a "maximum offset tolerance" of 30 m, using the "Douglas-Peucker simplification algorithm" *(36)*. All analyses were conducted with ArcGIS 10.2. A road model tool was created with the ArcGIS model builder to facilitate the process. For the purpose of comparison, an alternative map of roadless areas was developed with a 5-km buffer to all roads (Fig. S2).

For area calculations, roadless areas were projected with the World Cylindrical Equal Area Projection. Spatial calculations and maps were made with ArcGIS Version 10.2. Protected area coverage of roadless areas was calculated based on IUCN categories of protected areas, including (a) IUCN categories Ia, Ib and II, and (b) other protected areas classified as IUCN categories III to VI (IUCN & UNEP-WCMC 2015). Protected area data sets for each country were downloaded and processed singularly instead of using the global protected area file due to inconsistencies in the global data set.

D. Data processing - Ecological Value Index of Roadless Areas (EVIRA)

There are manifold and partially contrasting approaches for defining the conservation values of given areas. Attempts at conservation priority setting have been classified as reactive and proactive *(37)*, some approaches focus on patterns rather than processes; however, in times of rapid environmental change, there are good arguments for especially targeting ecological functionality and biological viability *(9, 38).* Therefore, we chose a functional priority-setting approach that is not based on merely anthropocentric values, such as use value or aesthetics, but comprises indicators that are defined in line with principles of modern ecosystem theory. In this context, we especially consider the capability of ecosystems to self-order and regulate abiotic and biotic conditions, which is greatly based on the capacity of uptaking and storing eco-exergy *(39, 40).* Specifically, exergy has been used for analyzing and indicating ecosystem health *(41-46).* As key attributes of ecosystem growth and development, Jørgensen (2006) *(42)* and Jørgensen et al. (2000) *(43)* proposed biomass, information and network as main growth forms of ecosystems.

To assess the conservation value of roadless areas, a corresponding additive index (Ecological Value Index of Roadless Areas, EVIRA) was created. Three indicators were chosen (for individual and more specific rationale of indicators see Table S5):

(1) Roadless area patch size: A larger roadless area patch size indicates less human disturbance, lower edge effects, higher populations of road-sensitive species, as well as higher ecological integrity and self-regulating capacity.

(2) Thiessen connectivity into all directions for roadless area patches: We describe connectivity (and degree of isolation), as the ratio between the size of a roadless area patch and its surrounding Thiessen polygon. A Thiessen (or Voronoi) polygon describes the area around a sample point or area where any position taken from inside the polygon is closer to the sample point/area than to any of the other sample points/areas *(47)*. To create Thiessen polygons Euclidean distance was calculated with the formula:

$$d(x,y) = \sqrt{\sum_{i=1}^{n}(x_i - y_i)^2}$$

The larger the Thiessen connectivity value, the closer neighboring roadless patches can be found. This is important for the integrity of landscape-scale processes (e.g., genetic exchange of metapopulations and endemics with narrow geographic ranges confined to roadless areas).

(3) Ecosystem Functionality Index *(9)*: This weighted, additive dimensionless index comprises vegetation density, tree height, carbon storage, species richness of vascular plants, plant functional richness and slope. Functionality is defined as “the state of ecosystems, characterized by inherent structures,

> ecological functions and dynamics, that provide ecosystems with both, the necessary efficiency and resilience to develop without abrupt change of system properties and geographical distribution, and allows for flexible response to external changes" *(9)*.

All indicators (Roadless area patch size, Thiessen connectivity, Ecosystem Functionality Index) were rasterized and adjusted in resolution and projection. A resolution of 0.002 (equally to 0.2 km) was chosen. ArcGIS 10.2 was used for projection, resolution and rasterization. All indicators were normalized between 0 and 100 and a weighted additive index was calculated using the software Insensa-GIS *(48)*. Thiessen connectivity into all directions and roadless areas patch size were weighted with 25%, whereas ecosystem functionality was weighted with 50%.

E. Sensitivity analysis for the Ecological Value Index of Roadless Areas (EVIRA)

Index construction always involves steps such as indicator selection and weighting. In order to transparently highlight the sensitivity of EVIRA to changes in these steps, we performed a statistical sensitivity analysis. Three different index versions were produced using jackknifing, ten of them using random weight variation within defined borders (connectivity into all directions and roadless area patch size 10-50%; ecosystem functionality index 30-70%) and one using equal weighting. Within the jackknifing procedure, three versions were created where each indicator was removed iteratively from the index calculation procedure. Overall 14 different index versions were created to perform the sensitivity analysis.

Pearson and Spearman rank correlation coefficients were calculated for the three indicators and EVIRA (Table S6). Significant and highly positive Spearman rank and Pearson correlation coefficients were found between the Ecosystem Functionality Index (EFI) (*9*) and EVIRA (Spearman r= 0.818; $p<0.0001$; Pearson r= 0.881; $p<0.0001$; Table S6). This is likely to be a consequence of the original weighting scheme of EVIRA, where EFI was given a weight double as high as the two other indicators. A high positive and significant Spearman rank correlation was also detected for roadless area patch size and EVIRA (Spearman r= 0.768; $p<0.0001$; Table S6). Therefore, EFI and roadless area patch size are the best single indicators for the final index output.

Mean values over all 14 index variations are shown in figure S6 with the highest values represented in blue and low values shown in orange. Similar to the original EVIRA, highest mean values are recorded for the Amazon, followed by the tundra and taiga of the northern and eastern lowlands of Siberia, as well as south-east Asian tropical rain forests.

The coefficient of variation was calculated over all 14 index variations to evaluate the variability of EVIRA (Fig. S9). Most parts of Australia show high levels of variation, as well as parts of Africa and central- and southwest Asia. The overall pattern is that regions with relatively high index values tend to have a lower coefficient of variation, whereas areas with high levels of variation tend to occur in regions with low index values. This results in a high confidence in the prediction of the ecological value, especially of those areas with high EVIRA values. A negative correlation coefficient between EVIRA and the coefficient of variation was detected (Spearman rank correlation: -0.97; Pearson correlation: -0.94). The volatility highlights the areas which were most frequently assigned a high index value (>70% of the maximum value) within the 14 different index variations

(Fig. S7). Very high readings were found for the sites with highest roadless area patch size as well as parts of Southeast Asia.

The proportion of area that changes its index value by less than 25%; between 25-50%; between 50-75%; and more than 75%, was explored for the equal weight method, and the three different index versions created by the jackknifing procedure (Fig. S8). Indicator selection seems to have a stronger effect on the output than the weighting scheme. More than half of the area changes its index value between 50 to 75% when connectivity into all directions was removed from the index, and 19% of the areas changed its index value by more than 75%. The exclusion of EFI showed that more than 60% of the area changed its index value between 25 and 50%. The removal of roadless area patch size (18% change in category 25-50%) and applying an equal weighting scheme (5% change in category 25-50%) did not change the index output significantly.

F. <u>Policy analyses: synergies and conflicts between conservation of roadless areas and conservation and sustainability agendas</u>

The "Aichi Biodiversity Targets" of the Convention on Biological Diversity (CBD) are part of the "Strategic Plan for Biodiversity 2011-2020" *(12)*. They circumscribe the United Nations' central agenda for the conservation of the Earth's diversity of life. They were adopted in October 2010 and comprise 20 targets that are grouped into five Strategic Goals. Seventeen "Sustainable Development Goals" (SDGs) have been defined within "Transforming our world: the 2030 Agenda for Sustainable Development" of the United Nations *(14)*, adopted in September 2015. They replace eight "Millennium Development

Goals” that were pursued from year 2000 to 2015 *(49)*. The SDGs are associated with 169 targets. Work on underlying indicators is ongoing; nevertheless, the latest report can provide direction for the interpretation of the goals and their respective targets *(50)*.

Specifically, our analyses of the global sustainability agendas aim at identifying potential synergies, conflicts and ambivalences between roadless areas conservation and the achievement of conservation and sustainability goals in the policy framework of the United Nations. In addition, these analyses indicate imminent conflicts among goals within the respective policy frameworks, particularly those concerning the global sustainability agenda. Furthermore, a considerable number of conservation and sustainability targets also were found to be ambivalent.

The calculation of conflict-synergy scores for the SDGs (Table S10) and the Aichi Strategic Goals (Table S11) is based on a simple index composed of individual scores attributed to all corresponding targets to which roadless areas are in some way applicable. We excluded the targets related to governance in general (marked by a combination of number and letter, e.g. “13.a”) from the analysis, thus reducing the number from 169 to 126. The individual scores for targets can have three discrete values:

- -1 (indicated by blue color): conservation of roadless areas is in conflict with the achievement of the target;
- 0 (yellow): conservation of roadless areas has an ambivalent relationship with the achievement of the target; and

- 1 (green): conservation of roadless areas is in synergy with the achievement of the target.

Roadless areas do not relate to a number of targets; these targets are therefore excluded from the analysis (indicated by grey color). The conflict-synergy score for a goal is calculated as the mean of all values for corresponding targets. The scores can, thus, vary between -1 and +1. They are classified as follows:

- <-0.5 (indicated by blue color): conflicts with goal prevail;
- -0.5 to 0.5 (yellow): mixture of synergies and conflicts with goal; and
- >0.5 (green): synergies with goal prevail.

The conflict-synergy scores for goals are also visualized by the colors in the large boxes of Tables S10 and S11.

SUPPLEMENTARY FIGURES (S1-S11)

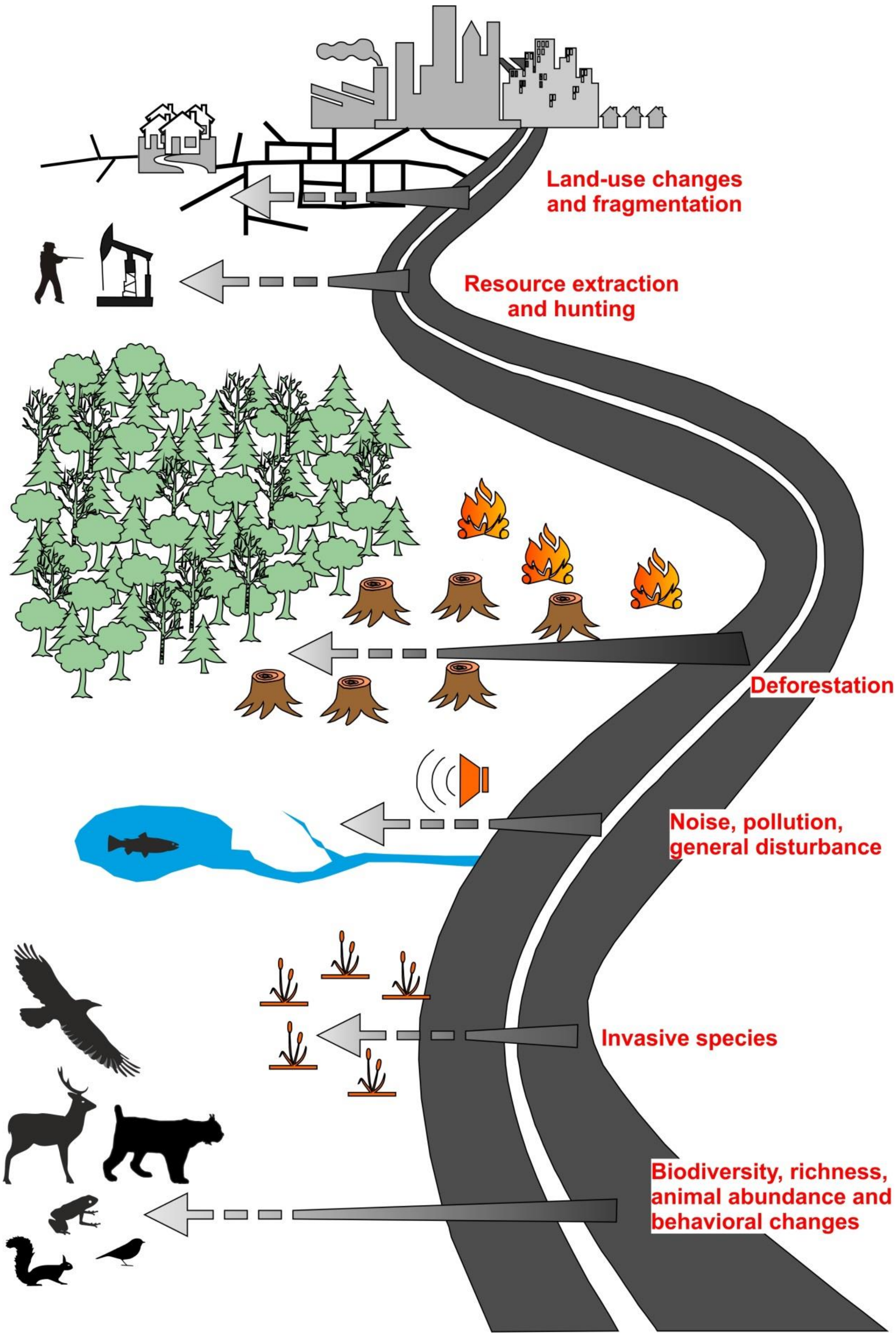

Fig. S1. Schematic representation of different categories of road impacts on biodiversity. These impacts decrease with the distance from the road. Road effects generally attenuate beyond one kilometer distance from the road (see literature review in table S2). One kilometer was therefore selected as a buffer to identify roadless areas as those areas relatively free from road disturbances.

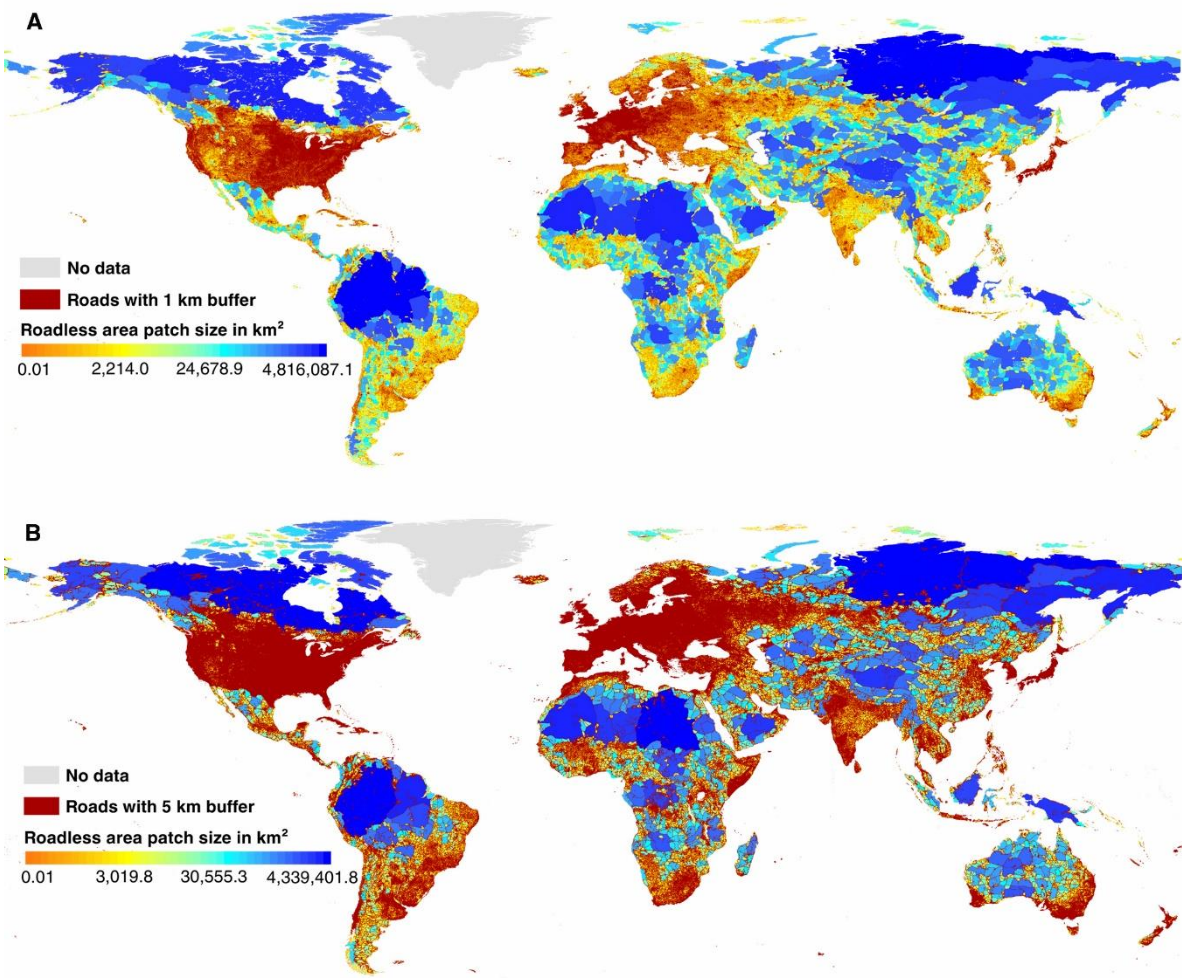

Fig. S2. The global distribution of roadless areas based on a (A) 1-km and a (B) 5-km buffer to all roads included in the OpenStreetMap data set (11/2013).

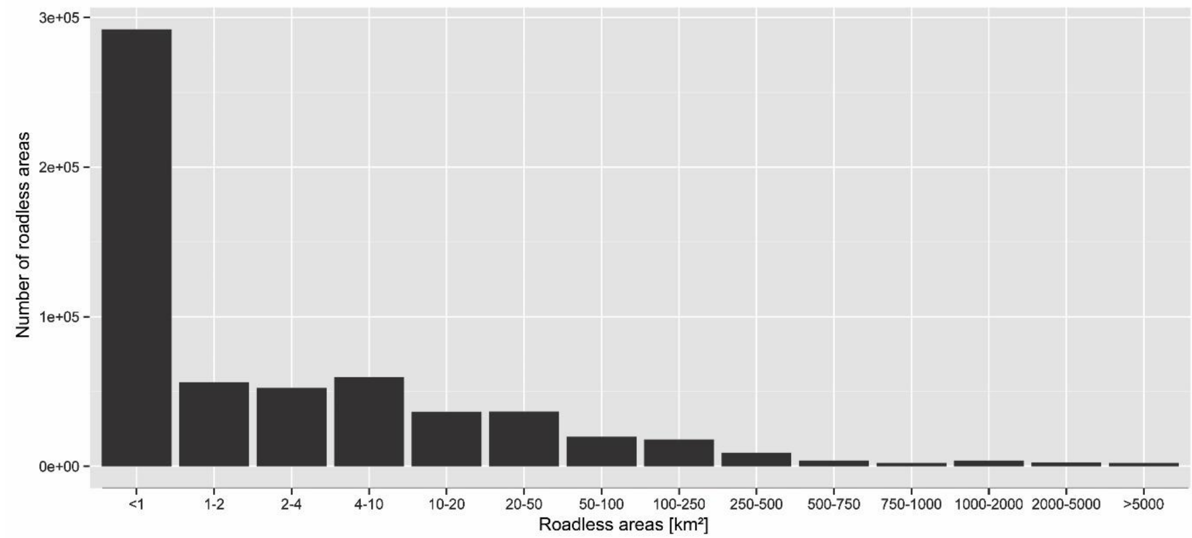

Fig. S3. Frequency of global roadless areas size classes based on 1-km buffer to all roads included in the OpenStreetMap data set (11/2013).

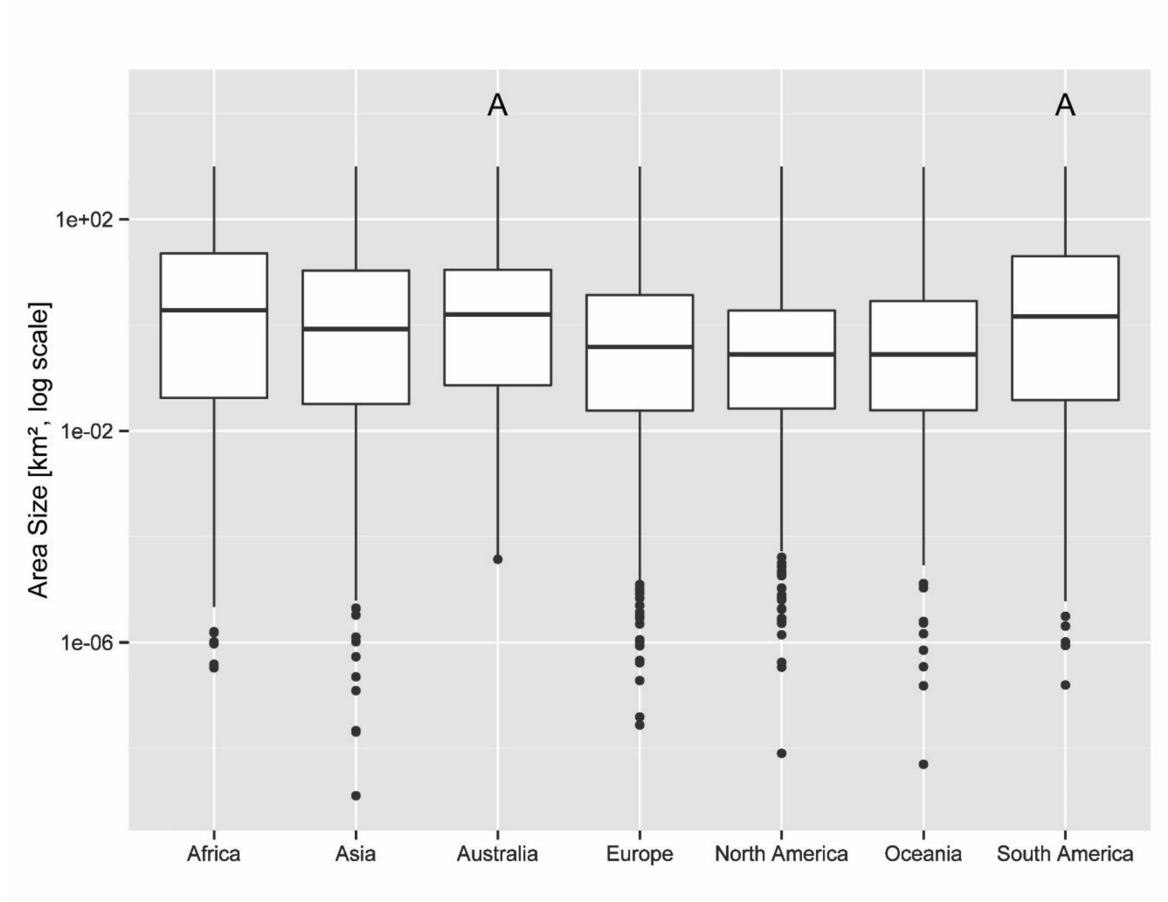

Fig. S4. Sizes of roadless areas across continents based on 1-km road buffer using the OpenStreetMap data set (11/2003) (Pairwise Wilcox test; "A" indicates that the corresponding distributions are not significantly different; p<0.001).

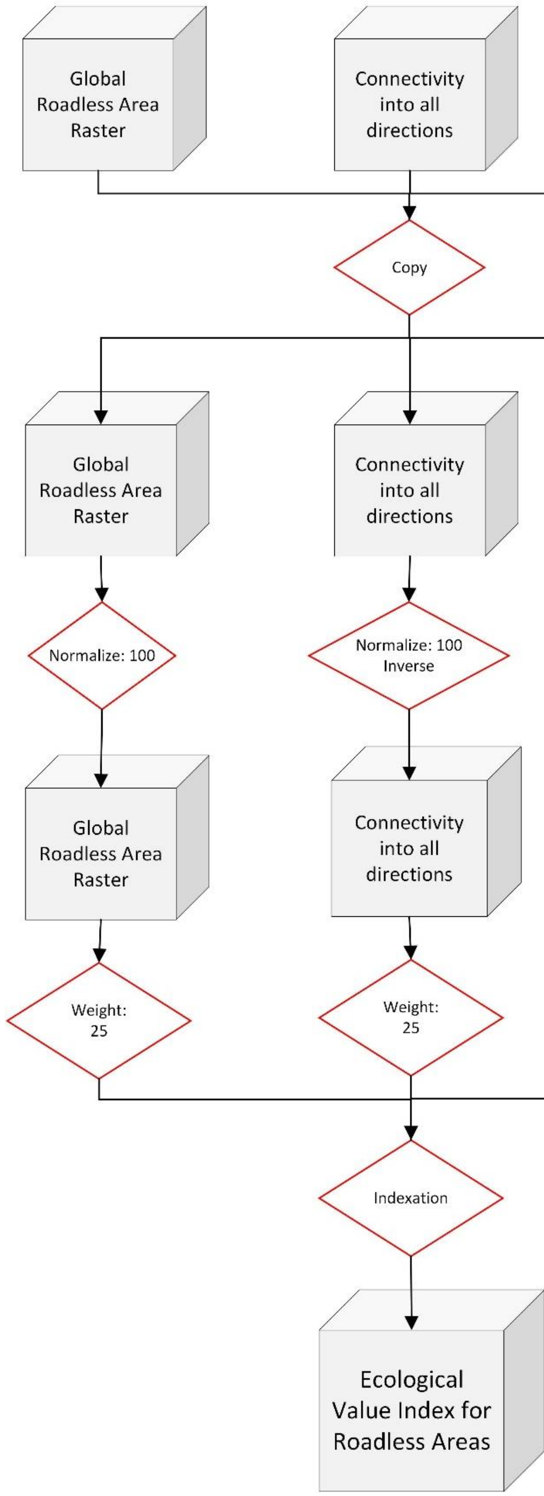

Fig. S5. Workflow of the indexation process for creating the *Ecological Value Index of Roadless Areas* (EVIRA).

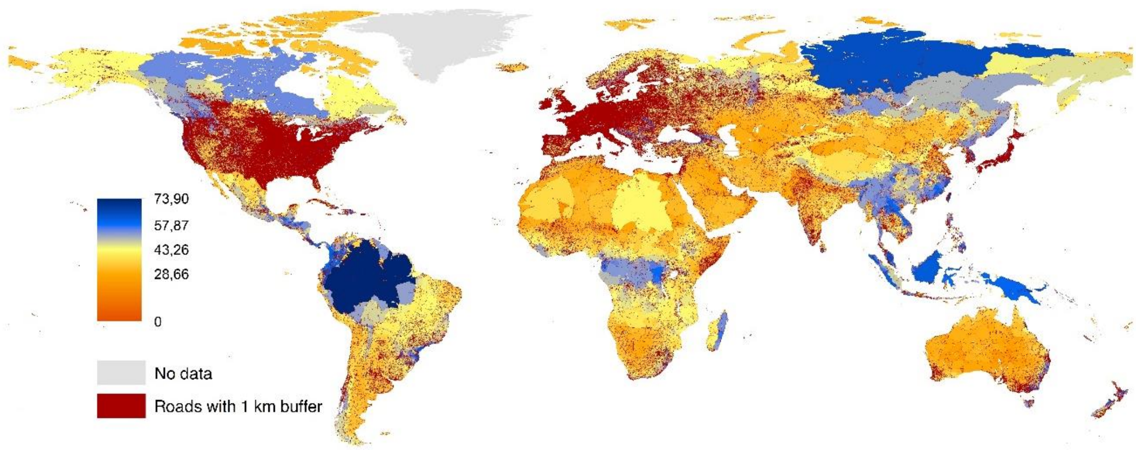

Fig. S6. Global map of mean values over 14 different index variations for the Ecological Value Index of Roadless Areas (EVIRA). Class breaks were calculated using the Jenks breaks algorithm.

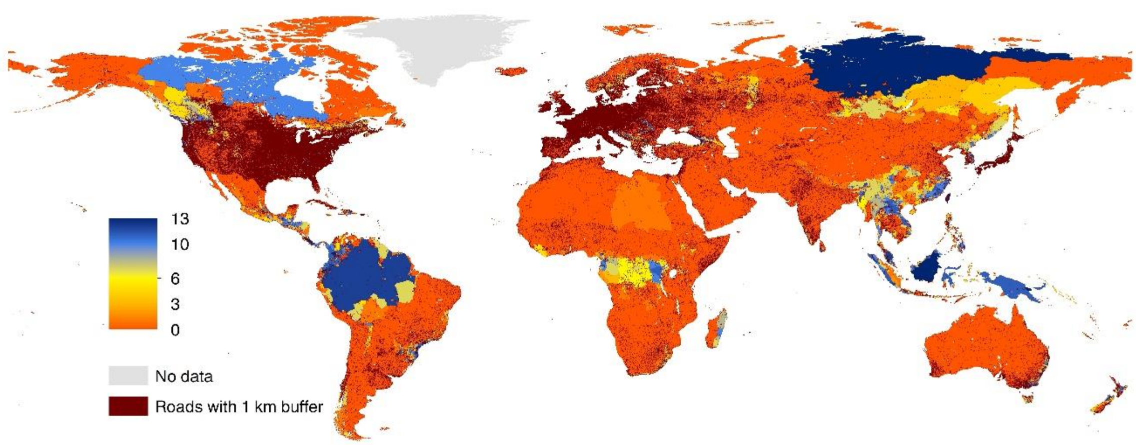

Fig. S7. Global map of volatility (frequency of that the value achieved at least 70% of the maximum index value) of the ecological value index of roadless areas (EVIRA) over all 14 index variations.

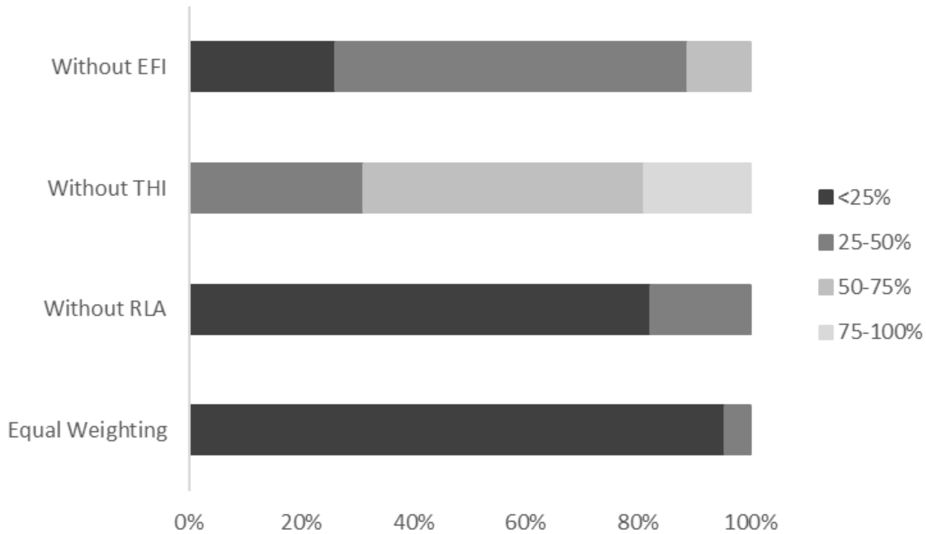

Fig. S8. Proportion of global area whose EVIRA value is changing < 25%, 25-50%, 50-75% and >75%, as shown by the sensitivity analysis. The three indicators making up the EVIRA index are the Ecosystem Functionality Index (EFI), the Thiessen connectivity into all directions (THI) and the Roadless area patch size (RLA).

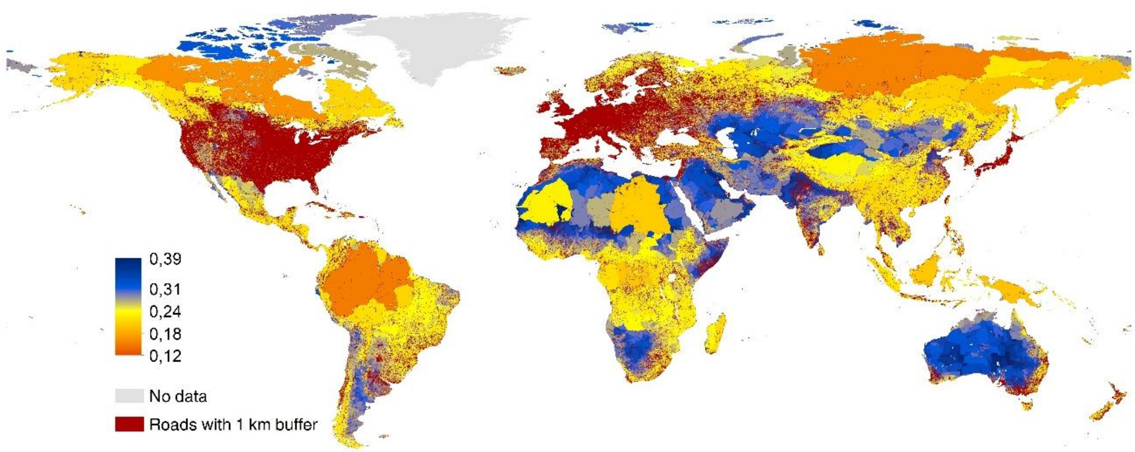

Fig. S9. Mean statistical sensitivity of the Ecological Value Index of Roadless Areas (EVIRA) as overall coefficient of variation of 14 index variations.

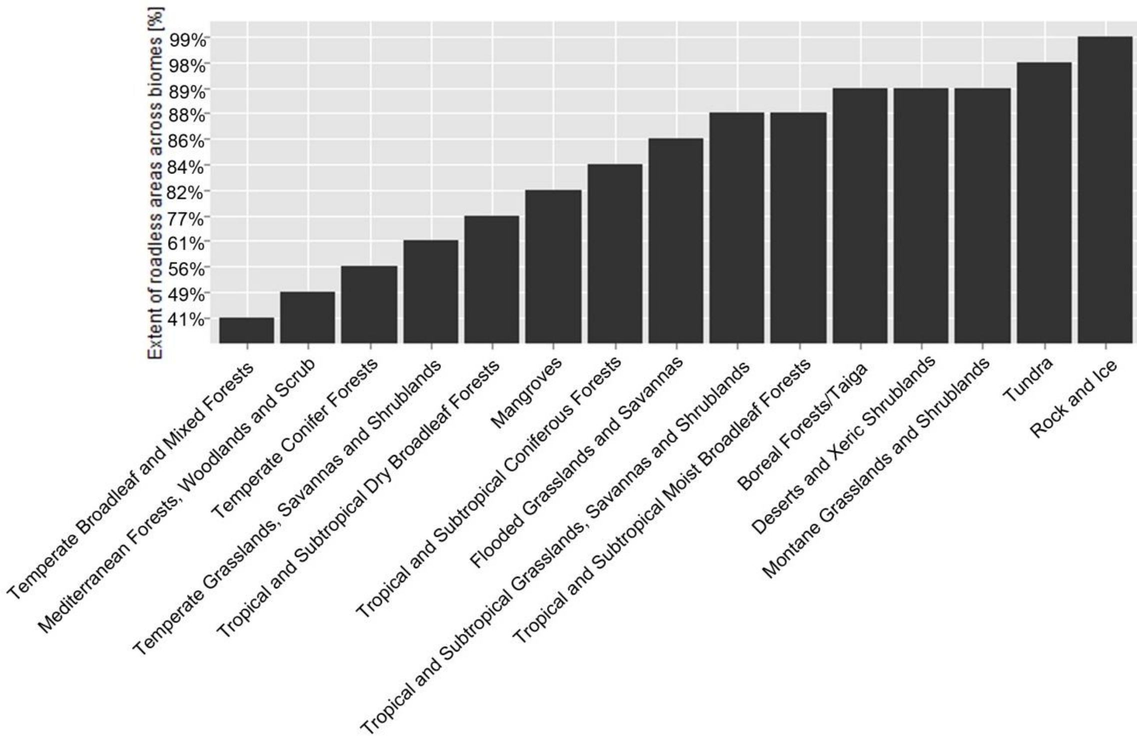

Fig. S10. Extent of roadless areas across biomes (without freshwater bodies, Antarctica and Greenland) according to classification by Olson et al. (2001) *(51)* and based on 1-km buffer to all roads included in the OpenStreetMap data set (11/2013).

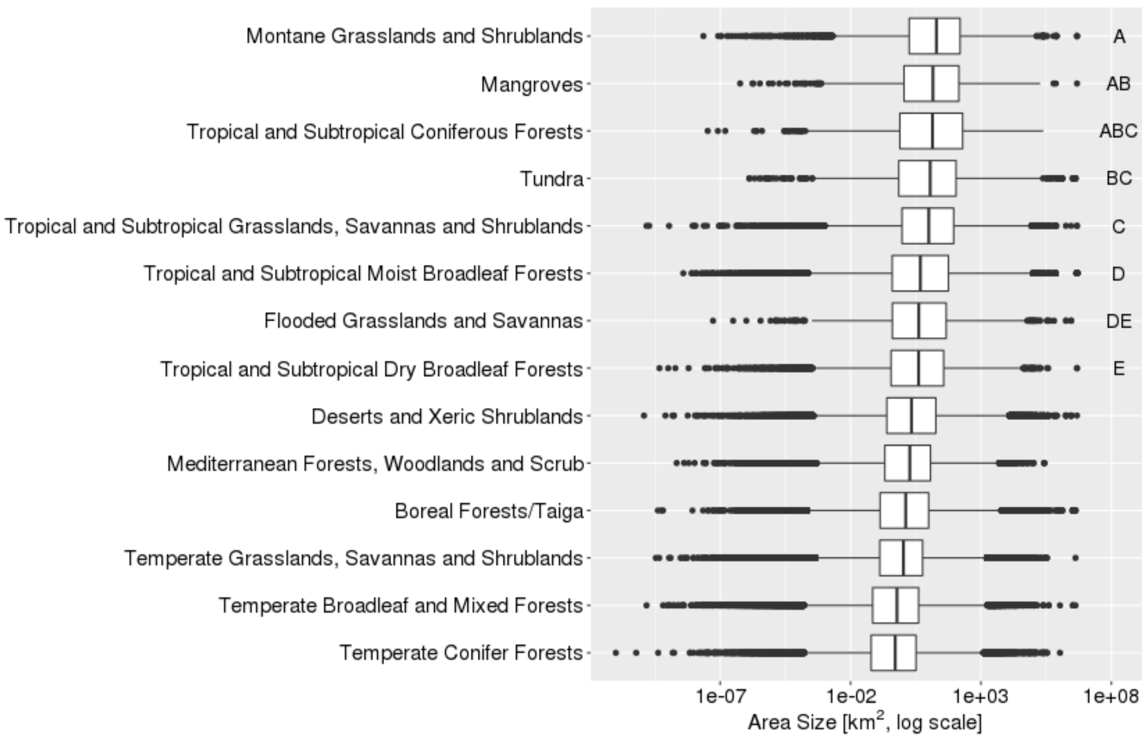

Fig. S11. Size distribution of roadless areas across different biome types assessed with a 1-km road buffer using the OpenStreetMap data set (11/2003) (Pairwise Wilcox test; if biomes share the same capital letters, then corresponding distributions are not significantly different; p<0.001).

SUPPLEMENTARY TABLES (Table S1-S11)

Table S1. Extent of 1-km-buffer roadless areas for Sabah, Malaysia, comparing three different road data sets (OpenStreetMap 11/2013, CIESIN 2013, CIFOR 2014).

	Roadless areas (km²)	**Roadless areas coverage (% of the territory of Sabah)**
Sabah total area	73,841.91	
Roadless areas using OSM data	66,944.69	91
Roadless areas using CIESIN data	68,271.54	92
Roadless areas using CIFOR data	29,700.56	40

Table S2. List of studies documenting or assuming road-effect zones or investigating the spatial influence of road effects. Studies are ordered according to the most important effect described (some studies dealt with more than one effect).

Road type or data	**Study system and location**	**Road effect tested**	**Effect description**	**Spatial range of influence of the road effect**	**Reference**
CHANGES IN ANIMAL ABUNDANCE, DENSITY AND POPULATION SIZE					
Highway, secondary, rural and cyclist road	Polders, farming areas, reclaimed marshland (Netherlands)	Changes in population density of four bird species	Population density increases with distance from the road for black-tailed godwit (*Limosa limosa*) and the lapwing (*Vanellus vanellus*), but not the other species	Up to 1,800 m	*(52)*
Highway	Willow coppices and shrubs (central Netherlands)	Density of territorial males of willow warblers (*Phylloscopus trochilus*)	Lower density of territorial males, lower presence of older males, 50% higher proportion of yearling males and 50% lower success of yearling males in the road zone Total annual output of males/ha 40% lower in the road zone	Road zone assumed as 200 m from the road; intermediate between 200-400 m, and control 400 m	*(53)*
Paved major roads with different traffic volume	Deciduous and coniferous woodland crossed by main roads (Netherlands)	Breeding density of woodland birds	Reduced density in 60% of the species adjacent to roads, due to noise	The maximum reduction of car noise at 200 m from the road The majority of the species (75%) showed maximum effect distances	*(54)*

				between 100 and 1,500 m For all species combined, the effect distances varied between: - 40-1,500 m and 70-2,800 m for roads with 10,000 and 60,000 cars/day, respectively, in deciduous woodland - 50-79 m and 100- 1,750 m for roads with 10,000 and 60,000 cars/day, respectively, in coniferous woodland	
Paved major roads with different traffic volume	Open moist grassland (N and W Netherlands)	Breeding densities of bird species, including waders	Most species had reduced density close to the road; this effect was very strong for the summed density of all species	For the density of all species combined, the disturbance distance was 120 m and 560 m for 5,000 and 50,000 cars/day, respectively. Among species, disturbance distance varied between 20-1,700 m at 5,000 cars/day, and 75- 3,530 m at 50,000 cars/day At 5,000 cars/day, 7 out of 12 species had an estimated population loss of 12-56% within 100 m of roads. At further distances, such reduction occurred in the black-tailed godwit (*Limosa limosa*, 22% in the 0-500 m zone), and the oystercatcher (*Haematopus ostralegus* 44% up to 500 m and 36% for 0-1,500 m zone). At 50,000 cars/day all species showed an estimated population loss of 40-74% within 100 m of the road and >10% at 0-500 m. Five species showed reductions of 14-44% up to 1,500 m	*(55)*
All roads	Rural area (Ontario, Canada)	Effect of traffic on population abundance of green frogs (*Rana clamitans*) and leopard frogs (*Rana pipiens*)	Negative effect of traffic density on leopard frog abundance (more vagile species), but not on green frog abundance	Leopard frog population density negatively affected by traffic density within a radius of 1.5 km	*(56)*
Highway	Desert (California, USA)	Tortoise activity and presence	Tortoise signs increasing with distance from the highway edge	Tortoise populations depressed in a zone extending at least 400 m from the road	*(57)*
Unpaved roads, mostly	Lowland tropical	Abundance of	Most species responded	Effects measures up to 1.2	*(58)*

from oil and logging companies	rainforest (SW Gabon)	mammal species	negatively to roads	km from the road	
Low-traffic road within forest	Deciduous forest (USA)	Change in abundance of salamander species	Reduction in salamander abundance	>35 m	*(59)*
Highway	Protected forest and commercial timberland (Adirondack Mountain, New York, USA)	Impact of road de-icing salts on the reproduction of adults and growth and survival of embryonic and larval of spotted salamander (*Ambystoma maculatum*) and wood frog (*Rana sylvatica*)	High concentration of salt reduced amphibian species survival close to the road (decline of embryo and larvae survival rate) A demographic model predicting population size decrease due to exposure to road salt (embryo and larva mortality effect); stronger effect closer to the road	Salt traveled up to 172 m from the highway into wetlands The negative effect of road salt on population sizes up to 200 m	*(60)*
Highway	Desert (Utah, USA)	Abundance and density of small mammals	No clear abundance, density, or diversity effects relative to distance from the road Species-specific response	No road-effect zone measured up to 400 and 600 m from the road in each of the two study years	*(61)*
All road types and also other infrastructure	Various; meta-analysis of 49 studies on 234 mammal and bird species	Road avoidance and reduced population density of birds and mammals	Mammal and bird population densities declined with their proximity to infrastructure Stronger avoidance in open areas compared to forested areas Habitat- and species-specific response	Up to about 1 km for birds, and up to about 5 km for mammal populations	*(62)*
Paved highway	Boreal forest (Canada)	Population density of brook charr (*Salvelinus fontinalis*) in streams	Population density differed markedly between upstream and downstream sites near highway crossings (of intermediate and low passability)	Up to 800 m from highway	*(63)*
Phantom road	Fir forest and cherry bushes (Idaho, USA)	Simulated traffic noise effect on bird abundance	Serious (25%) decline in bird abundance and almost complete avoidance by some species between noise-on and noise-off periods along the phantom road; such effect was not detected at control sites	Control sites at ca 800 m	*(64)*
Highway	Mountainous area with shrub-steppe vegetation (Ghamishloo Wildlife Refuge, Iran)	Loss of suitable habitat and disruption of the distribution pattern of two ungulate species, the goitered gazelle (*Gazella subgutturosa subgutturosa*) and the wild sheep	51% and 10% of high quality habitat unavailable for gazelle and sheep, respectively, due to road construction Presence points increased with road distance	Large increase in presence at > 3km from the road	*(65)*

		(*Ovis orientalis isphahanica*)			
Highways and national roads	Mediterranean agricultural landscape and cork oak woodland (Alentejo, Portugal)	Likelihood of owl species (barn owls *Tyto alba*, tawny owls *Strix aluco* and little owls *Athene noctua*) occurrence	Higher probability of owl occurrence at longer distance from major roads, particularly for barn owl	Owl presence occurred at further distances (1,591 ± SD 960 m) than absences (1,097 ± SD 826 m)	*(66)*
Paved interstate and county roads	Desert (Mojave, California, USA)	Signs of Mojave Desert tortoise presence (*Gopherus agassizii*)	Tortoise signs increased significantly with distance from roads	Reductions in signs extended farther from the high-traffic interstate than from the smaller, lower-traffic county roads (306 m versus 230 m)	*(67)*
Wide paved and minor unpaved roads	Mediterranean scrubland, dunes and wetlands (Doñana Biosphere Reserve, S Spain)	Presence probability of two ungulates, red deer (*Cervus elaphus*) and wild boar (*Sus scrofa*)	Presence probabilities for both species increased with the distance to the nearest road, in most cases were unpaved roads with negligible traffic volume	At 180 m from the nearest road, wild boar presence probability was lower than 0.2, and for red deer was lower than 0.7	*(68)*
MODIFICATION OF ANIMAL BEHAVIOR					
Highway	Willow coppices and shrubs (central Netherlands)	Breeding dispersal of male willow warblers (*Phylloscopus trochilus*)	Higher proportion of yearlings dispersing and longer dispersal distance in the road-zone	Road zone assumed as 200 m from the road; intermediate between 200-400 m, and control 400 m	*(69)*
Highway and major railroad line	Mountain areas covered mostly with mixed coniferous forest, valleys and prairies (Montana, USA)	Movements of grizzly bears (*Ursus arctos*)	Highway crossing frequency declined exponentially with increasing traffic volume Avoidance of areas close to the highway	Bears strongly avoided areas within 500 m of the highway (asymptote within the 500-600 m category)	*(70)*
Roads in rural areas	Steppe (Patagonia, Argentina)	Flying and feeding behavior of scavenger species	Flying activity and carcass detection was greater near roads (500 m buffer) Andean condors (*Vultur gryphus*) and black-chested buzzard-eagles (*Geranoaetus melanoleucus*) fed far from roads, while other species fed close to roads	Optimal distance for feeding activities for condors and eagles was 3,110 and 10,460 m from the road, respectively, and for the other species, from 218 to 365 m	*(71)*
Paved and unpaved roads	Steppe (Patagonia, Argentina)	Andean condor (*Vultur gryphus*) behavior at carcasses	In the patches far from roads many more condors came to feed, the average time spent per individual was longer, the proportion of time spent vigilant was lower, and the amount of food left uneaten on the carcasses was lower	Up to 350 m	*(72)*

Two-lane roads	Arid shrublands and grasslands (California, USA)	Changes in survival, reproduction, space use, den-site selection, prey availability, and diet of San Joaquin kit foxes (*Vulpes macrotis mutica*)	No effects of the distance to the road on survival, reproduction, litter size, space-use patterns and diet	No effects from 0 m to > 1,760 m from the road	*(73)*
Several types, from highways to unpaved roads	Lentic water bodies including ponds, lakes, dams, and quiet pools within streams (S Victoria, Australia)	Traffic noise effect on the pitch of advertisement calls in two species of frogs, the southern brown tree frog (*Litoria ewingii*) and the common eastern froglet (*Crinia signifera*)	Tree frogs call at a higher pitch in traffic noise and shift the call frequency	Maximum noise at 40 m from highway	*(74)*
Paved roads	Various, review of 25 studies on 13 raptor species	Raptor nest location	Meta-analysis showed an overall positive impact on the displacement of nests from roads Big raptors nesting in trees exhibited greater displacement distances from nests to roads than big raptors nesting in cliffs Distance from nests to roads increase 20–30% compared to control random points	The absolute magnitude of the displacement distance of raptor nests ranged between 200 and 800 m from the road, and 1,400 m for tree nesting raptors of big size, such as large eagles and vultures	*(75)*
Highway and railway line	Mixed woodland (Buunderkamp, Netherlands)	Traffic noise and effects on vocal activity and reproductive success of great tits (*Parus major*)	Traffic noise strongly decreased with distance from the motorway and varied with the time of day, season and weather conditions Noise levels affected negatively the reproductive success of great tits (smaller clutches and fewer fledged chicks in noisier areas)	Average drop of 20 dB SPL in sound levels over less than 500 m from the road Over 400 m from the motorway, mainly bird vocal activity influenced variation in sound levels in the 4 kHz band	*(76)*
Highway	Road verges, bushes, open fields, intermittent trees, woodland (UK)	Bat activity and diversity	Total bat activity, the number of species and the activity of *Pipistrellus pipistrellus* (the most abundant species) were all positively correlated with distance from the road	Activity and diversity increased up to 1.6 km either side of the road	*(77)*
Several road types (paved roads, gravel roads, unimproved roads, truck trails and ATV trails)	Montane ecosystem (Rocky Mountains, Canada)	Alteration of red deer (*Cervus elaphus*) behavior	Deer close to roads decreased their feeding time and increased vigilance and time spent travelling More evident when traffic surpasses 12 vehicles per day	Switch into a more-alert behavior closer than 500 m to roads with more than 12 vehicles/day Twice longer foraging bouts, 20% increase in feeding time, 23% vigilance decrease and 10% decrease in travelling	*(78)*

				time in deer >1 km from roads	
Forest and main roads	Fir–beech forests (Dinaric Mountains, Slovenia)	Home-range size of red deer (*Cervus elaphus*)	Home-range size increased as the distance of main roads from the edge of the home range increased	Home range stabilizes at ca 1,800 m from the road	*(79)*
Highway and dirt roads	Tropical forest in metropolitan area (SE Brazil)	Scavenger removal of experimentally-placed carcasses	High carcass removal for both road categories, with a peak during the day on the highway and at night on dirt roads	Road-effect zone as assumption: >1 km from the highway there is no effect of highway on the carcass removal rate in dirt roads	*(80)*
Forest roads	Scrublands and oak and mixed forests, and portions of natural grasslands, and agricultural areas (central and northern Greece)	Rendezvous site selection by wolves (*Canis lupus*)	Rendezvous sites were located away from forest roads (most important factor at home-range scale)	Wolves selected rendezvous sites farther from forest roads (mean= 435 m, range=73–1,614 m)	*(81)*
Paved and unpaved roads for visitors use	Open grasslands, bush, savanna and woodlands (Kruger National Park, South Africa)	Behavioral response and local spatial distribution of impala (*Aepyceros melampus*)	Impalas change their local spatial distribution near paved and well-traveled roads; unpaved roads largely unaffected their local distribution Greater tolerance distances on paved roads compared to unpaved roads. More flight response in unpaved roads Few flight response (19.5%); habituation may exist	Mean flight distance from the road 30.5 m (range 0–154) vs 35.0 m (range 0–215) for those animals that did not respond. Animals avoid close proximity (first 10 m) to paved roads	*(82)*
REDUCTION OF SPECIES RICHNESS AND DIVERSITY					
Two-lane roads	Mosaic of forest, shrubland and pastures, among 12 cities and close to cities (NW Madrid, Spain)	Abundance and species richness patterns of the native avifauna in fragmented landscape	Total number of bird species, total bird abundance and number of threatened species was negatively influenced by the distance to the nearby roads The abundance of urban-exploiter bird species increased closer to roads	In general, significant threshold distances averaged 300 m for roads, but varied among parameters Mean species richness was lowest <110 m from the road and highest >1,030 m Number of threatened species decreased <400 m from road Highest bird abundance at 290-540 m from the road in deciduous forest areas Abundance of urban exploiters increased if roads <510 m	*(83)*
Paved roads	Wetlands (Southern	Richness of four different wetland	Plant, bird, and herptile species richness diminishes	Strongest relationships at distances up to 1,000 to	*(84)*

	Ontario, Canada)	taxa (birds, mammals, herptiles, and plants)	with increasing density of paved roads on adjacent lands	2,000 m from the wetland edge Critical distance for plants is between 1 and 2 km from the wetland edge; for birds, between 0.5 and 1 km, and for herptiles and mammals at least 2 km	
Unpaved forest roads	Forest (S Appalachian Mountains, Tennessee, USA)	Abundance and richness of the macroinvertebrate fauna of the soil and leaf-litter depth	Reduced both the abundance and the richness of the macroinvertebrate soil fauna and the depth of the leaf-litter	Effects on faunal abundance and leaf-litter depth up to 100 m into the forest (max distance tested), whereas persists to 15 m	*(85)*
Unpaved forest roads	Temperate deciduous forest (USA)	Change in the distributions of understory plants, and site variables (species cover, canopy cover, litter depth and cover, and bare ground)	Richness and diversity of native species were lower on roadsides Exotic species were most prevalent near roads Roads created a disturbance corridor that affected site variables	Native species richness back to normal levels after 5 m distance Prevalence of exotic species and effects on site variables up to 15 m	*(86)*
Highways (plus other anthropogenic barriers)	Desert regions (California, USA)	Genetic diversity in metapopulation of desert bighorn sheep (*Ovis canadensis nelson*)	Reduction in the relative gene flow among study populations Decline in genetic diversity at a rate of 0.4% per year	Barrier effect distance (at which relative gene flow decrease equivalently) estimated at c. 40 km	*(87)*
Several road types (highway, paved rural road, unpaved dirt road)	Second-growth forest (Orange County, New York, USA)	Diversity, abundance and species density of carrion beetles	No consistent effects of distance from road on the diversity, abundance or species density of beetles across road types Forests near highways and paved rural roads were less diverse than near dirt roads	No effect up to 120 m from the roads (suggestion that road effect can permeate further)	*(88)*
Highway	Rural area (Ontario, Canada)	Anuran species richness and relative abundance for seven species	Species richness and abundance declined closer to the road Suggestion that new roads should be at least 500 m from wetlands (conservative estimate of the road-effect zone for species richness), but greater buffer distances recommended (at least 3,000 m for leopard frogs *Rana pipiens*)	Road-effect zones of 250–1,000 m for four of seven species and species richness, and well beyond 1,000 m for two species. Breakpoint at approximately 450-800 m from the highway for species richness; 200-300 m for the spring peeper (*Pseudacris crucifer*), American toad (*Bufo americanus*), and gray treefrog (*Hyla versicolor*); 600–1,000 m for the wood frog (*Rana sylvatica*); and 1,100 to 2,400 m for the chorus frog (*Pseudacris triseriata*)	*(89)*

High-traffic paved roads	Boreal forest (Canada)	Change in breeding bird occurrence	Bird species richness increased with increasing distance from roads Traffic noise declined with distance from the roads	Bird species richness reached a maximum at about 350 m from the road Traffic noise reached a minimum at about 450 m from the roads	*(90)*
Low-traffic unpaved roads	Tropical rainforest (Amazon, Ecuador)	Change in species richness and diversity of amphibians, butterflies and birds	Amphibian richness and understory bird richness and diversity decreased near roads Butterfly and overall diurnal bird richness increased near roads Taxon-specific response to roads	Up to 200 m from the road for butterflies, up to 250 m for amphibians and up to 350 m for birds	*(91)*
PROMOTION OF INVASIVE SPECIES					
Paved roads	Grasslands (California, USA)	Native and exotic plant diversity	In non-serpentine grasslands the percentage cover by native species, the percentage of species that were native, and the number of native grass species increased with distance from roads, while the cover by exotic species and number of exotic forb species decreased No effect of road proximity in serpentine grasslands	Native cover was greatest in sites >1,000 m from roads (23%) and least in sites 10 m from roads (9%) Percentage of species that were native was significantly greatest in sites >1,000 m from roads (44%) and least in those 10 m from roads (32%)	*(92)*
Paved roads	Grasslands (California, USA)	Survival and biomass of the invasive plant yellow starthistle (*Centaurea solstitialis*)	In non-serpentine grasslands, *Centaurea* survival and biomass was greater in sites closer to roads No effect of road proximity on the performance of planted *Centaurea* on serpentine soil	Survival and biomass greater in near (10 m) than in distant (>1,000 m) plots	*(93)*
All types, from highways to dirt roads, typically two-lane dirt and paved roads	Mature sugar maple-dominated forests (W Great Lakes, Minnesota and Wisconsin, USA)	Extent and patterns of earthworm invasion	Distance to the nearest road was the best predictor of earthworm invasion in Wisconsin Negative relationship between the distance to the nearest road and the presence of four taxonomic groups, except *Dendrobaena* which had positive	The invasion of the *Lumbricus–Aporrectodea* assemblage generally extends nearly 1,200 m from roads. The probability of occurrence does not decline below 50% until 470-930 m, and to 5% until the nearest road is > 1,300 m away Probability of finding *Dendrobaena* alone increases with road distance crossing 50% at >1,540 m.	*(94)*
Paved and forest roads	Deciduous forest (Maryland, USA)	Presence and percent cover of invasive plant species	More invasive species close to roads; sites containing three or more invasive species observed along paved roads Spread rates are higher in roadsides; roadside populations occupied a larger patches and expand more	Effects measured up to 150 m from the road; the range of influence is greater following the spread of the species	*(95)*

			rapidly		
High, medium and low traffic roads	Dry deciduous forest (India)	Presence of invasive plants	Increase in the presence of invasive plant species near roads, especially in medium and high traffic roads	Up to 100 m (not measured further)	*(96)*
Primary roads	Terrestrial, freshwater and marine ecosystems (NW Europe, encompassing Great Britain, France, Netherlands and Belgium)	Distribution of invasive species (72, including 17 terrestrial plants, 19 terrestrial animals, 17 freshwater and 19 marine organisms)	Roads promote the dispersal of non-native species Proximity to roads was a particularly important driver for plant species distribution	Maximum probability of invasion of two plants, the Kudzu (*Pueraria lobata montana*) and Kahili ginger (*Hedychium gardnerianum*) within 2 km from roads	*(97)*
INDUCING DEFORESTATION					
Highways	Tropical rainforest (Amazon, Brazil)	Deforestation through forest conversion to crops, pastures and secondary forest	Deforestation has claimed 29-58% of the forests within 50 km of paved roads	More than two-thirds of Amazon deforestation within 50 km of major paved highways	*(98)*
Highways and unpaved roads	Tropical rainforest and adjoining woodlands and savannas (Amazon, Brazil)	Deforestation	Proximity to roads, particularly to highways, increased deforestation	Deforestation rose mostly sharply within 50-100 km of highways and within 25-50 km of unpaved roads	*(99)*
Paved and unpaved roads	Tropical rainforest (Amazon, Brazil)	Deforestation spillover	Deforestation rises in sites that lack roads but are in the same county as site with a new paved or unpaved road	100 km	*(100)*
State and federal roads, some private roads	Tropical rainforest (Amazon, Brazil)	Deforestation fires (measured by hot pixels)	Exponential declines in hot pixel frequency with increasing distance from roads Fewer deforestation fires within protected areas than outside	Almost 90% fires were ≤10 km from roads	*(101)*
Paved and unpaved roads	Tropical rainforest (Southern Amazon, Peru, Brazil, Bolivia)	Deforestation	Deforestation rates drop with distance from major roads, although the distance before this drop off appears to relate to degree of road paving at regional level	45 km for roads where paving is complete; 18 km where paving is underway	*(102)*
Highway	Cerrado Savannas (Brazil)	Deforestation and habitat degradation	Deforestation increases closer to the roads, with pasture growing near the road, and forest cover growing further away	32.6% loss of Cerrado up to 9 km from the highway	*(103)*
Official and unofficial roads	Tropical rainforest (Amazon,	Deforestation	Deforestation was much higher near roads Protected areas near roads had	Nearly 95% of all deforestation occurred within 5.5 km of roads	*(104)*

	Brazil)		lower deforestation than did unprotected areas near roads	Highways begin to have a rapidly diminishing influence only at 32 km	
CHANGE OF LANDSCAPE PATTERNS AND FRAGMENTATION					
All road network, mainly composed of minor roads	17 townships across three ecoregions of forested landscapes (n. Wisconsin, USA)	Changes in landscape patterns and road density in a six-decade study period	Substantial changes in landscape patterns Road density doubled and the immediate area affected by roads increase twofold (5% to 10%). Reduction of median, mean and largest roadless patch size by a factor of four. Increases in housing density and fragmentation	Road-effect zone as assumption: 15 m	*(105)*
FACILITATION OF RESOURCE EXTRACTION AND HUNTING					
Road for oil extraction and access from rivers	Amazon Basin (Yasuní Biosphere Reserve, Ecuador)	Probability of hunting by the Waorani indigenous group	Spatial extent of hunting doubled in the presence of road, and include remote areas	Mean distance walked from a point of access (road, river) to a kill site was 1.36 km (SD=1.18), and the maximum distance was 7 km (99% records <5 km)	*(106)*
NOISE INCREASE					
Busy roads (and other sources of noise)	Various (review paper)	Effect of noise (sound pressure level) on response curve of species occupancy (general model)	Spatial propagation of elevated noise levels from a point source (such as a single car, which decays at a spreading loss of 6 dB or more per doubling of distance, line sources (such as a busy highway) lose only 3 dB per doubling of distance	The sound pressure level of noise decreases with increasing distance but may not reach "baseline" ambient levels until ~1 km away (this distance will vary depending on noise source and the environment)	*(107)*
VARIOUS					
Highway	Suburban landscape, including swamps, streams, wetlands, deciduous forest, open-fields, residential areas (Massachusetts, USA)	Alteration of streams, wetland drainage, road salt reaching water bodies, invasion by exotic species, changes in habitat and movement patterns of large mammals such as moose *Alces alces* and deer *Odocoileus virginianus*, forest and grassland birds, and amphibians	The effects of all factors extended >100 m from road. Moose corridors, road, avoidance by grassland birds and road salt extended >1 km	The road-effect zone averages approximately 600 m wide and is asymmetric	*(108)*
Highways, secondary and	Various, all	Estimation of the percentage of	One-fifth of the U.S. land area is ecologically affected by	Road-effect zone as	*(109)*

primary roads	USA	land ecologically affected by the public road system	public roads system	assumption: primary roads (10,000 vehicles/day): 305 m in woodland and 365 m in grassland primary roads (50,000 vehicles/day): 810 m in natural ecosystems in urban areas secondary roads: 200 m	

Table S3. Extent and amount of roadless areas (5-km-buffer) per continent using the OpenStreetMap data set (11/2003) (without Antarctica, Greenland, and freshwater bodies).

	Asia	Africa	North America	South America	Europe	Australia	Oceania	Global land
Total area (million km²)	44.32	29.70	21.51	17.64	9.75	7.64	0.43	130.00
Total roadless area cover (million km²)	28.62	19.36	9.88	11.09	1.30	5.09	0.11	75.45
Percentage of roadless coverage (%)	64.58	65.19	45.93	62.89	13.33	66.62	25.58	58.04

Table S4. Extent and amount of roadless areas (1-km buffer) per continent using the OpenStreetMap data set (11/2003) (without Antarctica and Greenland, and freshwater bodies).

	Asia	Africa	North America	South America	Europe	Australia	Oceania	Global land
Total area (million km²)	44.32	29.70	21.51	17.64	9.75	7.64	0.43	130.00
Total roadless area cover (million km²)	38.83	26.53	13.20	15.52	4.06	6.75	0.27	105.16
Percent roadless cover	87.60	89.30	61.39	88.00	41.64	88.26	63.87	80.28
Mean roadless area patch size (km²)	308.69	522.51	59.69	418.07	47.85	248.58	47.85	176.94
Maximum roadless patch size (million km²)	4.23	2.88	3.33	4.82	0.24	0.27	0.03	4.82
Median roadless patch size (km²)	2.85	6.75	0.48	4.81	0.85	2.98	0.84	1.07
Total no. roadless patches	101,992	50,770	221,197	37,124	153,323	24,216	5,691	594,312
No. roadless patches >1 km²	63,555	36,223	86,112	24,817	73,148	15,673	2,699	302,227
No. roadless patches >5 km²	43,854	27,237	36,787	18,420	40,268	10,178	1,463	178,207
No. roadless patches >10 km²	35,274	22,864	23,502	15,431	28,363	7,782	1,073	134,289
No. roadless patches >50 km²	18,356	12,992	7,609	9,189	9,561	3,223	453	61,383
No. roadless patches >100 km²	13,124	9,505	4,580	6,893	5,210	2,055	295	41,662
No. roadless patches >1000 km²	3,077	2,187	769	1,653	432	539	49	8,706

Table S5. Rationale of indicators used for *Ecological Value Index of Roadless Areas* (EVIRA).

Indicators	Rationale	Description
Roadless area patch size	Large roadless areas provide a much wider range of ecological benefits than smaller ones where road edge effects impact a larger share of the roadless patch (see Table S2).	Habitat fragmentation and corresponding negative environmental changes have been extensively treated in many studies (a comprehensive overview is given by Bennett et al. (2010) *(110)*. The impacts do not just relate to gene flow, population viability and loss of (less dispersive) species in habitat fragments, but also to ecosystem functioning. For example, there is certain evidence related to nutrient cycling, dung removal, pollination, and seed dispersal *(111)*. "The impacts of fragmentation on ecosystem functioning are often exacerbated by synergistic effects such as interactions with the matrix and increased hunting pressure in fragmented forests" *(111)*. There is growing evidence that certain species avoid areas with even minimal anthropogenic disturbance *(112, 113)*, which is another argument for conservation of large roadless areas. Especially in tropical regions, many species exist at rather low population densities, are seasonal migrants (often across different altitudinal belts) following scarce resources, or otherwise require large habitats for maintaining viable populations *(114-116)*.
Thiessen connectivity into all directions for roadless area patches	The larger the Thiessen connectivity value, the closer neighboring roadless patches can be found. This is important for the integrity of ecological landscape-scale processes (e.g., genetic exchange of populations confined to roadless areas).	Roaded forest ecosystems, for instance, are far more vulnerable than intact ones to predatory logging, wildfires, illegal mining, exotic species invasions, and other anthropogenic threats *(7, 114)*.
Ecosystem Functionality Index	Ecosystem Functionality is defined as the state of ecosystems, characterized by inherent structures, ecological functions and dynamics, that provide ecosystems with both, the necessary efficiency and resilience to develop without abrupt change of system properties and geographical distribution, and allows for flexible response to external changes.	This Ecosystem Functionality Index has been published by Freudenberger et al. (2012a) *(38)*.
comprising the following sub-indicators:		
- Vegetation density	Vegetation density is an indicator for biomass and the ecosystems' ability to dissipate incoming solar energy. Furthermore, a higher number of primary producers increase the capture of solar energy thereby improving ecosystem functionality.	Rationale from Freudenberger et al. (2012a, b) *(9, 38)*. Further references and sources provided in the corresponding methods sections.
- Tree height	Tree height is used as an indicator for biomass as well as structural complexity of an ecosystem. Old-growth forest conditions and complex vegetation stratification including foliage layering is dependent on tree height, thereby enhancing biodiversity and ecosystem functioning. Furthermore, it plays an important part in the absorption of solar radiation and in moderating microclimatic conditions.	Rationale from Freudenberger et al. (2012a, b) *(9, 38)*. Further references and sources provided in the corresponding methods sections.
- Carbon storage	Carbon storage is considered as an indicator for biomass and the ability of ecosystems to dissipate incoming solar energy. Areas with	Rationale from Freudenberger et al. (2012a, b) *(9, 38)*. Further references provided in the corresponding methods sections.

	higher carbon storage are also characterized by more intensive interactions with the atmosphere and higher regulating capacity.	
- Species richness of vascular plants	Species richness is considered to represent functional and structural redundancy, which is relevant for the resistance and resilience of ecosystems to e.g. climate change. Additionally, species richness is also associated with complex trophic structure and higher cycling rates of biomass, energy and information.	Rationale from Freudenberger et al. (2012a, b) *(9, 38)*. Further references and sources provided in the corresponding methods sections.
- Plant functional richness	Plant functional richness is an indicator derived from modelling survival probabilities of different plant functional types under climate change. Ecosystems with higher functional species richness are more likely to adapt to environmental change and therefore increase the adaptive capacity of an ecosystem.	Rationale from Freudenberger et al. (2012a, b) *(9, 38)*. Further references and sources provided in the corresponding methods sections.
- Slope	Topographical heterogeneity is connected to habitat diversity and species richness. At macro-scale habitat diversity increases along altitudinal gradients. Geographical barriers increase opportunities for allopatric speciation, and contribute to the genetic information that is stored within an ecosystem.	Rationale from Freudenberger et al. (2012a, b) *(9, 38)*. Further references and sources provided in the corresponding methods sections.

Table S6. Pearson (dark grey) and Spearman rank (light grey) correlation coefficient matrix for the three indicators of the ecological value index for roadless areas (EVIRA). All correlation coefficients are highly significant with p<0.0001. Correlation coefficients with values higher than 0.7 are displayed in bold.

	Ecological value index of roadless areas (EVIRA)	Roadless area patch size	Thiessen connectivity into all directions	Ecosystem functionality index (EFI)
Ecological value index of roadless areas (EVIRA)	1.000	**0.768**	-0.005	**0.818**
Roadless area patch size	0.488	1.000	-0.006	0.260
Thiessen connectivity into all directions	-0.272	-0.875	1.000	-0.002
Ecosystem functionality index (EFI)	**0.881**	0.155	0.048	1.000

Table S7. Distribution of roadless areas (1-km buffer) across anthromes (km^2) (according to Ellis et al. 2010 (*10*); analysis based on OpenStreetMap data set 11/2013).

Anthrome classes	South America	Central and North America	Europe	Asia	Africa	Australia	Oceania	Global	Share of global roadless areas (%)
Urban	4,007	4,387	2,374	32,332	9,058	706	263	53,129	0.05
Mixed settlements	18,372	18,749	5,295	233,664	93,038	1,070	1,556	371,746	0.36
Rice villages		444		1,561,288	358			1,562,090	1.50
Irrigated villages	9,099	18,415	8,092	917,304	31,193			984,105	0.94
Rainfed villages	48,983	70,791	48,853	1,307,198	514,561		85	1,990,474	1.91
Pastoral villages	67,829	16,127	1,748	233,641	195,302			514,649	0.49
Residential irrigated croplands	34,121	50,856	52,030	401,213	47,493	497	191	586,40	0.56
Residential rainfed croplands	453,081	324,541	779,233	2,209,022	1,853,242	7,405	6,051	5,632,575	5.39
Populated croplands	567,180	302,940	531,100	1,484,977	606,286	70,433	15,408	3,578,326	3.43
Remote croplands	161,957	345,517	21,507	360,306	135,530	391,144	7,822	1,423,783	1.36
Residential rangelands	1,252,057	177,381	62,984	1,404,975	3,314,670	9,205	2,844	6,224,116	5.96

Populated rangelands	2,800,656	572,493	261,741	3,430,646	4,634,380	67,350	27,188	11,794,455	11.29
Remote rangelands	2,214,349	737,996	94,936	5,999,912	2,294,862	6,047,983	76,368	17,466,406	16.72
Residential woodlands	230,507	141,898	106,706	1,322,994	1,343,634	4,246	20,478	3,170,464	3.04
Populated woodlands	1,464,277	490,479	709,214	2,397,132	2,134,048	29,333	60,523	7,285,006	6.97
Remote woodlands	2,182,821	485,807	201,057	1,241,981	448,189	29,731	27,679	4,617,265	4.42
Inhabited treeless and barren lands	781,593	248,646	49,804	2,183,217	1,665,865	508	1,056	4,930,688	4.72
Wild woodlands	2,710,257	5,929,872	829,528	7,534,326	332,290	71,611	17,868	17,425,751	16.68
Wild treeless and barren lands	484,370	2,976,033	171,235	4,345,674	6,858,975	1,771	444	14,838,501	14.21

Table S8. Protection status of roadless areas (1-km buffer) per continent (without Antarctica, Greenland, and large freshwater bodies) based on WDPA 2014 and OpenStreetMap (11/2003).

	Asia	**Africa**	**North America**	**South America**	**Europe**	**Australia**	**Oceania**	**Global land**
Protected areas cover (all categories) (km^2)	4,977,721	4,112,914	2,646,754	4,087,773	1,510,183	1,196,688	93,123	18,625,157
Protected area cover (%)	11.2	13.8	12.3	23.2	15.5	15.7	21.8	14.2
Roadless areas in IUCN categories (km^2)	3,989,458	2,056,657	2,146,627	2,364,065	410,437	1,074,445	72,177	12,113,866
Percent IUCN coverage of roadless areas	9.0	6.9	10.0	13.4	4.2	14.1	17.0	9.3
Strictly protected areas (IUCN I & II) (km^2)	1,029,356	1,028,218	1,511,100	997,502	272,877	589,763	33,848	5,462,664
Strictly protected areas (IUCN I & II) (%)	2.3	3.5	7.0	5.7	2.8	7.7	7.9	4.2
Roadless areas in strictly protected areas (IUCN I & II) (km^2)	966.322	969.151	1.370.853	974.208	180.903	525.068	28.492	5.014.999

Roadless areas strictly protected (IUCN I & II) (%)	2.2	3.3	6.4	5.5	1.9	6.9	6.7	3.8
Protected areas (IUCN III-VI) (km²)	3,215,796	1,194,583,55	1,006,467,51	1,450,552,58	701,944,89	581,476,89	54,291,11	8,205,112,91
Protected areas (IUCN III-VI) (%)	7.3	4.0	4.7	8.2	7.2	7.6	12.7	6.3
Roadless areas in protected areas (IUCN III-VI) (km²)	3,023,136	1,087,506	775,773	1,389,857	229,534	549,377	43,683	7,098,867
Roadless areas in protected areas (IUCN III-VI) (%)	6.8	3.7	3.7	7.9	2.3	7.2	10.2	5.4

Table S9. Extent and coverage of roadless areas of 1-km buffer under strict protection (IUCN I-II) category, according to their Ecological Value Index of Roadless Areas (EVIRA) using the OpenStreetMap data set (11/2003).

EVIRA values	North America (km^2)	South America (km^2)	Asia (km^2)	Africa (km^2)	Europe (km^2)	Australia (km^2)	Oceania (km^2)	Global (km^2)
0 - 13	0	0	0	0	0	0	0	0
14 - 28	109,7	8,0	5,430	6,092	1,700	50,525	2.2	63,868
29 - 33	86,441	9,367	98,425	269,842	2,042	274,650	855	741,622
34 - 37	81,286	20,640	108,467	201,490	13,496	82,089	36	507,500
38 - 42	75,476	45,810	81,685	240,560	44,801	29,444	106	517,883
43 - 47	454,357	64,917	100,975	85,371	66,762	23,597	417	796,396
48 - 53	204,952	151,089	173,866	50,750	40,796	11,856	15,446	648,755
54 - 58	444,939	132,629	147,985	88,619	7,878	34,984	8,074	865,107
59 - 64	17,582	31,144	105,544	25,579	2,437	16,871	3,466	202,623
65 - 80	3,617	518,198	143,008	0.0	227	82	0.3	665,132
Sum	**1,368,760**	**973,802**	**965,384**	**968,299**	**180,140**	**524,099**	**28,401**	**5,008,886**

Table S10. Synergies and conflicts between conservation of roadless areas and the United Nations' Sustainable Development Goals (SDGs) and their corresponding targets. Left column: Assessment of goals (large boxes): grey: at most weak synergies and conflicts with goal, blue: conflicts with goal prevail, yellow: mixture of synergies and conflicts with goal, green: synergies with goal prevail. Assessment of targets (insert boxes): grey: not applicable, blue: conflict, yellow: ambivalent relationship, green: synergy. Numbers in italics: target numbers. Bold number at bottom: conflict-synergy score of goals. →: reference to target(s).

Sustainable Development Goals and targets	Brief analysis of synergies and conflicts between conservation of roadless areas and Sustainable Development Goal targets
Goal 1. End poverty in all its forms everywhere *1* *2* *3* *4* *5* **-0,5** Compare AICHI BIODIVERSITY TARGETS 2, 14.	**Synergies:** The SDGs explicitly acknowledge the importance of integrating ecosystem and biodiversity values into poverty reduction strategies and accounts (compare to 15.9). In remote areas inhabited by indigenous or traditional people in the developing world, where governance is weak, road development may trigger uncontrolled frontier expansion and associated poverty. In the Amazon, frontier expansion through road construction has fostered large-scale economic activities (e.g. oil extraction, livestock and soy production), but often at the expense of the local communities. Road development in the region is associated to dire conflicts over land and natural resources *(117, 118)*. A better planning of the road development process and a prioritization of roadless areas for conservation purposes can help to reduce risks related to poverty (→ targets 1.1, 1.2, 1.4). In the Amazon, for instance, a more sensitive proposed development strategy should focus on strengthening governance in areas where roads have been established for a long time (and human population is relatively large and human development indices are low), while leaving more remote areas roadless or with roads unpaved *(119)*. Functional ecosystems, as they exist in roadless areas, effectively reduce human exposure to environmental shocks and disasters, including climate-related extreme events (such as floods: e.g., *(120)*, water scarcity: e.g., *(121)*, compare goal 6, fires: e.g., *(122);* → target 1.5). It is of great importance to maintain ecosystem functionality on the landscape scale, e.g. by prioritizing conservation of roadless areas around the headwaters of rivers against extreme fluctuations in run-off along the densely populated and intensively managed tailwater.
	Conflicts: Poverty often is related to the lack of access to markets and employment options (compare goal 8), health (compare goal 3) and education infrastructure (compare goal 4; *(123-126)*). Case studies have shown how roads significantly reduce poverty and increase consumption growth (→ targets 1.1, 1.2, 1.4; *(127-129)*). Reduced mobility also hampers organizational capacities, especially in remote rural areas, where it is difficult for poor people to meet and coordinate activities. In general, poor people will ask for better roads and mobility. Goal 9 explicitly addresses the relevance of infrastructure (see below). The conservation of roadless areas seems to represent a serious conflict and obstacle to development – if this development is thought along conventional lines and without exploring more sustainable alternatives for providing mobility.

Goal 2. End hunger, achieve food security and improved nutrition and promote sustainable agriculture *1* *2* *3* *4* *5* **-0,6** Compare AICHI BIODIVERSITY TARGETS 7, 8, 14.	**Synergies:** In remote regions, as they are found in parts of the western Amazon forests, the subsistence of many indigenous communities depends on forest products. However, new roads built into previously remote areas of low human population density have often triggered conversion of forest to croplands and pastures *(130)* and unsustainable exploitation of wildlife that can then be marketed easily as bushmeat in cities. Bushmeat can thus become scarce for residents who rely on this protein source *(131, 132)*. Functional ecosystems, as they exist in roadless areas, effectively reduce human exposure to environmental shocks and disasters, including climate-related extreme events (→ target 2.4; compare goal 1).
	Conflicts: At many places of the world, undernourishment increases with distance from roads and with it from markets and health services, among others *(133)*. Hunger can also be promoted by limited options for reaching poor rural people with food aid and development assistance (*(134)*; → targets 2.1-2.3, 2.5; compare goals 1, 3, 4, 6, 9).
Goal 3. Ensure healthy lives and promote well-being for all at all ages *1* *2* *3* *4* *5* *6* *7* *8* *9* **-0,1** Compare AICHI BIODIVERSITY TARGETS 6, 14.	**Synergies:** In general, roadless areas guarantee high ecosystem functionality (compare goal 1) and with it a variety of ecosystem services that are fundamental to people's health. Among others, tropical forest-dwelling indigenous communities use a variety of medicinal forest plants that can become scarce in the course of road construction and subsequent deforestation *(135)*. Roadless areas exclude deaths and injuries from road traffic accidents (→ target 3.6) as well road and traffic-related hazardous chemicals and air, water and soil pollution and contamination (*(136, 137)*; → target 3.9; compare goal 6). Road development in the Amazon and Indonesia has been shown to be associated with the spread of diseases (*(117);* → target 3.3). Abrupt contact with modern life-styles via new roads increases the vulnerability of formerly remote human populations to drug abuse and alcohol consumption (*(138);* → target 3.5).
	Conflicts: Remote rural populations mostly have reduced access to health care and medical assistance (*(133);* → targets 3.1, 3.2, 3.4, 3.7, 3.8).
Goal 4. Ensure inclusive and equitable quality education and promote lifelong learning opportunities for all *1* *2* *3* *4* *5* *6* *7* **-0,9** Compare AICHI BIODIVERSITY TARGETS 1.	**Synergies:** Experiencing wilderness has become an important element of education. While roadless areas are less accessible by motorized ways, they provide opportunities for this kind of education (*(139)* compare goal 8: nature tourism).
	Conflicts: With increasing distance from roads, access to "quality" education becomes more difficult. Among others, remote rural populations often lack literacy in the use of emerging technological devices (computers, internet etc., *(140);* → targets 4.1-4.7).
Goal 5. Achieve gender equality and empower all women and girls	**Synergies and conflicts: -**
Goal 6. Ensure availability and sustainable management of water and sanitation for all *1* *2* *3*	**Synergies:** Roads significantly harm the integrity and functionality of ecosystems and several services they provide to people (compare goal 1). Roads (including their construction) and traffic have been known for a long time as a source for water pollution (*(141);* → targets 6.1, 6.3, 6.5, 6.6).

4 *5* *6* **0,4** Compare AICHI BIODIVERSITY TARGETS 6, 8, 14.	**Conflicts:** In general, remote rural populations often have reduced access to technology, infrastructural development and assistance. It is cost-efficient, and practical for maintenance, to install water and sewer systems in the course of road construction (→ targets 6.1, 6.2).
Goal 7. Ensure access to affordable, reliable, sustainable and modern energy for all *1* *2* *3* **-0,5** Compare AICHI BIODIVERSITY TARGETS 7, 8, 14.	**Synergies:** none. **Conflicts:** In general, remote rural populations often have reduced access to technology and infrastructural development (compare goal 6). Electric wires can relatively easily be installed and maintained along roads (→ targets 7.1, 7.2). However, small-scale renewable (solar, wind) energy plants can be an alternative with additional advantages (low cost, energy autonomy; → target 7.2).
Goal 8. Promote sustained, inclusive and sustainable economic growth, full and productive employment and decent work for all *1* *2* *3* *4* *5* *6* *7* *8* *9* *10* **-0,3** Compare AICHI BIODIVERSITY TARGETS 2.	**Synergies:** Roadless areas can contribute substantially to slowing down environmental degradation (→ target 8.4; compare goal 15, 13). In addition, certain micro- and small enterprises can arise in spite of relatively great distances from roads (→ target 8.3) – or even depend on remoteness (nature tourism, e.g., *(142);* → target 8.9). It has been shown for the Amazon region that road development is associated with slave labor (*(118);* → target 8.7). Facilitated access to markets by roads may not always improve the income levels of poor people, as they will not be able to afford goods such as cars and petrol. **Conflicts:** Ease of mobility of people and goods promotes economic productivity and growth (*(143)*; → targets 8.1, 8.2; compare goals 9, 1). Young people of remote rural areas mostly have reduced access to good education and training opportunities (compare goal 4) and subsequently lower chances on the labor market (*(144);* → target 8.6).
Goal 9. Build resilient infrastructure, promote inclusive and sustainable industrialization and foster innovation *1* *2* *3* *4* *5* **-0,3** Compare AICHI BIODIVERSITY TARGET 2.	**Synergies:** Upgrades of roads in the existing network can be a cost-efficient and environmentally less problematic alternative to building new roads (*(4);* → target 9.4). **Conflicts:** Economic development, especially in developing economies or those in transition, depends on an effective road network (*(143)*; → targets 9.1, 9.2; compare goals 8, 1).
Goal 10. Reduce inequality within and among countries *1* *2* *3* *4* *5* *6* *7* **-0,7**	**Synergies:** none. **Conflicts:** Modern road traffic has increased the mobility of people and goods, but comes with an increased risk of accidents (*(145);* → target 10.7). Roads have a variety of homogenizing effects - in terms of biological diversity (e.g., aided dispersal of invasive species: *(146)*, culturally (*(147);* → target 10.2) etc. Economically, road building provides poor rural societies a better access to economic dynamics and is thus a standard element of economic development strategies (*(143)*; → target 10.1; compare goals 9, 8, 1).

Goal 11. Make cities and human settlements inclusive, safe, resilient and sustainable *1* *2* *3* *4* *5* *6* *7* **0,5** Compare AICHI BIODIVERSITY TARGET 14.	**Synergies:** "Indigenous peoples in voluntary isolation" request participation in road and human settlement planning and want to be exempted from any such development *(117)*. Targeting roadless areas will help concentrate development in urban areas and their immediate surroundings (*(105)*; → target 11.3). Failing to do so regularly results in "contagious development", i.e., unleashing a positive feedback of road construction and intensive land-use in a formerly road-free landscape *(4, 7)*. Remote areas, which provide vital ecosystem services to cities, can thus be kept functioning (→ target 11.5; compare goal 13, 1, 2). The status of natural heritage sites ("Criteria for the assessment of Outstanding Universal Value": vii, ix and x; *(148)*) is vitally coupled with remoteness (→ target 11.4).
	Conflicts: Further road construction may be deemed necessary to provide convenient access to public transport for a larger part of the population. However, people in remote rural regions may not be able to pay for public transport (*(149)*; → target 11.2).
Goal 12. Ensure sustainable consumption and production patterns *1* *2* *3* *4* *5* *6* *7* *8* **1,0** Compare AICHI BIODIVERSITY TARGET 4.	**Synergies:** Road construction and maintenance consume significant amounts of material (and energy) and thus enlarge the national and per capita material footprint (*(150)*; → target 12.2). Including roadless and other important areas for biodiversity and ecosystem services for people would make sustainability reports of companies *(151)* more diagnostic and could thus provide guidance for the adoption of sustainable practices (→ target 12.6).
	Conflicts: none.
Goal 13. Take urgent action to combat climate change and its impacts *1* *2* *3* **1,0** Compare AICHI BIODIVERSITY TARGETS 15, 10, 14.	**Synergies:** Functional ecosystems, as they exist in roadless areas, strengthen the resilience and adaptive capacity of human societies to climate-related hazards and natural disasters (→ target 13.1; compare goals 1-3). Roadless areas conservation would thus form a meaningful element of policies, strategies and planning for climate change adaptation (*(2)*; → target 13.2). Road construction and maintenance (with cement production being a relevant source of greenhouse gas emissions *(152)*) as well as traffic *(153)* also contribute large shares to overall greenhouse gas emissions. Policies, strategies and planning for climate change mitigation should therefore strive to reduce these activities to the lowest level possible (→ target 13.2).
	Conflicts: none.
Goal 14. Conserve and sustainably use the oceans, seas and marine resources for sustainable development *1* *2* *3* *4* *5* *6* *7* **1,0** Compare AICHI BIODIVERSITY TARGET 6.	**Synergies:** Considerable river sediment loads can result from road construction and erosion along roads *(121)*. Runoff from subsequent development, such as logging in mountain areas *(154)*, or agriculture, can also impact rivers and, finally, estuaries and near-coast marine waters (→ target 14.1).
	Conflicts: none.

Goal 15. Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss *1* *2* *3* *4* *5* *6* *7* *8* *9* **1,0** Compare AICHI BIODIVERSITY TARGETS 5, 11, 15, 12, 10.	**Synergies:** The conservation of roadless areas represents an effective and inexpensive means to conserving terrestrial and inland freshwater biodiversity and ecosystem services (*(2, 4);* → targets 15.1, 15.4, 15.5, 15.7, 15.8). This includes halting deforestation (*(98);* → targets 15.2) and combating desertification (*(155);* → targets 15.3). The inclusion of roadless areas would be a meaningful contribution to integrating ecosystem and biodiversity values into national and local planning as well as development processes, as is already the case in the United States of America and Germany (*(2, 4);* → targets 15.9). The present study demonstrates roadless areas are a tangible and transparent indicator for environmental accounting (→ target 15.9). **Conflicts:** none.
Goal 16. Promote peaceful and inclusive societies for sustainable development, provide access to justice for all and build effective, accountable and inclusive institutions at all levels *1* *2* *3* *4* *5* *6* *7* *8* *9* **1,0**	**Synergies:** Road development in the Brazilian Amazon is associated with an increase in homicide rate (*(118)*; → target 16.1). **Conflicts:** none.
Goal 17. Strengthen the means of implementation and revitalize the global partnership for sustainable development *1* *2* *3* *4* *5* *6* *7* *8* *9* *10* *11* *12* *13* *14* *15* *16* *17* *18* *19* **-1,0**	**Synergies:** none. **Conflicts:** Roads connect national economies (compare goal 8) and thus facilitate import-export traffic across borders (→ target 17.11), especially for landlocked regions or countries *(156)*.

Table S11. Synergies and conflicts between conservation of roadless areas and the United Nations' Aichi Strategic Goals and Biodiversity Targets. The color scheme indicates the level of synergy or conflict of goals and targets with roadless areas conservation (green: synergies prevail; grey: not applicable; yellow: ambivalent relationship). The numbers in insert boxes represent the conflict-synergy score of goals.

Aichi Strategic Goals and Biodiversity Targets	Brief analysis of synergies and conflicts between conservation of roadless areas and Aichi Biodiversity Targets
Strategic Goal A: Address the underlying causes of biodiversity loss by mainstreaming biodiversity across government and society **0,5**	
Target 1. By 2020, at the latest, people are aware of the values of biodiversity and the steps they can take to conserve and use it sustainably. Compare Sustainable Development Goal 4.	On the one hand, pristine ecosystems, such as they occur in roadless areas, are key for effective biodiversity conservation *(2)*. In agreement with modern concepts of sustainable land use, such as in biosphere reserves, these ecosystems are an essential element of sustainable use of the overall landscape *(157)*. Remote roadless areas provide opportunities for learning about natural ecosystems, i.e., wilderness (see goals B and C). On the other hand, roadless areas reduce accessibility of nature in general, thus making it more difficult to value biodiversity emotionally.
Target 2. By 2020, at the latest, biodiversity values have been integrated into national and local development and poverty reduction strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems. Compare Sustainable Development Goals 9, 8, 1.	While road infrastructure is related to economic growth and poverty alleviation *(158, 159)*, it has a crucial impact on biodiversity loss (see goal C), which in turn is directly linked with poverty aggravation *(160, 161)*. In remote areas inhabited mostly by indigenous or traditional people, road development may increase the spread of diseases, trigger conflicts over land and natural resources, and disrupt the traditional modes of production (which then have to compete with the global market), ultimately pushing these people towards poverty *(117, 162)*. The role of road development on poverty alleviation is hence conflicting, which calls for a better planning integrating roadless areas prioritization for biodiversity maintenance towards poverty alleviation.
Target 3. By 2020, at the latest, incentives, including subsidies, harmful to biodiversity are eliminated, phased out or reformed in order to minimize or avoid negative impacts, and positive incentives for the conservation and sustainable use of biodiversity are developed and applied, consistent and in harmony with the Convention and other relevant international obligations, taking into account national socio economic conditions.	Road transport receives between one- and two-thirds of worldwide conventional subsidies that are harmful in the long run to both the economy and the environment *(163)*. Road transport sector figures among the five most prominent sectors receiving such perverse subsidies *(164)*. An outstanding example refers to road infrastructure subsidies in the Amazon that have led to cattle ranching, extensive deforestation and biodiversity loss *(165)*. Alternatively, the integration of roadless areas into governmental policies could help in reducing and eliminating a substantial part of the harmful subsidies for the road transport sector.
Target 4. By 2020, at the latest, Governments, business and stakeholders at all levels have taken steps to achieve or have implemented plans for sustainable production and consumption and have kept the impacts of use of natural resources well within safe ecological limits. Compare Sustainable Development Goal 12.	Roadless areas, and relatively undisturbed areas in general, are of high resilience and ecosystem functionality *(2)*. Conserving these areas therefore contributes to maximizing ecosystem functionality of the wider landscape - they are an essential element of its sustainable use (compare targets 1, 7).
Strategic Goal B: Reduce the direct pressures on biodiversity and promote sustainable use **0,8**	
Target 5. By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced. Compare Sustainable Development Goal 15.	Road development is a major driver of habitat loss and fragmentation *(166)*. Roads act as barriers for species *(167)* and deforestation has been shown to increase along roads [*(98)*, Table S2]. Conserving roadless areas therefore directly helps to achieve this target.
Target 6. By 2020 all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem based approaches, so that overfishing is avoided, recovery plans and measures are in place for all depleted species, fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems and the impacts of fisheries on stocks, species and ecosystems are within safe	Roads facilitate the accessibility to remote terrestrial or freshwater ecosystems and increase the efficiency of natural resources exploitation and exportation, which are often depleted above their safe ecological limits *(1)*. For instance, a single road construction has been reported to have severe effect to a lake trout population, due to improved access for fishermen *(168)*. In addition, roads, their construction and traffic emit water pollutants *(137, 141)*. Similarly, road construction and roads can produce large sediment loads in rivers, particularly detrimental in wetlands and mountain areas.

ecological limits. Compare Sustainable Development Goals 14, 6, 3.	Roads also open up a landscape for logging and agriculture, and resulting runoff equally enters rivers *(154)*. Large part of this sediment ends up in estuaries and coastal waters.
Target 7. By 2020 areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity. Compare Sustainable Development Goal 2.	On one side, roadless areas exclude certain types of local development and even sustainable land use. And to keep up with demand for natural resources, any additional roadless area may require the intensification of land use in developed areas. On the other side, conservation of functional ecosystems, as they are still found in roadless areas, is essential for the larger landscape to stay functional. From this perspective, the remaining roadless areas can be seen as key elements of sustainably managed landscapes (compare targets 1, 4, 8).
Target 8. By 2020, pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity. Compare Sustainable Development Goals 6, 2.	Agricultural intensification might be necessary to make up for setting aside roadless areas (compare target 7). This might lead to increased use of fertilizers and pollution. It should be noted, however, that in many developing countries in particular there is a large amount of degraded land that can be restored and replace set-asides. However, conservation of roadless areas as relatively pristine ecosystems are a cost-efficient way of maximizing the provisioning of regulating ecosystem services such as nutrient uptake and water purification *(121)*.
Target 9. By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment.	Road density is a strong correlate of spatial patterns in biological invasions *(146)*. Limiting road development in roadless areas can, therefore, help to directly reduce the spread of invasive species (Table S2).
Target 10. By 2015, the multiple anthropogenic pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning. Compare Sustainable Development Goals 13, 15.	Roadless areas often represent areas with large carbon pools and sequestration potential. Furthermore, they represent areas of high ecosystem functionality important for climate regulation and long-term climate change adaptation. The conservation of roadless areas, thus, helps to mitigate and adapt to the impacts of climate change *(2, 4)*. Regarding marine ecosystems in particular, roadless areas prevent road-related sediment and agricultural runoff from impacting near-shore waters (compare target 6).
Strategic Goal C: To improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity **0,3**	
Target 11. By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes. Compare Sustainable Development Goal 15.	The conservation of roadless areas directly contributes to the conservation of valuable terrestrial ecosystems for biodiversity conservation. These areas also typically provide a wide array of ecosystem services, especially regulating services, and do this in large quantities. Furthermore, the conservation of these unfragmented and pristine areas directly contributes to the target of increasing connectivity. Conservation of the functionality of the watershed is highly dependent on the preservation of vegetation cover *(169)*, which benefits from conservation of roadless areas.
Target 12. By 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained. Compare Sustainable Development Goal 15.	Threatened species typical of anthropogenically disturbed ecosystems, such as old cultural landscapes in Europe and elsewhere, depend on certain semi-intensive, often historical, land use regimes (*170*). Therefore, in human-modified landscapes, the conservation of roadless areas in cases may be found little useful, or even counterproductive, to the target of improving the conservation status of some species. At the same time, other species (e.g., some amphibians) may experience reduced mortality in the absence of roads. After all, most threatened species are endangered by man-made loss of pristine ecosystems *(171)*. Roadless areas can retain populations of threatened species, supporting the native flora and fauna and buffering changes in the environmental conditions. Roadless areas which are large enough to host source populations can then serve as the origin for recolonization of areas where threatened species had disappeared *(172)*.
Target 13. By 2020, the genetic diversity of cultivated plants and farmed and domesticated animals and of wild relatives, including other socio-economically as well as culturally valuable species, is maintained, and strategies have been developed and implemented for minimizing genetic erosion and safeguarding their genetic diversity.	For one thing, on-farm conservation and use of cultivated species often requires the application of rather extensive agricultural practices *(173)*. This could lead to competition for area between the conservation of roadless areas and more extensive agricultural practices for the preservation of the diversity of cultivated plants and animals. Then again, wild relatives of domesticated plant and animal species can often only be found in pristine natural areas *(174)*.
Strategic Goal D: Enhance the benefits to all from biodiversity and ecosystem services **1,0**	
Target 14. By 2020, ecosystems that provide essential services, including services related to water, and	Functional ecosystems, as they exist in roadless areas, provide large quantities of many ecosystem services, especially of regulating services.

contribute to health, livelihoods and well-being, are restored and safeguarded, taking into account the needs of women, indigenous and local communities, and the poor and vulnerable. Compare Sustainable Development Goals 6, 11, 1, 2, 3, 13.	They effectively reduce human exposure to extreme environmental events [e.g., fires, *(122)*]. Remote areas are often also of high value especially to indigenous and traditional people *(117)*. Remote areas also provide vital ecosystem services to poor city dwellers, such as purification and stable provisioning of water *(121)*.
Target 15. By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification. Compare Sustainable Development Goals 15, 13.	Roadless areas comprise relatively little disturbed areas. Many of these harbor large carbon pools and sinks, e.g., peatlands and intact forests in tropical and boreal regions *(175)*. Furthermore, they provide many regulating ecosystem services and high ecosystem functionality and are, therefore, crucial for ecosystem-based adaptation to climate change (see above targets 1, 4, 7). They also provide a natural buffer against increasing desertification through maintenance of vegetation cover *(155)*.
Target 16. By 2015, the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization is in force and operational, consistent with national legislation.	
Strategic Goal E: Enhance implementation through participatory planning, knowledge management and capacity building **1,0**	
Target 17. By 2015 each Party has developed, adopted as a policy instrument, and has commenced implementing an effective, participatory and updated national biodiversity strategy and action plan.	
Target 18. By 2020, the traditional knowledge, innovations and practices of indigenous and local communities relevant for the conservation and sustainable use of biodiversity, and their customary use of biological resources, are respected, subject to national legislation and relevant international obligations, and fully integrated and reflected in the implementation of the Convention with the full and effective participation of indigenous and local communities, at all relevant levels.	Indigenous communities are most vulnerable to the impacts of road development. Road construction in former roadless areas can cause traditional environmental knowledge loss and even a depopulation of indigenous communities *(176)*. Indigenous people may lose their land *(177)*, or use it less after road construction *(178)*, benefit less from biological resources and face an alteration of traditional roles and practices *(179)*.
Target 19. By 2020, knowledge, the science base and technologies relating to biodiversity, its values, functioning, status and trends, and the consequences of its loss, are improved, widely shared and transferred, and applied.	Natural ecosystems, as they still exist in remote roadless areas, are unique learning sites not only for education (see above target 1). They also provide important insights into ecosystem properties and processes such as biomass stocks, ecological dynamics, or resistance and resilience to natural disturbances *(180)*.
Target 20. By 2020, at the latest, the mobilization of financial resources for effectively implementing the Strategic Plan for Biodiversity 2011-2020 from all sources, and in accordance with the consolidated and agreed process in the Strategy for Resource Mobilization, should increase substantially from the current levels. This target will be subject to changes contingent to resource needs assessments to be developed and reported by Parties.	

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Fig. S1. Geofabrik, http://www.geofabrik.de, OpenStreetMap ODbL. [Accessed (13/11/2013)].

Fig. S2. & Fig. S3. Nature Conservancy, Terrestrial Ecoregions: http://maps.tnc.org/gis_data.html [Accessed (15/08/2014)].

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Table S1. (a) Geofabrik, http://www.geofabrik.de, OpenStreetMap ODbL. [Accessed (13/11/2013)]. (b) D. Gaveau, S. Sloan, E. Molidena, H. Yaen, D. Sheil, N.K. Abram, M. Ancrenaz, R. Nasi, M. Quinones, N. Wielaard, E. Meijaard, Four decades of forest persistence, clearance and logging on Borneo. *PLoS ONE* **9(7)**, e101654 (2014). doi: 10.1371/journal.pone.0101654. (c) Center for International Earth Science Information Network - CIESIN - Columbia University, and Information Technology Outreach Services - ITOS - University of Georgia. 2013. Global Roads Open Access Data Set, Version 1 (gROADSv1). Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). http://dx.doi.org/10.7927/H4VD6WCT. [Accessed (06/10/2015)].

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SYNTHESIS & INTEGRATION

Spotted Owls and forest fire: a systematic review and meta-analysis of the evidence

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Abstract. Forest and Spotted Owl management documents often state that severe wildfire is a cause of recent declines in populations of Spotted Owls and that mixed-severity fires (5–70% of burned area in high-severity patches with >75% mortality of dominant vegetation) pose a primary threat to Spotted Owl population viability. This systematic review and meta-analysis summarize all available scientific literature on the effects of wildfire on Spotted Owl demography and ecology from studies using empirical data to answer the question: How does fire, especially recent mixed-severity fires with representative patches of high-severity burn within their home ranges, affect Spotted Owl foraging habitat selection, demography, and site occupancy parameters? Fifteen papers reported 50 effects from fire that could be differentiated from post-fire logging. Meta-analysis of mean standardized effects (Hedge's *d*) found only one parameter was significantly different from zero, a significant positive foraging habitat selection for low-severity burned forest. Multi-level mixed-effects meta-regressions (hierarchical models) of Hedge's *d* against percent of study area burned at high severity and time since fire found the following: a negative correlation of occupancy with time since fire; a positive effect on recruitment immediately after the fire, with the effect diminishing with time since fire; reproduction was positively correlated with the percent of high-severity fire in owl territories; and positive selection for foraging in low- and moderate-severity burned forest, with high-severity burned forest used in proportion to its availability, but not avoided. Meta-analysis of variation found significantly greater variation in parameters from burned sites relative to unburned, with specifically higher variation in estimates of occupancy, demography, and survival, and lower variation in estimates of selection probability for foraging habitat in low-severity burned forest. Spotted Owls were usually not significantly affected by mixed-severity fire, as 83% of all studies and 60% of all effects found no significant impact of fire on mean owl parameters. Contrary to current perceptions and recovery efforts for the Spotted Owl, mixed-severity fire does not appear to be a serious threat to owl populations; rather, wildfire has arguably more benefits than costs for Spotted Owls.

Key words: adaptive management; evidence-based decision making; meta-analysis; mixed-severity fire; Spotted Owls; *Strix occidentalis;* systematic review; wildfire.

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Introduction

Wildfires are major natural disturbances in forests of the western United States, and native plants and animals in this region have been coexisting with fire for thousands of years of their evolutionary history (Pierce et al. 2004, Power et al. 2008, Marlon et al. 2012). Western forest fires typically burn as mixed-severity fires with each fire resulting in a mosaic of different vegetation burn severities, including substantial patches (range, 5–70% of burned area; mean, 22%) of high-severity fire (Beaty and Taylor 2001, Hessburg et al. 2007, Whitlock et al. 2008, Williams and Baker 2012, Odion et al. 2014*a*, Baker 2015*a*). High-severity fire (high vegetation burn severity) kills most or all of the dominant vegetation in a stand (>75% mortality; Hanson et al. 2009, Baker 2015*a*, *b*) and creates complex early seral forests, where standing dead trees, fallen logs, shrubs, tree seedlings, and herbaceous plants comprise the structure (Swanson et al. 2011, DellaSala et al. 2014). Post-fire vegetation processes (i.e., succession) then commence according to the pre-fire vegetation, local wildfire processes, propagules from outside the disturbance, and the dynamic biotic and abiotic conditions at the site (Gutsell and Johnson 2006, Johnson and Miyanishi 2006, Mori 2011).

Spotted Owls (*Strix occidentalis*) occur in western U.S. forests and have been intensively studied since the 1970s (Fig. 1). The species is strongly associated with mature and old-growth (i.e., late-successional) conifer and mixed conifer–hardwood forests with thick overhead canopy and many large live and dead trees and fallen logs (Gutiérrez et al. 1995). Its association with older forests has made the Spotted Owl an important umbrella indicator species for public lands management (Noon and Franklin 2002). The scientific literature has established that the optimal habitat for Spotted Owl nesting, roosting, and foraging is provided by conifer and mixed conifer–hardwood forests dominated by medium (30–60 cm) and large (>61 cm) trees with medium (50–70%) to high (>70%) canopy cover (Gutiérrez et al. 1995). The populations of all three subspecies have declined due to widespread historical and ongoing habitat loss, primarily from logging mature and old-growth forests favored by the owls for nesting and roosting (Seamans et al. 2002, Forsman et al. 2011, USFWS 2011, 2012, Conner et al. 2013, Tempel and Gutiérrez 2013, Dugger et al. 2016).

Research on Spotted Owl in fire-affected landscapes did not begin until the early 2000s, and much of what scientists previously understood about habitat associations of Spotted Owl was derived from studies in forests that had generally not experienced recent fire, and where the non-suitable owl habitat was a result of logging (Gutiérrez et al. 1992, Franklin et al. 2000, Seamans et al. 2002, Blakesley et al. 2005, Seamans and Gutiérrez 2007, Forsman et al. 2011, Tempel et al. 2014). Because Spotted Owls are associated with dense, late-successional forests, it has often been assumed that fires that burn at high severity are analogous to clear-cut logging and have a negative effect on population viability. It has become widely believed among wildlife management professionals that severe wildfire is a contributing cause of recent Spotted Owl population declines (USFWS 2011, 2012, 2017), and many land managers believe that forest fires currently pose the greatest risk to owl habitat and are a primary threat to population viability (Davis et al. 2016, Gutiérrez et al. 2017). These beliefs result in fuel-reduction logging projects in Spotted Owl habitat (USDA 2012, 2018) which the USDA Forest Service and US Fish and Wildlife Service state are actions consistent with Spotted Owl recovery (USDA 2012, 2018, Gutiérrez et al. 2017, USFWS 2017). Narrative literature reviews have attempted to summarize the effects of fire on Spotted Owl (Bond 2016, Gutiérrez et al. 2017), but evidence-based conservation decisions should be based upon systematic, transparent reviews of primary literature with quantitative meta-analysis of effects (Sutherland et al. 2004, Pullin and Stewart 2006, Pullin and Knight 2009, Koricheva et al. 2013).

The following systematic review and meta-analysis summarize all available published scientific literature on the effects of wildfire on aspects of Spotted Owl demography (survival, recruitment, and reproduction), site occupancy, and habitat selection, from studies using empirical data to answer the question: How does fire, especially mixed-severity fire with substantial patches of high-severity fire within their home ranges, affect Spotted Owl demography, site occupancy, and habitat selection in the first few post-fire years?

Methods

Literature search

I conducted a systematic review of the primary scientific literature and used meta-analyses and meta-regression to examine the evidence for the direct effects of wildfire on Spotted Owl demography, site occupancy, and habitat selection. My subject was Spotted Owls; the intervention was

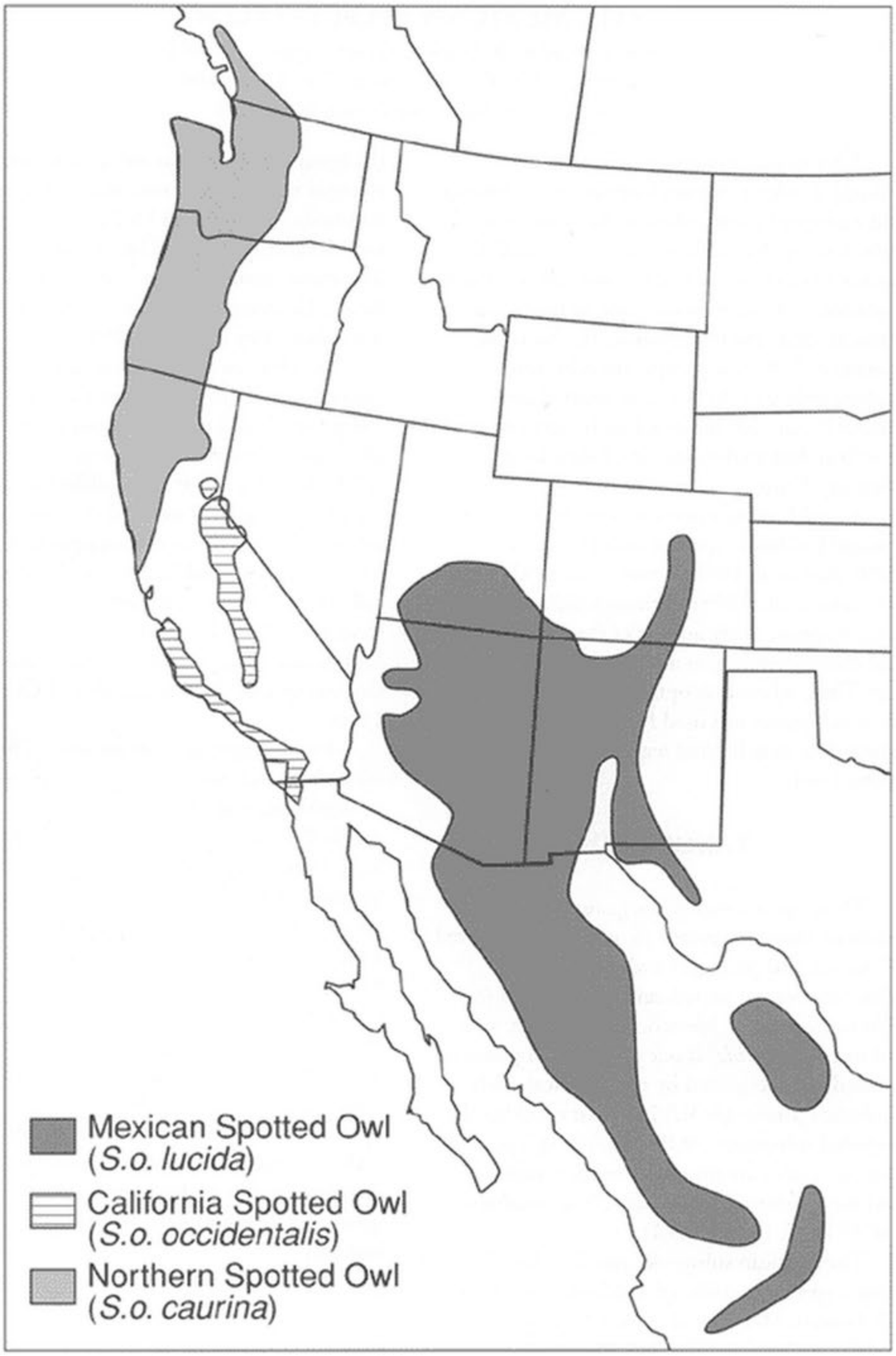

Fig. 1. Range map for the three subspecies of the Spotted Owl (*Strix occidentalis*).

wildfire; the outcomes were change or difference in estimates of demography, site occupancy, and habitat selection probabilities; and the comparator was pre-fire estimates or control estimates from unburned areas (Pullin and Stewart 2006). I searched the following electronic databases on 1 April 2018: Agricola, BIOSIS Previews, ISI Web of Science, and Google Scholar. Search terms were as follows: spotted AND owl AND *fire, Strix AND occidentalis AND *fire. My search included papers published in any year.

I used a threefold filtering process for accepting studies into the final systematic review. Initially, I filtered all articles by title and removed any obviously irrelevant material from the list of articles found in the search. Subsequently, I examined the abstracts of the remaining studies with regard to possible relevance to the systematic review question, using inclusion criteria based on the subject matter and the presentation of empirical data. I accepted articles for viewing at full text if I determined that they may contain information pertinent to the review question or if the abstract was ambiguous and did not allow inferences to be drawn about the content of the article. Finally, I read all remaining studies at full text and either rejected or accepted into the final review based upon subject matter (Pullin and Stewart 2006, Koricheva et al. 2013). Studies that only modeled effects of simulated fires on Spotted Owl habitat and demography were not considered here.

Because post-fire logging often occurred, I also recorded effects of this disturbance where they were reported. I believe all studies in the final review were generally comparable because time since fire and percent of high-severity burn were similar among studies (Tables 1, 2), and the high number of non-significant results reported indicates little to no publication bias exists in this topic (Tables 1, 2; Appendix S1: Fig. S1). I considered the basic sampling unit of all studies to be the central core of the owl breeding-season territory (~400 ha, or a circle with radius 1.1 km centered on the nest or roost stand) because this is the spatial and temporal scale for sampling used in almost all Spotted Owl studies. In contrast, Spotted Owl year-round home ranges vary according to latitude and dominant vegetation, but range from 300 to 11,000 ha, or circles with radius 1.0–5.9 km (Zabel et al. 1992). I considered forest fires to affect the landscape scale (~10,000 ha/decade), but that fires would affect numerous individual owl breeding-season territories (1200 ha) and year-round home ranges (300–19,000 ha) in various ways.

Meta-analyses and meta-regression

I evaluated all final review papers and included all papers where effects of fire were reported and could be differentiated from other disturbances such as post-fire logging. I extracted evidence by reading every paper and tabulating all quantified results from text, tables, and figures (Table 1). I noted the mean ($\bar{x}$) and variation (SD) of burned and unburned groups for all significant and non-significant parameters, the parameters being estimated, sample sizes (n = number of owl breeding sites in burned and unburned groups), amount of high-severity fire in the total fire perimeter and/or within the owl territory core areas examined, time since fire (years), amount of post-fire logging that occurred, subspecies (California = *Strix occidentalis occidentalis*, Mexican = *Strix occidentalis lucida*, or northern = *Strix occidentalis caurina*), and whether the result was statistically significant (as defined in each paper).

I conducted all analyses in R 3.3.1 (www.r-project.org). For meta-analysis, I noted or calculated the mean, variance (SD), and sample size for burned (treatment) and unburned (control) groups. I calculated raw effect sizes as mean differences ($\bar{x}_{\text{burned}} - \bar{x}_{\text{control}}$) and signs (positive or negative) for all reported effects, regardless of their statistical significance. Most papers reported effect sizes as probabilities (occupancy, survival, and foraging habitat selection) so raw effect sizes were scaled between negative and positive one with a mean of zero, making comparison among studies easy. When papers reported multiple effects (e.g., occupancy and reproduction, or survival and recruitment), I recorded each effect individually. Where papers did not report any effect size for a parameter determined to have no significant effects from fire, I included a zero to represent the presence of no significant effect and to avoid a significance bias in the meta-analysis. I stratified data by subspecies (California, Mexican, or northern) and parameter type according to whether the study estimated site occupancy, foraging habitat selection (substratified into selection for low-, moderate-, and high-severity burned forest), and demographic rates (substratified into survival, reproduction, and recruitment). I performed meta-analyses on parameters for which ≥4 estimates existed from ≥4 different fires.

I used three quantitative methods for evaluating the evidence (Koricheva et al. 2013): a random-effects meta-analysis of mean effect sizes as

Table 1. Summary of systematic review of studies examining effects of fire on Spotted Owls.

No.	Ref.	Sample size	HOD	Time since fire	Context	Fire effects (* = statistically significant, NS = non-significant)	Fire	Any effect	Signif. effect	Post-fire logging
1	Bond et al. (2002)	21 owls in 11 burned sites	OD	1 yr post-fire	No effect on survival, site fidelity, mate fidelity, or reproduction. 50% of territories burned 36–88% high severity, 50% burned mostly low –moderate severity, unknown amount of post-fire logging	No significant effects. (3% higher survival NS, 1% lower site fidelity [occupancy] NS, 26% higher repro NS)	0/+/−	+0.032 −0.013 +0.259	na	na
2	Jenness et al. (2004)	33 burned and 31 unburned breeding sites	OD	1-yr study, 1–4 yr post-fire	No effect on occupancy from fire or amount of high-severity fire. No effect on reproduction. 55% of burned territories area burned, 18% at high severity, unknown amount of post-fire logging	No significant effects from fire. (14% lower occupancy NS, 7% lower repro in burn NS)	0/−	−0.14 −0.07	na	na
3	Bond et al. (2009)	Seven radioed owls from four burned sites	H	1-yr study, 4 yr post-fire	Owls preferred burned forest for foraging, especially high-severity burned forest. Owls preferred roost sites burned at low severity and avoided unburned sites and sites burned at moderate and high severity. 69% of foraging area burned, 13% at high severity, <3% post-fire logging	Positive effect from fire on foraging habitat selection (+42%, +42% +33%*), negative and positive effect of fire on roosting nesting habitat selection (+29%, −13%, −28%*)	+/−	+0.33 +0.42 +0.42 +0.29 −0.13 −0.28	+0.33 +0.42 +0.42 +0.29 −0.13 −0.28	na
4	Bond et al. (2010)	Five radioed owls in occupied burned sites	H	1-yr study, 4 yr post-fire	Three of five owls occupied burned forest over winter	No significant effects, perhaps some positive effect	0/+	na	na	na
5	Clark et al. (2011)	11 radioed owls in burned and post-fire logged sites, 12 in unburned sites	D	2-yr study, 3–4 yr post logging	No effects on survival. Reduced survival in salvage-logged areas relative to owls in unburned forest. 14% high severity, 21% post-fire logged	Negative survival effect from combined effects of fire and post-fire logging (−0.07 NS)	?	na	na	−0.07
6	Roberts et al. (2011)	16 burned and 16 unburned survey areas	O	1-yr study, 2–14 yr post-fire	No effect of fire on survey area occupancy. 14% of survey area burned at high severity, little to no post-fire logging	No significant effect from fire. Possible negative effect from basal area and canopy cover model (−26% lower occupancy in burned survey area NS)	0/−	−0.260	na	na

(Table 1. *Continued*)

No.	Ref.	Sample size	HOD	Time since fire	Context	Fire effects (* = statistically significant, NS = non-significant)	Fire	Any effect	Signif. effect	Post-fire logging
7	Lee et al. (2012)	41 burned and 145 unburned breeding sites	O	11-yr study, 1–7 yr post-fire from six large fires	No effect on occupancy probability. 32% high severity. Unknown amount of post-fire logging	No significant effect from fire, perhaps a slightly positive effect (4% higher occupancy in burned sites NS)	0/+	+0.041	na	na
8	Bond et al. (2013)	Seven radioed owls	H	1-yr study, 4 yr post-fire	Owls in burned forest have same size or smaller home ranges than owls in unburned forest. 69% of foraging area burned, 13% at high severity, 3% post-fire logging	No significant effect from fire, possible positive effect (HR size 12% smaller in burned area NS)	0/+	+0.12	na	na
9	Clark et al. (2013)	40 burned and salvage-logged sites and 103 unburned sites	O	13-yr study, 1–4 yr post-fire	Lower site occupancy on salvage-logged sites relative to unburned sites. 11% high severity, 13% post-fire logged	Negative effect on occupancy from combined fire and post-fire logging (−0.39*)	?	na	na	−0.39
10	Lee et al. (2013)	71 burned and 97 unburned breeding sites, post-fire logging on 21 of the burned sites	O	8-yr study, 1–8 yr post-fire	No effects from fire or logging. Burned site occupancy 17% (10% for pairs) lower than unburned sites. Post-fire logged sites occupancy 5% lower than unlogged burned sites. 23% high severity in burned sites, 59% logged in post-fire logged sites	No significant effect from fire, negative effect (17% lower any occupancy, 10% lower pair occupancy in burn NS) Same data as ref. no. 14	0/−	−0.171 −0.107	na	−0.05
11	Ganey et al. (2014)	Four radioed owls	H	1-yr study, 4–6 yr post-fire	Owls moved to burned forest over winter. Burned wintering sites had 2–6 times more prey biomass relative to unburned core areas. 21% high severity, unknown amount of post-fire logged	Positive effect from fire	+	na	na	na
12	Tempel et al. (2014)	12 burned, 62 unburned sites	DO	20-yr study of survival and reproduction, 6-yr study of occupancy.	No effect on survival, reproduction, or site extinction. Reported a negative effect of fire on colonization rate, but colonization parameter was faulty due to low sample size and zero colonization events. Unknown amount of high-severity fire, unknown amount of post-fire logging	No significant effect from fire. Possible negative effect from fire (6% lower occupancy when fire frequency doubled in simulations that assumed zero post-fire colonizations)	0/−	0 0 −0.060	−0.060	na

(Table 1. *Continued*)

No.	Ref.	Sample size	HOD	Time since fire	Context	Fire effects (* = statistically significant, NS = non-significant)	Fire	Any effect	Signif. effect	Post-fire logging
13	Lee and Bond (2015*a*)	45 burned breeding sites	O	Rim Fire, 1-yr study, 1 yr post-fire	Higher burned-site occupancy rates than any published unburned area. 100% high-severity fire in territory surrounding nest and roost sites reduced single owl occupancy probability 5% relative to sites with 0% high severity. Amount of high-severity fire did not affect occupancy by pairs of owls. In fire perimeter: 37% high severity, no post-fire logging	Positive (17% higher occupancy rates*). Small negative effect on site occupancy (3% lower occupancy in burn*). No significant effect on pair occupancy	+/0	+0.175 −0.04 0	+0.175	na
14	Lee and Bond (2015*b*)	71 burned and 97 unburned breeding sites, post-fire logging on 21 of the burned sites	OD	8-yr study, 1–8 yr post-fire	Occupancy of high-quality sites (previously reproductive) that burned was 2% lower than unburned sites. Occupancy of high-quality sites that were post-fire logged was 3% lower. Occupancy of low-quality sites (previously non-reproductive) was 19% lower in burned vs. unburned sites and 26% lower after post-fire logging. Fire did not affect reproduction. 23% high severity in burned sites, 59% logged in post-fire logged sites	Negative effect on site occupancy (2% and 19% lower*), No significant effect on reproduction	−/0	−0.02 −0.19 0	−0.02 −0.19	−0.03 −0.26
15	Bond et al. (2016)	Eight radioed owls in five sites	H	2-yr study, 3–4 yr post-fire	Owls used forests burned at all severities in proportion to their availability, with the exception of significant selection for moderately burned forest farther from core areas. 23% high severity, <5% post-fire logging	No significant effect from fire (3% lower probability of use in high-severity burn NS), some positive effect (15% higher probability of use of low-severity burn NS, 10% higher probability of use in moderate-severity burned forest NS, 3% higher probability of use of moderate severity away from the core*)	0/+	−0.03 +0.15 +0.10	+0.033	na

(Table 1. *Continued*)

No.	Ref.	Sample size	HOD	Time since fire	Context	Fire effects (* = statistically significant, NS = non-significant)	Fire	Any effect	Signif. effect	Post-fire logging
16	Comfort et al. (2016)	23 radioed owls in post-fire logged area	H	2-yr study, 3–4 yr post logging	Scale-dependent effects of logging (+/−). Owls selected a moderate amount of hard edges around logged stands. 14% high severity, 21% post-fire logged	Positive and negative effect from post-fire logging created edges	?	na	na	+/−
17	Jones et al. (2016)	30 burned sites, 15 unburned sites, nine radioed owls in seven sites	OH	23-yr study, 1 yr post-fire	Negative effects from high-severity fire. Positive effect of low- to moderate-severity fire. 64% high-severity burn, 2% post-fire logging	>50% high-severity burned sites had lower occupancy (−0.49*), <50% high-severity burned sites had higher occupancy (+0.07 NS). High-severity burned habitat was avoided (−0.307*), low-tomoderate-severity burn was preferred (+0.04 NS)	+/−	+0.070 −0.490 −0.307 +0.04	−0.490 −0.307 +0.04	na
18	Tempel et al. (2016)	43 burned sites and 232 unburned sites in four study areas	O	19-yr study, examined 3-yr post-fire effects	No effects of fire. One study area had positive effect of fire. Lower site extinction probability correlated with proportion of site where wildfire reduced canopy >10%. 1% of all territories burned, unknown amount of post-fire logging	No significant effect from fire, some positive effect (1% lower extinction rate in burned sites NS)	0/+	+0.003 0 0 0	na	na
19	Eyes et al. (2017)	13 radioed owls in eight sites (14 owl-year data sets)	H	3-yr study, 1–14 yr post-fire	No effect of fire on foraging habitat selection, owls foraged in all burn severities in proportion to their availability. 6% high severity, little to no post-fire logging	No significant effect from fire. Possibly negative effect (6% lower probability of use for highest burn severity NS; 3% lower use of moderate severity NS)	0/−	−0.06 −0.03	na	na
20	Rockweit et al. (2017)	193 burned and 386 unburned encounter histories from 28 burned (8, 2, 4, 14) and 70 unburned sites	D	26-yr study, 4–26 yr post-fire	Four fires had different effects. Generally, fires reduced survival and increased recruitment. 10%, 12%, 16%, and 48% high severity, no post-fire logging reported	Two fires had no significant effects on survival or recruitment. Two fires had reduced survival (−0.17 and −0.30*), one had increased recruitment (+0.22*)	0/+/−	−0.03 −0.10 −0.17 −0.30 +0.01 +0.02 +0.04 +0.22	−0.17 −0.30 +0.22	na

(Table 1. *Continued*)

No.	Ref.	Sample size	HOD	Time since fire	Context	Fire effects (* = statistically significant, NS = non-significant)	Fire	Any effect	Signif. effect	Post-fire logging
21	Hanson et al. (2018)	54 burned sites in eight fires that were occupied immediately before fire, before–after comparison	O	14-yr study, 1 yr post-fire	Eight large fires (4 included in Tempel et al. 2016). Four groups: 20–49% and 50–80% high-severity fire; and <5% and ≥5% post-fire logging within 1500 m of site center. Mean 63% high severity in core areas, mean 17% logged if ≥5% of core was post-fire logged Compared burned site occupancy with unburned occupancy from Tempel et al. (2016)	No significant effect from fire, significant negative effect of post-fire logging (3% reduction in occupancy if 50–80% of core burned high-severity fire NS, 52% reduction in occupancy from ≥5% post-fire logging*)	0/-	−0.017 −0.013	na	−0.52

Notes: HOD indicates habitat selection (H), occupancy (O), or demographic (D) parameters were estimated. A question mark (?) indicates confounded fire and post-fire logging effects, so fire effects could not be estimated.

the standardized difference in means (Hedge's *d*; Hedges and Olkin 1985); multi-level linear mixed-effects models (hierarchical models) meta-regression of time since fire and percent of high-severity fire in the study area as covariates to explain heterogeneity in mean effect sizes (Hedges and Vevea 1998, Nakagawa and Santos 2012); and a random-effects meta-analysis of variation to examine differences in parameter variances due to fire with effect sizes as the natural logarithm of the ratio between the coefficients of variation (lnCVR; Nakagawa et al. 2015). For analyses, I used the metafor package of R (Viechtbauer 2010) and used function metacont for random-effects meta-analyses, function rma.mv for multi-level linear mixed-effects model meta-regression, and function rma for random-effects meta-analysis of variation (Viechtbauer 2010). Study within geographic area was included as multi-level random effects to properly estimate study site- and region-specific variation and to account for repeated measurements (pseudo-replication) within a study or region. Regions were defined as Sierra Nevada, southern California, national parks, not California, and the Eldorado density study area (because several studies used data from there).

I used all three methods at three levels: on all parameters, on three main groups of parameters (occupancy, foraging habitat selection, and demography), and on subgroups of habitat selection (for low-, moderate-, and high-severity burned forest) and demography (survival, reproduction, and recruitment). In meta-analyses, I used z tests to determine if effects were significantly different from zero (95% confidence interval excluded zero). In meta-regression, z tests determined whether intercepts or slope coefficients were significantly different from zero. I quantified heterogeneity among effects as Cochran's Q (Hedges and Olkin 1985) and I^2 (Higgins and Thompson 2002). I used a funnel plot and the rank correlation test (Kendall's τ) to assess publication bias (Begg and Mazumdar 1994).

Results

Literature search

I found 21 papers reporting empirical evidence relevant to direct fire effects on owls (Table 1). Three papers presented data from a study area which was extensively logged post-fire and results did not discriminate between effects of fire and post-fire logging (Clark et al. 2011, 2013, Comfort et al. 2016), so these three papers were not included in meta-analyses with the meta-analysis set of papers that were not confounded

Table 2. Summary statistics for published effects of mixed-severity fire on Spotted Owls (*Strix occidentalis*) 1987–2018 used in meta-analysis.

Ref no.	Study	Subspecies	Region	Parameter	*n* burned	*n* unburned	Raw effect size (mean difference)	Significant (in study)	Time since fire (yr)	Percentage of high-severity fire in burned territories
1	Bond (2002)	CNM	NotCal	Occupancy	18	100	−0.013	na	1	30
1	Bond (2002)	CNM	NotCal	Reproduction	7	100	0.259	na	1	30
1	Bond (2002)	CNM	NotCal	Survival	21	100	0.032	na	1	30
2	Jenness (2004)	M	NotCal	Occupancy	33	31	−0.14	na	2.5	16
2	Jenness (2004)	M	NotCal	Reproduction	33	31	−0.07	na	2.5	16
3	Bond (2009)	C	SN	Foraging High	7	7†	0.42	0.42	4	13
3	Bond (2009)	C	SN	Foraging Low	7	7†	0.33	0.33	4	13
3	Bond (2009)	C	SN	Foraging Mod	7	7†	0.42	0.42	4	13
6	Roberts (2011)	C	NP	Occupancy	16	16	−0.26	na	8	12
7	Lee (2012)	C	SN	Occupancy	41	145	0.041	na	4	32
10	Lee (2013)	C	SoCal	Occupancy	71	97	−0.171	na	4.5	23
10	Lee (2013)	C	SoCal	Occupancy	71	97	−0.107	na	4.5	23
12	Tempel (2014)	C	Eldorado	Occupancy	12	62	−0.06	−0.06	3	23‡
12	Tempel (2014)	C	Eldorado	Reproduction	12	62	0	na	3	23‡
12	Tempel (2014)	C	Eldorado	Survival	12	62	0	na	3	23‡
13	Lee (2015*a*)	C	SN	Occupancy	45	45	−0.04	na	1	37
13	Lee (2015*a*)	C	SN	Occupancy	45	45	0	na	1	37
13	Lee (2015*a*)	C	SN	Occupancy	45	145	0.175	0.175	1	37
14	Lee (2015*b*)	C	SoCal	Occupancy	71	97	−0.19	−0.19	4.5	23
14	Lee (2015*b*)	C	SoCal	Occupancy	71	97	−0.02	−0.02	4.5	23
14	Lee (2015*b*)	C	SoCal	Reproduction	71	97	0	na	4.5	23
15	Bond (2016)	C	SoCal	Foraging High	8	8†	−0.093	na	3.5	15
15	Bond (2016)	C	SoCal	Foraging High	8	8†	−0.035	na	3.5	16
15	Bond (2016)	C	SoCal	Foraging High	8	8†	0.092	na	3.5	9
15	Bond (2016)	C	SoCal	Foraging Low	8	8†	0.115	na	3.5	15
15	Bond (2016)	C	SoCal	Foraging Low	8	8†	0.167	na	3.5	9
15	Bond (2016)	C	SoCal	Foraging Low	8	8†	0.169	na	3.5	16
15	Bond (2016)	C	SoCal	Foraging Mod	8	8†	−0.042	na	3.5	15
15	Bond (2016)	C	SoCal	Foraging Mod	8	8†	0.033	0.033	3.5	16
15	Bond (2016)	C	SoCal	Foraging Mod	8	8†	0.102	na	3.5	9
17	Jones (2016)	C	Eldorado	Foraging High	9	9†	−0.307	−0.307	1	19

(Table 2. *Continued*)

Ref no.	Study	Subspecies	Region	Parameter	*n* burned	*n* unburned	Raw effect size (mean difference)	Significant (in study)	Time since fire (yr)	Percentage of high-severity fire in burned territories
17	Jones (2016)	C	Eldorado	Foraging Mod	9	9†	0.04	+0.04	1	19
17	Jones (2016)	C	Eldorado	Occupancy	14	15	−0.490	−0.490	1	64
17	Jones (2016)	C	Eldorado	Occupancy	16	15	0.07	na	1	19
18	Tempel (2016)	C	SN	Occupancy	12	78	0	na	4	23‡
18	Tempel (2016)	C	Eldorado	Occupancy	14	60	0	na	4	23‡
18	Tempel (2016)	C	SN	Occupancy	3	63	0	na	4	23‡
18	Tempel (2016)	C	NP	Occupancy	14	31	0.003	0.003	4	23‡
19	Eyes (2017)	C	SN	Foraging High	13	13†	−0.06	−0.06	7	6
19	Eyes (2017)	C	SN	Foraging Mod	13	13†	−0.03	−0.03	7	6
20	Rockweit (2017)	N	NotCal	Recruitment	8	8	0.01	na	12.5	10
20	Rockweit (2017)	N	NotCal	Recruitment	2	2	0.02	na	6.5	16
20	Rockweit (2017)	N	NotCal	Recruitment	4	4	0.04	na	4	48
20	Rockweit (2017)	N	NotCal	Recruitment	14	14	0.22	0.22	2	12
20	Rockweit (2017)	N	NotCal	Survival	4	4	−0.30	−0.3	4	48
20	Rockweit (2017)	N	NotCal	Survival	14	14	−0.17	−0.17	2	12
20	Rockweit (2017)	N	NotCal	Survival	2	2	−0.10	na	6.5	16
20	Rockweit (2017)	N	NotCal	Survival	8	8	−0.03	na	12.5	10
21	Hanson (2018)	C	SN	Occupancy	13	201	−0.017	−0.017	1	63
21	Hanson (2018)	C	SN	Occupancy	15	201	0.013	0.013	1	35

Notes: Study indicates first author and year. Subspecies are C, California (*Strix occidentalis occidentalis*); N, northern (*Strix occidentalis caurina*); M, Mexican (*Strix occidentalis lucida*); CNM, study included all subspecies. Regions are SN, Sierra Nevada, California (except El Dorado study area and national parks); SoCal, southern California; Eldorado, El Dorado study area in Sierra Nevada, California; NotCal, not California Spotted Owl subspecies; NP, national parks. Parameters: habitat selection (foraging or roosting) in low-, moderate-, (mod) or high-severity burned forest; occupancy, recruitment, reproduction, and survival. Sample sizes (*n*) are number of breeding site territories burned and unburned. Raw mean effect size is $\bar{x}_{\text{burned}} - \bar{x}_{\text{control}}$, significant repeats effects that the individual study determined was statistically significant. Time since fire is the median number of years between the fire and the parameter estimate(s). Percent high-severity fire in burned study territories is the mean relevant to the estimate, or the grand mean if percentage of high severity was not reported (see ‡).

† Habitat selection occurred within territories that contained a mosaic of burn severities and unburned forest.

‡ Percent high-severity fire was not reported for burned territories only for all territories burned and unburned, so the grand mean of reported percentages was used.

by extensive post-fire logging (Table 2). All 21 papers are summarized in Appendix S1.

Fifteen of the 18 papers in the meta-analysis set reported evidence explicitly pertaining to mixed-severity wildfires that burned during the past few decades and which included proportions of high-severity burn characteristic of this fire regime, while three reported evidence from an undifferentiated mix of wildfire and prescribed fires. The studies reported varying amounts of high-severity fire, a defining feature of mixed-severity fires, and the burn severity type that is most responsible for vegetation changes in wildfires, with an overall mean percent of high-severity fire of 26% (standard error [SE] = 3.6, range 6–64) within the study area. Because almost all the studies in this review reported on effects from recent wildfires (all fires burned in the past 30 yr, mean time since fire = 4 yr, SE = 1.1, range 1–26), the reported effects are representative of natural mixed-severity fires as they burned through currently existing forest structure, fire regime, and climate conditions. Papers reported effects of fire on site occupancy (11), foraging habitat selection (4), reproduction (4), apparent survival (3), overwinter roosting habitat selection (2), site fidelity (1), mate fidelity (1), breeding-season nesting and roosting habitat selection (1), home-range size (1), and recruitment (1). Sample sizes measured as number of burned sites were variable among studies (demography CV = 122%, site occupancy CV = 56%, and habitat selection CV = 24%).

Meta-analyses

Meta-analysis of 50 reported effects on occupancy, foraging habitat selection, and demographic rates found effect sizes and signs were variable (Table 2 and Fig. 2), with high heterogeneity among effects ($Q = 1091$, df = 51, $P < 0.0001$; $I^2 = 95.3\%$). Funnel plot (Appendix S1: Fig. S1) and rank correlation test (Kendall's $\tau = 0.108$, $P = 0.27$) showed no publication bias or unusual heterogeneity. Sample sizes (n = number of reported effects) were variable among parameter types (Fig. 3). The number of reported effects were occupancy = 20; demography = 14; and foraging habitat selection = 16. The number of reported effects by demography subtype were survival = 6; reproduction = 4; and recruitment = 4. The number of reported effects by habitat selection subtype were low-severity burned forest = 4; moderate-severity burned forest = 6; and high-severity burned forest = 6.

The mixed-effects model meta-analysis of fire effects on Spotted Owl parameters grouped by type (occupancy, demography, and foraging habitat selection), and subtypes of demography (survival, reproduction, and recruitment) or foraging habitat selection (selection for low-, moderate-, and high-severity burned forest), found mixed-severity fire has generally no significant effect on Spotted Owls (Fig. 3a). Mean overall raw effect size was positive (+0.001), but weighted mean Hedge's d from the random-effects model was not significantly different from zero (Fig. 3a, 95% confidence interval included zero). Mean raw effect sizes were negative for occupancy (−0.060), demography (−0.006), and survival (−0.095), but no Hedge's d value for these three negative effects was significantly different from zero (Fig. 3a). Mean raw effect sizes were positive for reproduction (+0.047), recruitment (+0.073), foraging habitat selection (+0.083), selection of high-severity (+0.004), moderate-severity (+0.087), and low-severity burned forest (+0.195), but Hedge's d values were not significantly different from zero for any of these positive effects, except for significant selection of low-severity burned forest (Fig. 3a).

Variation was generally higher among parameter estimates from burned areas compared to estimates from unburned areas (mean $CV_{burned} - CV_{unburned} = 23\%$; range 4–57%). The mixed-effects meta-analysis of variation in fire effects on Spotted Owl parameters (lnCVR) found mixed-severity fire resulted in significantly higher variation in parameter estimates in all parameters and in occupancy, demography, and survival (Fig. 3b). There was significantly lower variation in estimates of foraging habitat selection probability for low-severity burned forest (Fig. 3b).

Meta-regression

Meta-regression of all standardized mean effects found significant effect of time since fire (Table 3), and a nearly significant effect of percent high-severity burn in territory cores (Table 3), so those effects were included in parameter-specific meta-regressions. Subspecies was not a significant factor (Table 3), so effects from different subspecies were pooled in subsequent parameter-specific analyses.

Meta-regression of occupancy probability found no significant immediate effect of fire on occupancy (intercept not significantly different from zero; Table 4). There was a significant negative effect of time since fire (Fig. 4, Table 4), but no effect of percent high-severity fire in study territories (Table 4). The negative effect of time since fire was sensitive to one study (Roberts et al. 2011), and when that study was omitted, the effect disappeared.

Meta-regression of demographic parameters found a significant positive effect on recruitment immediately after the fire (intercept significantly different from zero), but the effect diminished

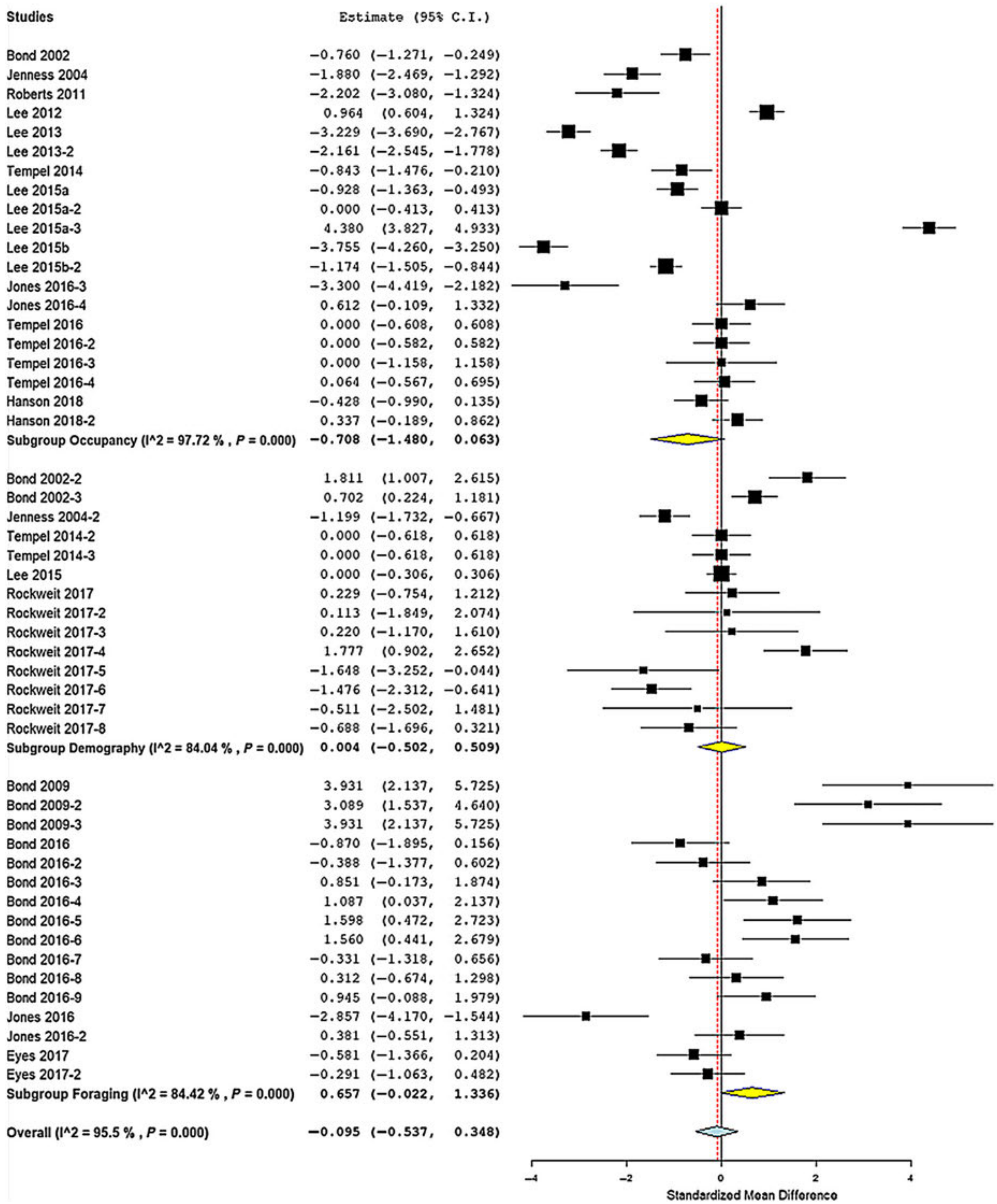

Fig. 2. Forest plot of effect sizes for 50 Spotted Owl (*Strix occidentalis*) parameters (grouped into occupancy, demography, and foraging habitat selection) affected by mixed-severity wildfire as standardized mean difference (Hedge's *d*) between burned and unburned samples. Studies and parameters are listed in Table 2.

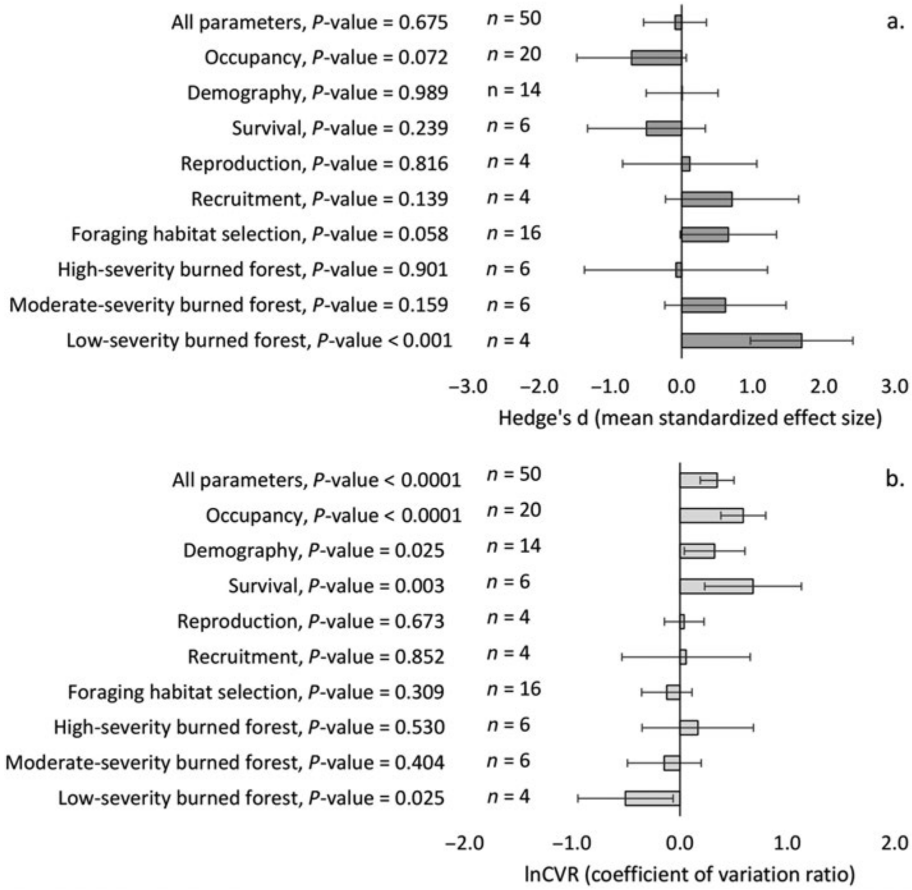

Fig. 3. Results of mixed-effects meta-analyses of mixed-severity fire effects (n = 50 effects from 21 studies) on Spotted Owl (*Strix occidentalis*) parameters grouped by type (occupancy, demography, and foraging habitat selection) and subtype of demography (survival, reproduction, and recruitment), or habitat selection (selection for low-, moderate-, and high-severity burned forest). (a) Hedge's d is standardized mean effect size, and error bars are 95% confidence intervals. The only significant effect (95% confidence intervals excluded zero) was a positive effect of habitat selection for low-severity burned forest. (b) lnCVR is the natural logarithm of the ratio between the coefficients of variation, a measure of differences in variation of parameter estimates between burned and unburned areas. Mixed-severity fire resulted in significantly higher variation in parameter estimates in all parameters, occupancy, demography, and survival, and significantly lower variation in habitat selection for low-severity burned forest.

with time since fire (Fig. 5, Table 4). Reproduction intercept was not significantly different from recruitment (Table 4), and not significantly different from zero (z = −0.218, P = 0.86), but reproduction was significantly positively correlated with the percent of high-severity fire in owl territories (Fig. 5, Table 4). Survival was significantly lower than recruitment (Table 4), but survival intercept was not significantly different from zero (z = −0.052, P = 0.97). There were no significant survival effects of time since fire or percent of high-severity fire (Table 4).

Meta-regression of foraging habitat selection parameters found a significant positive selection for low- and moderate-severity burned forest, with high-severity burned forest used in proportion to its availability, but not avoided (Fig. 5, Table 4). Time since fire did not affect foraging habitat selection during the period covered by the studies I examined (up to 7 yr), and the

Table 3. Results from multivariate mixed-effects meta-regression model of mixed-severity fire effects (n = 50 effects from 21 studies) on Spotted Owl (*Strix occidentalis*) parameters related to occupancy, demography, and foraging habitat selection.

Covariates	β	SE	z	P
Intercept (California subspecies)	1.601	1.070	1.497	0.134
Time since fire	−0.199	0.099	−2.017	**0.044**
Percentage of area high-severity fire in study territories	−0.044	0.023	−1.866	0.062
Mix of California, northern, Mexican subspecies	0.467	1.592	0.294	0.769
Mexican subspecies	−1.947	1.608	−1.211	0.226
Northern subspecies	0.360	1.571	0.229	0.819

Notes: SE, standard error. Time since fire was significant, and percent high-severity burn in territory cores was nearly significant, so those effects were included in parameter-specific meta-regressions. Subspecies was not a significant factor, so effects from different subspecies were pooled in subsequent parameter-specific analyses. Bold values are significant at alpha = 0.05.

amount of high-severity fire did not affect habitat selection overall (Table 4).

Post-fire logging had negative effects on Spotted Owls in 100% of the papers that examined this disturbance and where effects from fire and post-fire logging could be differentiated, with large effect sizes (−0.18 occupancy, −0.07 survival).

Discussion

This systematic review and summary of effects from the primary literature indicated Spotted Owls are usually not significantly affected by mixed-severity fire as 83% of all studies and 60% of all effects found no significant impact of fire on owl parameters. Meta-analysis of mean effects found no significant effects of fire on owls, except a positive effect on foraging habitat selection for low-severity burned forest. Meta-regression indicated significant positive effects in recruitment, reproduction, and foraging habitat selection for low- and moderate-severity burned forest. Meta-regression found a significant negative effect of time since fire on occupancy probability. Meta-analysis of variation found mixed-severity fire resulted in greater parameter variation overall, and specifically in occupancy, demography, and survival, and significantly less variation in foraging habitat selection for low-severity burned forest.

Table 4. Table of model coefficients from multi-level linear mixed-effects model meta-regression for effects of mixed-severity fire on Spotted Owls 1987–2018.

Coefficient	β	SE	z	P
Occupancy				
Intercept	1.854	1.115	1.662	0.096
Time since fire	−0.512	0.216	−2.375	**0.018**
Percentage of area high-severity fire in study territories	−0.036	0.022	−1.645	0.100
Demography				
Intercept (Recruitment)	2.328	1.152	2.021	**0.043**
Time since fire (Recruitment)	−0.153	0.065	−2.347	**0.019**
Percentage of area high-severity fire in study territories	−0.032	0.022	−1.466	0.143
Reproduction	−6.479	3.337	−1.942	0.052
Survival	−2.558	1.206	−2.121	**0.034**
Time since fire (reproduction)	0.034	0.422	0.081	0.936
Time since fire (survival)	0.101	0.112	0.900	0.368
Percentage of area high-severity fire (reproduction)	0.234	0.109	2.142	**0.032**
Percentage of area high-severity fire (survival)	0.031	0.033	0.924	0.356
Foraging habitat selection				
Intercept (High severity)	1.167	2.926	0.399	0.690
Time since fire	−0.061	0.529	−0.115	0.908
Percentage of area high-severity fire in study territories	−0.084	0.068	−1.236	0.216
Low severity	1.936	0.732	2.644	**0.008**
Moderate severity	0.777	0.321	2.416	**0.016**

Note: SE, standard error. Bold values are significant at alpha = 0.05.

These results represent Spotted Owl responses to mixed-severity wildfires that burned within the past 30 yr with representative proportions of high-severity fire in a landscape mosaic. Additionally, because most of the studies in this review reported on effects from wildfire, rather than prescribed fire, the fires and their effects are representative of wildfires as they burned through currently existing forest structure, fire regime, and climate conditions. Several studies have reported that fires during the past few decades have been larger and more severe than the historical mean (Miller and Safford 2012, 2017, Mallek et al. 2013, Steel et al. 2015), but others have disputed this

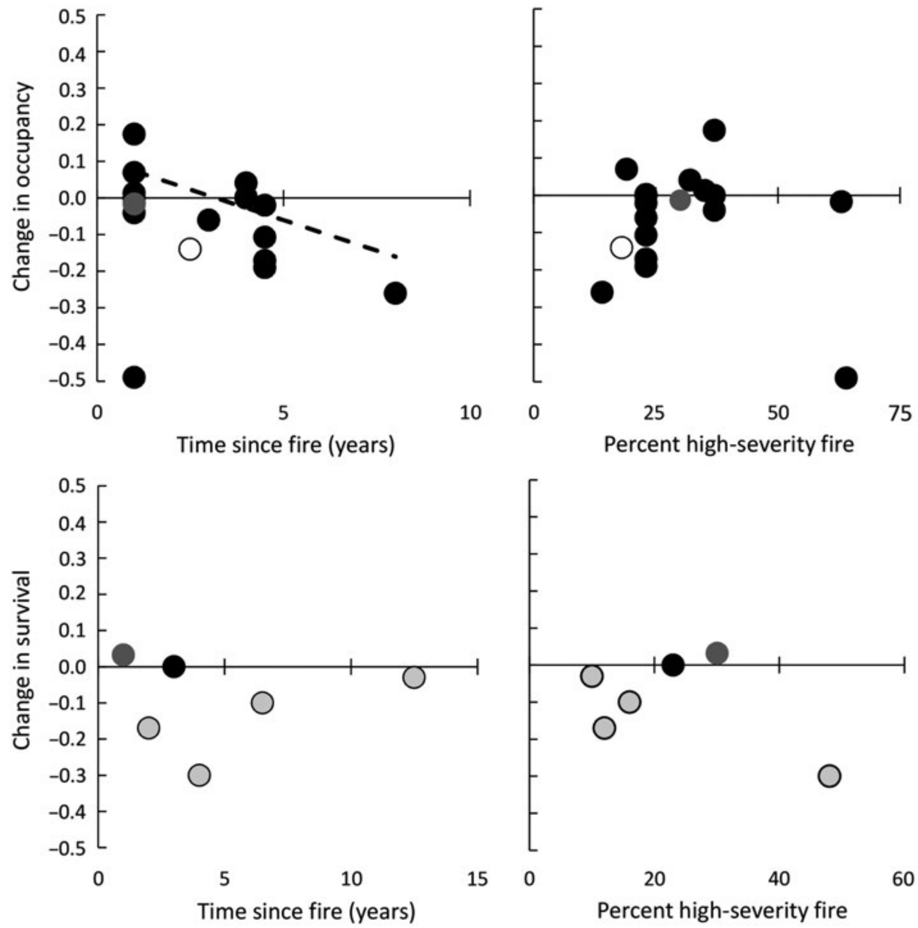

Fig. 4. Results of multi-level linear mixed-effects models (hierarchical models) meta-regression of time since fire and percent of high-severity fire in the study area as covariates to explain heterogeneity in effect sizes from mixed-severity fire on Spotted Owl (*Strix occidentalis*) parameters of breeding site occupancy and survival. The only significant effect was a reduction in occupancy with increasing time since fire, but the effect was sensitive to one study. Symbols indicate subspecies: filled black circles, California; white circles with black outline, Mexican; light gray circles with black outline, northern; and dark gray circles, all three subspecies.

point (Odion and Hanson 2006, Hanson et al. 2009, Odion et al. 2014*a*, Baker 2015*a*). Regardless of what is correct about trends in fire severity, Spotted Owls appear fairly resistant and/or resilient to effects from recent hot, large fires, wherever these fires fall in the long-term range of variability for size and amount of high-severity burn. This is corroborated by the meta-regressions that explicitly quantified the relationship between amount of high-severity fire and Spotted Owl parameters and found only a positive significant correlation (reproduction). My finding of no significant negative relationships between amount of high-severity fire and Spotted Owl parameters demonstrates that large high-severity fire patches, including territories that burn 100% at high severity as was seen in sites within several of the studies in this review, do not have unequivocally negative outcomes for Spotted Owls.

Contrary to current perceptions, recovery efforts, and forest management projects for the Spotted Owl (USFWS 2011, 2012, 2017, USDA 2012, 2018, Gutiérrez et al. 2017) mixed-severity fire as it has been burning in recent decades does not appear to be an immediate, dire threat to owl populations that require landscape-level fuel-reduction

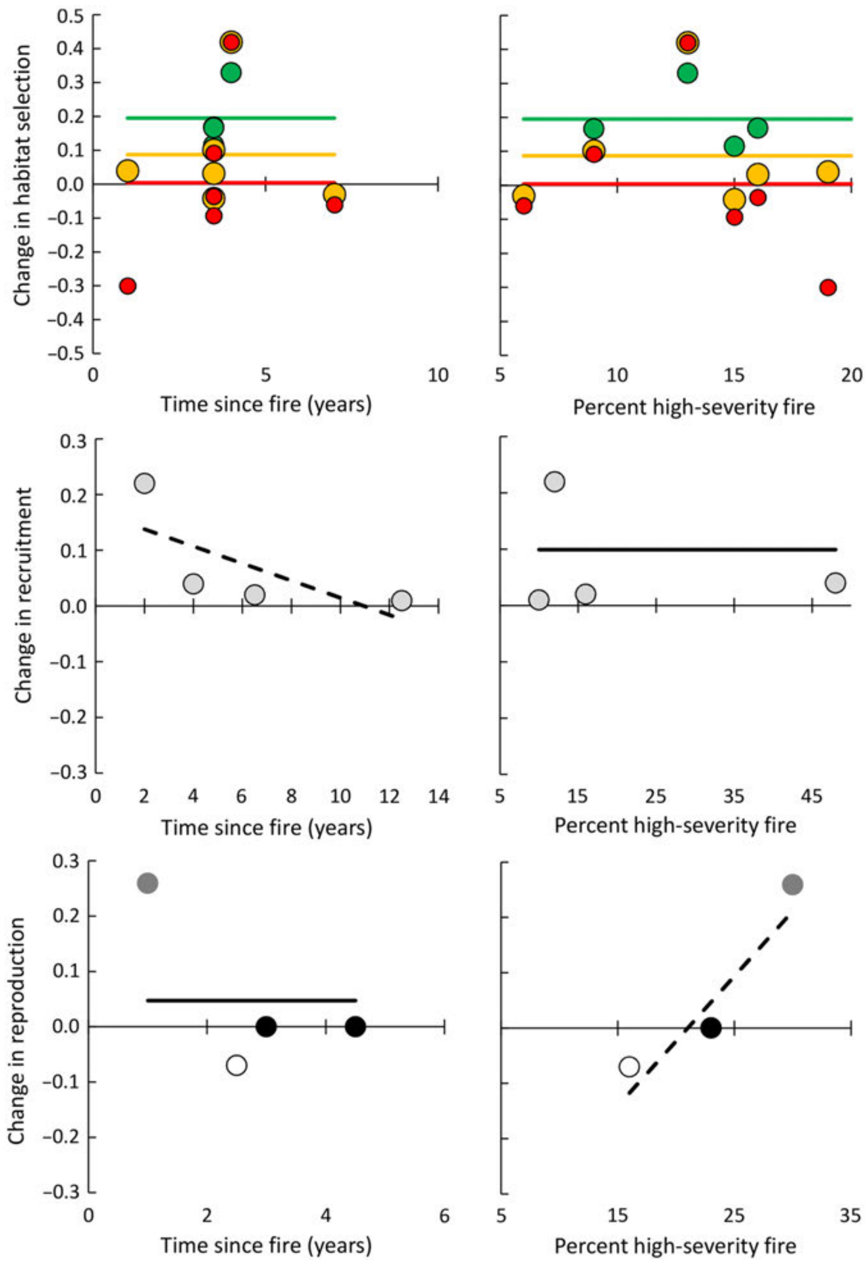

Fig. 5. Results of multi-level linear mixed-effects models (hierarchical models) meta-regression of time since fire and percent of high-severity fire in the study area as covariates to explain heterogeneity in effect sizes from mixed-severity fire on Spotted Owl (*Strix occidentalis*) parameters of foraging habitat selection, recruitment, and reproduction. Significant effects included positive selection for low- and moderate-severity burned forest for foraging, increased recruitment immediately post-fire that diminished with increasing time since fire, and increased reproduction with a positive correlation with amount of high-severity fire. In top two panels, all studies were California subspecies, and colors indicate forest in different burn severity categories: green, low severity; orange, moderate severity; red, high severity. In bottom four panels, symbols indicate subspecies: filled black circles, California; white circles with black outline, Mexican; light gray circles with black outline, northern; and dark gray circles, all three subspecies.

treatments to mitigate fire severity. Empirical studies reviewed here demonstrated that wildfires can generally have no significant effect, but effects can include improved foraging habitat, reduced site occupancy, and improved demographic rates. Most territories occupied by reproductive Spotted Owl pairs that burn remain occupied and reproductive at the same rates as sites that did not experience recent fire, regardless of the amount of high-severity fire in core nesting and roosting areas.

To place my results into perspective, mixed-severity fire typically affects (≥50% vegetation basal area mortality) a very small portion (0.02–0.50%) of Spotted Owl nesting and roosting habitat per year (Odion et al. 2014*b*, Baker 2015*b*, Stephens et al. 2016). Breeding sites that experienced a typical mixed-severity burn mosaic can be expected to have occupancy probability reduced by −0.06 on average. A 0.06 decline in occupancy is less than typical annual declines in occupancy rates observed in the Sierra Nevada in the absence of large fires (Jones et al. 2016: Fig. 3f). In comparison, post-fire logging caused a mean occupancy probability reduction of −0.18.

Post-fire logging is likely to be partially responsible for some of the negative effects attributed to high-severity fire in the studies reviewed here (Tempel et al. 2014, Jones et al. 2016, Rockweit et al. 2017, Hanson et al. 2018). Because Spotted Owl studies typically characterize territory vegetation only in the breeding core area within 1.1 km of the nest, these studies ignore habitat changes and alterations in the year-round home-range area that can extend up to 5.9 km from the nest (Zabel et al. 1992). Spotted Owl habitat protections have generally not included areas beyond 1 km from the nest, a management policy that has not contributed to population recovery.

Complex early seral forests created by fire differ from post-fire salvage-logged forests in that dead trees remain on-site, providing perching sites for hunting owls as well as food sources and shelter for numerous wildlife species (Hutto 2006, Swanson et al. 2011, DellaSala et al. 2014). Longitudinal studies also indicated that burned breeding sites where owls were not detected immediately after fire were often recolonized later (Lee et al. 2012, 2013, Tempel et al. 2016), and this review shows burned forest habitat is used for foraging, demonstrating the mistake of concluding severely burned sites or habitats are lost to Spotted Owls or require restoration (Davis et al. 2016). A recent global meta-analysis found post-fire logging is generally not consistent with ecological management objectives (Thorn et al. 2018).

This review on fire and Spotted Owls forms one portion of the evidence base for data-driven forest management. A recent systematic review of thinning and fire found 56 studies addressing fuel treatment effectiveness in real (not simulated) wildfires from eight states in the western United States (Kalies and Kent 2016). There was general agreement that thin + burn treatments (thinning immediately followed by burning) had some positive effects in terms of reducing fire severity, while treatments by burning or thinning alone were less effective or ineffective (Kalies and Kent 2016). There is also evidence that doing nothing can achieve many forest restoration goals related to age structure and fuels' density (Zachmann et al. 2018). Additional systematic reviews are needed to examine (1) the quantifiable risk of fire to Spotted Owl habitat, as there are disparate lines of evidence regarding whether fire is impeding the recovery of late-seral-stage forests; and (2) the impacts of fuel treatments on Spotted Owl demography and site occupancy. Thinning immediately followed by burning to reduce wildfire risk may or may not have adverse effects on Spotted Owls (Franklin et al. 2000, Dugger et al. 2005, Tempel et al. 2014, 2016, Odion et al. 2014*b*), but the evidence presented here indicates fire itself has arguably more benefits than costs to the species and thus suggests thinning is not necessary.

The results presented here should serve to guide management decisions, but also should be understood as limited by the available data. The sample sizes of number of estimated effects from mixed-severity fire on survival and recruitment were small and limited mainly to the northern subspecies. There were also very few studies from the Mexican subspecies. A few studies presented effect sizes that were influential on results, especially meta-regression results (Roberts et al. 2011), so studies examining longer times since fire are needed. We encourage future studies to increase sample sizes of each parameter and to provide a more balanced sample of studies from all subspecies, and over longer time frames.

Management Implications

The preponderance of evidence presented here shows mixed-severity forest fires, as they have burned through Spotted Owl habitat in recent decades under current forest structural, fire regime, and climate conditions, have no significant negative effects on Spotted Owl foraging habitat selection, or demography, and have significant positive effects on foraging habitat selection, recruitment, and reproduction. Forest fire does not appear to be a serious threat to owl populations and likely imparts more benefits than costs for Spotted Owls; therefore, fuel-reduction treatments intended to mitigate fire severity in Spotted Owl habitat are unnecessary. These findings should inform revisions to planning documents to consider burned forest, including large patches of high-severity burned forest, as useful habitat that imparts significant benefits to Spotted Owls. Forest and wildlife planning documents promote a diverse mosaic of heterogeneous tree densities and ages (USFWS 2017, USDA 2018), the very conditions created by mixed-severity wildfire, and it follows that heterogeneous post-fire structure would lead to greater variation in some Spotted Owl parameters, as was observed in the meta-analysis of variation. Planning documents (USFWS 2011, 2012, 2017, Gutiérrez et al. 2017, USDA 2018) claiming that forest fires currently pose the greatest risk to owl habitat and are a primary threat to population viability appear outdated in light of this review.

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Supporting Information

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Newly discovered landscape traps produce regime shifts in wet forests

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We describe the "landscape trap" concept, whereby entire landscapes are shifted into, and then maintained (trapped) in, a highly compromised structural and functional state as the result of multiple temporal and spatial feedbacks between human and natural disturbance regimes. The landscape trap concept builds on ideas like stable alternative states and other relevant concepts, but it substantively expands the conceptual thinking in a number of unique ways. In this paper, we (*i*) review the literature to develop the concept of landscape traps, including their general features; (*ii*) provide a case study as an example of a landscape trap from the mountain ash (*Eucalyptus regnans*) forests of southeastern Australia; (*iii*) suggest how landscape traps can be detected before they are irrevocably established; and (*iv*) present evidence of the generality of landscape traps in different ecosystems worldwide.

altered ecosystem processes | old growth

ECOLOGY

In many environments worldwide, key drivers of ecosystem change interact and reinforce one another to trigger cascades of ecosystem modification that are difficult or impossible to reverse (1–3). These cascades are often referred to as regime shifts (4–6). Examples of significant regime shifts include overfishing and trophic cascades in marine predator–prey systems (7) and human disturbance-driven losses of detritivore populations and subsequent changes in the decomposition of organic matter (8). Regime shifts are almost always identified in retrospect, making it difficult to know how to avoid them in advance and problematic to reverse their effects. Therefore, understanding of the mechanistic processes by which regime shifts occur may provide opportunities to change resource management and avoid irreversible and undesirable ecological changes.

In this paper, we describe the "landscape trap" concept, of which the outcome is a regime shift triggered by a series of feedback processes resulting from interacting natural and anthropogenic disturbances. We define a landscape trap as that wherein entire landscapes are shifted into a state in which major functional and ecological attributes are compromised. These shifts in a landscape lead to feedback processes that either maintain an ecosystem in a compromised state or push it into a further regime shift in which an entirely new type of vegetation cover develops. Landscape traps are large-scale ecological phenomena that arise through a combination of altered spatial characteristics of a landscape coupled with synergistic interactions among multiple human and natural disturbances. Thus, changes in the frequency and spatial contagion of large-scale disturbances are the key interacting factors driving entire landscapes into an undesirable and potentially irreversible state (i.e., landscape trap). We demonstrate the concept with examples involving spatial and temporal feedback between logging and fire in forest ecosystems and also provide examples of landscape traps in other environments.

Like other kinds of ecological traps, the landscape trap concept shares characteristics like shifts between alternative stable states and multiple feedback processes (9). However, a focus at a landscape scale and on temporal and spatial changes in disturbances sets the landscape trap concept apart from other kinds of ecological traps and regime shifts, such as population traps and extinction vortices in small populations of animals (10) and elevated rates of animal species loss below threshold levels of native vegetation cover (11).

To the best of our collective knowledge, the landscape trap concept has not been previously reported, yet we argue that landscape traps may be more prevalent in ecosystems around the world than currently recognized. Common ingredients contributing to landscape traps are (*i*) overharvesting of natural resources in a landscape; (*ii*) climate change effects on species' life histories and/or the frequency and severity of ecological disturbances; (*iii*) major changes in the spatial characteristics of landscapes; (*iv*) feedback loops between the changed environmental conditions and other major stressors; and (*v*) severely impaired ecological functions of a landscape in an altered state, such as, for example, reduced populations of species and habitat suitability, reduced carbon storage, and reduced water and timber production. The interaction of these factors is shown in a conceptual model in Fig. 1.

We suggest that landscape traps exist in many ecosystems. For example, logged tropical rainforests in parts of Asia have become more fire-prone (12). Postfire salvage logging in some of these rainforests, in turn, changes the vegetation composition toward more fire-prone grassland taxa. Additional fire further degrades fire-sensitive remnant rainforest, eventually leading to a regime shift to exotic fire-promoting grasslands, limiting opportunities for the vegetation to revert to tropical rainforest (13). Such kinds of interrelationships between logging and altered fire regimes are widespread in tropical rainforests in many other parts of the world, including South America and Africa (14), as are relationships between logging and exotic fire-prone grasses (15).

Temperate forests are not immune to such traps. In moist temperate forests of western North America, logging-related alterations in stand structure increase the risk for both occurrence and severity of subsequent wildfires through changes in fuel types and conditions (16, 17). High-severity wildfires kill young trees planted following previous logging operations. This necessitates reforestation efforts, but these young stands are susceptible to being killed in subsequent recurring high-severity fires (16). Similar kinds of relationships between logging regimes and altered fire regimes have been reported in a range of forest types elsewhere around the world (reviewed in 18).

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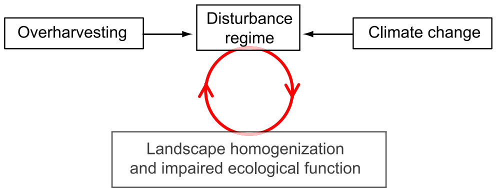

Fig. 1. Conceptual model of a landscape trap. The trap results from the reinforcing feedback loop shown in red.

Results and Discussion

Specific Example of a Landscape Trap: Mountain Ash Forests of Victoria, Southeastern Australia. The specific example of a landscape trap that we present comes from the mountain ash (*Eucalyptus regnans*) forests of southeastern Australia in the central highlands of Victoria. The likely regime shift is from landscapes dominated by old-growth forests that are 200–450 y of age to those dominated by young fire-prone forests that do not survive to become old growth. Evidence comes from new spatial information following massive wildfires in 2009, perhaps the most economically destructive in Australian history (19), coupled with understanding that has emerged from 28 y of extensive field information and associated data analyses in mountain ash forests (20).

The central highlands of Victoria support ∼121,000 ha of mountain ash forest. These are spectacular forests with old-growth trees reaching 90 m or more in height (14). Mountain ash forests persist only within a particular fire regime (*sensu* 21). Before European settlement over 150 y ago, the fire regime was infrequent severe wildfire that occurred in late summer (22). Young seedlings germinate from seed released from the crowns of burned mature trees to produce a new even-aged stand (20). Wildfires may be stand-replacing, because the young trees regenerating after fire belong to a single age cohort (23). When the interval between stand-replacing disturbances is less than 20–30 y (which is the period required for trees to reach sexual maturity and begin producing seed) (24), stands of mountain ash forest will be replaced by other species, particularly wattle (*Acacia* spp.) (20).

In the past century, a new disturbance regime (logging) has been added to the previous natural fire regime. Large areas of mountain ash have been subject to timber and pulpwood harvesting (Fig. 2). In the past 40 y, the traditional method of logging has been clear-cutting, in which all merchantable trees within a 15- to 40-ha area are cut in a single operation (25). Following clear-cutting, logging debris is burned to create a bed of ashes in which the regeneration of a new eucalypt stand takes place, often by artificial reseeding. The vast majority of mountain ash landscapes have become dominated by large areas of regrowth forest with small areas of old forest embedded within them. Old-growth mountain ash forest (*sensu* 20) typically covers less than 3% of the majority of the 3,000- to 6,000-ha wood production forest blocks in the central highlands; however, in some cases, it is less than 1% (20). Indeed, following more than a century of logging and wildfires in 1926, 1932, 1939, 1983, and, most recently, 2009, ∼1.1% of the entire mountain ash forest estate is now in an old-growth stage. This landscape is in stark contrast to mountain ash landscapes 100–150 y ago, which historical accounts (e.g., 26), coupled with stand reconstruction work relating to tree age and stem diameters of large dead (snag) trees remaining within young stands (27), suggest were dominated by large areas of old growth, possibly as high as 60–80% total cover in the central highlands of Victoria (20) (Fig. 2).

Development of a Landscape Fire-Trap in Mountain Ash Forests. The interacting effects of wildfire, logging, and the combination of wildfire and logging (i.e., salvage logging) (*sensu* 28) are creating a previously unrecognized landscape trap in which the disturbance dynamics of "trapped" mountain ash forest landscapes are markedly different from those before European settlement (Figs. S1 and S2). The core process underlying this landscape trap is a positive feedback loop between fire frequency/severity and a reduction in forest age at the stand and landscape levels, leading to an increased risk for dense young regenerating stands repeatedly reburning before they reach a more mature state (Fig. 3). The landscape trap will potentially create irreversible changes in disturbance dynamics, forest cover, landscape pattern, and vegetation structure, and thereby lead to a major regime shift or alternative state. We explain below the evidence for the positive feedback process that underpins this landscape trap (Fig. S2) and discuss why it is historically unprecedented and why it is beginning to dominate the contemporary landscape.

Positive feedback loop between reduced forest stand age and fire. Young stands of mountain ash forest are created by natural regeneration following wildfire. Detailed on-site measurements following the 2009 wildfires have revealed that young forest burns at higher severity than mature forest. We suggest this is for four key reasons:

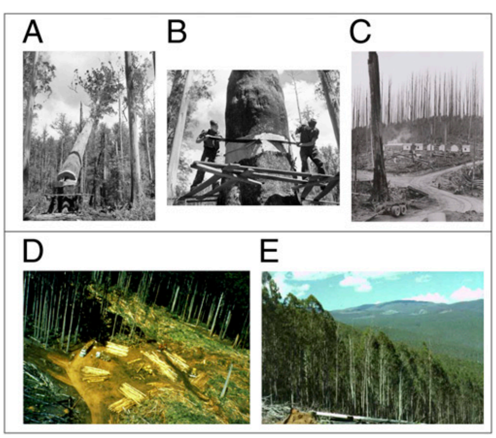

Fig. 2. Photo montage showing historical logging in extensive stands of old-growth forest (*A*–*C*) and extensive clear-cut areas of forest cut in the past 10 y (*D* and *E*) in the mountain ash forest in the central highlands of Victoria. (Photos courtesy of National Archives of Australia, State Library of Victoria and D.B.L.)

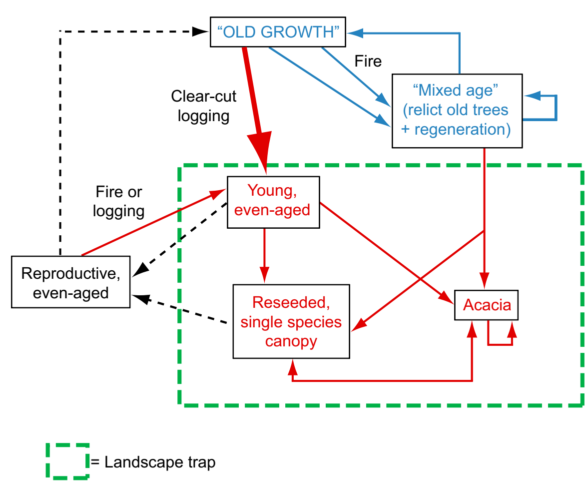

Fig. 3. Development of a landscape trap in the mountain ash forests of the central highlands of Victoria.

i) Young regenerating stands of mountain ash trees are characterized by densely spaced regrowth saplings. There can be several million eucalypt seedlings per hectare soon after a fire or logging. Through processes of rapid natural self-thinning, this declines to ∼400 stems per hectare at 40 y and 40–80 stems per hectare in mature forest after 150–200 y (29). The marked reduction in the number of stems per unit area over time is primarily attributable to competition-derived death and collapse of small suppressed pole and sapling trees, which add greatly to the density of the vegetation in young regrowing forests but do not generally occur in mature and old-growth mountain ash forests (30). Densely spaced stands of regrowth saplings, coupled with the subsequent natural processes of rapid self-thinning that characterize the early stages of stand regeneration in mountain ash forests, create significantly more fine and medium fuels than in old forests (31).

ii) The closely spaced crowns in densely stocked young stands are readily susceptible to carrying a crown fire. This is in contrast to old-growth stands, which are characterized by large relatively well-spaced trees with open crowns and small lateral subcrowns (24).

iii) Trees in young stands are shorter than those in old-growth stands. The flame height needed to scorch or consume the canopy in young stands is therefore significantly lower than in old-growth stands (22).

iv) Young forests support significantly smaller diameter logs on the ground than old-growth stands (32). Such smaller diameter logs support significantly less dense and luxuriant moss mats than larger diameter fallen trees. Moss mats hold large amounts of water (1,100% of dry weight) (33); they play a significant role in moisture retention within logs, and thereby may reduce the risk for burning.

Why has this positive feedback loop not occurred historically? Before European settlement, frequent, widespread, high-severity wildfires in mountain ash forests would have been suppressed by a combination of extended periods of wet climatic conditions and the absence of the intensive human disturbances resulting from clear-cut logging. This favored a negative feedback loop between forest age and fire, enabling young forest to mature into a less fire-prone state that was not conducive to widespread high-severity wildfire (Fig. S1).

Why is this positive feedback loop now beginning to develop? Two major changes have occurred relatively recently to favor the positive feedback loop: reduced forest age in mountain ash forests and increased fire frequency (Fig. 3 and Figs. S1 and S2). First, there has been a 25% reduction in rainfall in southeastern Australia over the past few decades (34). Second, logging has converted more than 90% of formerly old forest to young regenerating stands. Young forest resulting from clear-cut logging has two added elements of fire proneness: (*i*) fine fuels created by logging operations are added to those from the collapse of small-diameter stems and shedding of branches during natural self-thinning and self-pruning processes in densely stocked regenerating stands, and (*ii*) the spatial pattern of stand age classes in mountain ash landscapes has been altered, with an increased prevalence of young densely stocked forest and a significantly reduced area of (mesic) old-growth forest. This, in turn, has increased the fire contagion in the landscape.

Codes of logging practice and the practical logistics of harvesting operations mean that clear-cutting is applied to flatter and more accessible parts of mountain ash landscapes. However, these places are also where old-growth stands were formerly most likely to occur. Evidence for this comes from work in closed-water catchments of the central highlands of Victoria, where there were no confounding effects of past and present human disturbances that would have otherwise obscured key spatial patterns of forest age classes (22). Before the 2009 wildfires, old growth mountain ash occupied a subset of the overall environmental domain of mountain ash per se, typically within a narrow band of mesic sites rather than ridges or steep slopes. This environmental domain was not only favorable for tree growth but interacted with spatial differences in natural disturbance regimes (35). Mesic sites support taller trees. They are also places where both the fire frequency and the intensity of past wildfires were attenuated (22). Former areas of old-growth forest on flat terrain have now been converted to young regenerating stands and are spatially connected to young burned or logged forest on midslopes and ridges. Importantly, the more widespread that young logged and regenerated forest becomes, the greater is the risk for increasing spatial contagion in the spread of wildfire through landscapes (31), because moist remnant areas that would have slowed or halted the spread of fire (and formerly supported old forest) have been converted to young forest. Spatial contagion in recurrent high-severity fire may therefore reinforce a pattern of increasing homogeneity in the cover of young forest in a landscape (Fig. S2). This pattern occurs because some areas of fire refugia (e.g., flat plateau, deep south-facing valley floors) that were traditionally characterized by a long absence of fire (particularly high-severity fire) and supported stands of multiaged forest or old-growth forest (35) become more susceptible to being burned by high-severity conflagrations that spread from adjacent more flammable logged and young regenerating areas (Figs. S1 and S2). Notably, although natural disturbance regimes often increase heterogeneity in many landscapes (36), the opposite frequently occurs in areas subject to landscape trap phenomena, in which the combination of human and natural disturbance regimes can lead to increased landscape homogeneity.

Research in moist forests around the world suggests that other factors associated with logging may increase susceptibility of young regenerating forests to being burned or reburning at high severity. For example, the large quantities of logging slash created by harvesting operations can sustain fires for longer than fuels in unlogged forest (12). Similarly, lightning strike ignition is more likely to occur in harvested stands because of increased fine fuels resulting from logging slash, and this effect may remain for 10–30 y following logging (37). Finally, the removal of trees by logging creates microclimatic conditions that lead to increased drying of understory vegetation and the forest floor, and a correspondingly elevated fire risk (38).

Once a mountain ash forest landscape is dominated by widespread areas of young fire-prone forest, the elevated risk for high-severity spatially contagious fire decreases the probability that the landscape can return to its former mature state, particularly under the drier and warmer conditions associated with climate change. Hence, the dynamics of trapped mountain ash forest landscapes are different from those in the past (>100 y ago) (Fig. 3 and Figs. S1 and S2). The current set of interacting disturbance regimes of fire, logging, and postfire (salvage) logging did not exist before European settlement. Importantly, there is a major asymmetry in the period during which mountain ash forest ecosystems have coevolved with natural disturbances (>20 million y) compared with the 20–100 y during which the interacting human and natural disturbance regimes have produced a landscape trap.

End point: Regime shift? The positive feedback cycle of widespread young regenerating stands and frequent high-severity wildfire means that either extensive areas of trapped young mountain ash forest will be maintained or a further regime shift will occur in which a new type of vegetation cover develops, particularly wattle (*Acacia* spp.) (Fig. 3 and Figs. S1 and S2). Once mountain ash has been eliminated from an extensive area, it recolonizes slowly because the seed released from the crowns of burned mature trees disperses ∼1.5–2.0 crown heights from a source

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tree and successful regeneration (fire) events may occur every 30–400 y. Therefore, the regeneration niche, which is a key part of the life cycle of mountain ash (39), is maladapted to the altered landscape conditions and altered fire regime created by recurrent logging and wildfire. Recurrent high-frequency wildfire may result in repeatedly burned areas that were formerly dominated by mountain ash being colonized by other eucalypt species that do not depend on seedling regeneration but, instead, recover after wildfire via strategies like epicormic resprouting [e.g., shining gum (*Eucalyptus nitens*), messmate (*E. obliqua*)].

Irrespective of whether mountain ash forest landscapes remain trapped as widespread, young, fire-prone stands or undergo a regime shift to extensive areas dominated by *Acacia* spp. and other species, such changes will result in significant impairment of ecological functions like carbon storage, water production (40, 41), and biodiversity conservation. For example, neither young small-diameter mountain ash trees nor *Acacia* spp. support the cavities that are crucial nesting and denning sites for many species of animals. They also lack critical structural features, such as extensive bark streamers, that are key foraging microhabitats for wildlife (42). These changes in vegetation structure are likely to lead to irreversible losses in habitat suitability for ∼40 species of vertebrates in mountain ash forests that are dependent on large 120- to 150+-y-old trees with hollows.

Avoiding a Landscape Trap in Mountain Ash Forests of Victoria. Three important strategies are needed to reduce the problems created by the landscape trap in the mountain ash forests of Victoria. First, large (>1,000 ha) areas of currently unburned forest need to be retained, wherein the number of anthropogenic stressors is reduced. The area of green forest was reduced dramatically by the 2009 wildfires; hence, relative biodiversity, carbon storage, and water production values of remaining unburned forest have increased. However, such uncommon areas of unlogged forest are increasingly sought after for timber and pulpwood harvesting because (*i*) they are among the declining number of places suitable for cutting as a consequence of past fires and past (prefire) logging operations, (*ii*) there are legislated guarantees to provide logging contractors with forest to cut for timber and pulpwood (43), and (*iii*) cutting burnt forest (i.e., salvage logging) has major negative environmental impacts and long-term effects on forest recovery and forest biodiversity (28). Targeting limited remaining areas of unburned forest for logging depletes the overall amount of these forests, with long-term economic implications for harvest contractors. Increased logging pressure on green areas has other ecological implications: Remaining areas of green forest are important refugia for biodiversity following wildfires and are critical for underpinning postfire ecological recovery (32). Legislative and other impediments to reducing harvest levels highlight the existence of management and socioeconomic traps within landscape traps, and these need serious and timely review.

A second strategy to avoid the development of a landscape trap in the now highly fire-prone mountain ash landscapes of Victoria is to recalculate the sustained yield to accommodate future losses of timber resulting from the inevitable burning of some parts of forest landscapes. This strategy has the advantage of not overcommitting remaining unlogged green forest in the event of wildfires, thereby resulting in more conservative management of natural resources and more explicit recognition of the uncertainty created by major natural disturbances.

Given the extent of recently burned forest in Victoria, a third important strategy to reduce the risks for development of a landscape trap is to try to limit the amount of future fire. Although mountain ash trees are dependent on fire to promote regeneration, fires have been extensive in the past 25–100 y; another fire in the coming 20 y within currently young regenerating stands is likely to lead to a major regime shift (Fig. 3). Reducing the amount of fire in mountain ash forests is a significant challenge. Broad-area prescribed burning is not a viable management option because high levels of moisture in the vegetation and large quantities of biomass make planned fires extremely difficult to control (20). However, prescribed burning as part of a regime of fire can be an appropriate management option in drier forest types that are adjacent to mountain ash forests. Carefully applied strategic burning in such drier environments may help to reduce the extent of spatial contagion in wildfire that occurs in these areas and, in turn, reduce the risk for adjacent stands of mountain ash forest being burned (44).

Examples of Landscape Traps in Ecosystems Other Than Forests. We contend that landscape traps may be prevalent in many ecosystems. For example, climate change and overfishing have facilitated the conversion of subtidal kelp (*Macrocystis pyrifera*) forests in Tasmanian coastal waters to "barrens" habitat resulting from overgrazing by the sea urchin *Centrostephanus rodgersii*. Ocean warming and altered circulation patterns have enabled the poleward spread of this sea urchin (45), and overfishing of predators, such as the southern rock lobster (*Jasus edwardsii*), has enabled *C. rodgersii* to establish high-population density barrens that result in the loss of biodiversity and a reduction in the productivity of fisheries and contribute to the decline of such predators as *J. edwardsii* (46). Aquatic environments where water quality can be radically altered by nutrient inputs from human activities (e.g., 47) also are susceptible to the development of landscape traps.

Grazing on public lands in the western United States has been blamed for reducing biodiversity and, together with exotic weeds, may have led these grassland ecosystems into a landscape trap that produces a plant community from which there is no going back (48). Livestock grazing in western United States may have reduced the abundance of preferred plant species while subjecting the soil to weed invasion, such that large areas are now degraded rangelands in the same manner illustrated in eastern Australia by the "woody weed" problem in semiarid woodlands (49). Introduced grasses, such as cheatgrass (*Bromus tectorum*), can similarly move grassland communities in the intermountain western United States into a regime change that is nearly impossible to reverse (50, 51). A lack of reversible change may be best illustrated by landscape traps in regions heavily impacted by disturbances like mountaintop mining (52).

Concluding Comments

We suggest that strategies and management interventions are needed to reduce the probability of landscape traps developing (Fig. 4). One approach is to recognize that landscape traps can exist and identify the suite of spatial and temporal characteristics that can combine to give rise to them, including (*i*) exploitation of the natural resources in a landscape through unsustainable levels of harvesting; (*ii*) alteration in the spatial characteristics of landscapes, including modifications to the frequency and severity of ecological disturbances; (*iii*) feedbacks between altered environmental conditions and other major anthropogenic stressors;

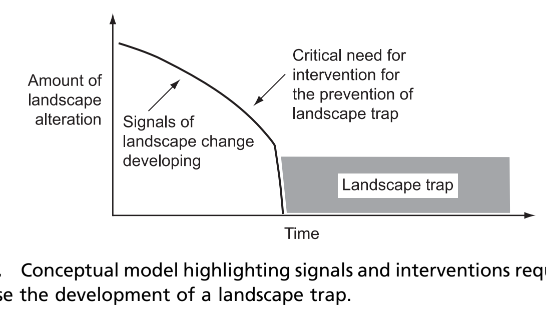

Fig. 4. Conceptual model highlighting signals and interventions required to reverse the development of a landscape trap.

and (*iv*) severely impaired landscape processes and functions. A second approach is to limit the number of anthropogenic stressors in landscapes and reduce the potential for negative interactions among multiple stressors. This may equate to a more conservative approach to the harvesting of natural resources or, in other cases, application of management strategies that reduce feedbacks (e.g., fuel reduction through prescribed burning). Sustained yields of natural resources also may need to be rapidly reassessed following catastrophic events to avoid overcommitting remaining intact areas and further increasing the risk for creating a landscape trap.

We suggest that the need for proactive management to prevent the development of landscape traps is critical, given that (*i*) landscape traps might be at increased risk for development in response to significant "events" like major natural disturbances, which are likely to become more frequent, more severe, or both under rapid climate change in many regions (e.g., 53, 54), and (*ii*) marked asymmetry exists between the rapidity with which landscape traps may develop and the prolonged time scales (hundreds to thousands of years) that characterize natural ecological processes and natural disturbance regimes.

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CONSERVATION ECOLOGY

Report

USDA Forest Service Roadless Areas: Potential Biodiversity Conservation Reserves

Colby Loucks[1], Nicholas Brown[2], Andrea Loucks[3], and Kerry Cesareo[1]

ABSTRACT. In January 2001, approximately 23 x 10^6 ha of land in the U.S. National Forest System were slated to remain roadless and protected from timber extraction under the *Final Roadless Conservation Rule*. We examined the potential contributions of these areas to the conservation of biodiversity. Using GIS, we analyzed the concordance of inventoried roadless areas (IRAs) with ecoregion-scale biological importance and endangered and imperiled species distributions on a scale of 1:24,000. We found that more than 25% of IRAs are located in globally or regionally outstanding ecoregions and that 77% of inventoried roadless areas have the potential to conserve threatened, endangered, or imperiled species. IRAs would increase the conservation reserve network containing these species by 156%. We further illustrate the conservation potential of IRAs by highlighting their contribution to the conservation of the grizzly bear (*Ursos arctos*), a wide-ranging carnivore. The area created by the addition of IRAs to the existing system of conservation reserves shows a strong concordance with grizzly bear recovery zones and habitat range. Based on these findings, we conclude that IRAs belonging to the U.S. Forest Service are one of the most important biotic areas in the nation, and that their status as roadless areas could have lasting and far-reaching effects for biodiversity conservation.

INTRODUCTION

In January 2001, the Clinton administration promulgated its *Roadless Area Conservation Rule,* which states that 237,000 km^2 of inventoried roadless areas (IRAs) within the U.S. National Forest System will remain roadless and protected from timber extraction (USDA Forest Service 2000). These lands represent 31% of the National Forest System or 2.5% of the total U.S. land base (DeVelice and Martin 2001). They would increase the amount of strictly protected land area in the United States in IUCN categories I–III from 4.8 to 8.5%. Beyond these most basic statistics, few studies have analyzed the potential contribution of IRAs to biodiversity conservation (Martin et al. 2000, DeVelice and Martin 2001).

DeVelice and Martin (2001) assessed the extent to which IRAs could contribute to building a representative network of conservation reserves in the United States. Using ecoregions as their unit of analysis (Ricketts et al. 1999), they found that IRAs could potentially expand ecoregional representation, increase the area of reserves at lower elevations, and increase the size of conservation areas to provide refuge for wide-ranging species. However, in their assessment they did not evaluate the contribution of IRAs toward the conservation of biodiversity and populations of specifically threatened, endangered, or imperiled species.

The lands belonging to the USDA Forest Service contain more than 80% of mammal and reptile species and more than 90% of the bird, amphibian, and fish species in the United States, including many that have been extirpated from large portions of their presettlement ranges (USDA Forest Service 1997). According to the NatureServe database, more than 1400 of these species have been designated as threatened and endangered (TE) species under the *Endangered Species Act* (ESA). The *Forest Service Roadless Area Final Environmental Impact Statement* identified approximately 400 TE or proposed species found on USDA Forest Service land and an estimated 220 (55%) that are directly or indirectly associated with IRAs (USDA Forest Service 2000). IRAs provide or influence designated critical habitat for at least 30 of these species (USDA Forest Service 2000).

However, the ESA list is not a complete listing of imperiled species. There are numerous species that are globally rare or threatened with extinction but for

[1]World Wildlife Fund; [2]NatureServe; [3]Pinchot Institute

various reasons do not appear on the ESA TE species list. Many of these species also occur on USDA Forest Service land. To fill this gap, we supplemented the TE species list with species categorized as critically imperiled or imperiled according to NatureServe's central database.

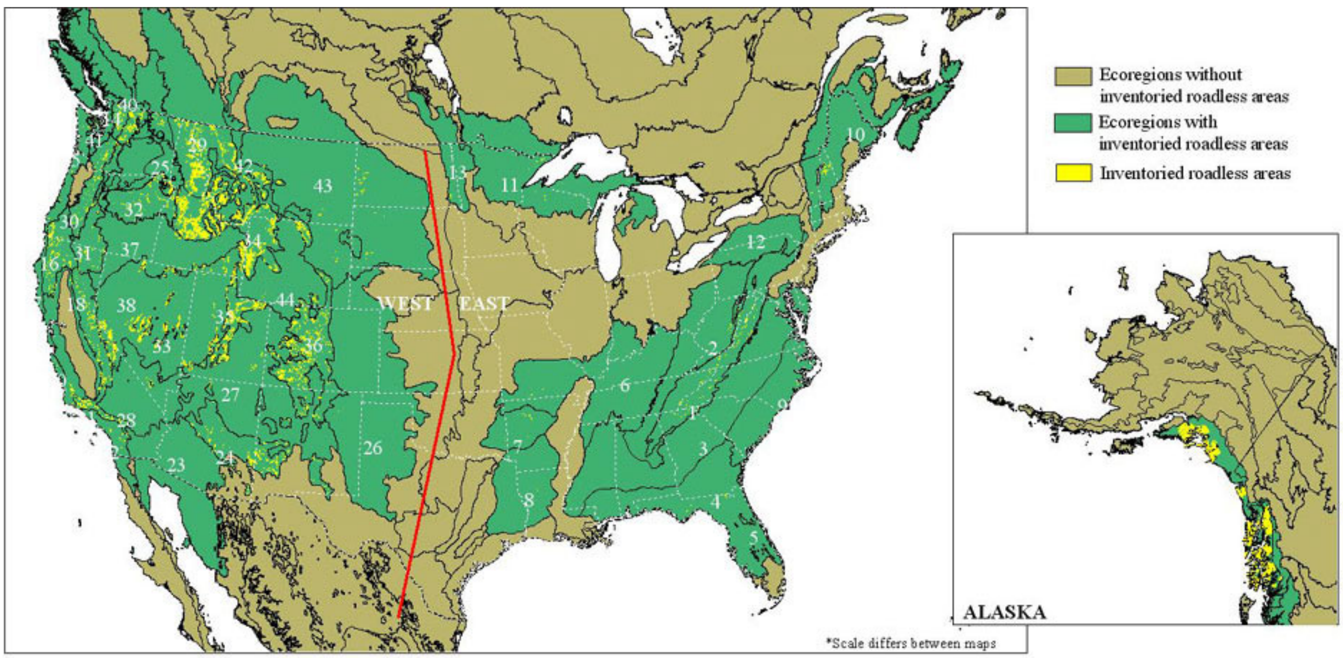
The objective of this paper is to assess three critical questions associated with IRAs:

> Is there a high concordance between IRAs and ecoregions of particular biodiversity values?
>
> Do IRAs overlap with threatened, endangered, or imperiled species?
>
> Is there potential for IRAs to assist in the conservation of wide-ranging species, such as the threatened grizzly bear (*Ursos arctos horribilis*), in the conterminous United States?

METHODS

We obtained the spatial coverages of the inventoried road areas (IRAs) in vector format from the roadless area conservation Web site (Table 1).

Fig. 1. Overlap of USDA Forest Service inventoried roadless areas (IRAs) with ecoregions that contain USDA Forest Service lands. The bold line indicates the separation of IRAs into three geographic regions: east, west, and Alaska.



1. Appalachian-Blue Ridge Forests
2. Appalachian Mixed Mesophytic Forests
3. Southeastern Mixed Forests
4. Southeastern Conifer Forests
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16. Klamath-Siskiyou Forests
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18. Sierra Nevada Forests
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20. California Interior Chaparral & Woodlands
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25. Palouse Grasslands
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42. Montana Valley and Foothill Grasslands
43. Northwestern Mixed Grasslands
44. Wyoming Basin Shrub Steppe
45. Northern Pacific Coastal Forests
46. Pacific Coastal Mountain Tundra & Ice Fields

Table 1. Data sources. All data web data sources were accessed in February 2001.

Database name	Source
USDA Forest Service roadless area database	http://roadless.fs.fed.us/documents/feis/data/gis/coverages/index.shtml
World Wildlife Fund ecoregions database	Ricketts et al. 1999
NatureServe central databases	NatureServe
Protected areas database	Conservation Biology Institute and World Wildlife Fund
Grizzly bear recovery area boundaries	U.S. Fish and Wildlife Service and University of Montana

Ecoregions

As seen in Fig. 1 and Table 1, we evaluated the potential benefit of IRAs for biodiversity conservation using the ecoregions and biological importance rankings provided in Ricketts et al. (1999). Using ArcView 3.2, we combined the IRAs and ecoregion coverages, both in vector format. To facilitate interpretation, we separated our analysis into three geographic regions, i.e., the eastern United States, the western United States, and Alaska, following the methodology used by DeVelice and Martin (2001).

Ricketts et al. (1999:7) defined an ecoregion as " ... a relatively large area of land or water that contains a geographically distinct assemblage of natural communities." Ecoregions were selected as the units of analysis because they integrate ecological, biological, and geographic considerations into land-use decision making and are being used to establish priorities for large-scale conservation efforts (Omernik 1995*a,b*, Ricketts et al. 1999, Groves et al. 2002). Where ecoregions extend into either Canada or Mexico, we included only those portions within U.S. boundaries for all analyses. Although we would have preferred to maintain ecoregional contiguity, the spatial nature of USDA Forest Service lands and the applicability of the *Endangered Species Act* required strict adherence to political boundaries.

Ricketts et al. (1999) classified the biological importance of each ecoregion based on species distribution, i.e., richness and endemism, rare ecological or evolutionary phenomena such as large-scale migrations or extraordinary adaptive radiations, and global rarity of habitat type, e.g., Mediterranean-climate scrub habitats. They used species distribution data for seven taxonomic groups: birds, mammals, butterflies, amphibians, reptiles, land snails, and vascular plants (Ricketts et al. 1999). Each category was divided into four rankings: globally outstanding, high, medium, and low. The rankings for each of the four categories were combined to assign an overall biological ranking to each ecoregion. Ecoregions whose biodiversity features were equaled or surpassed in only a few areas around the world were termed "globally outstanding." To earn this ranking, an ecoregion had to be designated "globally outstanding" for at least one category. The second-highest category, or continentally important ecoregions, were termed "regionally outstanding," followed by "bioregionally outstanding" and "nationally important" (Ricketts et al. 1999). Although our analyses focused on those ecoregions characterized as globally and regionally outstanding, even the lowest category, nationally important, contains important biodiversity in a local context.

Threatened, endangered, and imperiled species

Currently, public land managers are required to monitor populations of threatened and endangered (TE) species and, where appropriate, develop management plans to conserve these populations and their habitat requirements (U.S. Fish and Wildlife Service 1973). Previous studies have analyzed the distribution of TE species based on counties, or boroughs in Alaska, and identified high-concentration areas of TE species and associated habitats (Dobson et al. 1997, Flather et al. 1998, Stein et al. 2000). Despite their valuable findings, these previous studies were limited by the coarse level of spatial resolution and the use of political units of disparate sizes. To avoid similar limitations with our analysis, we use data of a finer resolution to identify levels of concordance between the locations of IRAs and TE species.

The NatureServe central database (Table 1) provided the finer-resolution data for the identification of the locations of TE species. Data for this database are developed by state natural heritage programs and managed by NatureServe. Natural heritage programs have documented and tracked the occurrence of threatened, endangered, and imperiled species for nearly 30 yr (Jenkins 1985, 1988, 1996). The system assigns global conservation status ranks known as "element global ranks" or "G-RANKS" to species and communities that are intended to estimate the extent of their imperilment or vulnerability. Conservation status ranks are assigned based on an assessment of rarity, the extent of recent decline of populations, threats, biological fragility, and other factors (Stein et al 2000). The most imperiled species and communities are ranked G1, and the most stable ones are ranked G5.

The NatureServe central database includes fields for federal ESA listing status and for global conservation status. We selected records of species that are federally listed as threatened or endangered (TE) according to the U.S. Fish and Wildlife Service or the National Marine and Fisheries Service and those that are ranked by NatureServe as critically imperiled (G1) or imperiled (G2). The output file was a vector file of 109,125 occurrences of species with G1 or G2 rankings or federal ESA listings. These occurrences were collated into 7.5-min quadrangles from the U.S. Geological Survey. The largest quadrangles, in the southern part of the United States, are 179 km^2. We used two data products for our analyses. The first contains only TE species (Fig. 2), and the second contains TE, G1, and G2 species (Fig. 3). The spatial resolution of the locational data varied according to the equipment and methodologies that natural heritage programs used in collecting the data. However, the maximum uncertainty for the data set was less than the area of a quadrangle grid cell.

Fig. 2. Threatened and endangered (TE) species distributions by the 7.5-min quadrangles of the U.S. Geological Survey.

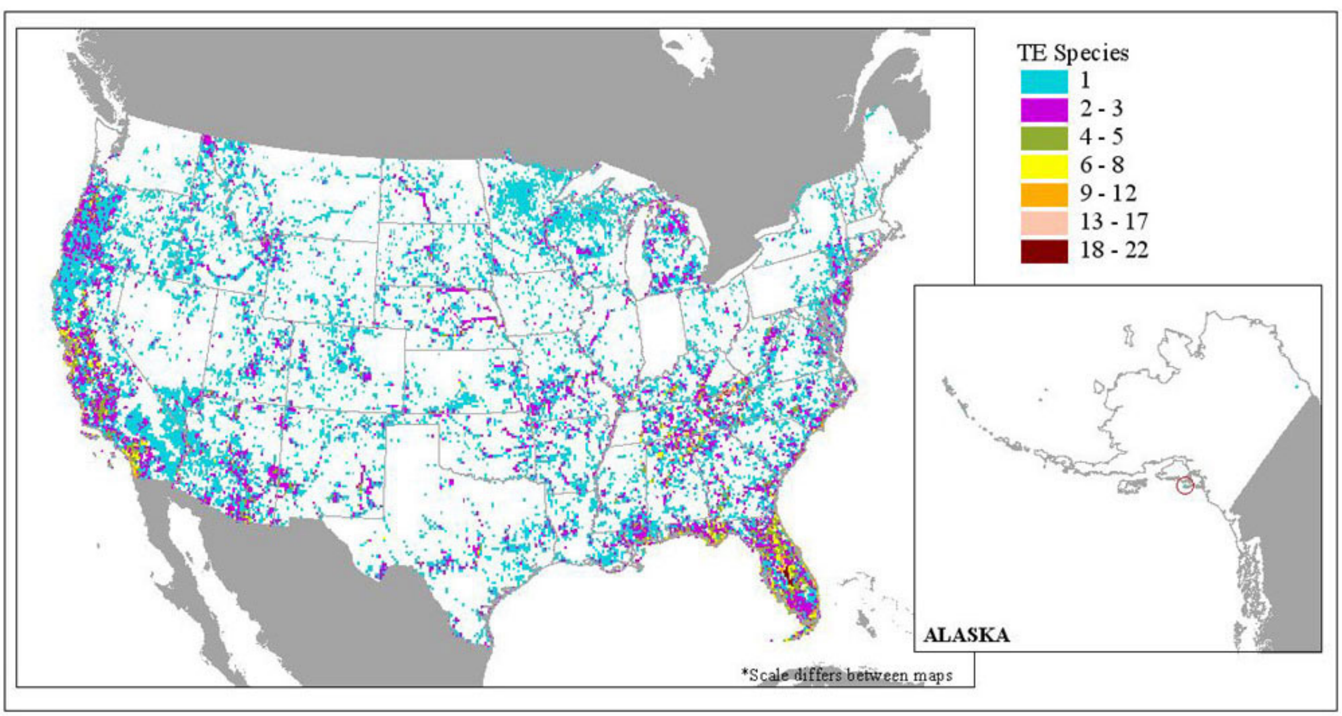
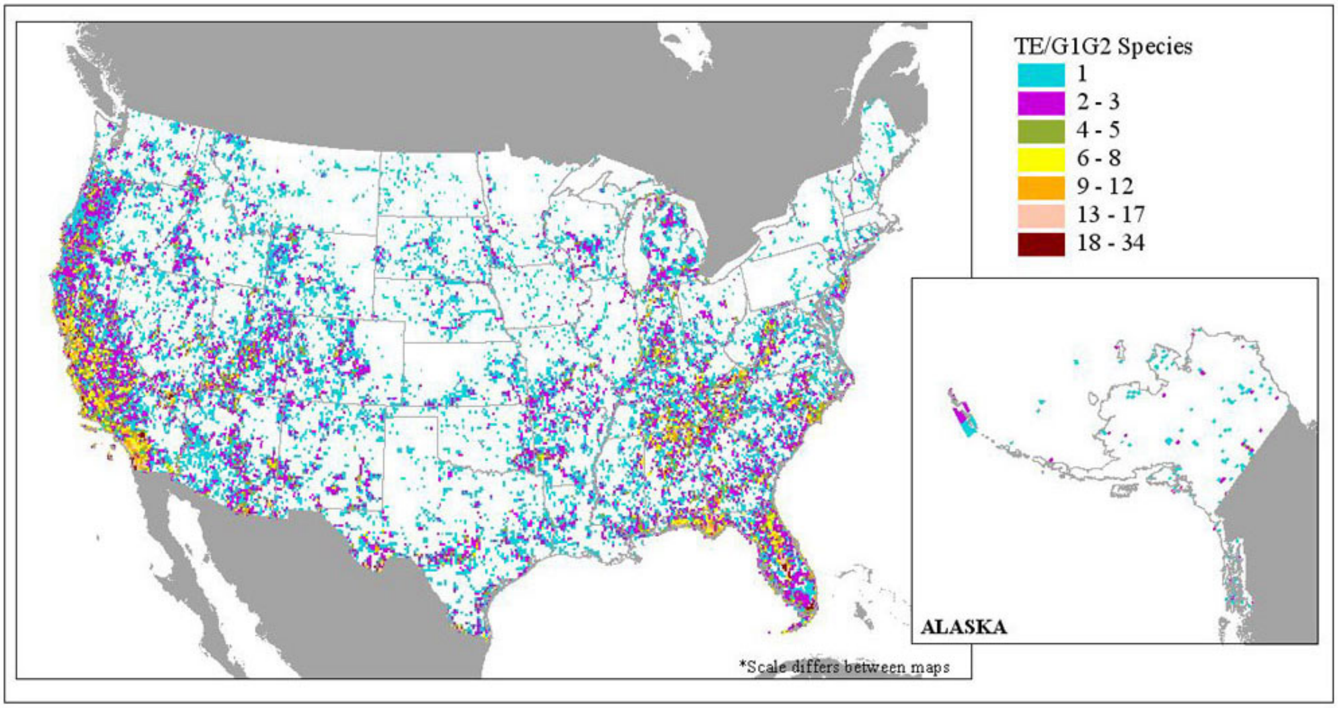


Fig. 3. Threatened and endangered (TE) species and critically imperiled (G1) and imperiled (G2) species distributions by the 7.5-min quadrangles of the U.S. Geological Survey.



The TE, G1, and G2 data sets demonstrate only a moderate degree of overlap. These discrepancies occur partly because the NatureServe system evaluates only biological factors, whereas species are assigned to federal listings for both scientific and political reasons. There are 75,000 occurrences of TE species, but only 27,000 are ranked G1 or G2 by the NatureServe system. Of the 1409 ESA-listed TE species in the NatureServe database, 1109 are ranked G1 or G2. Conversely, there are 5997 species ranked G1 or G2 that are not classified as TE species. Of the 61,000 occurrences of G1 and G2 species recorded in the NatureServe database, more than 33,000 occurrences lack a TE species designation. One of the reasons for the disparity between the high concordance of species but the low concordance of occurrences is the fact that certain species are wide-ranging. For example, the grizzly bear, which is a threatened species but not a G1 or G2 species, is recorded often across its wide range, so that it accounts for far more records than a narrow endemic species that is both TE and listed as G1 or G2.

The NatureServe database contains information gaps (Table 2). However, although the missing data for Idaho, Montana, and Washington are critical for the conservation of individual species, the lack of them served only to make our analysis a more conservative estimate of the potential contributions of IRAs to species conservation. There are no IRAs in Massachusetts and only one in Maine, with a total area of 24 km^2.

We overlaid both the TE species and TE/G1–G2 species databases with the uniquely named IRAs to identify the percentage of IRAs that contain known occurrences of TE or G1–G2 populations. In instances where multiple quadrangles containing species occurred within a single IRA unit, we erred on the conservative side and used only the quadrangle that contained the most species, i.e., we assumed that multiple quadrangles would contain the same species.

We also analyzed the relative increase in conservation reserves that IRAs would confer to TE and TE/G1–G2 species. We overlaid the TE and TE/G1–G2 databases with a conservation area database compiled by the Conservation Biology Institute and World Wildlife Fund (Table 1). This database includes all federal, state, county, and municipal public lands and some private lands. The private lands have not been systematically surveyed and do not include conservation easements. We used only lands that are classified for strict biodiversity conservation, which we define as those designated as categories I–III by the IUCN. Category I is for Strict Nature Reserves/Wilderness Areas, category II covers National Parks, and category III includes National Monuments (The World Conservation Union 1978, The World Conservation Union 1994). Hereafter we refer to the areas that meet these criteria as "conservation reserves." We did not include protected-area categories IV–VI, which allow road building, timber harvesting, and other extractive activities in our analysis. Of 78 x 10^6 ha of National Forest land, 14 x 10^6 ha are designated as National Wilderness Areas, and an additional 2.5 x 10^6 ha are classified as Special Designated Areas that are IUCN category I reserves. The remaining 61.5 x 10^6 ha of National Forest land, which are not classified as conservation reserves, are governed by periodic management plans that may allow or restrict resource uses and extraction.

Table 2. Gaps in data available for this study.

State	Missing data
Idaho	Fish data
Maine	Animal data
Massachusetts	All data
Montana	Canada lynx, bull trout, gray wolf data
Washington	Most animal data

Grizzly bear case study

Finally, because national analyses can obscure important details of individual species, we also analyzed the potential contribution of IRAs to grizzly bears (*Ursos arctos horribilis*), specifically in relation to the regions designated as grizzly bear recovery areas by the U.S. Fish and Wildlife Service (Table 1). We overlaid these grizzly bear recovery zones with the IRAs to assess the concordance of these areas. We chose grizzly bears because they are a federally listed threatened species in the conterminous United States and require large and contiguous habitat areas to survive.

All spatial databases were in vector format and put into a common projection prior to the overlap analysis. All spatial estimates derived from our analyses were obtained by summarizing the area of overlap of the respective GIS databases. One caveat of our methodology is that the combination of multiple GIS layers may lead to the propagation of spatial errors and increased uncertainty (Flather et al. 1998, Heuvelink 1998). This concern is a generalized methodological one. Our errors are no greater or smaller than those of any similar analysis that uses multiple spatial data from multiple sources. The TE species databases, protected areas database, and IRA coverages represent a vast collection of data from many sources. It is likely that errors are associated with each of these layers. However, most of our analyses were conducted at a sufficiently broad scale that we believe the error rate is not large enough to affect our ultimate conclusions.

RESULTS

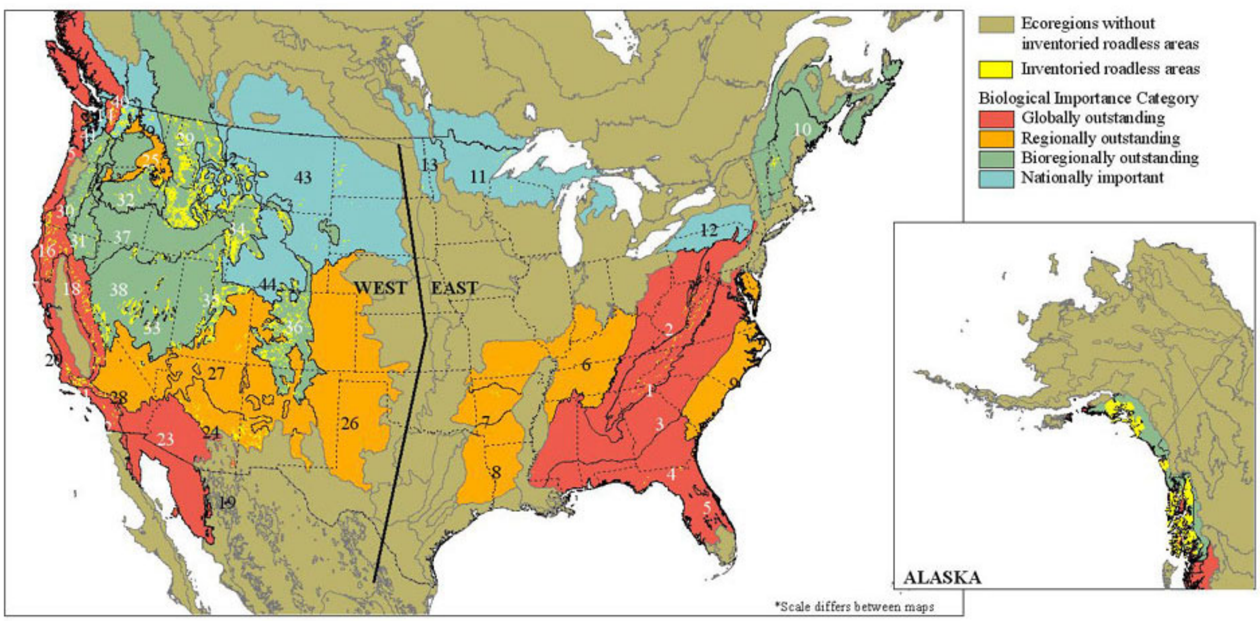
Ecoregions

Across the United States, we found that more than 20% of inventoried roadless areas (IRAs) were located within ecoregions that have been classified as globally outstanding (Table 3, Fig. 4). In the eastern region, approximately 70% of the IRAs are found in globally or regionally outstanding ecoregions (Table 3, Fig. 4). More than 50% of these forests occur in two Appalachian ecoregions, the Appalachian-Blue Ridge forests and the Appalachian mixed mesophytic forests. Both are considered globally outstanding for their diverse endemic species, which range across many taxa (Stephenson et al. 1993, Ricketts et al. 1999). The vast majority of the IRAs in eastern forests are less than 10.1 km^2 in size, and few are adjacent to existing wilderness areas (DeVelice and Martin 2001).

Table 3. Distribution of inventoried roadless areas (IRAs) by category of ecoregion biodiversity as per Ricketts et al. (1999). The percentage is the percentage of IRAs that fall into that particular category.

Biodiversity category	km^2	Percentage
Globally outstanding	50,221	21.2
Regionally outstanding	12,648	5.4
Bioregionally outstanding	164,600	69.5
Nationally important	9268	3.9

Fig. 4. Overlap of USDA Forest Service inventoried roadless areas and ecoregions classified by biological importance (see Ricketts et al. 1999).



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In the western region, IRAs are found predominantly in bioregionally outstanding ecoregions, with only 18% in globally or regionally outstanding ecoregions (Table 3, Fig. 4). Although globally and regionally outstanding IRAs are found mainly in the states of California, Oregon, Washington, and Arizona, the intermountain west contains most of the nation's bioregionally and nationally important IRAs. Western IRAs are on average larger than eastern IRAs, and the vast majority are adjacent to existing wilderness areas. If the IRAs were combined with the wilderness areas, the western forests would contain 34 of the 45 largest contiguous areas of strictly protected forests in the United States (DeVelice and Martin 2001).

Table 4. Comparison of the degree of overlap between inventoried roadless areas (IRAs) and quadranges containing threatened or endangered (TE) species or quadranges containing TE species that are also ranked as highly imperiled (G1–G2) by the IUCN. The mean number of TE or TE/G1–G2 species present in each IRA is given.

Region	Total no. of IRA units†	No. of IRA units with TE species quadrangles (% of total)	Mean no. of species‡	No. of IRA units with TE/G1–G2 species quadrangles (% of total)	Mean no. of species‡
Eastern United States	286	201 (70.3)	2.1	228 (79.7)	4
Western United States	2159	1317 (61.0)	1.6	1692 (78.3)	2.9
Alaska	150	2 (1.3)	1	88 (58.6)	1.3

†Units are defined by each named inventoried roadless area.
‡Where multiple quadrangles occurred in a single IRA unit, we used only the quadrangle with the greatest number of species.

Threatened, endangered, and imperiled species

Of the 2595 IRA units, approximately 58% of them overlap with TE species quadrangles (Table 4). When separated into geographic regions, the IRAs in the eastern and western United States demonstrate overlaps of 70.3 and 61.0%, respectively. Of the IRAs that contain TE species, the mean number of TE species found in IRAs is highest in the east (2.1 species) and lowest in Alaska (1.0 species).

When G1–G2 species are included in the analysis, both the number of IRAs that contain TE/G1–G2 species and the mean number of species of concern found in each IRA increase (Table 4). In sum, approximately 77% of the IRAs overlap with quadrangles that contain species at risk. The Alaska region contains the largest increase in IRAs when G1–G2 species are included, increasing to 58.6 from 1.3%. The west increases to 78.3%, and the east increases to 79.7%. However, the east shows the largest increase in mean number of TE/G1–G2 species found in IRAs, increasing from 2.1 to 4.0 species (Table 4).

The IRAs could also contribute a significant amount of land area to existing conservation reserves for both TE and TE/G1–G2 species in all geographic regions (Table 5). The largest increase in area and the greatest percent increase in conservation reserves are found in the western United States, with the exception of the 100% increase from the single quadrangle in Alaska. IRAs would contribute to a 96% increase in available habitat in conservation reserves for TE species, whereas the inclusion of G1–G2 species expands that increase to 210%. Although the eastern region would see similar but more modest gains, habitat in conservation reserves in the Alaska region would increase 113% for TE/G1–G2 species (Table 5). Overall, IRAs would increase the conservation reserve network containing TE, G1, or G2 species by 156%.

Table 5. The concordance of occurrences of threatened or endangered (TE) species or of TE species that are also classified as highly imperiled (G1–G2) by the IUCN with the existing conservation reserve network (IUCN I–III) and inventoried roadless areas (IRAs).

Region	No. of TE species quadrangles in IUCN I–III conservation reserves	No. of TE species quadrangles in IRAs	Percent increase	No. of TE/G1–G2 species quadrangles in IRAs	No. of TE/G1–G2 species quadrangles in IUCN I–III conservation reserves	Percent increase
Eastern United States	995	217	22	1027	431	42
Western United States	1752	1679	96	2200	4627	210
Alaska	0	1	100	38	43	113

Grizzly bear case study

As seen in Fig. 5, the inclusion of IRAs in the existing system of conservation reserves in Washington, Idaho, Montana, and Wyoming shows a strong concordance with the grizzly bear recovery zones of the U.S. Fish and Wildlife Service, as well as bear habitat range (Martin et al. 2000, USDA Forest Service 2000). In total, the six grizzly bear recovery zones include approximately 15,300 km^2 of IRAs. Approximately 24,750 km^2 of almost contiguous IRAs surround the Salmon-Selway (Bitterroot) Recovery Zone (SSRZ), which has already been designated a wilderness area and assigned to IUCN category I.

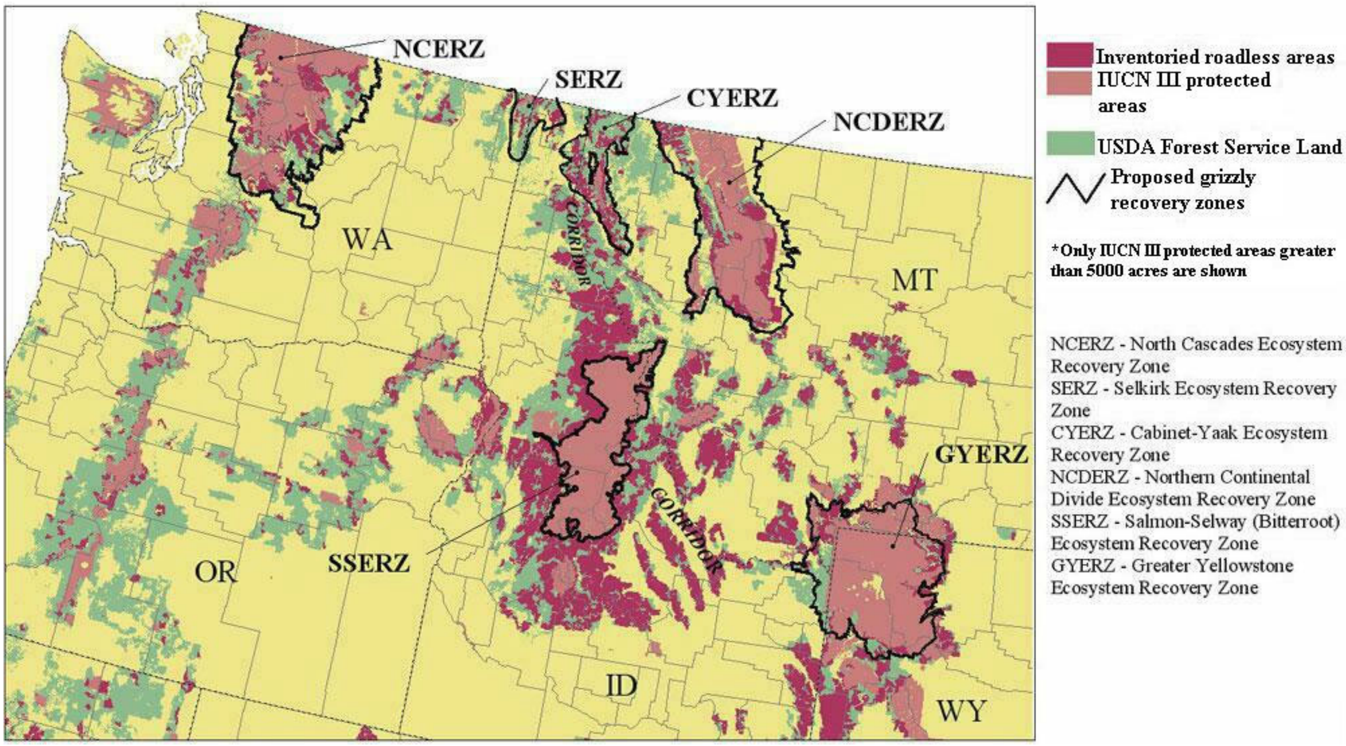
DISCUSSION

Our analyses found that one-quarter of the inventoried roadless ares (IRAs) are found in globally or regionally outstanding ecoregions, and that they have the potential to provide important habitat for numerous species, including threatened, endangered, and imperiled species. This conclusion is further illustrated by an investigation of the potential benefit of IRAs to grizzly bear conservation.

Based on these findings, the assignment of IRAs to IUCN category III or higher could increase the area of conservation reserves in the United States from 4.8 to 8.5%. This broad national conclusion has different implications depending on geographic region. For example, whereas fewer than 3% of the IRAs are found in the eastern United States, the vast majority of eastern IRAs are found in the ecoregions with the greatest amount of biodiversity and the least amount of existing protection. In addition, despite the fact that western forests currently have some of the highest existing protection levels in the United States, Scott et al. (2001) found that many existing reserves in the United States are concentrated in areas of high elevation and low soil productivity. Therefore, despite the current levels of perceived protection, the nation's biological diversity may be under-represented in the current system, particularly in the mountainous west (Scott et al. 2001). DeVelice and Martin (2001) have shown that approximately 40% (about 91,300 km^2) of the IRAs are at an elevation below 1500 m and that 35% of the total IRAs are adjacent to designated wilderness areas. The combination of increased protection of forest habitat and the potential increase in size of conservation reserves would have a positive effect on the conservation of large mammals in the western United States.

The purpose of the *Endangered Species Act* is to " ... provide a means whereby the ecosystems upon which endangered and threatened species depend may be conserved ... " (U.S. Fish and Wildlife Service 1973). The act further directs that " ... all Federal departments shall seek to conserve endangered species and threatened species." In this regard, many IRAs function as biological refugia for terrestrial and aquatic species, including numerous threatened, endangered, and imperiled species. The maintenance of natural values in IRAs could contribute to their long-term viability (Brown and Archuleta 2000). IRAs contain more than 220 TE species, i.e., approximately 25% of listed or proposed animal species and 13% of listed plant species (USDA Forest Service 2000).

Fig. 5. Overlap of USDA Forest Service inventoried roadless areas and grizzly bear recovery zones.



Among TE species, 88% are imperiled by habitat destruction and degradation (Wilcove et al. 1998). Dobson et al. (1997) found that, if the habitats of TE species were more extensively protected, a large number of them would be efficiently conserved. Our analysis showed that the vast majority of IRAs hosted TE or G1–G2 imperiled species and that, by adding the IRAs to the existing conservation reserve system, the conservation of species at risk and their habitat could be better realized. Although we recognize that not all threatened, endangered, or imperiled species require lands free of active land management to survive, limiting the human footprint by placing IRAs off limits to road construction and maintenance, resource extraction, and other development activities could provide a counterpoint to the multiple-use activities taking place elsewhere within the National Forest System.

Furthermore, although there may be duplicate species populations within IRAs or existing conservation reserves, the high level of endangerment of these species should predicate that we conserve as many populations as possible. Therefore, the potential issues of complementarity or duplication of species across IRAs should not diminish the contribution that IRAs could make to conserving species at risk. Our analyses have shown that, despite the small size and extent of IRAs in the eastern United States, they contain a greater number of endangered or imperiled species across more IRAs than do the west and Alaska. However, many of the western IRAs are missing data or have not been surveyed. This error of omission serves only to emphasize that our findings are a conservative estimate of potential species endangerment particularly in IRAs in Alaska and the western United States.

Top carnivore species, such as the grizzly bear, often have the largest species-level area requirements in an ecosystem and maintain ecological structures and resilience by top-down trophic interactions. They need large, contiguous habitat blocks to persist, and there must be landscape connectivity among core areas to ensure sufficient habitat for viable populations (Soulé and Noss 1998, Carroll et al. 2001). As a result of these requirements, large reserves are necessary to maintain populations of these wide-ranging species. Woodroffe and Ginsberg (1998) recently estimated that habitats of 20,000 km^2 are needed to provide a 90% chance for the long-term survival of the grizzly bear in the wild. Indeed, only those wilderness areas that were 20,000 km^2 or larger in 1920 still support grizzly bears today (Mattson and Merrill 2002). The 40,000 km^2 of IRAs in and near designated grizzly recovery zones in the northern Rockies will help improve the long-term habitat viability for grizzly bears in the region (Martin et al. 2000, USDA Forest Service 2000).

Carroll et al. (2001) proposed the need for a comprehensive conservation strategy for carnivores in the Rocky Mountains that considers the requirements of several species, including grizzly bear, wolverine, fisher, and lynx. The regions where these four species overlap show a strong concordance with grizzly bear recovery zones. IRAs may benefit all of these species by providing expanded and buffered habitat and, in turn, secure the ecological integrity of those ecosystems (Terborgh and Soulé 1999, Conner et al. 2000, Martin et al. 2000). If grizzly bear populations remain limited by the size and configuration of current conservation reserves, their long-term survival in the conterminous United States cannot be assured (Mattson and Merrill 2002).

Bruner et al. (2001) found a clear relationship between the existence of a viable and well-connected system of conservation reserves and biodiversity conservation. Because of the stable long-term ownership tenure associated with USDA Forest Service lands, as opposed to privately held forests, many of these forested areas contain a wealth of biological diversity. Historically, land within the Forest Service has been managed under a multiple-use strategy, with timber extraction being a main component of many of these plans. However, multiple-use management may not ensure the protection of the full range of biodiversity, because anthropogenic habitat degradation and destruction are the primary causes of biodiversity loss (Ehrlich 1988, Myers 1988, Wilcove et al. 1996, Haila 1999, Wood 2000).

Setting aside IRAs for stricter protection from extractive or economically driven activities may indeed meet many biological objectives, e.g., integration of fish and wildlife values and watershed and forest health, consistent with the agency's multipurpose agenda. In addition, IRAs may also contribute invaluable benchmarks to gauge ecological changes on managed U.S. Forest Service lands. A representative system of natural habitats, set aside from active management, would allow natural ecological processes, including a full suite of existing native species, to survive free of human activities. Without strict conservation areas that represent all forest habitat types, it will be difficult to make objective assessments on the sustainability of forest management (Noss and Cooperrider 1994, Norton 1999, Noss et al. 1999). Based upon our analyses, we conclude that IRAs support many at-risk species and thereby greatly contribute to the conservation of biodiversity throughout the United States. For some species with only a few remaining populations, the strict and permanent protection of IRAs may represent the final, critical refuge.

Responses to this article can be read online at:
http://www.consecol.org/vol7/iss2/art5/responses/index.html

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Fire history and fire–climate relationships along a fire regime gradient in the Santa Fe Municipal Watershed, NM, USA

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ABSTRACT

The Santa Fe municipal watershed provides up to 40% of the city's water and is at high risk of a stand-replacing fire that could threaten the water resource and cause severe ecological damage. Restoration and crown fire hazard reduction in the ponderosa pine (PP) forest is in progress, but the historic role of crown fire in the mixed-conifer/aspen (MC) and spruce-dominated forests is unknown but necessary to guide management here and in similar forests throughout the southwestern United States. The objective of our study was to use dendroecological techniques to reconstruct fire history and fire–climate relationships along an elevation, forest type, and fire regime gradient in the Santa Fe River watershed and provide historical ecological data to guide management. We combined systematic (gridded) sampling of forest age structure with targeted sampling of fire scars, tree-ring growth changes/injuries, and death dates to reconstruct fire occurrence and severity in the 7016 ha study area (elevation 2330–3650 m). Fire scars from 141 trees (at 41 plots) and age structure of 438 trees (from 26 transects) were used to reconstruct 110 unique fire years (1296–2008). The majority (79.0%) of fires burned during the late spring/early summer. Widespread fires that scarred more than 25% of the recording trees were more frequent in PP (mean fire interval $(MFI)_{25\%}$ = 20.8 years) compared to the MC forest (31.6 years). Only 24% of the fires in PP were recorded in the MC forest, but these accounted for a large percent of all MC fires (69%). Fire occurrence was associated with anomalously wet (and usually El Niño) years preceding anomalously dry (and usually La Niña) years both in PP and in the MC forest. Fire in the MC occurred during more severe drought (mean summer Palmer Drought Severity Index; PDSI = −2.59), compared to the adjacent PP forest (PDSI = −1.03). The last fire in the spruce forest (1685) was largely stand replacing (1200 ha, 93% of sampled area), recorded as fire scars at 68% of plots throughout the MC and PP forests, and burned during a severe, regional drought (PDSI = −6.92). The drought–fire relationship reconstructed in all forest types suggests that if droughts become more frequent and severe, as predicted, the probability of large, severe fire occurrence will increase.

1. Introduction

Large areas of forest throughout the southwestern United States (Arizona, New Mexico and adjacent areas) are unnaturally dense due a century of fire exclusion, and are consequently at high risk of historically unprecedented large crown fires (Covington and Moore, 1994; Allen et al., 2002). Given limited resources for treatment, a triage approach must be adopted to identify areas with high resource value or that are located strategically within the larger landscape. Historical ecological data describing the range of variability of disturbance regimes and their climatic controls are vital to guide forest restoration (Swetnam et al., 1999), particularly when facing the additional challenge of a changing climate (Millar et al., 2007).

The upper Santa Fe River watershed, New Mexico is arguably the most at risk, high-profile municipal watershed in the southwestern U.S. Santa Fe is the oldest state capital, founded on the Santa Fe River in the early 17th century (Debuys, 1985). Sitting at 2137 m elevation on the alluvial plane of a steep, forested, montane watershed, Santa Fe is inextricably linked to the ecosystem services (e.g., drinking water) and natural hazards (e.g., fire and floods) associated with the wildland urban interface. Surface water that originates high in the spruce-fir forests of the Pecos Wilderness Area is regulated through a system of reservoirs that provides up to 40% of the city's water supply (Grant, 2002). The population in Santa Fe County has tripled in recent decades (1970–2007; USCB, 2009), overtaxing the already limited water

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supply. Like much of the West, there has not been a widespread fire in the ponderosa pine (PP) and mixed-conifer (MC) forests of the watershed for 130 years, increasing the area at high risk of crown fire beyond the spruce-fir forests, where they naturally occur.

Recent, large, crown fires in near by watersheds have produced runoff and erosion events two orders of magnitude greater than pre-fire events (Veenhuis, 2002). This type of event in the Santa Fe watershed could destroy the water supply infrastructure and flood the historic heart of the city. The threat of catastrophic fire sparked years of contentious public debates, which ultimately led to U.S. Congressional earmarks of seven million dollars to fund planning and implementation of crown fire hazard reduction and forest restoration in the lower elevation PP forests (USDA, 2001). However, the ecological role of fire and the consequences of fire exclusion in the upper elevation mesic MC and spruce-fir forest types remain largely unknown, and it is these forests that cover the majority of the area that supplies the main reservoir.

1.1. Gradients: elevation, forest types and fire regimes

Gradients (e.g., elevation and vegetation) are common in terrestrial ecosystems and are a valuable way to study how ecological processes vary across a range of conditions (Whittaker and Niering, 1965; Whittaker, 1967). In the southwestern U.S. fire is a keystone ecological process that has affected vegetation across ecosystem gradients for hundreds to thousands of years (Swetnam and Baisan, 1996; Allen, 2002; Anderson et al., 2008). The size, frequency and severity of fire over time define the fire regime (Agee, 1993). Fire regimes are commonly classified by the extremes of the fire severity gradient (low severity or high severity). Recently, the term *mixed-severity fire regime* has been described as including a range of fire severities across a spatially complex mix of forest patches, including unburned, low, moderate, and high severity fire (Agee, 2005).

At landscape scales (1000–100,000 ha; watersheds to mountain ranges), fire can move across gradients of elevation, forest types and likely, between fire regimes. The PP forest type in the southwestern U.S. is a classic low severity, high frequency fire regime (Swetnam and Baisan, 1996). Subalpine spruce-fir forests in the southern Rocky Mountains exemplify the other extreme: a high severity, low frequency fire regime (Romme and Knight, 1981; Sibold et al., 2006). The steep topography of the southwestern U.S. juxtaposes these two forest types (representing the extremes of the fire severity gradient) in close proximity (<10 km separation) along a continuous elevation gradient with continuous fuels. MC forests are intermediately located between PP and spruce-fir (Dick-Peddie, 1993). Lower elevation, xeric, MC forests historically burned with low severity, but less frequently than PP (Dieterich, 1983; Brown et al., 2001). Some upper elevation, mesic MC forests have evidence of high severity fire (Fule et al., 2003; Margolis et al., 2007; Margolis, 2007). Historically, drought synchronized fire occurrence within and between low and high severity fire regimes regionally (Swetnam and Baisan, 1996; Margolis et al., 2007), but there is limited research examining connectivity between low and high severity fire regimes along a continuous forest gradient in a single, continuous landscape (Fule et al., 2003).

The implications of low and high severity fire regime connectivity are important given the well-documented changes associated with fire exclusion in ecosystems of the southwestern U.S. Over a century of fire exclusion in PP forests of the region has dramatically increased forest density and the risk of large crown fires (Covington et al., 1997; Allen et al., 2002). While there is historical evidence of high severity fire patches in some MC forests (Fule et al., 2003), increased forest density in other MC forests due to fire exclusion has increased the size of forest patches at risk of crown fire (Fule et al., 2003; Cocke et al., 2005; Heinlein et al., 2005; Margolis et al., 2007).

There is comparatively less information about the effects of fire exclusion on forest density, composition, and crown fire risk in the upper elevation spruce-fir forests of the region (Fule et al., 2003; Cocke et al., 2005). It is generally thought that a century of fire exclusion has not had dramatic impacts in these naturally dense forest types (Sibold et al., 2006), because high elevation, high severity forest fire regimes burn at long (centennial-scale) return intervals (Turner and Romme, 1994). Evidence of decreased fire frequency during the fire suppression period, compared to previous centuries has been observed in some subalpine forests of the Southern Rockies (Kipfmueller and Baker, 2000), but not others (Sibold et al., 2006). If forest ecosystems along steep elevation gradients are connected by fire spread across vegetation and fire regime gradients, then a century of fire exclusion in the lower elevation pine-dominated and MC forests is likely to have affected the high elevation, high severity forest fire regimes as well.

The semi-arid climate of the southwestern U.S. is highly variable, with frequent (2–7 years) wet/dry oscillations that are partially driven by multiple ocean-atmosphere oscillations, particularly the El Niño Southern Oscillation (ENSO; Diaz and Markgraf, 2000). Fire–climate analyses indicate that moisture variability largely explains patterns of fire occurrence in tree-ring reconstructed and contemporary records in low- and mid-elevation forests of the southwestern U.S. (Swetnam and Betancourt, 1990; Crimmins and Comrie, 2004). Warmer temperatures in recent decades have increased the length of the fire season, resulting in more large fires throughout the western U.S. (Westerling et al., 2006). The established link between climate variability and fire, coupled with predicted warmer global temperatures (IPCC, 2007) and drier conditions in the southwestern U.S. (Seager et al., 2007) has led to predictions of more large fires in the future (Westerling et al., 2006). Better understanding of the link between climate variability and fire occurrence along the elevation gradient of forest types and fire regimes is necessary to proactively manage our forests with a science-based approach, in the face of climate change.

The goal of the research is to provide essential historical ecological data across a gradient of forest types and fire regimes to guide management in the upper Santa Fe watershed and similar upper montane forest types in the region. Our first objective was to reconstruct fire history (frequency, severity, and size) along an elevation, vegetation and fire regime gradient in the upper Santa Fe Watershed. Our second objective was to reconstruct and compare historic fire–climate relationships between forest types. Our third objective was to test for evidence of direct connectivity of fire regimes along the fire severity gradient from low, to mixed, to high severity.

1.2. Study area

The study area encompasses the upper Santa Fe River watershed (7016 ha), which includes the headwaters located within the U.S. Forest Service Pecos Wilderness Area (Fig. 1). The watershed is located on the west slope of the Sangre de Cristo Mountains, northeast of the city of Santa Fe, NM, near the southern terminus of the Southern Rocky Mountains. The upper watershed has been closed to the public since 1932 to protect the water supply for the city of Santa Fe (USDA, 2001). Elevation ranged from 2237 m at the lowest point in the stream channel to 3847 m on the peaks that rise above tree-line and define the headwaters of the basin. Tree-ring samples were collected from 2328 m to 3650 m.

The climate is semi-arid and continental. Precipitation peaks during summer monsoon convective storms (July–August), and winter snowpack is common except during extreme drought years

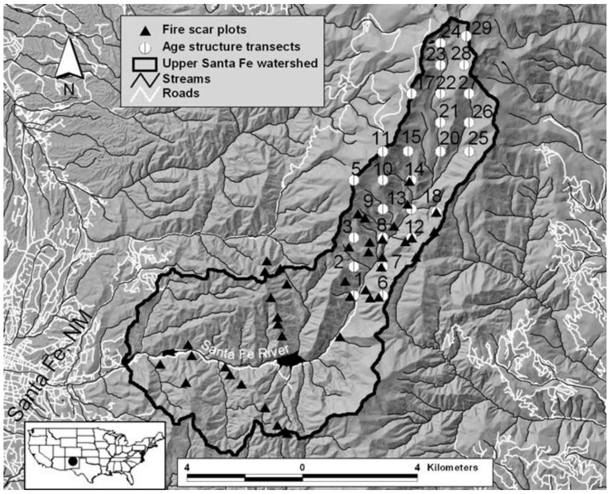

Fig. 1. Location of gridded age structure transects (numbered) and fire scar plots used to reconstruct fire history in the upper Santa Fe watershed, NM. The upper watershed, containing only age structure transects, is spruce-dominated forest. The lower watershed, containing only fire scar plots, is PP. The middle-elevation forest area where both age structure and fire scars overlap is MC.

(e.g., 2002). Temperature (1972–2005) at Santa Fe, NM (2060 m) ranged from an annual average minimum of 2.3 °C to an annual average maximum of 18.2 °C. Total annual average precipitation was 34.8 cm and total annual average snowfall was 44.2 cm (Western Regional Climate Center, www.wrcc.dri.edu). Fire occurrence records were available for 222 fires (1970–2003) from the ranger district containing the watershed and the adjacent district to the east (Española and Pecos/Las Vegas Ranger Districts). The majority (93%) of fires occurred between May and August, peaking in July, but monthly area burned peaks in May and June during the dry foresummer. Eighty percent of all fires were started by lightning (USFS, unpublished data).

Along the elevation gradient, forest types transitioned from PP dominated forests in the lower part of the study area, to MC in the middle elevations, to spruce-fir in the upper elevations. The spruce-fir type was composed of Engelmann spruce (*Picea engelmannii* Parry) and corkbark fir (*Abies lasiocarpa* [Hook] Nutt. var. *arizonica* [Merriam] Lemmon), but Engelmann spruce was dominant in all locations sampled and often present in pure stands. This general upper elevation forest type is hereafter referred to as "spruce-dominated." The MC forest was relatively diverse and species composition varied largely by aspect. The following species were present in various combinations in this forest type, listed in order of abundance of the dominant tree: Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco.), ponderosa pine (*Pinus ponderosa* Lawson), southwestern white pine (*Pinus strobiformis* Engelm), quaking aspen (*Populus tremuloides* Michx.), white fir (*Abies concolor* [Gord. and Glend.] Lindl. Ex Hildebr.), and Engelmann spruce. There were no conspicuous, large (>50 ha) stands dominated by quaking aspen that might indicate recent stand-replacing fire patches. The lower section of the study area was dominated by ponderosa pine, with associated species including Colorado pinyon pine (*Pinus edulis* Engelm.), Rocky Mountain juniper (*Juniperus scopulorum* Sarg.), Gambel oak (*Quercus gambelii* Nutt.), Douglas-fir, white fir, and southwestern white pine.

2. Methods

2.1. Tree-ring fire history methods

A combination of tree-ring methods was necessary to reconstruct fire history along the elevation, vegetation and fire regime gradient. Fire scar-based methods (Dieterich and Swetnam, 1984; Swetnam and Baisan, 1996) were used to reconstruct surface fire frequency, seasonality, and extent for the PP and dry MC portions of the watershed where fire scars were present. However, in the upper elevation spruce-dominated and mesic MC forests, fire-scarred trees were rare or non-existent because: (1) high severity, high intensity, stand-replacing crown fires destroy (kill and burn) direct tree-ring evidence of past fires, (2) the thin bark of spruce and fir species is more susceptible to being fatally girdled, even by low-intensity fire, thus leaving no evidence of the most recent fire (e.g., fire scars), and/or (3) long fire return intervals may allow the rare fire-scarred trees to heal over so that open fire scar wounds are not visible.

In forest types where fire scars are not abundant, age structure-based fire history methods are commonly applied (Heinselman, 1973; Agee, 1993; Johnson and Gutsell, 1994). These methods largely rely upon the establishment dates of tree cohorts that regenerate following stand-replacing fire events to date the fire and determine aerial extent of the stand-replacing patches. Thus, in the spruce-dominated forest type we used age structure sampling methods to reconstruct the age of the dominant, presumably oldest trees, thereby estimating the time since the last fire (Kipfmueller and Baker, 1998). Satellite imagery, aerial photography, and field observations were used to identify any potential post-fire forest patches.

Age structure data alone may not be sufficient to determine if the forest patch was a post-stand-replacing fire cohort and ultimately date the fire. Unlike lodgepole pine (*Pinus contorta*) or quaking aspen, spruce and fir trees may take years to decades to regenerate following stand-replacing fire (Antos and Parish, 2002). This is likely due to a combination of variability in seed sources, dispersal, and post-fire weather and climate. The precision of fire dates derived strictly from forest age will not be annual because of lagged regeneration. Decadal precision of fire dates can be sufficient when calculating area-based estimates of fire frequency (e.g., natural fire rotation; Heinselman, 1973), due to the long return intervals (100 years to >400 years) of forest crown fire regimes (Turner and Romme, 1994). Annually precise stand-replacing fire dates may be reconstructed if fire scars, fire-killed trees or injured trees are present in adjacent forest stands, or on the perimeter of unburned patches (Johnson and Gutsell, 1994; Margolis et al., 2007). Annually resolved fire dates are necessary for inter-annual fire–climate analyses, which can provide specific climate information associated with the relatively rare, but important, stand-replacing fire events.

In forest types such as MC, where a combination of low severity and high severity fire occurs (i.e., mixed-severity fire), it is necessary to use a combination of fire scar and age structure-based methods to reconstruct the fire history (Agee, 2005). For example, some stands within the MC zone had no fire scars or potentially fire-killed trees, and stand age was the best evidence of past fire. Alternatively, other stands had abundant fire scars (i.e., evidence of repeated, low severity surface fire) and no clear post stand-replacing fire cohorts.

2.2. Sampling design

In the PP forests of the lower portion of the study area we used a targeted approach to locate and collect fire scars. Targeted fire scar sampling in Southwestern ponderosa pine provides similar estimates of fire frequency compared to systematic sampling, particularly for widespread fires, with the added benefit of providing a longer record (Van Horne and Fule, 2006). Samples were primarily collected at 50 m-radius plots along two transects; a north-facing transect, and a south-facing transect. These transects were located in the middle of the PP zone and followed a series of ridges that extended from the river up to the respective watershed boundaries. From the ridgetop location we searched both slopes that descended from the ridge and adjacent slopes that could be seen from the ridge. Additional plots were located within the PP zone to provide broader spatial coverage of the PP forest fire history. One group of plots was located west (downstream) of the transects in the area surrounding the lower reservoir. An additional plot was located east (upstream) of the transects, above the second reservoir. The resulting spatial patterning of the plots was determined by a combination of our search effort, topography and the presence of fire-scarred material.

In the high elevation spruce-dominated forests, a systematic, gridded age structure sampling approach provided the best evidence of fire history (i.e., "time since last fire"; Johnson and Gutsell, 1994). We generated a 1 km grid beginning with a random location in the study area (Fig. 1). The grid was oriented along cardinal directions to facilitate navigation in the field. Two grid points (24 and 28) initially fell within unforested vegetation types and were relocated 50 m inside the nearest forested area.

In the middle-elevation, MC forest evidence of fire was present as both fire scars and post-fire tree regeneration cohorts. We extended the 1 km-spaced age structure grid into the MC zone, and because fire scars were only present at 5 of 12 MC age structure transects we used a targeted approach to locate and sample fire scars in this forest type. In the MC forest, fire-scarred trees were most abundant on the relatively flat ridges, apparently because of lower fire severity that allowed trees to survive fires that were otherwise stand-replacing on the adjacent steep slopes. We searched and sampled ridges with the goal of obtaining a relatively even spatial distribution of fire scar plots and to maximize the length of the fire history record. The final spatial distribution of the fire scar sample plots was ultimately determined by the location of fire-scarred trees, in part determined by topography and chance, and therefore is not evenly distributed in space.

In the topographically complex mountains of the semi-arid southwestern U.S., elevation and aspect can be important variables mediating vegetation type (e.g., Whittaker and Niering (1965)) and fire regimes (e.g., Iniguez et al. (2008)). To ensure that the distribution of aspect class (N, S, E, W) at our gridded, age structure transects was proportional to the relative abundance of aspect classes in the study area we stratified the sampling grid by aspect class. The percent of sample points in the four primary aspect classes was distributed similar to the percent of land area in each aspect (Table 1), with a slight over-sampling of east-facing slopes and under-sampling of the south-facing slopes.

2.3. Field sampling

Where multiple fire-scarred trees were present we used a plot-based field sampling approach. A plot was sampled where two or more fire-scarred trees were located less than 15 m apart. The plot center was located between the samples. Samples from multiple fire-scarred trees were collected within a 50 m search radius that defined the plot. Collecting multiple trees within a plot increased the probability of recording all fires that actually occurred in that area. This is necessary because trees are imperfect recorders of fire and individual trees may not record all fires (as fire scars) that burned around the tree (Dieterich and Swetnam, 1984). Wedges and cross-sections were collected from fire-scarred logs, stumps and rarely from live trees with a cross-cut saw in the MC forest (within the Pecos Wilderness Area) and with a chainsaw in the PP forests using standard procedures (e.g., Arno and Sneck (1977)).

To determine stand age at the gridded age structure transects in the spruce-dominated and MC zones we sampled the 20 largest (diameter at breast height (dbh)) trees along a 100 m by 20 m belt transect. The transect was centered on the grid point and the long axis was oriented parallel to the contour of the slope (i.e., sideslope). To determine tree age, increment cores were collected as close to the base of the tree as possible (<0.3 m). We angled the borer down to intersect the estimated location of the root crown in an attempt to sample all the years of tree growth. We re-sampled trees until we extracted a core containing rings estimated to be within 10 years of the pith.

2.4. Lab methods

All tree-ring samples were sanded with progressively finer sandpaper until the ring structure was visible and then crossdated using standard dendrochronological procedures (Stokes and Smiley, 1968). For fire scar samples, we determined the calendar year of the scar and the season of fire occurrence by analyzing the relative position of each scar within the annual growth ring: dormant season, early earlywood, middle earlywood, late earlywood, latewood, or unknown (Baisan and Swetnam, 1990). Predominant occurrence of spring or early summer fires in northern New Mexico and the southwestern U.S. is widely supported by fire seasonality data from observed 20th century fires in the region (Barrows, 1978), locally, and from hundreds of tree-ring reconstructed fires (Swetnam and Baisan, 1996). Based on our observations and conventional season of montane fire occurrence in the region, all fire years with fire scars recorded only in the dormant season were assigned to the spring/summer of the next year (ring).

For age structure samples, we estimated the date of the first year of growth (pith) for increment cores that did not contain the pith ring, using a concentric circle pith estimator (Applequist, 1958). Cores that were estimated to be greater than 30 years from the pith ring or that had no curvature in the inner rings were not included in the age structure data. Because cores were collected at a downward angle to intercept the root crown, the error associated with the age to core height was assumed to be within the resolution of the age class bins (10 years) and was not estimated.

A qualitative description of the initial tree-ring growth of cored trees (open, average, or suppressed) was recorded to provide information regarding the growth environment when trees established (Romme and Knight, 1981). Spruce and fir species are shade tolerant and are able to survive in low light conditions under canopies, but the growth rates in these conditions can be very slow (i.e., "suppressed"). Growing conditions for trees germinating in an open forest, such as following a stand-replacing fire, would be more favorable and should be indicated as relatively wide initial ring widths (i.e., open). This information was combined

Table 1
Aspect class of land area and age structure transect grid in the MC and spruce-dominated forests.

Aspect class	Area (ha)	Area (% of total)	Age structure transect (#)	Age structure transect (% of total)
Flat (0% slope)	0.22	0.01	0	0
N (315–45°)	221.11	8.29	3	11.54
E (45–135°)	773.59	28.99	9	34.62
S (135–225°)	852.73	31.95	6	23.08
W (225–315°)	821.03	30.77	8	30.77

with tree ages and fire scar dates to determine if trees were likely part of post stand-replacing fire cohorts.

2.5. Data analysis

The fire scar data were entered into a database and analyzed using FHX2 software (Grissino-Mayer, 2001). Because fire scar return intervals are rarely normally distributed and more commonly fit a Weibull distribution (Grissino-Mayer, 1999), we tested for the fit of the Weibull model (Kolmogorov–Smirnov (K-S) test) and estimated the Weibull Median Probability Interval (WMPI). Central tendency parameters (mean, median and WMPI) of fire frequency were calculated for five "filtered" subsets of the composite fire history data for (1) the PP forest and (2) the MC forest. The following filtered subsets of reconstructed fires were used for the analysis: (1) all fires, (2) fires recorded by a minimum of 2 trees, (3) a minimum of 2 trees and >10% of recording trees, 10% scarred, (4) a minimum of 2 trees and >20% of recording trees, 20% scarred and (5) fires recorded by a minimum of 2 trees and >25% of recording trees, 25% scarred. "Recording trees" refers to previously fire-scarred samples that have intact wood (i.e., not burned away or missing pieces) and an open wound (not covered by bark) during the time period in question. Many montane conifers have thick bark that protects trees from damage to the cambium by fire. These full-bark trees may not record fires as fire scars, while the same fire is recorded on adjacent trees with pre-existing open "cat face" fire scar wounds.

Filtering the fire scar data by the percent of recording trees scarred is used to infer relatively large, spreading fires, as compared to less widespread fires that only scar a relatively small number (percent) of trees (see discussion in Swetnam and Baisan, 2003). Widespread fires are thought to be more ecologically important because of the extent of the effects. Too few fire-scarred trees were present on the landscape and/or collected to confidently allow plot-based fire interval analysis (e.g., Iniguez et al., 2008). In addition, high severity fire in parts of the MC forest killed and burned evidence of prior fires at individual plots, so fire dates from all plots were combined to make a site composite for each forest type (Dieterich, 1980). We also chose not to analyze fire intervals for individual trees (point intervals), because our attempt to extend the record back in time by targeting remnant wood resulted in many samples having an incomplete record due to being burned and/or eroded. This was particularly the case in the MC zone of the wilderness area, where a majority of samples were remnant wood. Given these limitations of a relatively long record, we still are confident that the percent of trees recording fire is a good indicator of widespread vs. localized fires and that the widespread fires that we focus on are the most robust to variability in sampling (Van Horne and Fule, 2006). Because fire intervals vary over time with changes in fuels and climate (e.g., Swetnam (1993)), central tendency statistics (e.g., MFI) oversimplify historic fire regimes. We report additional statistics (e.g., minimum and maximum fire intervals) and interpret these data in terms of fire management to provide a better understanding of the historic range of variability of the fire regime.

To test for differences in historical fire frequency between the PP and MC forests we used the Student's *t*-test to compare MFI, the Folded-*f* test to analyze differences in variance, and the K-S test to analyze differences in distributions (FHX2, Grissino-Mayer, 2001). Because these tests assume that the data are normally distributed, the data are transformed to the standard normal distribution (i.e., mean of zero and a standard deviation of one) before the comparisons (Grissino-Mayer, 2001). To quantify synchrony of burning (i.e., fire spread) between the PP and the MC forests we counted the number of coincident fire years between the two forest types, and calculated the percent of all fire years in each forest type that were synchronous between forest types. As a more robust test of synchrony we used Chi-squared analysis to test for independence between MC fire years and PP fire years (1550–1880) for all filtered subsets of fire years.

We used superposed epoch analysis (SEA; Baisan and Swetnam, 1990) to test for inter-annual relationships between variability in four tree-ring reconstructed measures of climate and fire occurrence in (a) the PP forest and (b) the MC forest. The tree-ring reconstructed climate variables included (1) Palmer Drought Severity Index (PDSI), (2) annual precipitation from El Malpais, NM, (3) an index of El Niño/Southern Oscillation (ENSO), and (4) an index of the Pacific Decadal Oscillation (PDO).

PDSI is a commonly used measure of available moisture (Palmer, 1965). Summer (June–August) PDSI is a good indicator of moisture conditions prior to and during the southwestern U.S. fire season and is highly correlated with variability in historical fire occurrence (Swetnam and Baisan, 2003) and 20th century fire occurrence (Crimmins and Comrie, 2004). A 2.5° gridded tree-ring reconstruction of summer PDSI exists for much of North America and in the southwestern U.S. it extends hundreds of years prior to the 20th century instrumental climate data (Cook et al., 2004). PDSI gridpoint 133 is nearest to our study site and is used in the SEA analysis. A tree-ring based precipitation reconstruction from El Malpais National Monument, in west-central NM (Grissino-Mayer, 1996), was also used as a sub-regional climate variable.

Indices of Pacific Ocean-atmosphere oscillations (e.g., ENSO and PDO) that have been shown to affect climate variability in the southwestern U.S. (Diaz and Markgraf, 2000; Brown and Comrie, 2002) were also used as variables in the SEA analysis. As a proxy index for ENSO we used the tree-ring reconstructed Niño3 index (Cook, 2000) of winter (December–February) sea surface temperature (SST) from the eastern equatorial Pacific Ocean (5°N–5°S, 90°–150°W). Positive (negative) Niño3 index values represent warm (cool) SST's - El Niño (La Niña).

We used the (D'Arrigo et al., 2001) annual PDO index reconstruction derived from temperature sensitive tree-ring sites from coastal Alaska (5) and Oregon (1), and two tree-ring reconstructed PDSI grid points in northern Mexico. Positive (negative) index values of PDO correspond with warm (cold) phases of the primary mode of variability in Pacific Ocean SST's polewards of 20°N (Mantua et al., 1997).

To test whether drier conditions were associated with fire in the MC forest than in PP, we compared mean PDSI during widespread (25% scarred) and "all fire" years with a *t*-test. To test whether widespread fires occurred on drier years than "all fire" years we compared mean PDSI between fire years for each vegetation type with a *t*-test.

3. Results

3.1. Fire scars—PP

In the PP zone (1600 ha search area) we crossdated a total of 442 fire scars from 76 trees at 20 locations, for a total of 99 unique fire years (Tables 2 and 3). The PP fire scar record covers 709 years (1296–2004), with fire scars recorded from 1331 to 1966 (Fig. 2, Table 2). The period 1550–1880 was chosen for fire interval analysis as the best compromise between record length and sample depth.

The season of fire occurrence was determined for 331 (75%) of the fire scars (Table 4). The remaining fire scars were in poor condition or were in rings too narrow to accurately determine the season. When fire scar season could be determined, the most frequent occurrence (69%) was in the dormant (D) season (i.e., between ring boundaries). All but 3% of the remaining fire scars were recorded in the earlywood (E) portion of the ring and the

Table 2
Fire scar record statistics.

Forest type	Search area (ha)	Plots (#)	Fire-scarred trees (#)	Fire scars (#)	Unique fire years (#)	Full record (years)	Fire scar record (years)	Fire interval analysis (years)
PP	1600	20	76	442	99	1296–2004	1331–1985	1550–1880
MC	1200	21	65	139	35	1337–2008	1339–1879	1495–1880
Spruce-dominated	1200	26	0	–	–	–	–	–

Table 3
Upper Santa Fe watershed fire scar dates (all fires). Fire years recorded in both forest types indicated in bold.

Century	<1500	1500s	1600s	1700s	1800s	1900s
PP (1296–2004)	1331, 1398, 1415, 1434, **1445**, 1479, **1495**	1503, **1516**, **1522**, 1532, **1542**, 1551, 1558, **1562**, 1573, 1580, **1587**, 1591	1600, 1604, 1605, 1606, **1608**, 1612, 1616, 1617, **1619**, **1622**, 1623, **1624**, 1628, 1631, 1633, 1636, 1638, 1642, 1644, 1646, 1648, **1654**, 1656, 1659, 1661, 1664, 1672, 1676, 1683, **1685**, 1687, 1696,	**1700**, 1705, **1715**, 1719, 1724, 1725, **1729**, **1737**, 1739, 1742, **1748**, 1763, **1773**, 1778, 1779, 1784, 1786, 1788, 1794, **1795**	1803, 1805, 1808, 1809, 1810, 1814, **1819**, 1823, 1825, 1826, 1831, 1835, **1842**, 1858, **1860**, 1864, 1867, 1877, **1879**, 1883, 1885, 1886, 1893	1902, 1904, 1911, 1931, 1946, 1966
MC (1337–2008)	1399, 1444, **1445**, **1495**	1500, **1516**, **1522**, **1542**, 1546, **1562**, 1579, **1587**, 1599	**1608**, 1614, **1619**, **1622**, **1624**, **1654**, **1685**	**1700**, **1715**, 1716, **1729**, 1730, **1737**, **1748**, **1773**, **1795**	**1819**, 1820, **1842**, 1857, **1860**, **1879**	

Table 4
Fire scar seasonality reconstructed from the relative position of the fire scar in the tree-ring. Period of record: PP, 1296–2006 and MC, 1337–2008.

Scar position	Number of fire scars (PP/MC)	Percent of scars with season determined (PP/MC)
Dormant	229/39	69.2/42.4
Early earlywood	40/27	12.1/29.3
Middle earlywood	35/15	10.6/16.3
Late earlywood	18/11	5.4/12.0
Latewood	9/0	2.7/0.0

majority of those were in the first third of the earlywood (early earlywood, EE). The remaining 3% of the fires were recorded in the latewood (A) portion of the tree-ring. Overall, 81% of the fires in the PP zone were burning in the beginning of the growing season (May or June; D or EE).

The fire frequency of the reconstructed PP fire regime was highly variable through time (Fig. 2), and cannot adequately be described by one metric (e.g., MFI). The fire interval data (1550–1880) were not normally distributed (K-S d-statistic = 0.438, $p < 0.001$) and were fit with the Weibull model (K-S d-statistic = 0.132, $p = 0.144$). Increasingly exclusive filters increased the fire interval central tendency statistics by eliminating the (small) fires recorded by only a few trees, such that the WMPI increased from 3.8 years (all fires) to 18.8 years (25% scarred; Table 5). MFI was similar and ranged from 4.3 years (all fires) to 20.8 years (25% scarred). Thus, somewhere within the 1600 ha PP search area there was a fire recorded by at least one tree approximately every four years, on average, and relatively widespread fires scarring more than 25% of the trees occurred approximately every 18–21 years, on average. The minimum fire interval ranged from 1 year (all fires) to 7 years (25% scarred). The maximum fire interval ranged from 16 years (all fires) to a fire-free period of 63 years (1779–1842, 25% scarred). No widespread fires (25% scarred) occurred in the 20th century.

3.2. Fire scars—MC

In the mixed–conifer/aspen forests (1200 ha search area) we crossdated a total of 139 fire scars from 65 trees at 21 locations, for a total of 35 unique fire years (Tables 2 and 3). The MC fire scar record covers 672 years (1337–2008) with fire scars recorded between 1399 and 1879 (Fig. 2, Table 2). The period from 1495 to 1880 was chosen for fire interval analysis.

The season of fire occurrence was determined for 92 (66%) of the fire scars (Table 4). Based on the observed dominance of earlywood fires and the absence of latewood fires we used the same convention as in the ponderosa zone to assign fires only recorded in the dormant season to the spring/summer of the next year ($n = 7$). The majority (72%) of the fire scars dated to the season in the MC zone were burning in the spring or early summer (May or June; D or EE).

The MC fire interval data (1495–1880) were fit with the Weibull model (K-S d-statistic = 0.103, $p = 0.897$). The WMPI ranged from 10.3 years (all fires) to 27.8 years (25% scarred, Table 5). MFI was similar and ranged from 12.4 years (all fires) to 31.6 years (25% scarred). Minimum fire intervals ranged from 1 year for all fires, to 6 years for widespread (25% scarred) fires. Maximum fire intervals ranged from 31 years for all fires, to 94 years for widespread fires. No widespread fires (25% scarred) occurred in the 20th century. Further comparisons of fire intervals among the 5 filtered datasets and between vegetation types are discussed later in the paper.

Table 5
Fire interval analysis statistics for the PP (1550–1880) and the MC forests (1495–1880) for five filtered subsets of fire years (e.g., 20% = fires recorded by >20% of the recording trees).

Filter	Intervals (#) PP/MC	Mean fire interval (years) PP/MC	Median fire interval (years) PP/MC	Weibull median probability interval (years) PP/MC	Minimum interval (years) PP/MC	Maximum interval (years) PP/MC
All fires	76/31	$4.3^{+}/12.4^{+}$	4.0/12.0	3.8/10.3	1/1	16/31
>2 Trees	48/18	$6.8^{+}/21.3^{+}$	5.0/16.5	5.8/18.9	1/6	20/71
10%	34/18	$9.1^{+}/21.3^{+}$	7.0/16.5	8.0/18.9	1/6	25/71
20%	17/14	$17.1^{+}/27.4^{+}$	15.0/22.5	15.0/24.4	7/6	63/94
25%	14/11	20.8/31.6	15.5/25.0	18.8/27.8	7/6	63/94

$^{+}$ Indicates significantly different ($p < 0.05$) MFI between PP and MC (Student's t-test).

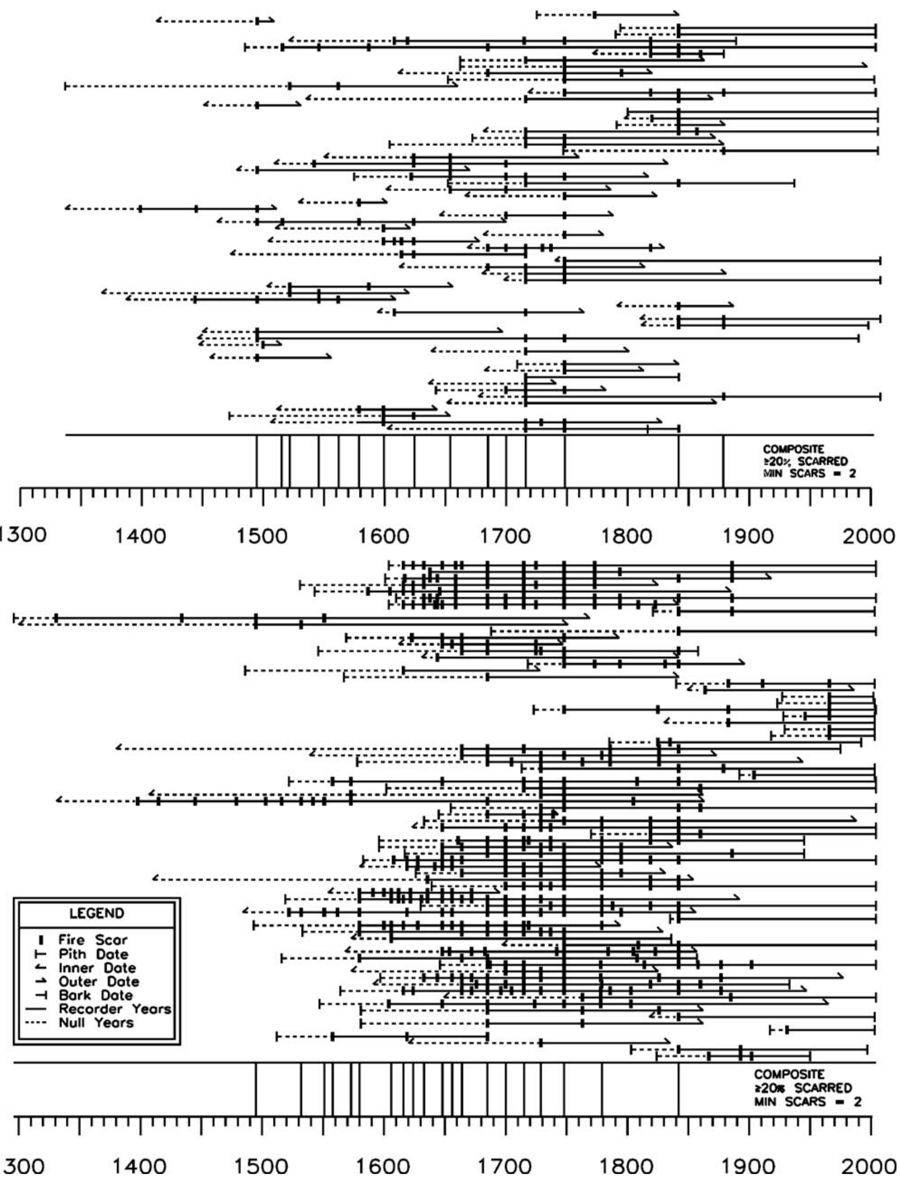

Fig. 2. Historical fire occurrence recorded by fire scars (1296–2004) in the PP forest (bottom) and MC forest (top) of the Santa Fe watershed. Each horizontal line is a tree and each vertical line is a dendrochronologically crossdated fire scar. The fire occurrence composite (bottom of each fire chart) indicates "widespread" fires recorded by a minimum of 2 trees and at least 20% of the trees recording fire.

3.3. Fire scars—spruce-dominated forest

No fire scars or any other direct evidence of fire (e.g., charred wood) were encountered at or between the age structure transects in the spruce-fir zone (1200 ha search area). Fire history in this vegetation type is presented in the age structure section.

3.4. PP vs. MC

The number of fire scars and individual fire years in the PP zone was approximately three times greater than that in the MC forest (Table 2). Historic fire intervals (1550–1880) in the PP zone were significantly shorter than in the MC forest for four of the five filtered subsets of fire years (all fires, ≥2 trees scarred, 10% scarred, and 20% scarred, Table 5). Although the MFI in the MC zone for the 25% scarred class (31.6 years) was approximately 10 years longer than in the ponderosa zone (20.8 years), the values were not statistically different (Student's *t*-test with equal variance, *t*-statistic = −1.780, $p = 0.092$).

Twenty-four fire years were synchronous between the two forest types (Table 3). Multiple synchronous fire years occurred every century from the 1400s to the 1800s. The number of synchronous fire years was greater than that would be expected by chance for all fire years ($\chi^2 = 39.22$, $p < 0.005$) and widespread (25% scarred) fire years ($\chi^2 = 29.15$, $p < 0.005$, with Yates correction for continuity). Sixty nine percent of all fires in the MC forest were also recorded in the PP zone. Only 24% of all fires in the PP forest were recorded in the MC zone.

3.5. Age structure

All of the age structure transects were located in the MC and spruce-dominated forest. Age structure transects were classified as spruce-dominated ($n = 14$) if the plurality of dominant trees was

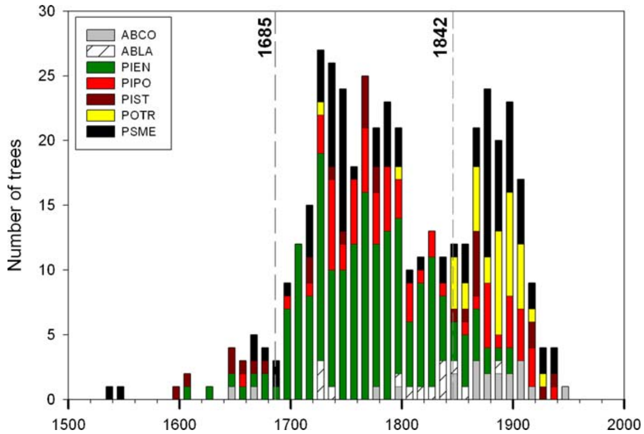

Fig. 3. Age structure and species composition of the dominant trees from the MC and the spruce-dominated forests. Data are from all age structure transects, in 10-year classes (plotted on the last year of the decade), and presented as estimated pith dates. Quaking aspen (POTR) was sub-dominant, but was sampled as a potential indicator of high severity fire. The last widespread fires with a stand-replacing component in the spruce-dominated (1685) and the MC forests (1842) are indicated as dashed lines.

Engelmann spruce. The remaining transects were classified as MC ($n = 12$). We collected 594 cores from 488 trees at 26 age structure transects (Fig. 1). We were not able to collect cores from all 20 dominant trees at 5 transects due to decomposed wood near the tree center and inclement weather. We were able to estimate pith dates for 438 (90%) of the sampled trees. Cores from the remaining 10% of the trees had no curvature in the inner rings or were estimated to be greater than 30 years from the pith so the number of rings to pith could not be estimated. The major cause for inadequate cores for pith estimation was decomposed wood near pith.

The collective age structure of dominant trees at all 26 transects in the MC and spruce-dominant forest has two recruitment peaks (i.e., a bi-modal distribution, Fig. 3). Less than 3% of the dominant trees established prior to 1650. A change in recruitment occurred in the late 1600s, increasing from a local minima of three trees (1681–1690) to the mode of 27 trees only 40 years later (1721–1730). This recruitment peak is dominated by Engelmann spruce. A second major tree recruitment pulse occurred in the mid-1800s. This younger recruitment peak is dominated by MC species. The recruitment peaks follow the last widespread fires in the MC (1842) and the spruce-dominated forests (1685) and there are relatively few trees dating to the decades prior to these two widespread fires.

The age structure at the individual transects illustrates both commonality and variability within and between the MC and the spruce-dominated forests (Figs. 4 and 5). The youngest MC stands all established after 1850 (1, 2, 6, 7) and were located nearest to the PP zone. The two oldest MC stands established circa 1600 (14, 18) and were located on rocky, relatively fire-protected sites near the upper MC/spruce ecotone. The youngest spruce-dominated stand began regenerating in the 1760s and the oldest trees date to the 1530s. The average age of the dominant trees in the spruce-dominated stands (mean [median] estimated pith date = 1769 [1763]) was approximately 60–100 years greater than in the MC forest (1829 [1861]).

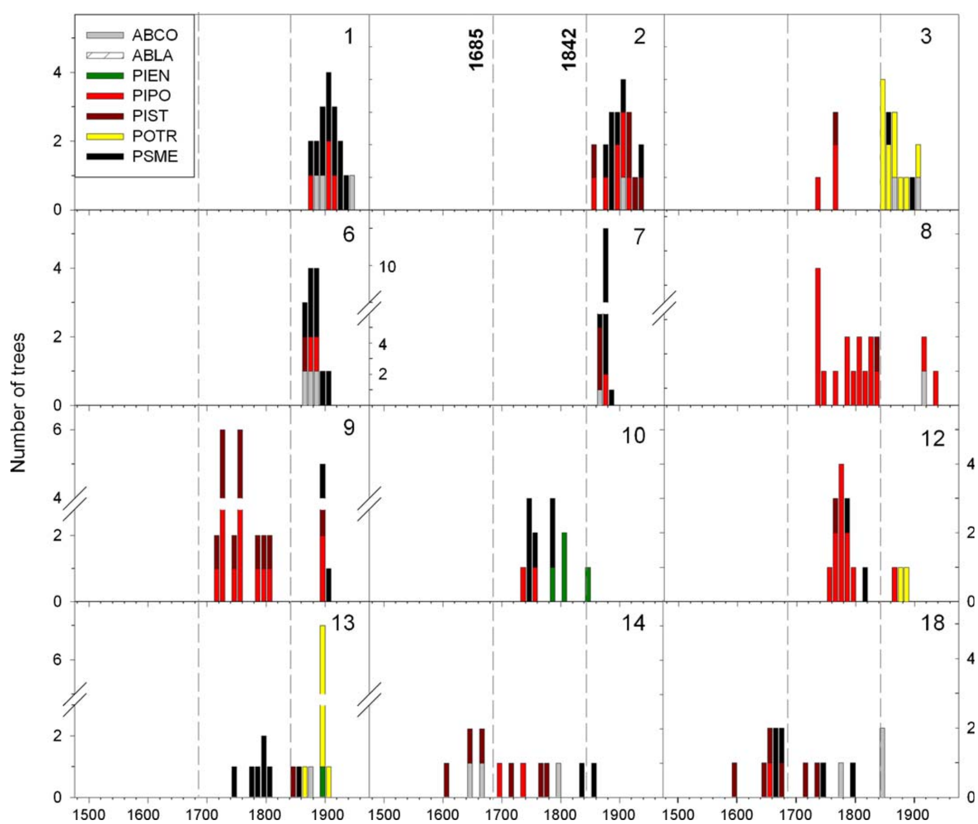

Fig. 4. Age structure and species composition of the dominant trees at individual age structure transects (e.g., 1) from the MC forest. Tree age data (estimated pith dates) are in 10-year classes. Note different scale for transects 7, 9, and 13.

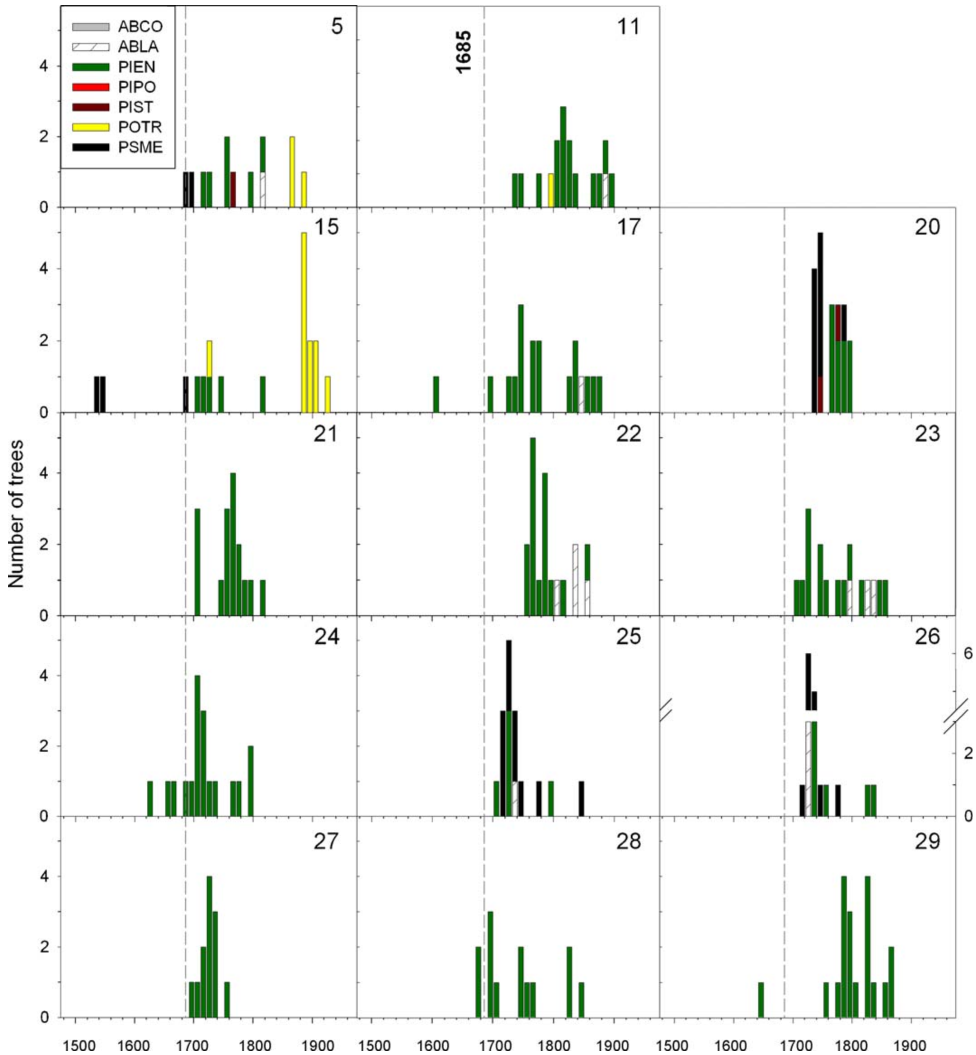

Fig. 5. Age structure and species composition of the dominant trees at individual age structure transects (e.g., 11) from the spruce (co-) dominated forest. Tree age data (estimated pith dates) are in 10-year classes. The lack of trees pre-dating the 1685 fire at 9 of the 14 transects suggests that this fire was largely stand-replacing in the upper, spruce-dominated portion of the watershed. Four of the five transects with trees surviving the fire (15, 17, 24, 28) had growth changes or injuries (i.e., traumatic resin ducts) in the tree-rings in 1685 (see Fig. 6).

3.6. Evidence of stand-replacing fire

The 1685 fire was recorded as fire scars by 57% ($n = 35$) of the recording fire-scarred trees at 68% ($n = 19$) of the recording fire scar plots throughout the MC and PP zones. Nine of the 14 spruce-dominated age structure transects and 10 of the 12 MC transects had no living trees that pre-date 1685. Four out of the five spruce-dominated transects that pre-date 1685 (15, 17, 24, and 28) had trees with growth changes or injuries/resin ducts in the tree-rings in 1685 (e.g., Fig. 6). The combination of age structure, growth changes/injuries, and widespread fire scar evidence indicates that the 1685 fire was relatively large and stand-replacing in the upper elevation forest.

The interpolated area of the 1685 fire within the upper Santa Fe watershed based on the spatial distribution of tree-ring evidence was 4730 ha. Approximately 25% of the reconstructed fire area was stand-replacing (1200 ha), all within the spruce-dominated zone (Fig. 7). It is likely that some of the younger forest stands below the spruce-dominated zone also burned with stand-replacing severity in 1685, but subsequent fires killed and burned any evidence of prior post-fire cohorts. We were conservative when reconstructing fire area and included these younger age structure transects as "not recording." The gaps between polygons in the reconstructed 1685 fire area are likely due to this lost record of fire.

Other fires that were widespread throughout the watershed (i.e., recorded by >50% of recording fire scar plots in the MC and PP forests, 1748, 1842; Fig. 2) were not recorded in the spruce-dominated forest. Age structure transects with many trees that

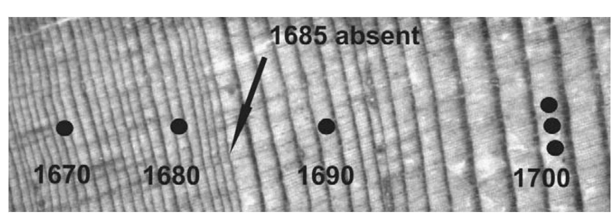

Fig. 6. Tree-ring growth release in a Douglass-fir core inferred to be a result of reduced competition due to tree mortality following the 1685 high severity fire.

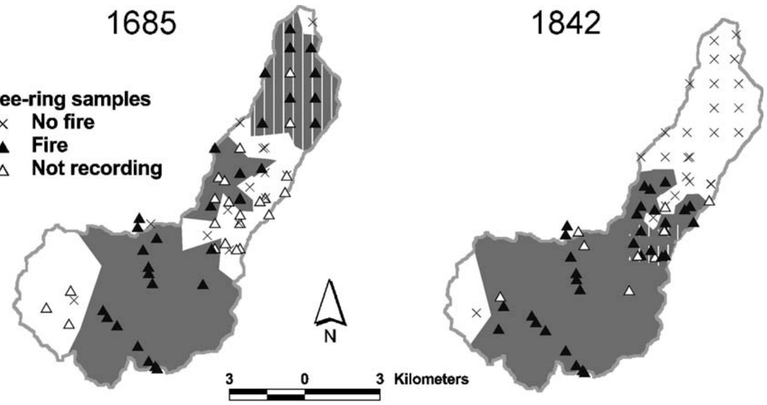

Fig. 7. Reconstructed fire area (gray) derived from thiesen polygon interpolation of tree-ring fire history data (fire scars, death dates, growth changes/injuries, and forest age structure). Areas with vertical white lines indicate stand-replacing fire patches.

pre-date these fires suggest that although these fires were widespread in the mid-elevation MC and lower pine forests, climate and/or fuel conditions were not suitable for fire spread into the mesic upper elevation spruce-dominated forest. It is possible however, that some widespread fires (e.g., 1716) may have burned with localized stand-replacing severity in the lower spruce-dominated forest and may explain the lack of trees in the early 1700s at some transects (e.g., 20).

3.7. Mixed-severity fire

There was evidence of mixed-severity fire in the MC zone. We use the term "mixed-severity" to indicate that some forest stands experienced high severity, stand-replacing fire (recorded as a tree recruitment pulse with no surviving trees) and other, adjacent stands experienced low-severity surface fire (recorded as fire scars). The landscape structure in the lower MC zone is such that north- and south-facing slopes are located on opposite sides of ridges. The youngest stands in the watershed (transects 1, 2, 6 and 7; Figs. 1 and 4) were on the more productive north- and east-facing slopes in this zone, near the ecotone with PP. These stands established in the mid-to-late 19th century and had the highest percentages of trees with "open" inner-ring growth (85–95%).

The 1842 fire was recorded as fire scars by 82% ($n = 24$) of the recording plots and 57% ($n = 42$) of the recording fire-scarred trees in the MC and PP forests. In addition to the four transects with no trees surviving the 1842 fire, three transects (3, 9, and 12) had growth changes or injuries/resin ducts in the tree-rings in 1842. Transect three had an aspen recruitment pulse beginning immediately following 1842 and the dominant PP trees that survived the fire had multi-year growth suppressions in the tree-rings beginning in 1843. The fire scar plot located less than 200 m southwest of age structure transect six had no samples post-dating 1842 and one of the fire-scarred trees had a bark-ring date of 1841.

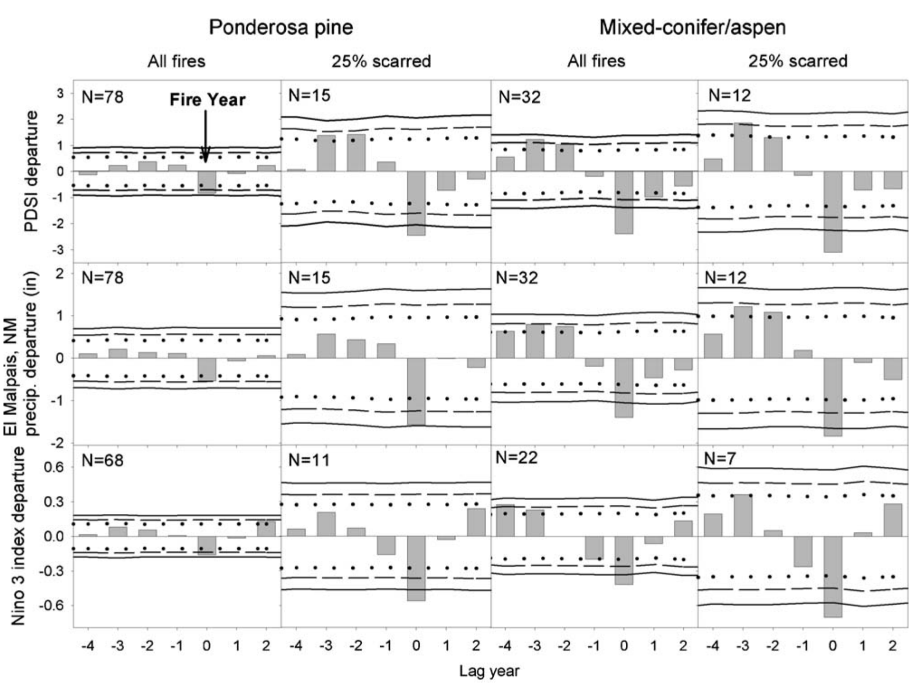

Fig. 8. Superposed epoch analysis for the PP and MC forests illustrating departure from the mean of reconstructed climate indices (PDSI, El Malpais, NM precipitation, and Niño3 index) for all fires and widespread (25% scarred) fires. Dotted, dashed and solid lines represent 95, 99, and 99.9% confidence intervals derived from 1000 Monte Carlo simulations; n, number of fire years.

The proximity of high severity (age structure) and low-severity (fire scar and tree-ring growth change) evidence within the lower MC zone indicates that the 1842 fire burned with mixed-severity within this forest type. Based on the multiple lines of fire evidence presented above, the reconstructed 1842 fire area within the upper Santa Fe watershed was 4642 ha (Fig. 7). The reconstructed stand-replacing fire area was 182 ha, consisting of multiple patches ranging from 34 ha to 110 ha.

3.8. Fire–climate

The results of the SEA indicate that all four filtered subsets of fire scar data (all fires, 10% scarred [not shown], 20% scarred [not shown], and 25% scarred) from both the PP and MC forests were significantly associated with negative (dry) departures during the fire year from mean summer PDSI and El Malpais, NM precipitation (Fig. 8). Fire occurrence in both forest types was also associated with positive (wet) departures from mean summer PDSI two to three years prior to the fire year. Fire occurrence was also associated with positive (wet) departures from mean annual precipitation at El Malpais, NM two to four years prior to the fire year in the MC forest. All sets of fire scar data in both forest types were associated with negative (cool ocean phase—La Niña) SST departures from the mean Niño3 index during the fire year. All fires and widespread (25% scarred) fires in the MC forest and 10% scarred fires in the PP forest (not shown) were associated with positive (warm ocean phase—El Niño) SST departures three to four years prior to the fire year. Fire occurrence in both forest types was not associated with inter-annual variations in PDO (results not shown). The period of analysis was the same used in the fire interval analyses, except when the reconstructed climate series was limiting (i.e., earliest date for reconstructed Niño3 index, 1600 and PDO index, 1700).

Although the results of the SEA indicate surprisingly similar inter-annual fire–climate relationships between the MC and the PP forest types, there were some differences. Mean summer PDSI associated with all fires in the MC forest (-2.59) was significantly drier than in the PP (-1.03; $t = 3.428$, $p < 0.001$, t-test with equal variance; SPSS 16.0). Widespread fire years (25% scarred) in the mixed/conifer aspen forests occurred on drier years (mean PDSI = -3.22) than in the PP forest (-2.57), but the difference was not statistically significant ($t = 0.798$, $p = 0.432$). Widespread fires occurred during drier years on average compared to all fires in the PP forest ($t = 2.498$, $p = 0.014$). The same was true in the MC forest, but the difference was not significant ($t = 0.896$, $p = 0.375$). The PDSI during the one reconstructed stand-replacing fire (1685) in the spruce-fir zone was -6.92.

4. Discussion

Fire in the upper Santa Fe River watershed historically spread between forest types and fire regimes. Low severity fires burned frequently in the PP forests. During sufficiently dry conditions fire spread up the watershed into the MC forests and burned with mixed-severity. During an extreme drought (1685), fire continued to spread into the highest elevation spruce-dominated forests and burned primarily with high severity. The connectivity of forests through fire, the removal of this important process, and historical evidence of large (100–1200 ha) stand-replacing fire patches in MC and spruce-dominated forests have important implications for both fire and water management in the upper Santa Fe watershed and similar forests throughout the region.

4.1. Human influence on the fire regime

Santa Fe was settled by the Spanish earlier than other locations in the southwestern U.S. (1600s), making this site unique. The most striking feature of the Santa Fe watershed fire scar record is the lack of widespread fire since the mid-to-late 19th century (Fig. 2). Fires stopped earlier (i.e., last widespread fire in the PP and MC, 1842) compared to the general pattern of circa 1900 fire exclusion in the southwestern U.S. (Swetnam and Baisan, 1996, 2003). The start of fire exclusion at a particular site has been linked to the timing of intensive land use practices (e.g., grazing and fuel wood collecting) by the Spanish and Anglo-American settlers (Savage and Swetnam, 1990; Baisan and Swetnam, 1997). Sheep herding in the vicinity of Santa Fe began in the 1600s, became a stable industry regionally by the mid-1700s, and peak numbers in the pre-American Civil War era were recorded in the 1820s and 1830s (Baxter, 1987). This early, intensive land use may have created a pattern of anomalously early fire exclusion (e.g., early 1700s, Sandia Mountains, NM; Baisan and Swetnam, 1997) on the east side of the Rio Grande valley along the Camino Real Spanish travel and settlement route. A long gap between widespread fires in the PP and MC forest in the Santa Fe watershed beginning in the 1700s may indicate initial effects of early grazing, but may also have a climatic explanation.

In specific locations in the southwestern U.S. the fire scar record has revealed periods of anomalously high fire frequency (e.g., repeated 1-year fire intervals) or a change in the seasonality of fire occurrence, indicating possible human ignitions (e.g., Chiricahua Mountains, Arizona; Seklecki et al. (1996)). Very few (<2%) latewood fires were recorded in the Santa Fe watershed and there was not evidence of anomalously high fire frequency, despite the long record of settlement. The high percentage of lightning-caused fires (80%, $n = 178$, 1970–2003) in the local area supports the general premise that sufficient lightning ignitions occur in the southwestern U.S. to account for the reconstructed frequency of fire occurrence (Allen, 2002).

4.2. Spruce-fir fire history

Very little fire history and/or age structure data exist for old-growth spruce-fir forests of Arizona and New Mexico. Fule et al. (2003) reconstructed a mixed-severity fire regime with surprisingly frequent small fires ($MFI_{all\ fires}$ = 2.6 years) and less frequent widespread fires ($MFI_{25\%}$ = 31.0 years) in a relatively low elevation (<2800 m) spruce-fir forest that contained a mix of species (including PP) on the north rim of the Grand Canyon, AZ. A higher elevation spruce-fir forest (average elevation 3200 m) in the San Francisco Peaks, AZ, has not burned catastrophically for over 200 years based on the age of the oldest trees (Cocke et al., 2005). Other high elevation (>3000 m) pure spruce-fir forests in the southern sky island region (Pinaleño Mountains, AZ, and Mogollon Mountains in the Gila Wilderness, NM) had not experienced significant stand-replacing disturbance for at least 300 years prior to the recent crown fires beginning in the late 1990s (Grissino-Mayer et al., 1995; Margolis, 2007). Multiple lines of tree-ring evidence suggest that the Pinaleño spruce-fir stand regenerated after a stand-replacing fire in 1685 (Grissino-Mayer et al., 1995; Margolis, 2007; Swetnam et al., 2009), the same year as the upper Santa Fe watershed. Drought conditions in 1685 were remarkably severe and widespread throughout the southwestern U.S. (Cook et al., 2004). This climate event synchronized these rare stand-replacing fire events, and potentially others, hundreds of kilometers apart.

4.3. Comparing the PP and MC fire regimes

Historical MFI was significantly shorter in PP compared to the higher elevation MC forest in four of the five filtered subsets of fire years (Table 5). Widespread fires in the MC forest occurred on average at intervals that were 10 years (50%) longer than in PP

(Table 5; PP $MFI_{25\%}$ = 20.8 years, MC $MFI_{25\%}$ = 31.6 years). The difference in fire frequency might be partially explained by a larger area in the PP zone (PP, 1600 ha vs. MC, 1200 ha), different sampling intensity (PP, 76 trees: MC, 65 trees) or the spatial distribution of samples. However, these sampling differences are relatively small and with sufficient sample numbers, 25% scarred MFI is robust to differences in sampling (Van Horne and Fule, 2006) and thus likely does not account for the magnitude of observed fire frequency differences. Regionally, MC forests burned less frequently than pine-dominant forests based on comparisons from dozens of Southwestern fire history studies (Swetnam and Baisan, 1996; Heinlein et al., 2005). $MFI_{25\%}$ of widespread fires at six other MC sites in New Mexico ranged from 16.0 years to 26.4 years (Swetnam and Baisan, 1996), which is shorter than the Santa Fe watershed (MC $MFI_{25\%}$ = 31.6 years). The relatively long MFI could be a result of settlement and land-use (e.g., grazing) by the Spanish beginning in the 1600s (Debuys, 1985), which could have reduced fine fuels and consequently fire occurrence in the watershed earlier than in other locations (e.g., Savage and Swetnam, 1990; Baisan and Swetnam, 1997).

An inverse relationship between fire frequency and elevation exists broadly across the montane forests of the western U.S. (Martin, 1982) and at individual sites (Caprio and Swetnam, 1995; Brown et al., 2001)), but site-specific topographic factors may weaken the relationship in some locations (Brown et al., 2001). A hypothesized mechanism for this pattern relates to increased moisture in the higher elevation forests and consequently less frequent occurrence of drought conditions severe enough to dry fuels sufficiently to sustain fire spread. Our results indicate that, on average, fires in the MC forest occurred during drier conditions compared to the adjoining lower elevation PP, providing quantitative support for this hypothesis (Fig. 8). Specifically, the grassy understory of the drier, relatively open PP forest was more likely to carry fire, even if fuels in the mesic mixed-conifer zone were not primed by drought for widespread fire.

4.4. Fire–climate relationships

The relationship between fire occurrence in MC and PP forests and drought during the fire year is intuitive and commonly observed in fire history reconstructions across fuel types in the southwestern U.S. (Fig. 8; Swetnam and Baisan, 1996). The relationship between fire occurrence and wet conditions in prior years is less intuitive, but also well replicated in pine-dominant forests of the southwestern U.S. from fire history studies (Baisan and Swetnam, 1990; Swetnam and Baisan, 1996) and the instrumental record (Crimmins and Comrie, 2004; Baisan and Swetnam, 1990) hypothesize that wet years increase fine fuels (e.g., grass and pine needles) that carry fire, which are burned during subsequent dry years.

This antecedent wet-year relationship is not present in high elevation sub-alpine forests and upper montane seral MC forests of the Southern Rockies (e.g., Sibold et al., 2006; Margolis et al., 2007). A similar drought-only fire–climate relationship exists at multiple MC fire history sites in the region (Swetnam and Baisan, 1996; Touchan et al., 1996). These more mesic, higher elevation forest types are generally not fuel-limited, but require more severe drought for fire occurrence than lower elevation forests.

Based on this prior research, the relationship between fire occurrence and antecedent wet years in the MC forests of the Santa Fe watershed was somewhat surprising (Fig. 8). This result suggests that variability in fine fuels may have been important for fire occurrence (i.e., the system was fuel-limited). But how can a fire regime with a 20- to 30-year mean return interval for widespread fires be fuel-limited? Twenty years in a MC forest should be sufficient to produce enough fuel to sustain fire spread, even in the semi-arid southwestern U.S. It is possible that due to the topographic heterogeneity of the landscape (opposing north and south-facing slopes), wet conditions followed by drought were needed to produce sufficient fuel on the drier south aspects to connect the more productive forest patches and allow fire to burn across aspect and forest types. Grazing could amplify the aspect-driven fuel discontinuity by further reducing fuels on the drier, grassy, south-facing slopes.

A second factor that may explain the wet lags in the MC SEA results is the connectivity of the MC forest to the adjacent, large, frequent burning PP forest. The PP forest in the Santa Fe watershed, similar to others throughout the southwestern U.S., had an herbaceous understory that fueled the frequent fires. As expected, historical fire occurrence in the Santa Fe watershed PP forest was associated with prior wet years that replenished this herbaceous fuel layer (Fig. 8). Prevailing wind direction and the tendency for fire to move upslope would push fires from the PP into the MC forest. Based on our analysis of fire synchrony, 24% of the PP fires spread to the MC forests, but these accounted for a large proportion (69%) of all fires in the MC forest (Table 3). Thus, if fires in PP were in part fueled by prior wet years, and it was sufficiently dry during the fire year, fires would continue to spread up the "fireshed" into the MC forest. The connectivity between forest types would indirectly link fire occurrence in the MC zone to antecedent wet years.

4.5. Landscape scale connectivity of fire regimes

By reconstructing fire history along an elevation, vegetation and fire regime gradient we were able to reconstruct evidence of the transition of fire regimes (and individual fires) from surface fire, to mixed-severity fire (e.g., 1842 fire), to widespread stand-replacing fire (e.g., 1685 fire) in a single watershed. We present multiple lines of evidence of connectivity between forest types and fire regimes through fire as a continuous process that moves across artificially drawn fire regime and vegetation boundaries (Caprio and Swetnam, 1995; Fule et al., 2003). An important implication of this connectivity is that by altering the fire regime in one location (forest type) there may be effects in other forest types. The disruption of the surface fire regime in the mid-elevation, pine-dominated forest throughout the southwestern U.S. (Swetnam and Baisan, 1996, 2003) may not only have serious consequences for that vegetation type (Allen et al., 2002), but is also likely to have effects all along vegetation/elevational gradients. In the Santa Fe watershed, early fire exclusion in PP (i.e., last widespread fire, 1842) from grazing followed by active fire suppression removed an important source of fires for the MC and the spruce-dominated forests. As a result, fire frequency was dramatically reduced in the upper elevation MC forest (Fig. 2).

4.6. Mixed-conifer/aspen forest change due to fire exclusion

Over 120 years of fire exclusion in the MC forest has contributed to changes in structure and composition similar to what occurred regionally and locally in PP and MC forests (Fig. 9). We present age structure data from two fire sensitive species (white fir and quaking aspen) as examples of changes in species composition in the MC forest that occurred coincidently with fire exclusion. Seventy-five percent of the dominant white fir in the MC zone recruited since the last widespread fire (1842; Figs. 3–5). Young white fir has thin bark, making them particularly sensitive to even low-intensity surface fire. In the absence of fire these trees survived to occupy a dominant canopy position, and because they are shade tolerant, they have continued to recruit in the understory, creating ladder fuels, and increasing crown fire hazard. This pattern has been documented in PP dominated systems (Allen

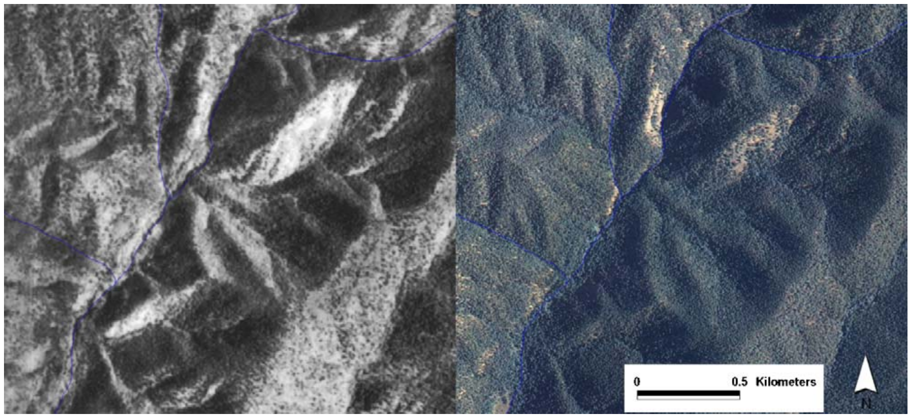

Fig. 9. Comparison of aerial photos (1935 on the left, 2005 on the right) from the MC forest of the Santa Fe watershed indicating a dramatic increase in forest cover on south- and southeast-facing slopes. Images encompass age structure plots 8, 12, and 7. Photos from the U.S.F.S. Santa Fe National Forest, courtesy of Julie Luetzelschwab.

et al., 2002) and other southwestern U.S. MC forests (Mast and Wolf, 2006).

Fire was historically an important determinant of quaking aspen mortality and natality in many upper elevation forests across the western U.S. (Kulakowski et al., 2006; Margolis, 2007; Margolis et al., 2007) and the cessation of fire has been identified as one cause of widespread stand-deterioration throughout its range (Bartos and Campbell, 1998; Kashian et al., 2007). In the Santa Fe watershed, only one (2.5%) quaking aspen stem pre-dated the last widespread fire (1842, Figs. 3–5). Quaking aspen recruitment pulses occurred at three transects following the last fire (3, 13, and 15). Conifers survived the fire at these locations, indicating mixed-severity fire effects by species (i.e., quaking aspen were top-killed and re-sprouted while the overstory conifers survived). This evidence of fire killing and regenerating quaking aspen stems at multiple locations throughout the MC forest illustrates the substantial effect of fire (occurrence and exclusion) on quaking aspen age structure.

4.7. Spruce-fir forest: potential for fire exclusion effects

In the high elevation spruce-fir forests of the region, limited research has assessed the potential for changes related to fire exclusion (Fule et al., 2003; Cocke et al., 2005). Cocke et al. (2005) recorded increased density in spruce-fir since 1876, but this is consistent with natural succession in this forest type. Because *Picea* and *Abies* species are shade tolerant and fires are infrequent, these forests naturally increase in density through time. Different approaches (e.g., examining effects of fire interval length on successional pathways) may be necessary to evaluate potential effects of fire exclusion in this forest type.

Changes in the length of fire-free intervals, even if they were naturally long, may affect successional pathways and forest composition (Romme and Knight, 1981; Kipfmueller and Kupfer, 2005). For example, Romme and Knight (1981) found that sites with naturally longer fire-free intervals and more rapid succession were dominated by spruce-fir forests, compared to sites with more frequent fire and slower succession, which were dominated by lodgepole pine. In the southwestern U.S., quaking aspen is the upper elevation tree species most likely to be sensitive to changes in the length of fire intervals. Seral quaking aspen in the upper montane forests of the region depend on stand-replacing fire for widespread regeneration and long-term perpetuation of the stand (Margolis et al., 2007). Following fire in these seral stands, the aspen-conifer successional pathway proceeds and shade tolerant conifer species regenerate under the canopy, eventually overtopping and shading out the aspen stems in the absence of fire (Dick-Peddie, 1993). Lengthening fire-free intervals in seral aspen stands beyond the life of the above-ground stems and the below-ground clonal root resources could potentially remove aspen from the site, affecting the long-term forest composition.

We hypothesize that although fire intervals were naturally long in high elevation forests of the southern Rocky Mountains, because fire historically spread between forest types, fire exclusion in the lower elevation forests has likely affected some high elevation forests. Future research should be designed to test for changes (e.g., altered successional pathways) resulting from fire exclusion. Upper elevation spruce-fir forests are naturally dense, so although forest density has been an indicator of change in PP and MC forests, it is not likely the best variable to test for change in the spruce-fir zone.

4.8. Will Santa Fe flood?

Large patches of high severity fire (>100 ha) historically occurred on some north-facing slopes in the MC forests of the Santa Fe watershed. The dramatic increase in forest density and canopy cover in these forests, evident from repeat photos (Fig. 9), has very likely increased the size of forest patches at risk of high severity fire. Areas that historically burned with mixed-severity (i.e., 100 ha patches of high severity fire adjacent to equally large low-severity patches) now are likely to burn as larger, contiguous high severity patches. This increased area of forest at risk of stand-replacing fire could subsequently result in a larger, historically unprecedented post-fire hydrologic response in this vital municipal watershed (e.g., Veenhuis, 2002).

One approach to evaluating post-fire flood risk would be to use a combination of our historical fire reconstructions and a hydrological model. The 1685 fire was the worst-case scenario in the spruce-dominated forest; 93% of the sampled spruce forest burned with stand-replacing severity (~1200 ha). The reconstructed spatial extent and location of low and high severity fire patches from this fire and others (e.g., 1842) could be used to populate a GIS-based hydrologic model such as The Automated Geospatial Watershed Assessment Tool (Goodrich et al., 2006). Alternatively, fire behavior and fire spread models (e.g., FARSITE) could be used to estimate the range of high severity patch sizes under current forest conditions for comparison with reconstructed patch size. The different fire scenarios (modeled and reconstructed) could then be used to populate the hydrologic model. Modeled post-fire runoff and erosion output would provide the best possible answer to the big question in the Santa Fe watershed: what will happen to the water supply when the forest burns?

5. Conclusions

Historical fire in the upper Santa Fe River watershed burned across gradients of elevation, forest types and fire severity. Widespread fires that burned up to 80% of the MC forest area occurred on average at intervals 10 years longer ($MFI_{25\%}$ = 31.6 years) than in the adjacent, lower elevation PP forest ($MFI_{25\%}$ = 20.8 years). The historical MC fire regime is best described as mixed-severity, where patches of stand-replacing fire greater than 100 ha were located adjacent to stands with evidence of repeated surface fire. The upper elevation spruce-dominated forest last burned in 1685 in a climate-driven stand-replacing fire that affected greater than 93% (1200 ha) of the sampled spruce forest and at least 68% of the MC and PP forests (total fire area, 4730 ha). This history of fire that includes natural stand-replacing patches in the upper elevation forests presents challenges for fire management in the watershed. Restoring the aspect-driven heterogeneity of fuels in the MC forest is both ecologically sound and would reduce the area at risk of crown that could threaten the water supply. Given the natural occurrence of large (>1000 ha) stand-replacing fire patches in the spruce-fir zone of the Pecos Wilderness Area, where fire hazard reduction treatment options are limited and would be ecologically unsound, hydrologic models should be used to develop a contingency plan for a large, high severity fire.

Climate variability has strongly influenced fire regimes for centuries in the montane forests of the southwestern U.S. (Swetnam and Betancourt, 1990; Swetnam and Baisan, 1996) and more broadly across western North America (Kitzberger et al., 2007). Fire synchrony between the MC and the PP forest during 24 individual fire years (69% of all MC fires) indicates both top-down control of fire occurrence by climate and connectivity between forest types and fire regimes. More severe drought was required on average for the higher elevation MC forest to burn (sometimes with mixed-severity), compared to the lower PP forest. The worst single-year drought in over 700 years (1685) was associated with the last major fire in the upper elevation spruce-dominated forests of the Santa Fe watershed and synchronized high severity fire in the upper elevations of multiple, distant mountain ranges. This evidence of a direct relationship between drought severity, fire occurrence, and fire severity in MC and spruce-dominated forests suggests that if temperatures continue to increase (IPCC, 2007) and droughts become more frequent and severe as predicted (Seager et al., 2007), the probability of large and severe fire occurrence will increase (Westerling et al., 2006). This emphasizes the urgency for creative and science-based fire and watershed planning and management in this and other fire prone, vitally important watersheds across the West.

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Drought, multi-seasonal climate, and wildfire in northern New Mexico

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Abstract Wildfire is increasingly a concern in the USA, where 10 million acres burned in 2015. Climate is a primary driver of wildfire, and understanding fire-climate relationships is crucial for informing fire management and modeling the effects of climate change on fire. In the southwestern USA, fire-climate relationships have been informed by tree-ring data that extend centuries prior to the onset of fire exclusion in the late 1800s. Variability in cool-season precipitation has been linked to fire occurrence, but the effects of the summer North American monsoon on fire are less understood, as are the effects of climate on fire seasonality. We use a new set of reconstructions for cool-season (October–April) and monsoon-season (July–August) moisture conditions along with a large new fire scar dataset to examine relationships between multi-seasonal climate variability, fire extent, and fire seasonality in the Jemez Mountains, New Mexico (1599–1899 CE). Results suggest that large fires burning in all seasons are strongly influenced by the current year cool-season moisture, but fires burning mid-summer to fall are also influenced by monsoon moisture. Wet conditions several years prior to the fire year during the cool season, and to a lesser extent during the monsoon season, are also important for spring through late-summer fires. Persistent cool-season drought longer than 3 years may inhibit fires due to the lack of moisture to replenish surface fuels. This suggests that fuels may become increasingly limiting for fire occurrence in semi-arid regions that are projected to become drier with climate change.

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1 Introduction

Over the past decade, wildfires have made headlines due to their increasing size, severity, and cost. Warming temperatures, drought, and earlier snowmelt—all consistent with projected future climate in the southwestern USA—have been linked to an increasing number of large fires (Dennison et al. 2014; Westerling 2016). The legacy of late nineteenth and twentieth century land use and forest management has also played an important role by increasing fuels that have led to recent megafires (10,000 ha to >100,000 ha), particularly in dry conifer forests (Stephens et al. 2014). Natural climate variability, in addition to human influences, has long been a primary driver of variability in wildfire occurrence, severity, and seasonality (Littell et al. 2016; Swetnam et al. 2016). Climate change will likely alter fire regimes globally, but the mechanisms and the directions of the effects are complex and will vary geographically (Moritz et al. 2012).

Understanding the relationships between climate variability and wildfire by analyzing instrumental and paleoecological data (e.g., tree rings or sediment charcoal) is increasingly valuable for fire management and modeling future fire regimes. Robust relationships have been established in North America between variability in instrumental period (twentieth and twenty-first century) and paleo (pre-twentieth century) fire records and a suite of climate variables, climate patterns, and ocean-atmosphere oscillations (Swetnam and Betancourt 1990; Westerling et al. 2006; Kitzberger et al. 2007; Marlon et al. 2012; Williams et al. 2015). However, fire-climate relationships are spatially and temporally complex, with significant variability within and among regions in fire and moisture seasonality, and lagging relationships that drive fire (Swetnam and Betancourt 1998; Littell et al. 2009; Keeley and Syphard 2016). To date, there is limited understanding of the impacts of the seasonality of moisture and persistent drought on wildfire size and seasonality.

The relationships between cool-season moisture and fire over past centuries (circa 1600–1900 CE) have long been established in the southwestern USA using data from fire-scarred trees (for background on fire scars, see Text S1). A pattern of one or two wet cool seasons followed by cool-season drought is consistently associated with fire occurrence in dry conifer forests of the region (Swetnam and Betancourt 1990; Swetnam and Betancourt 1998). In contrast, the role of summer moisture, delivered through the North American monsoon (NAM) and accounting for up to 50% of the annual precipitation in the southwestern USA, has not been well investigated. Limited research indicates a potential influence of the NAM on fire through increased fine fuels from prior wet monsoons (Crimmins and Comrie 2004; Text S2), or the possibility of monsoon drought leading to more monsoon-season fires (Grissino-Mayer and Swetnam 2000). Until recently, there have been no tree-ring proxies of summer moisture, but a large new network of partial ring-width chronologies now enables the reconstruction of both cool- and monsoon-season moisture in the southwestern USA (Griffin et al. 2013; Text S3).

In this study, we compile the largest known collection of fire scar data for a single mountain range and develop new reconstructions of cool- and monsoon-season moisture to investigate relationships between historical fire regimes and multi-seasonal climate in northern New Mexico. Our main research questions are (1) How do monsoon- and cool-season moisture variability affect fire occurrence, extent, and seasonality? and (2) What is the relationship between fire and prolonged drought? Our goal is to improve the understanding of fire-climate relationships in the past to help inform how climate change may impact fire regimes in the future.

2 Study area and data

The Jemez Mountains are located in northern New Mexico within NAM region 3 (Gochis et al. 2009; Fig. 1). Approximately 44% of the annual precipitation falls in the cool season (October–April) and 43% in the monsoon season (July–September). The warmest and driest months of the year are May and June, when the largest fires occur. Multiple large fires have burned in the Jemez Mountains in recent years, including the 2011 Las Conchas fire (63,400 ha). Vegetation in the Jemez Mountains ranges from grasslands at the lower forest border (~2000 m a.s.l.), to ponderosa pine and mixed conifer forests, to montane meadows and spruce forests at the highest elevations (~3000 m a.s.l.). The majority of the landscape was historically dry conifer forest that included ponderosa pine. The region has extensive networks of fire-scarred and climatically sensitive trees, making it an ideal location for tree-ring fire-climate analyses (Swetnam et al. 2016).

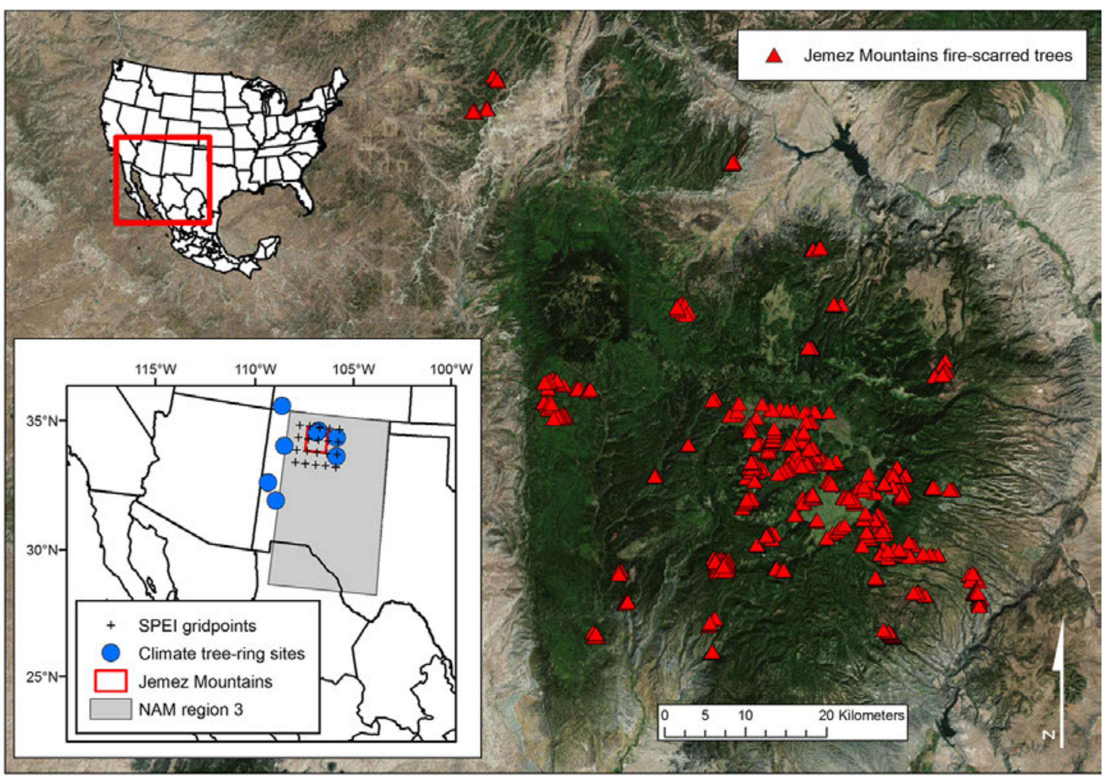

Fig. 1 Study area in southwestern North America focused on the North American monsoon (NAM) region 3. Inset map indicates the location of the climate-sensitive tree-ring sites, the standardized precipitation-evapotranspiration index (SPEI) gridpoints used in the climate reconstructions, the Jemez Mountains, and NAM region 3. The aerial photo is of the Jemez Mountains in New Mexico, which contain a network of 1343 fire-scarred trees

2.1 Tree-ring climate reconstructions

To reconstruct cool- and monsoon-season moisture, we used existing earlywood and adjusted latewood chronologies from 23 sites in New Mexico and southern Colorado located within and adjacent to the Jemez Mountains and NAM region 3 (Text S3). Adjusted latewood chronologies have the dependence of latewood growth on earlywood removed statistically (Griffin et al. 2011). We reconstructed the standardized precipitation-evapotranspiration index (SPEI), because fire is influenced by the combined effects of temperature and moisture that are integrated into SPEI (Williams et al. 2015). SPEI data were obtained from the Global SPEI Database, which uses monthly precipitation and potential evapotranspiration at a 0.5 degree spatial resolution. A regional time series was generated based on the average of 20 grid points centered on the Jemez study area (Text S3, Fig. 1). Monthly SPEI was averaged for the cool (October–April) and monsoon (July–August) seasons.

Reconstruction models were developed by calibrating earlywood chronologies with October–April SPEI and adjusted latewood chronologies with July–August SPEI separately, using stepwise regression (1896–2007). Models explained 67 and 52% of the total variance for October–April and July–August SPEI, respectively. Models met the assumptions of linear regression, and cross-validation statistics indicate reasonable skill. Details of regression results are in supplemental materials (Text S3, Table S1, Fig. S1). The October–April SPEI reconstruction extends 1594–2007 and July–August SPEI, 1599–2008. The relationship between the seasonal SPEI variables is preserved, for the most part, in the reconstructions. There is no relationship between the instrumental cool- and monsoon-season SPEI ($r = -0.09$, $p > 0.05$), but there is a weak correlation between the reconstructed cool- and monsoon-season SPEI in the instrumental period ($r = 0.23$, $p < 0.05$). Over the full common reconstruction period, 1599–2007, the cool and monsoon-season SPEI are uncorrelated ($r = 0.09$, $p > 0.05$).

2.2 Tree-ring fire history reconstructions

The tree-ring fire scar data were compiled from existing collections in the Jemez Mountains. The data cover approximately 300,000 ha of historically dry conifer forests that used to burn predominantly with low-severity fire. This network, the largest in North America for a single mountain range, is a compilation of 19 studies conducted over 40 years (Text S4). A total of 8588 fire scars from 1295 trees were dated to the year (1599—1899). Fire seasonality was determined for 77% of the scars ($n = 6581$) from the position of the scar within the annual ring. Categories for scar positions and their seasonal timing include: dormant (D—early spring); early, mid, and late earlywood (E, M, L—late spring through mid-summer); and latewood (A—late summer and fall). Most fire years historically had scars in multiple fire seasons (Fig. 2 and S2). Details of the fire scar seasonality methods are described in the supplementary materials (Text S1).

Fire scar data were compiled and analyzed with the "burnr" fire history package in R (Malevich et al. 2015; R Core Team 2015). Percent of recording trees scarred was used as a proxy for relative fire size (e.g., Farris et al. 2010). Fires recorded by a single tree were not included in the analysis. After 1899, the number of fires in the Jemez Mountains declines precipitously due to increased human land use, so the common period for the fire and climate data is 1599–1899. Native Americans influenced fire regimes through the mid-1600s in the southwest Jemez Mountains (Swetnam et al. 2016), which could affect fire-climate relationships in the early part of the record.

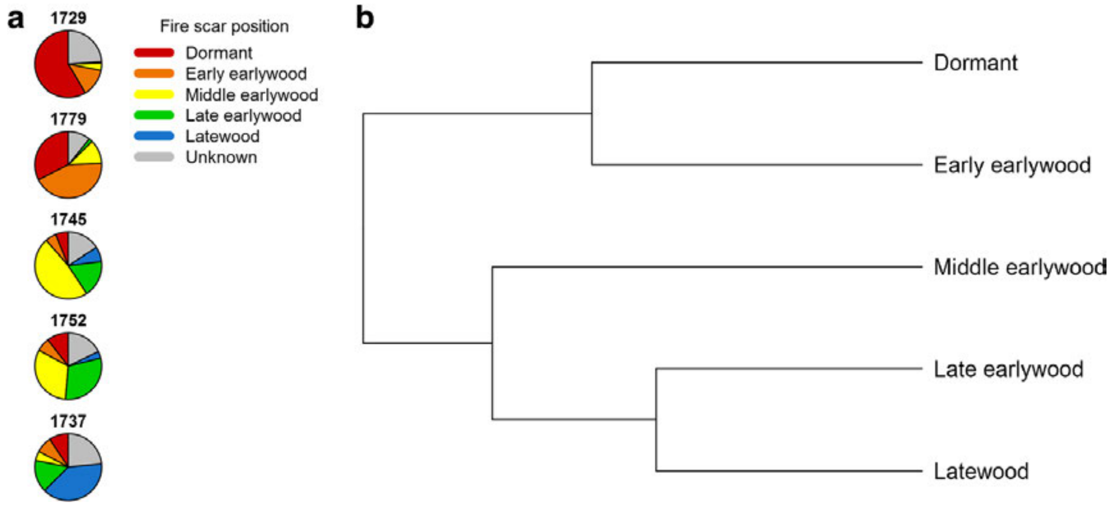

Fig. 2 **a** The proportion of trees scarred by fire in the Jemez Mountains in each fire scar position, or season, for five large fire years. The selected years have the largest number of fire scars in the spring dormant (1729) through late summer/fall latewood (1737) fire seasons in the 1700s. Note the inter- and intra-annual variability in the distribution of fire seasonality. **b** Cluster dendrogram of large, 95th percentile fire years by fire-scar position (n = 16 years for each scar position). Note the grouping of dormant and early earlywood (DE) fire years and middle earlywood, late earlywood and latewood fire years (MLA)

Analyses focused on the extreme fire years. Extreme large and small fire years were determined by the 95th and 5th percentile rank of the percent of recording trees scarred in a year (Table S2). Extreme fire years were first determined for all fires (combining all fire seasons, including unknown seasonality), and then for each of the five individual fire scar seasonalities (spring through fall). A total of 16 fire years fell within the 95th percentile (Fig. S2). The 5th percentile years were all years when no fires occurred.

3 Analysis methods

3.1 Multi-seasonal fire-climate analysis

We used superposed epoch analysis (SEA) to test whether fire occurrence and fire seasonality were associated with cool- and monsoon-season SPEI anomalies (Swetnam 1993). SEA is a compositing approach that uses block re-sampling and bootstrap simulations to evaluate the significance of the concurrence between fire event years and wet or dry conditions in the event year or lagged years. We examined 7-year blocks of cool- or monsoon-season SPEI spanning 4 years before, and 2 years after the fire year (year zero). We first used SEA to test whether cool- and monsoon-season SPEI anomalies were associated with all extreme large fire years and no fire years, and then for SPEI associations with the separate individual fire seasons.

To determine associations among the different fire scar positions, as well as relationships between fire-scar positions and seasonal climate, we used hierarchical cluster analysis of extreme large fire years for all five individual fire scar positions (hclust; R Core Team 2015). The analysis includes all possible combinations, not just adjacent scar positions. The groups that resulted from the cluster analysis were used as a framework for combining multiple fire scar positions for analyzing the relationships between sequences of cool- and monsoon-season moisture and the related fire scar positions, as well as the drought-fire analysis.

3.2 Drought-fire analysis

Reconstructed cool- and monsoon-season SPEI series were first analyzed to investigate characteristics of seasonal drought. This included the number and length of droughts (single and consecutive years with negative SPEI values) and comparisons of these metrics between cool- and monsoon-season droughts. The relationships between droughts and large fire years in the early (D and E) and mid-to-late (M, L, and A) fire seasons—as grouped by the cluster analysis—were then examined to determine (1) the length of droughts in which the large fires occurred and (2) the year in the drought that large fires occurred. On the basis of the SEA results, early-season fires were evaluated with cool-season droughts, and mid- to late-season fires were evaluated with both cool- and monsoon-season droughts.

We also assessed whether the driest decades of the cool- and monsoon-season SPEI reconstructions were associated with increased fire. Here, we relax the threshold for fires to include those with at least 2.5% of trees scarred (74th percentile, n = 79 fire years for early-season fires and 85th percentile, n = 46 fire years for mid- to late-season fires). Decadal dry periods were identified as the five driest non-overlapping decades for each climate season. Decades with the highest fire activity for early and mid- to late-season fires were defined as the five non-overlapping decades with the largest sum of the percent of recording trees scarred. These decadal measures of climate and fire were compared visually to assess the correspondence between the most active fire periods and the driest periods.

4 Results

4.1 Cool-season climate associated with large fire years

The SEA analysis for the largest fire years, regardless of fire season, highlights the importance of cool-season drought during the fire year (Fig. 3a, top row). The largest fire years were also associated with wet cool seasons 2 and 3 years prior to the fire year. No significant associations were found between all large fires and monsoon-season moisture, although a similar pattern of dry conditions during the fire year preceded by wet years is suggested (Fig. 3b, top row). Years without fire were associated with wet cool seasons in the fire year, but not with monsoon moisture.

When the largest fire years for each fire season are analyzed, several different fire-climate relationships are revealed. The SEA results indicate that early season (D and E) fires are most strongly associated with cool-season drought during the fire year, with the strength of the association decreasing by mid-summer through fall (Fig. 3a). Similarly, the importance of prior wet cool seasons associated with large fire occurrence decreases through the fire season; spring (D) fires are associated with two prior wet cool seasons 2 and 3 years before the fire year; early- to mid-summer fires (E, M, and L) are associated with one wet cool season 2 or 3 years prior to the fire year; and late-summer and fall (A) fires have no significant relationship with prior wet cool seasons.

4.2 Monsoon-season climate associated with large fire years

Monsoon-season drought during the fire year is significantly associated with large late season (L and A) fire occurrence (Fig. 3b). There is a suggestion of a similar relationship with the

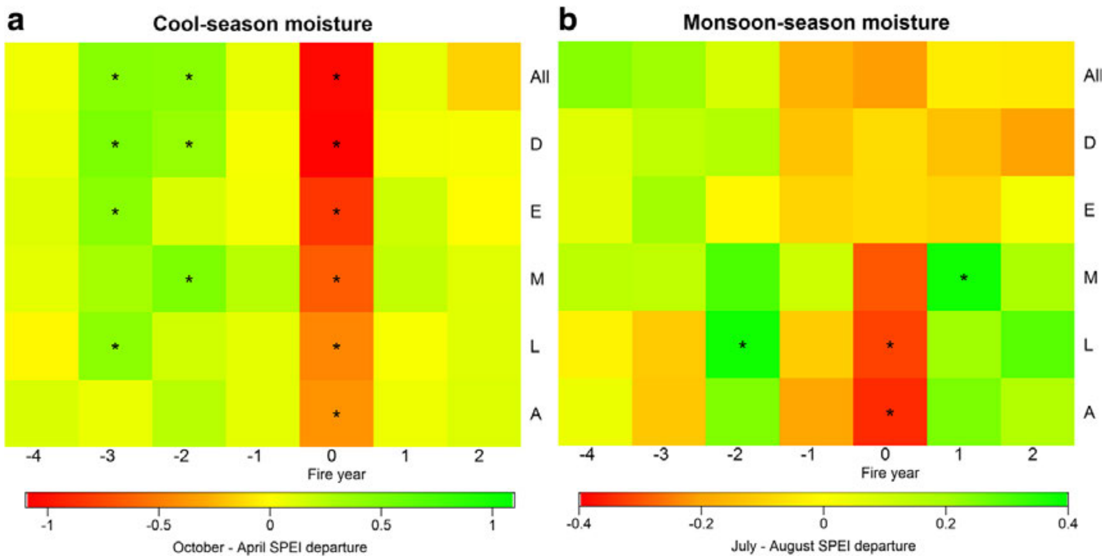

Fig. 3 Superposed epoch analysis of **a** cool-season moisture and **b** monsoon-season moisture by fire-scar position for large, 95th percentile fire years in the Jemez Mountains (n = 16 fire years for each seasonality, 1599–1899). All = all fire scar positions, D = dormant, E = early earlywood, M = middle earlywood, L = late earlywood, and A = latewood. SPEI = standardized precipitation-evapotranspiration index. *Asterisks* in cells denote significant departures from mean SPEI based on bootstrap simulations ($p < 0.05$)

monsoon and mid-season (M) fires. Since late-season fires are also associated with cool-season drought during the fire year, joint drought in the cool and monsoon seasons appears important for widespread late-season fires. Wet conditions in the cool and monsoon seasons 2 or 3 years prior to the fire year also appear to be important in this sequence favoring late-season fires. There is no significant association between monsoon moisture and D or E fires.

4.3 Fire seasonality patterns

The cluster analysis of the fire-scar seasonality of large fire years supports results from the SEA. There were two main groups of fire scar positions: (1) dormant and early earlywood (DE) and (2) middle earlywood, late earlywood, and latewood (MLA) (Fig. 2b). The patterns of fire-climate relationships from the SEA suggest a similar grouping of M, L, and A fires, particularly in association with monsoon moisture (Fig. 3b). This implies different climatic controls on spring and early summer (DE) fires compared with the mid-summer to fall (MLA) fires. Large early-season fires, by virtue of their timing, are strongly linked to cool-season drought, and they rarely continue to burn throughout the summer (Fig. 2 and S2). Whereas the largest MLA fires burn through the summer under dry monsoon conditions and consequently are associated with drought in both the cool and monsoon seasons. Mid-summer (M) fires are most likely to continue burning through the late summer and fall (L or A scars), but only during dry monsoons (e.g., differing fire-scar position distributions of the 1729 dormant fire year compared to the 1745 middle earlywood fire year, Fig. 2 and S2).

4.4 Relationships between persistent drought and fire

Results from the SEA and cluster analysis suggest that a sequence of both wet and dry years in the cool and monsoon seasons lead to large fires. Thus, a short-term drought might be most favorable for fire, while persistent (multi-year) droughts that do not include intervening wet

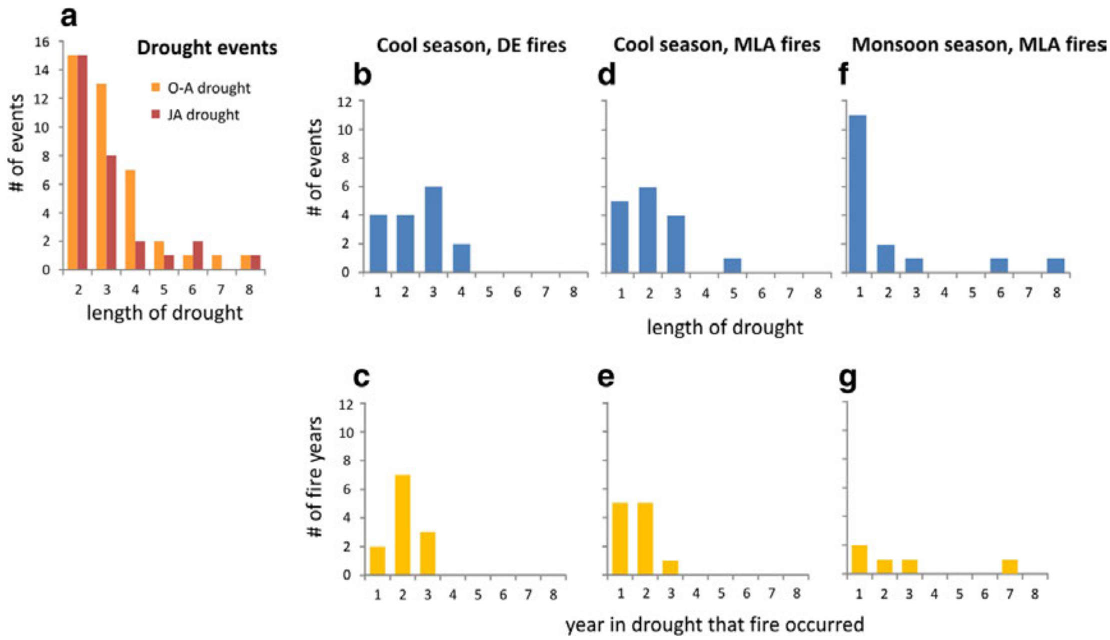

Fig. 4 **a** Numbers and length of droughts (consecutive years of negative SPEI) for October–April (*orange*) and July–August (*brown*) SPEI, 1599–1899. **b**, **d**, and **f** Lengths of the droughts in which the largest fire years occurred, and numbers of fire years corresponding to each drought length for cool season DE fires, cool season MLA fires, and monsoon season MLA fires. **c**, **e**, and **g** The year within the drought in which the fire occurred

conditions could inhibit large fires. The SPEI drought analysis revealed differences in the distributions of drought lengths between the cool and monsoon seasons (Fig. 4a). Single dry years are more common in the monsoon and multi-year droughts occur more frequently in the cool season. There were 25 cool-season droughts of 3 years or more, compared to 14 for the monsoon (Table S3). The longest cool-season droughts lasted up to 8 years. In order to explore the relationship between fire occurrence and drought length, we examined when large fires occurred relative to multi-year cool- and monsoon-season droughts.

Although there are cool-season droughts lasting 5 to 8 years, none of the largest early-season (DE) fires occur during these persistent droughts (Fig. 4). Of the 16 largest early season fire years, ten occurred within a 2- to 3-year cool-season drought, and two within a 4-year cool-season drought (Fig. 4b). The remaining four large early season fire years occurred during single-year cool-season droughts. Within a multi-year cool-season drought, fires only occurred in the first 3 years and primarily in the second year of the drought (Fig. 4c). This result generally supports the SEA, which indicates that conditions most strongly linked to large early-season fires include a wet year 2 or 3 years prior to the fire year, but not the year prior to the fire year.

Relationships between persistent cool-season droughts and the largest mid- to late-season fires (MLA) are similar to the early-season fires. Most of the mid- to late-season fires occur during droughts of 2 or 3 years (Fig. 4d). Almost all large mid- to late-season fires occur in the first 2 years of a cool-season drought (Fig. 4e).

Similarly, persistent monsoon-season drought was not related to large fire occurrence. Only 31% (5 of 16) of the large mid- to late-season fire years occurred during persistent monsoon droughts (Fig. 4f). These monsoon droughts lasted 2 to 8 years. All but one of these large fires occurred within the first 3 years of the persistent drought (Fig. 4g). Results for the monsoon droughts may reflect the fact that, compared to the cool season, monsoon droughts are more likely to occur as single years (Fig. 4a).

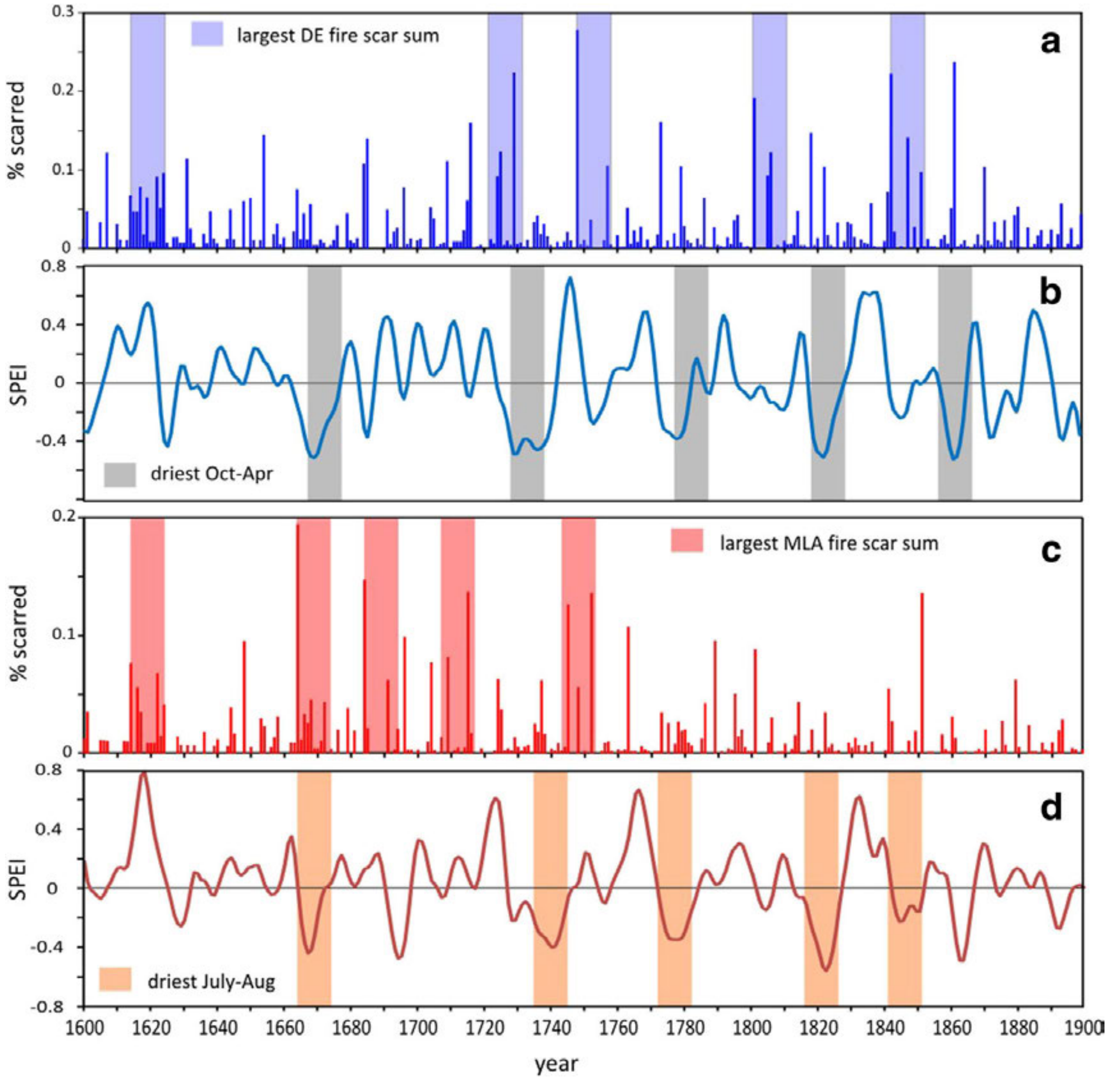

Fig. 5 **a** Percent of recording trees scarred by early-season (DE) fires in *dark blue bars*. *Light blue vertical bars* are the five non-overlapping decades with the largest sums of percent DE scarred trees. **b** October–April SPEI smoothed with a 10-year spline; *vertical bars* are the five non-overlapping decades with the lowest SPEI values. **c** Percent of recording trees scarred by mid- to late-season (MLA) fires in *dark red bars*. *Light red vertical bars* are the five non-overlapping decades with the largest sums of percent MLA scarred trees. **d** July–August SPEI smoothed with a 10-year spline; *vertical bars* are the five non-overlapping decades with the lowest SPEI values

When the driest decades of both SPEI seasons were assessed, they were not consistently related to decades of high fire occurrence. For early season (DE) fires, the five decades with the largest sum of percent trees scarred—periods of widespread fire—had little correspondence with the driest decades of cool-season moisture (Fig. 5a, b). Since both cool- and monsoon-season drought appear to influence mid- to late-season fires (Fig. 3), we compared dry decades for both seasons with high MLA fire decades. These dry periods are distributed across three centuries (Fig. 5b, d), whereas the decades with the largest MLA fire scar sums are concentrated in the first half of the record (Fig. 5c). As with DE fires, there is little correspondence between the decades with the most widespread MLA fires and the driest decades of cool-season SPEI. The one exception is the mid-1660s to mid-1670s (Fig. 5b, c). However, when looking at monsoon moisture, two of the five driest decades do overlap with high mid- to late-season fire scar sums. The mid-1660s is unique, with widespread mid- to late-season fires coinciding with some of the driest decades in both seasons. This period includes the year with the highest percent of trees scarred in the mid-to-late fire season, 1664.

5 Discussion

5.1 Multi-seasonal climate associated with large fire years

The largest early-season fires tend to occur when wet cool seasons are followed by cool-season drought. Years without fires occur after wet cool seasons, with no influence from climate in prior years. These results emphasize the historical importance of cool-season moisture for promoting conditions conducive to large fires in the dry conifer forests of the Jemez (Touchan et al. 1996), the southwestern USA (Swetnam and Betancourt 1998), and the western USA (Swetnam et al. 2016). Modern studies confirm the importance of cool-season wet-dry oscillations in the cool season for fire occurrence across the western USA, but highlight regional differences. Cool-season drought is an important predictor of twentieth century area burned in northern or mountainous ecoprovinces across the western USA, whereas wet cool seasons in prior years are also important in drier ecoprovinces (Westerling et al. 2003; Littell et al. 2009).

Our results are the first documented effects of the NAM on fire occurrence prior to the twentieth century. Monsoon moisture has the greatest effect on mid- to late-season (M, L, and A) fires. The monsoon must be dry for these mid- to late-summer and fall fires to be widespread, as hypothesized by Grissino-Mayer and Swetnam (2000). Large late-season fires may also depend on cool-season conditions, such that dual-season drought preceded by dual-season wet conditions are important for large late-season fire occurrence. Modern studies indicate that prior-year NAM moisture was associated with fires in Arizona and the Great Basin (Westerling et al. 2003; Crimmins and Comrie 2004; Littell et al. 2009). In these studies, wet summers 1 and 2 years prior to the fire likely increased fine fuels, such as grasses, that were important for fire spread.

The intra-annual distribution of fire seasonality derived from tree-ring fire scars provides additional insights into the effects of the monsoon on fire seasonality. The largest early-season fires appear to burn until the onset of the monsoon (Fig. 2 and S2). This is consistent with modern fires in the region, many of which are extinguished by monsoon moisture. Historically, many of the largest late summer and fall fires appear to have occurred when dry monsoons allowed relatively small early-season fires to continue to burn into the summer and fall. This is indicated by all of the largest late summer and fall fires having some proportion of trees scarred in the early (DE) fire seasons (see distribution of fire scar positions for large latewood fires in Fig. 2 and S2). It is also possible that some large late-season fires may have ignited during a dry monsoon season. Multiple ignitions over the fire season could confound these interpretations.

5.2 Persistent drought and fire

Analysis of cool- and monsoon-season droughts and fire occurrence indicates that, overall, fires most often occur during the first or second year of multi-year droughts. Long droughts do not appear to promote large fires in the later years of the drought. The occurrence of all but one large fire in the first 3 years of a drought is not surprising (Fig. 4c, e, and g), but reinforces the importance of short droughts for fires in the region. This is further supported by the sequence of climate conditions leading to fires, which include a wet cool season several years prior to the fire. Because of the key role of wet cool seasons 2 and 3 years prior to a large fire year, and to a lesser degree in the monsoon season, prolonged drought may actually limit the occurrence of large fires in dry conifer forests. Once these dry forests burn, they need moisture to replenish surface fuels before the area can burn again.

The decades with the driest cool seasons were not consistently related to periods of high fire occurrence. These dry decades do not provide the necessary periodic wet conditions that precede the biggest fire years. Fitch and Meyer (2016) also found that extended dry periods in the Jemez Mountains, going back multiple millennia, did not necessarily correspond with increased fire activity, likely due to fuel limitations. While extremely dry winters are a necessary component for the most widespread fires, regardless of fire seasonality, if dry conditions persist beyond several years, the chances of widespread fire likely diminish. This result of persistent drought reducing fire occurrence in a fuel-limited ecosystem supports observations of the importance of biomass variability for modeling fire regimes globally and their response to climate change (Krawchuk et al. 2009).

Overall, these results suggest that the strongest climatic controls over fire regimes in the Jemez Mountains were seasonal and inter-annual to sub-decadal in scale. Decadal fire-climate relations were generally weak. This suggests that fine fuel biomass production (grasses, tree needles, and cones), which can respond to these short time-scale variations in climate, was likely the most important mechanism of climatic influence. It is probably not coincidental that the El Niño-Southern Oscillation (the key synoptic climate control over wet-dry oscillations in the southwestern USA), the phenological cycle of ponderosa pine (*Pinus ponderosa*) needle and cone production, and the frequency of surface fires, all typically occur over time scales of about 2 to 7 years (Maguire 1956; Swetnam and Betancourt 1990). That is, natural wet-dry oscillations might readily entrain inherent (and evolved) vegetative and reproductive cycles of flammable fuel production, which in turn promote synchronized, extensive surface fires.

5.3 Insights from the past for future fire regimes

Projecting fire response to climate change in semi-arid, biomass-limited regions is challenging, and future fire regimes will likely vary temporally in accordance with biomass availability. Climate-driven changes in vegetation will further confound forecasts of future fire regimes. Williams et al. (2015) suggest that future increased drought and moisture stress will increase fire occurrence in the southwestern USA, until fuel becomes limiting. Our results suggest that in the semi-arid southwestern USA, fuel was historically limiting in dry conifer forests and that persistent cool-season drought actually reduced fire occurrence. This differs from wetter, more productive mixed-conifer, aspen, and spruce-fir forests that are not fuel limited and where prior wet years are not associated with fire occurrence, only severe drought during the fire year (Swetnam and Betancourt 1998; Margolis and Swetnam 2013). Fine and heavy fuel loads in dry conifer forests have increased significantly over the last century due to fire exclusion (Fulé et al. 1997), although mega-fires in recent decades are beginning to reduce these overabundant fuels in portions of the landscape (Stephens et al. 2014). As warming continues to increase drought stress and increase large fire occurrence, some of the drier ecosystems in the region may move back toward being fuel-limited, with consequences for forecasting future fire regimes.

A major uncertainty for future fire regimes in fuel-limited systems is future moisture variability. Forecasting future precipitation is particularly complex in the southwestern USA, because of the two seasons of moisture. Projected extended drying in the region, due to reduced cool-season moisture (e.g., Seager and Vecchi 2010) would likely continue to increase fire occurrence in coming decades. However, as biomass becomes limiting, fire occurrence could ultimately decrease in dry forests and woodlands where fine-fuels are important for fire spread. A transition to a shortened or a weak NAM (e.g., Cook and Seager 2013) could extend the fire season in the southwestern USA through the summer and into the fall, which is currently rare, but consistent with the tree-ring record. Failed monsoons could represent the scenario with the greatest fire occurrence in the near term, before moisture stress from increased temperature supersedes any potential increases in precipitation (e.g., Williams et al. 2013), and biomass becomes increasingly limiting to fire occurrence.

6 Conclusions

We present the first in-depth, landscape-scale analysis of historical multi-seasonal climatic controls of fire size and seasonality using tree rings. Our findings suggest different seasonal climate controls on early season and mid- to late-season fires, but in both cases, sequences of wet and dry conditions are critical for preconditioning forests to burn. Dry conditions in the year of the fire—dry in the cool season for early-season fires, and dry in the monsoon season for late-season fires—are critical. Equally important are wet conditions, particularly in the cool season, 2 to 3 years preceding the fire year. The importance of this sequence of wet and dry years has key implications for relationships between fire activity and drought, and our results indicate persistent drought is not associated with the largest fires or periods of high fire activity in this region.

Our results suggest that as moisture stress increases in the southwestern USA due to warming (Seager et al. 2007; Williams et al. 2013), large fire occurrence may decrease in some fuel-limited ecosystems. Many model projections of global fire response to climate change use multi-decadal climate “normals” and lack inter-annual or intra-annual climate variability (e.g., Krawchuk et al. 2009; Moritz et al. 2012). We demonstrate that inter- and intra-annual climate variability is an important control for large fire occurrence and fire seasonality in a semi-arid, monsoon-affected region of southwestern North America. Accurate projections of inter- and intra-annual moisture variability will likely be important to accurately model future fire in the southwestern USA, particularly due to the bimodal precipitation regime and a likely future increase in biomass limitations on fire occurrence (i.e., requiring wet conditions to produce fuels to burn). In the future, in semi-arid regions such as the southwestern USA, prolonged droughts driven by warming could decrease fire activity due to biomass limitations.

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Research Article

Historical Stand-Replacing Fire in Upper Montane Forests of the Madrean Sky Islands and Mogollon Plateau, Southwestern USA

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ABSTRACT

The recent occurrence of large fires with a substantial stand-replacing component in the southwestern United States (e.g., Cerro Grande, 2000; Rodeo-Chedeski, 2002; Aspen, 2003; Horseshoe 2, Las Conchas, and Wallow, 2011) has raised questions about the historical role of stand-replacing fire in the region. We reconstructed fire dates and stand-replacing fire patch sizes using four lines of tree-ring evidence at four upper montane forest sites (>2600 m) in the Madrean Sky Islands and Mogollon Plateau of Arizona and New Mexico, USA. The four lines of tree-ring evidence include: (1) quaking aspen (*Populus tremuloides*) and spruce-fir age structure, (2) conifer death dates, (3) traumatic resin ducts and ring-width changes, and (4) conifer fire scars. Pre-1905 fire regimes in the upper montane forest sites were variable, with drier, south-facing portions of some sites recording frequent, low-severity fire (mean fire interval of all fires ranging from 5 yr to 11 yr among sites), others burning with stand-replacing severity, and others with no evidence of fire for >300 yr. Reconstructed fires at three of the four sites (Pinaleño Mountains, San Francisco Peaks, and Gila Wilderness) had stand-replacing fire patches >200 ha, with maximum patch sizes ranging from 286 ha in mixed conifer-aspen forests to 521 ha in spruce-fir forests. These data suggest that recent stand-replacing fire patches as large as 200 ha to 500 ha burning in upper elevation (>2600 m) mixed conifer-aspen and spruce-fir forests may be within the historical range of variability.

Keywords: fire history, mixed conifer, quaking aspen, spruce-fir, tree ring

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INTRODUCTION

The number and duration of large fires in the western United States has increased in recent decades due in part to increasing temperatures (Westerling *et al.* 2006). In the southwestern US (Arizona, New Mexico, and proximate areas), many of the recent large fires included large (100 ha to >1000 ha) high-severity fire patches, which raises questions about the historical role of stand-replacing fire in the region. Many of the recent stand-replacing fire patches in the southwestern US have occurred in the overstocked, mid-elevation ponderosa pine (*Pinus ponderosa* C. Lawson) and dry mixed conifer forests, where extensive stand-replacing fires are unreported in the documentary records prior to *circa* 1950 (Cooper 1960, Allen *et al.* 2002). However, in the upper elevation (>2600 m) mixed conifer-aspen and spruce-fir forests, historical photographs and tree-ring data from seral quaking aspen (*Populus tremuloides* Michx.) stands provide direct evidence that fires with large (100 ha to >1000 ha) stand-replacing patches occurred in parts of the region as recently as the early twentieth century (Abolt 1997, Romme *et al.* 2001, Margolis *et al.* 2007).

Relatively little is known about pre-EuroAmerican settlement fire regimes (size, severity, frequency, and seasonality) of upper elevation forests in the southwestern US (Grissino-Mayer *et al.* 1995, Fulé *et al.* 2003, Margolis *et al.* 2007, Margolis and Balmat 2009). Extensive fire histories from upper montane and subalpine forests of southern Wyoming, Colorado, and northern New Mexico indicate that infrequent (>100 yr intervals) stand-replacing fire is a dominant disturbance in upper elevation forests of the southern Rocky Mountains (Kipfmueller and Baker 2000, Sibold *et al.* 2006, Margolis *et al.* 2007). Thus, it is logical to hypothesize that upper elevation mixed conifer-aspen and spruce-fir forests of the southwestern US outside of the southern Rocky Mountains potentially had a historical fire regime that included infrequent, relatively large (>100 ha) patches of stand-replacing fire.

Reconstructing Stand-Replacing Fire

Age-structure-based methods for reconstructing fire history were developed in coniferous subalpine and boreal forests of North America where stand-replacing fire regimes are dominant (Clements 1910, Heinselman 1973, Agee 1993, Johnson and Gutsell 1994). By definition, stand-replacing fires leave few or no surviving trees to record direct evidence of those fires within the highest burn severity patches (but note that fire-scarred survivors can sometimes be found on the edges of such patches; e.g., Margolis *et al.* 2007). Post-fire tree cohorts, assumed to have established soon after the fire, are the most common type of evidence used to date and map stand-replacing burns. In the Rocky Mountains, the assumption that there is typically rapid recruitment of a post-fire cohort (i.e., <5 yr) within stand-replacing burn patches is well supported in the case of quaking aspen, because it has evolved mechanisms for rapid regeneration, and has been commonly observed to do so following fires (Clements 1910, Patton and Avant 1970). Post-fire cohort evidence (dates and mapped perimeters) can be combined with the relatively rare direct conifer evidence of fire (e.g., fire scars, tree death dates, ring-width changes or traumatic resin ducts) to reconstruct annually resolved stand-replacing fire dates (Johnson and Gutsell 1994, Margolis *et al.* 2007).

In the current study, we separate the upper elevation forest into mixed conifer-aspen (2600 m to 3100 m) and spruce-fir (>3100 m) because of differing fire ecology, and potentially different fire regimes and use of differing fire history methods. Age-structure-based fire history methods in mixed conifer-aspen forests have been applied in a few studies in the southwestern US, primarily focusing on quaking aspen regeneration dates as a proxy for stand-replacing fire (Abolt 1997, Romme *et al.* 2001, Margolis *et al.* 2007). Romme *et al.* (2001) reconstructed a 140-year stand-replacing fire rotation period from aspen stand age in the La Plata Mountains of southwestern Colorado. They noted that the lack of fire-scarred trees in aspen stands was a limitation to dating past fires. Abolt (1997) used coincident aspen pith dates and conifer fire scars from lower elevations to date stand-replacing fire patches in mixed conifer forests of the Mogollon Mountains of southwestern New Mexico. Margolis *et al.* (2007) combined four lines of tree-ring evidence (aspen age structure, conifer fire scars, conifer death dates, and conifer injury dates) to reconstruct synchronous, drought-related stand-replacing fire dates and patch sizes from aspen stands embedded in upper montane mixed conifer and spruce-fir forests at a network of twelve sites in the upper Rio Grande Basin (New Mexico and Colorado). These studies indicate that, because of the unique fire ecology of quaking aspen (i.e., high sensitivity to being killed by fire and ability to re-sprout), the age structure from seral aspen stands is a potential indicator of historical stand-replacing fire in upper elevation forests in the southwestern US.

Fewer studies have evaluated the effectiveness of age-structure-based fire history methods in southwestern US spruce-fir forests. In the Pinaleño Mountains, Arizona, Grissino-Mayer *et al.* (1995) used intensive, but spatially limited, age structure sampling in spruce-fir forests, combined with lower elevation fire scars, to hypothesize that the spruce-fir zone regenerated following a stand-replacing fire. Due to limited spatial coverage of the sampling, stand-replacing fire area was not estimated. Fulé *et al.* (2003) used fire scars, tree age and species, and spatial patterns of forest stands to reconstruct fire-initiated tree groups at the plot scale (20 m × 50 m), which likely originated after severe eighteenth century fires in high-elevation forests (including aspen and spruce-fir) on the north rim of the Grand Canyon, Arizona. They were not able to identify distinct fire-created stands in the study area from aerial photos or satellite data, which differs from the stand-replacing fire history methods used in the Rocky Mountains. In the Santa Fe Watershed, New Mexico, Margolis and Balmat (2009) combined a systematic spatial grid sampling of spruce-fir age structure with conifer ring-width growth changes and conifer fire scars to conclude that approximately 90% of the spruce-fir zone (1200 ha) regenerated following stand-replacing fire. These studies provide evidence of past stand-replacing fires in spruce-fir forests in the southwestern US, but leave questions about patch sizes, variability between sites, and the ability to apply fire history methods from other regions and forest types.

Fire Patch Size and Severity

Fire patch size and severity have strong influences on the ecological effects of fire on terrestrial and aquatic systems. Stand-replacing fire patch size is a key determinant of post-fire vegetation composition and structure (Agee 1993, Turner *et al.* 1994, Turner and Romme 1994). Following the extensive (>250 000 ha) fires in Yellowstone National Park, Wyoming, in 1988, the size and severity of burn patches were shown to affect overall plant cover, tree seedling recruitment, and herbaceous recruitment (Turner *et al.* 1994). High-severity fires remove overstory vegetation and ground cover that dramatically affects watersheds and water resources by altering the important processes of evapotranspiration, interception, surface flow, and subsurface flow (Swanson 1981). The size of high-severity fire patches is important in determining the probability of fire-induced flooding or debris flows (Pearthree and Wohl 1991, Cannon and Reneau 2000). Recent, large stand-replacing fires in the southwestern US have produced runoff and erosion events as much as two orders of magnitude greater than pre-fire conditions (Veenhuis 2002).

High-severity (stand-replacing) fire patches are usually part of a "mosaic" of burn severities, within fire perimeters that include moderate- and low-severity surface fire patches, as well as unburned patches (Turner and Romme 1994). For example, less than half of the 1988 Yellowstone fires burned with high severity (Turner *et al.* 1994). Reconstructing the complex spatial patterns and wide range of burn severities of pre-twentieth century fires at high resolution (i.e., less than a few hectares) is not possible. However, the largest stand-replacing fire patches often leave a persistent and identifiable legacy in the form of tree ages and, less commonly, as conifer death dates, conifer fire scars, and tree-ring growth patterns in conifers injured by the fire. From these legacies, stand-replacing fire patch sizes and dates can be reconstructed and compared with recent fires even if overall size (extent) of the entire fire is unknown.

Research Objectives

Our primary objective was to use dendroecological methods to expand the upper elevation stand-replacing fire history network of Margolis *et al.* (2007) to four new sites in mixed conifer-aspen forests (2600 m to 3100 m elevation) in the Mogollon Plateau and Madrean Sky Island regions of the southwestern US, focusing on quaking aspen as a potential indicator of the dating and patch size of past stand-replacing fires. The secondary objective was to test the utility of using spruce-fir forest age structure to expand the reconstruction of stand-replacing fires above the local elevation range of quaking aspen (>3100 m) at two test sites. We did not attempt to reconstruct a complete inventory of all historical stand-replacing fire patches at these four sites; rather, we mapped and dated the largest and potentially most ecologically significant patches.

METHODS

Study Area

To expand the existing southwestern US network of upper elevation stand-replacing fire history sites of Margolis *et al.* (2007) beyond the upper Rio Grande Basin, we selected two sites on the Mogollon Plateau and two sites from the Madrean Sky Islands (Figure 1, Table 1). The sites were selected based on the presence of the largest seral aspen stands, which potentially represented historical stand-replacing fire patches. We used the regional gap analysis program vegetation map, USDA National Forest vegetation maps, black and white and color infrared digital ortho-rectified quarter-quadrangle photographs (DOQQs) and field surveys to map and verify the largest aspen patches on the Mogollon Plateau and Madrean Sky Islands on US Forest Service land. We set the minimum aspen patch size threshold at 5 ha to eliminate smaller patches. We targeted seral aspen stands embedded within conifers to eliminate self-replacing aspen and aspen within high-elevation parklands that likely experienced frequent surface fires (Jones and DeByle 1985).

The largest potential post-stand-replacing fire aspen patches on the Mogollon Plateau were in the San Francisco Peaks (SFP) and the Mogollon Mountains (Gila Wilderness, GIL; Table 1 and Figure 2). On the Mogollon Plateau, we chose GIL as our test site for age-structure-based fire history methods in spruce-fir (>3100 m) because the patches were smaller than at SFP and required less sampling. In the Sky Islands, the Chiricahua Mountains (CHI) and the Pinaleño Mountains (PIN) had the largest potential historical post-stand-replacing fire aspen patches (Figure 2). At PIN, aspen was not present in homogeneous patches; rather, aspen stems were scattered throughout the mixed conifer forest, potentially representing older stand-replacing fire patches that had infilled with conifers. The PIN contains the only spruce-fir forest in the Sky Islands, which we used as the second test site for spruce-fir fire history methods.

Mean elevation of the study sites was 2982 m and tree-ring samples were collected between 2694 m and 3257 m (Table 1). All sites are managed as US Forest Service wilderness areas except PIN, which is closed to the public to protect the endangered Mount Graham red squirrel (*Tamiasciurus hudsonicus grahamensis*). We did not see evidence of logging (e.g.,

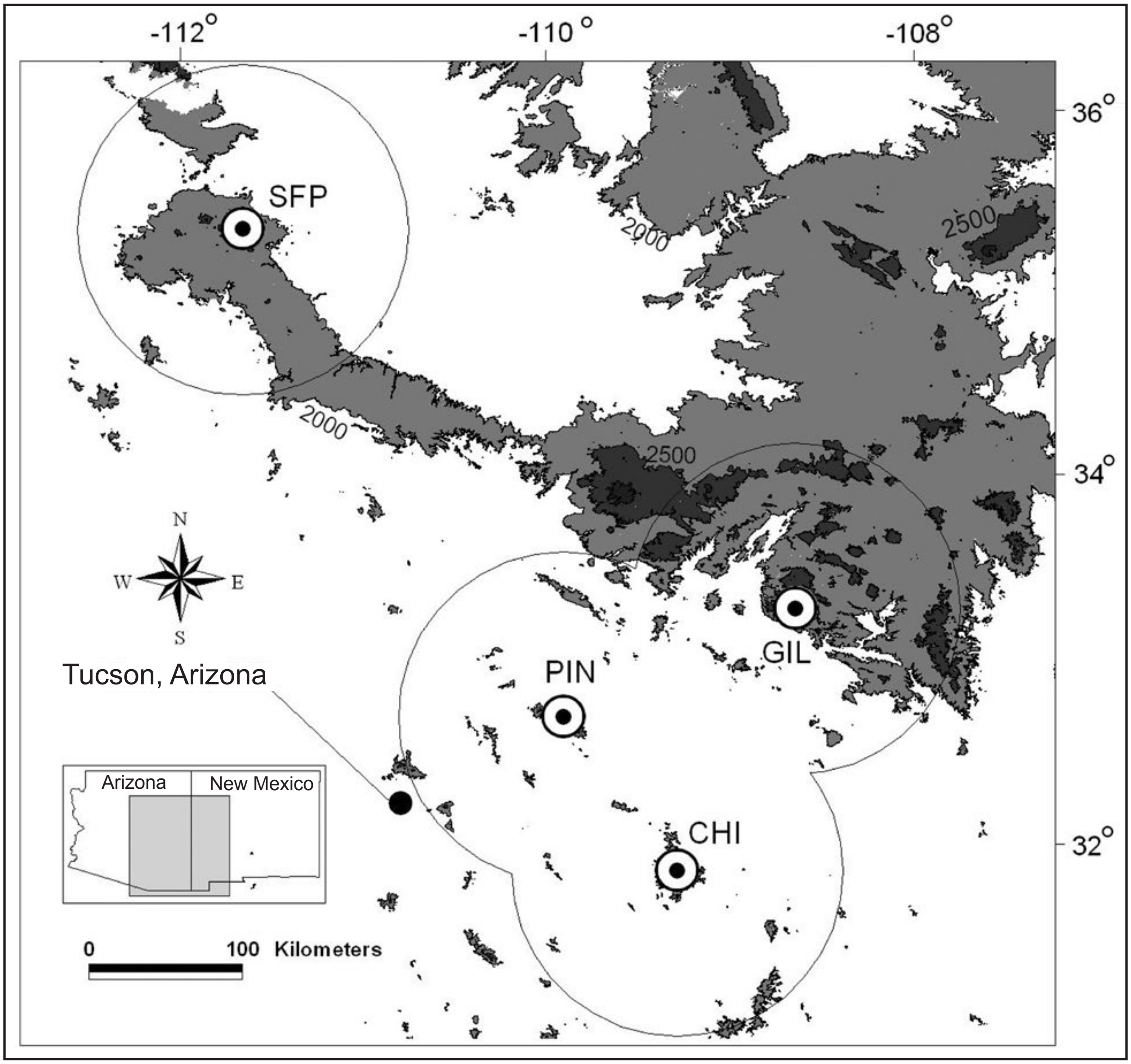

Figure 1. Map of site locations (e.g., SFP) in the Mogollon Plateau and the Madrean Sky Islands of Arizona and New Mexico, USA. Shading indicates major topographic features >2000 m in elevation at 500 m intervals. Large circles indicate the 100 km search radius around the fire history sites used to select recent fires (1984 to 2008) to quantify the size of recent stand-replacing fire patches.

Table 1. Site information for four upper elevation fire history sites from the Mogollon Plateau and Madrean Sky Islands, USA.

Site ID	Site name	Vegetation type[a]	Sampled aspen area (ha)	Sampled spruce-fir area (ha)[b]	Number of plots	Mean sample elevation (m)
CHI	Chiricahua Mountains	MC/S	139	--	26	2856
GIL	Mogollon Mountains	MC/SF	744	1639	32	3060
PIN	Pinaleño Mountains	MC/SF	0[*]	521	33	3057
SFP	San Francisco Peaks	MC/SF	990	--	25	2954

[a] MC = mixed conifer-aspen, SF = spruce-fir, S = spruce
[b] Spruce-fir was only mapped and sampled at two test sites (GIL and PIN).
[*] Distinctive seral aspen patches greater than 5 ha were not present.

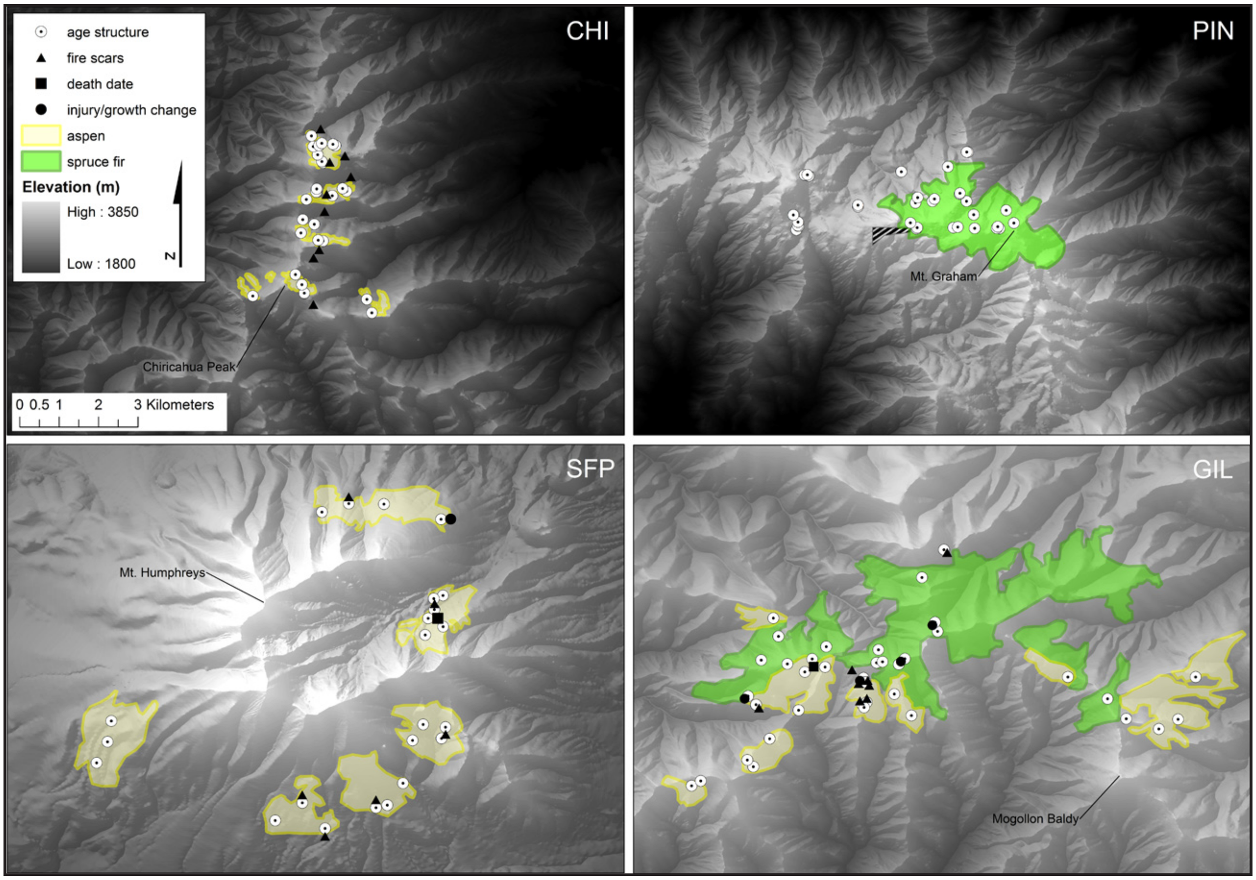

Figure 2. Tree-ring sample locations and analyzed aspen and spruce-fir stands at the study sites in the Chiricahua Mountains (CHI), Pinaleño Mountains (PIN), San Francisco Peaks (SFP), and Gila Wilderness (GIL) of the Mogollon Mountains. Hatched polygon at PIN indicates fire scar sample area from Grissino-Mayer *et al.* (1995).

stumps or skid trails) within the sampled stands. Fire exclusion resulting from late nineteenth century grazing followed by twentieth century fire suppression occurred at all sites, similar to most montane forests in the southwestern US (Dieterich 1980, Bahre 1985, Swetnam and Baisan 1996, Allen *et al.* 2002).

The general climate of the study area is continental with a bimodal precipitation regime. All sites receive an average of 40% to 50% of annual precipitation from summer (July to September) monsoon convective thunderstorms (1910 to 2009; http://www.prism.oregonstate.edu/). Average annual precipitation was similar amongst sites, ranging from 800 mm to 950 mm. Average annual maximum temperature ranged from 12.5 °C to 17 °C and minimum temperature ranged from 0° C to –4.5° C (1910 to 2009; http://www.prism.oregonstate.edu/). All sites receive winter snow, but snowpack varies widely from year to year depending on the winter storm track. The majority of area that burns in the study area occurs during a consistently dry and warm pre-monsoon period that begins in April or May and lasts through June (Barrows 1978). The potential severity and length of the fire season in the high-elevation forests of the region is largely a function of the snowpack and residual moisture that persists into the early summer pre-monsoon period.

The sampled seral quaking aspen stands at all four sites were located adjacent to mixed conifer or spruce-fir forests. The following conifer tree species were observed within and adjacent to the aspen stands, listed in descending order of occurrence: Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), southwestern white pine (*Pinus strobiformis* Engelm.), white fir (*Abies concolor* [Gord. & Glend.] Lindl. Ex Hildebr.), subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.), ponderosa pine, and Rocky Mountain bristlecone pine (*Pinus aristata* Engelm.).

Although all sites contained quaking aspen, there were differences between and within sites. Aspen patches in the two Sky Island sites were smaller than on the Mogollon Plateau (Table 1, Figure 2). This pattern can be partially explained by less land area in the aspen zone (2600 m to 3100 m) at the Sky Island sites (2927 ha in CHI, and 5945 ha in PIN) compared to Mogollon sites (7088 ha in SFP, and 7645 ha in GIL). Within-site differences in vegetation that could affect fire regimes were driven by aspect, with south-facing slopes containing drier, more open forests, and north-facing slopes generally supporting more mesic, denser forests.

Stand-Replacing Fire History Methods—Mixed Conifer-Aspen Forest

Our general sampling methods follow Margolis *et al.* (2007), in which large quaking aspen patches embedded in mesic mixed conifer and spruce-fir forests of the upper Rio Grande Basin were mapped and tree-ring dated with multiple lines of evidence to reconstruct stand-replacing fire patch sizes and dates. The four lines of evidence included 1) quaking aspen age structure, 2) conifer death dates, 3) conifer traumatic resin ducts or ring-width changes, and 4) conifer fire scars. All conifer death dates were bark-ring dates. Bark-ring dates indicate that either bark or other evidence of an intact outer ring (e.g., insect galleries) was present on the samples—this ensures that the outer ring dates are actual tree death dates.

Age structure plots were randomly located within each mapped aspen patch at a minimum density of three to four plots per 100 ha (e.g., SFP in Figure 2). Aspen patches were visually surveyed in the field to ensure plot locations were representative of the stand. Additional plots were added in the field at locations with conifer evidence of fire to verify stand boundaries, or to age potentially older trees (fire survivors) indicated by anomalously large diameter.

Aspen age structure plots had a 10 m fixed radius. Within the plots, we cored the two aspen stems with the greatest diameter at breast height (dbh). Trees were cored at <0.3 m core height until the pith was present in one sample at the plot. In a post-stand-replacing fire aspen stand, sampling two stems per plots at multiple plots within a patch has been shown to be sufficient to determine stand age (Margolis *et al.* 2007). This is because of the immediate asexual regeneration response of aspen following aboveground stem mortality, which creates a distinct recruitment pulse and a single-tiered, even-aged stand (Barnes 1966, Patton and Avant 1970). In upper montane seral aspen stands, subsequent regeneration is relatively rare and the dominant post-fire cohort is easily identified as the stems with largest dbh (Margolis *et al.* 2007). A more intensive sampling design would be necessary to fully describe a multi-cohort age structure, but this was not our goal. Post-fire quaking aspen regeneration can grow up to 1 m in the first year of growth (Jones 1975); thus, <0.3 m core height seems adequate to capture the first year of the aspen regeneration pulse (Margolis *et al.* 2007).

We searched within aspen patches and along the patch boundaries for conifers with potential direct evidence of fire (e.g., fire scars, conifer death dates, and ring-width changes and injuries). Cross sections and partial cross sections were collected with handsaws from remnant conifer logs, living trees, and standing dead snags with intact outer rings. Increment cores were collected from potentially injured live conifers without basal scars. Potential evidence of fire injury included char, scars on the undersides of branches, elevated crown base height, and unilateral loss of branches. Fire scars were not collected at PIN due to the existing fire scar collection located within our study site (Grissino-Mayer *et al.* 1995).

All tree-ring samples were prepared and crossdated according to standard dendrochronological procedure (Stokes and Smiley 1968). To estimate the date of the first year of growth (pith) for age structure increment cores that did not contain the pith ring, we used a concentric circle pith estimator (Applequist 1958). Dates from the four lines of tree-ring evidence were plotted together to determine fire dates (from conifer fire scars, death dates, and tree-ring growth changes and injuries) and stand-replacing fire patches (from age structure of aspen patches).

A mapped aspen patch was determined to represent the minimum extent of a previous stand-replacing burn patch if: 1) the oldest aspen estimated pith dates were associated with (≤5 years following) a fire event recorded by conifer death dates from within the patch or fire scars on surviving trees along the periphery of the patch, and 2) estimated aspen pith dates were part of a site-level (i.e., multi-patch) aspen recruitment pulse. The rarity and poor spatial coverage of fire-scarred trees at some sites (e.g., $n = 6$) prevented the use of percent-scarred filters to categorize and compare relatively widespread versus local fires between sites (Swetnam and Baisan 2003). Instead, we categorized fires recorded by ≥5 conifer samples at a site (e.g., conifer death dates, growth changes or traumatic resin ducts, and fire scars) as likely being more widespread than fires recorded by fewer trees.

Testing Fire History Methods in Spruce-Fir Forest

Within our study area, potential post-stand-replacing fire quaking aspen patches were generally found between 2600 m and 3100 m, and forests above 3100 m were generally dominated by spruce and fir. Pure spruce-fir forests with no living aspen stems would not be expected to contain quaking aspen regeneration following fire, so above 3100 m in this region, past stand-replacing fire patch size and dates cannot be estimated using post-fire aspen patches. We tested the utility of age structure fire history methods in spruce-fir forests at two sites, PIN and GIL. These sites were chosen because the relatively small size of the spruce-fir patches was more manageable for testing the efficacy of the methods. Therefore, the extensive spruce-fir stands at SFP were not sampled.

Aerial photographs and field observations were used to map spruce-fir patches and identify differences in texture, density, color, or differences in tree height, potentially representing fire boundaries (Johnson and Larsen 1991, Agee 1993, Johnson and Gutsell 1994). We were not able to identify any evidence of potential fire boundaries (e.g., discrete changes in canopy height) within the spruce-fir stands at PIN or GIL. Therefore, we treated each spruce-fir stand as a single potential stand-replacing fire patch.

In contrast to the predominance of asexual reproduction in aspen, spruce and fir trees recruit from seed, so the initial post-fire cohort can lag behind the fire date and may be distributed over decades (e.g., Antos and Parish 2002). Subsequent cohorts of these shade-tolerant conifers are able to regenerate under the canopy of the initial post-fire cohort. This multiple-aged structure makes the initial post-fire cohort in spruce-fir more difficult to identify with age or size structure data. We collected age structure samples using similar methods for dating aspen patches (see above), but with two differences to account for the differing fire ecology. First, we doubled the number of trees cored at each plot to include the four trees of largest dbh in order to account for the potentially complex age structure. Conifers were cored as low on the bole as possible and angled down to intersect the root crown and capture the earliest years of growth.

The second difference from the aspen age structure methodology was in the criterion to qualify as a stand-replacing fire patch. A spruce-fir patch was determined to be a post-stand-replacing fire patch if the oldest estimated conifer pith dates were ≤10 years following a fire recorded by conifer death dates from within the patch or by fire scars on the periphery of the patch. We increased the cut-off criteria to 10 yr (compared to 5 yr for aspen) to account for potentially lagged seedling recruitment (compared to immediate asexual regeneration in aspen). A 10-year lag window is likely conservative, given reports of greater than 50-year lags for subalpine forest regeneration following stand-replacing fire (Stahelin 1943). Because of relatively high fire frequency recorded by fire scars in some of the mixed conifer forests immediately below the spruce-fir stands, we determined that a 10-year lag would help to avoid spurious matches between fire scar dates and age structure that could be interpreted as stand-replacing fire dates. All tree-ring samples were collected in 2003 and 2004.

Historical Stand-Replacing Fire Patch Size

The aspen and spruce-fir patches that were dated to historical stand-replacing fires were used to derive minimum estimates of historical stand-replacing fire patch sizes. Patch area was calculated with a geographic information system (GIS). This data set provides the first estimate of historical stand-replacing burn patch sizes within two elevation and vegetation ranges at our study sites, including: 1) the aspen zone (2600 m to 3100 m) and 2) the spruce-fir zone (>3100 m).

RESULTS

Tree-ring dates from 178 aspen stems and 139 conifers were used to reconstruct upper montane fire history, including stand-replacing fire patch dates and sizes (Tables 2 and 3, Figures 3-6). Annually dated, direct conifer evidence of fire (e.g., fire scars and tree death dates) was used to reconstruct 77 new fires in addition to the existing fire dates (from Grissino-Mayer *et al.* 1995) for PIN. Across the four sites, 100 fires occurring on 87 unique fire dates were analyzed (1623 to 1904; Table 3). Twenty five percent of the fires (n = 25) were recorded by ≥5 conifers (including fire scars) at a site. An average of 59 % of all sampled aspen regenerated within five years after fire, ranging from 27 % to 89 % among sites. Three fires (1685 in PIN, 1879 in SFP, and 1904 in GIL) met our criteria for stand-replacing fire within mapped aspen or spruce-fir patches (Figures 4-6). Evidence of stand-replacing fire included aspen and conifer recruitment pulses, coincident conifer death and fire scar dates, and a lack of trees that survived (pre-date) the fire.

Table 2. Number of trees with crossdated tree-ring samples used to reconstruct fire history in the Mogollon Plateau and Madrean Sky Islands, USA.

Site ID	Aspen age structure	Conifer age structure	Conifer fire scar	Conifer death date	Conifer growth change or injury	Total
CHI	44	0	26	0	6	76
GIL	58	44	10	1	6	119
PIN	31	25	12*	0	0	68
SFP	45	0	6	1	2	54
Total	178	69	54	2	14	317

* Data from Grissino-Mayer *et al.* 1995 (PIN).

Table 3. Stand-replacing fires, fires recorded by ≥5 trees, and all additional fires reconstructed from multiple lines of tree-ring evidence.

Site	Stand-replacing fire dates	Fires recorded by ≥5 trees	All additional fires
CHI		1685, 1711, 1725, 1748, 1763, 1773, 1785, 1817, 1826, 1841, 1851, 1868, 1877, 1886	1654, 1661, 1688, 1697, 1698, 1700, 1701, 1703, 1709, 1716, 1721, 1723, 1727, 1733, 1737, 1739, 1749, 1752, 1760, 1765, 1775, 1779, 1787, 1789, 1794, 1798, 1800, 1805, 1806, 1807, 1818, 1822, 1835, 1838, 1840, 1848, 1849, 1859, 1863, 1875, 1883, 1894, 1903, 1904
GIL	1904	1904, 1748, 1773	1716, 1765
PIN*	1685	1685, 1773, 1785, 1819, 1842, 1858, 1871	1623, 1648, 1668, 1670, 1674, 1687, 1691, 1696, 1709, 1719, 1733, 1745, 1748, 1752, 1760, 1847
SFP	1879	1879	1752, 1773, 1809, 1818, 1836, 1840, 1847, 1851, 1855, 1857, 1860, 1863, 1876

* Fire scar data from Grissino-Mayer *et al.* 1995 (PIN).

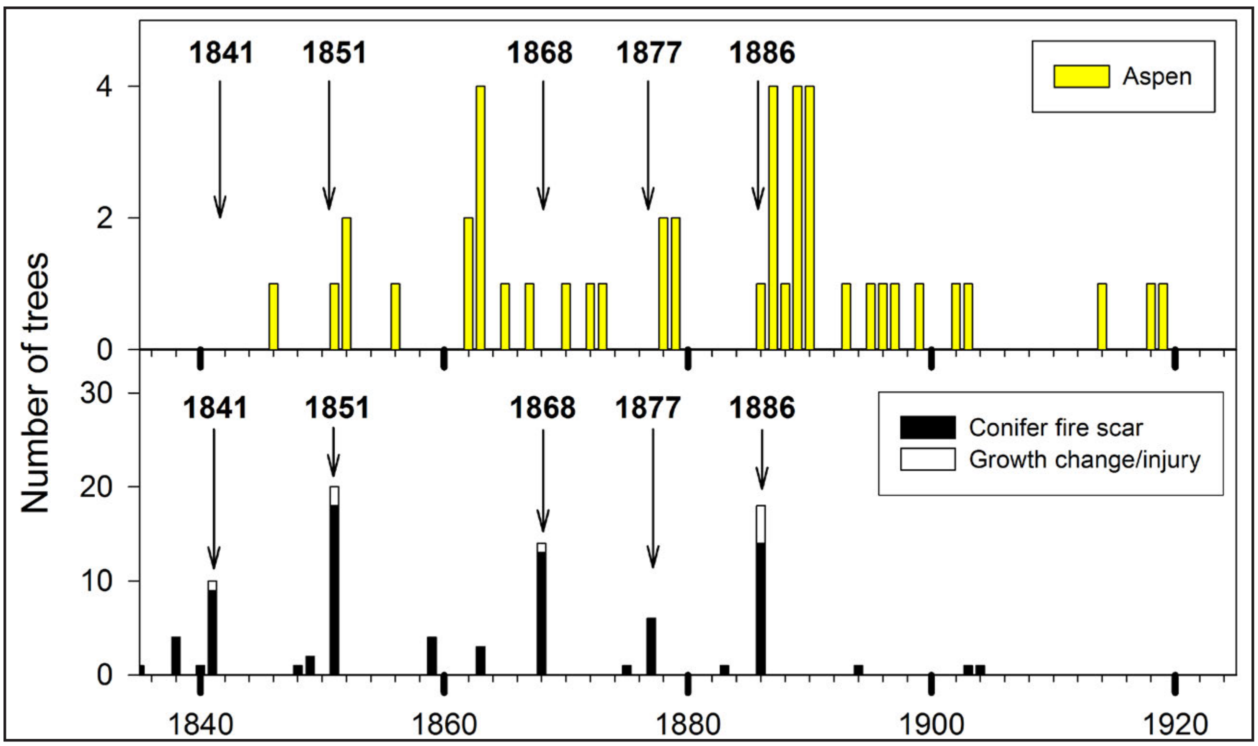

Figure 3. Chiricahua Mountains (CHI) estimated aspen pith dates (top) and direct conifer evidence of fire (bottom) in 1-year classes used to reconstruct fire history in the upper elevation forests. Years (e.g., 1886) indicate annually dated fire events recorded by ≥5 conifer trees, including fire scars.

Chiricahua Mountains

Eight small quaking aspen patches were mapped at CHI, totaling 139 ha (Table 1, Figure 2). No single post-stand-replacing fire quaking aspen cohort was present at CHI, but 89% of the aspen stems regenerated within five years after a fire (Figure 3). Surface fires recorded by conifers on south-facing slopes adjacent to the aspen stands were relatively

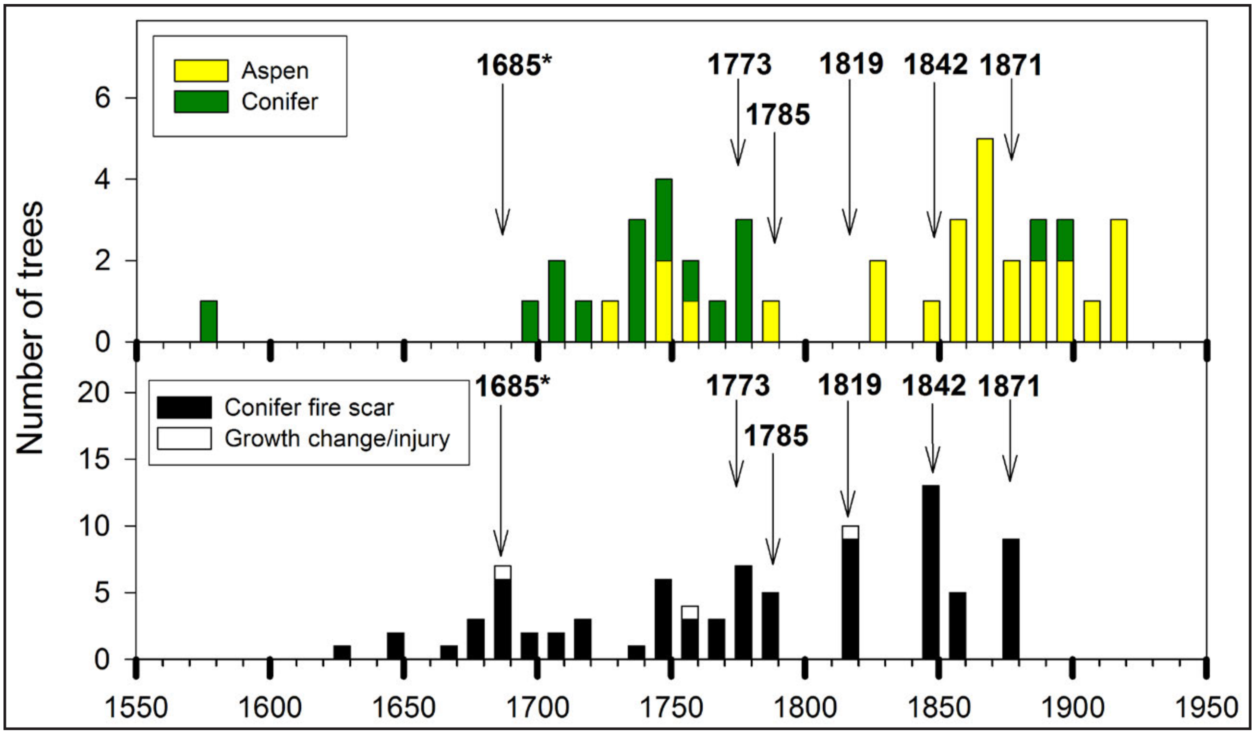

Figure 4. Pinaleño Mountains (PIN) estimated pith dates (top) and direct conifer evidence of fire (bottom) in 10-year classes used to reconstruct fire history in the upper elevation forests. Years (e.g., 1685) indicate annually dated fire events recorded by ≥5 conifer trees, including fire scars. * Indicates stand-replacing fire date. Fire scar data from Grissino-Mayer *et al.* (1995).

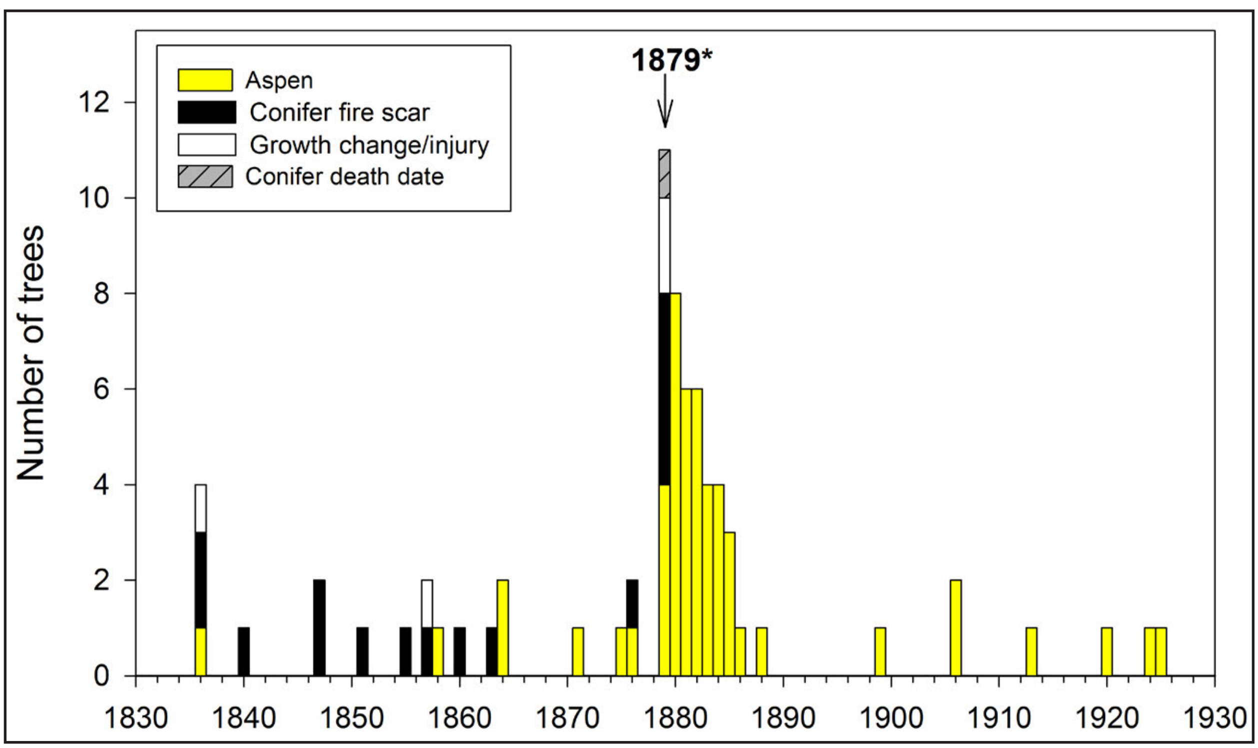

Figure 5. San Francisco Peaks (SFP) estimated aspen pith dates and direct conifer evidence of fire in 1-year classes used to reconstruct fire history in the upper montane forests. Years (e.g., 1879) indicate annually dated fire events recorded by ≥5 conifer trees, including fire scars. * Indicates stand-replacing fire date.

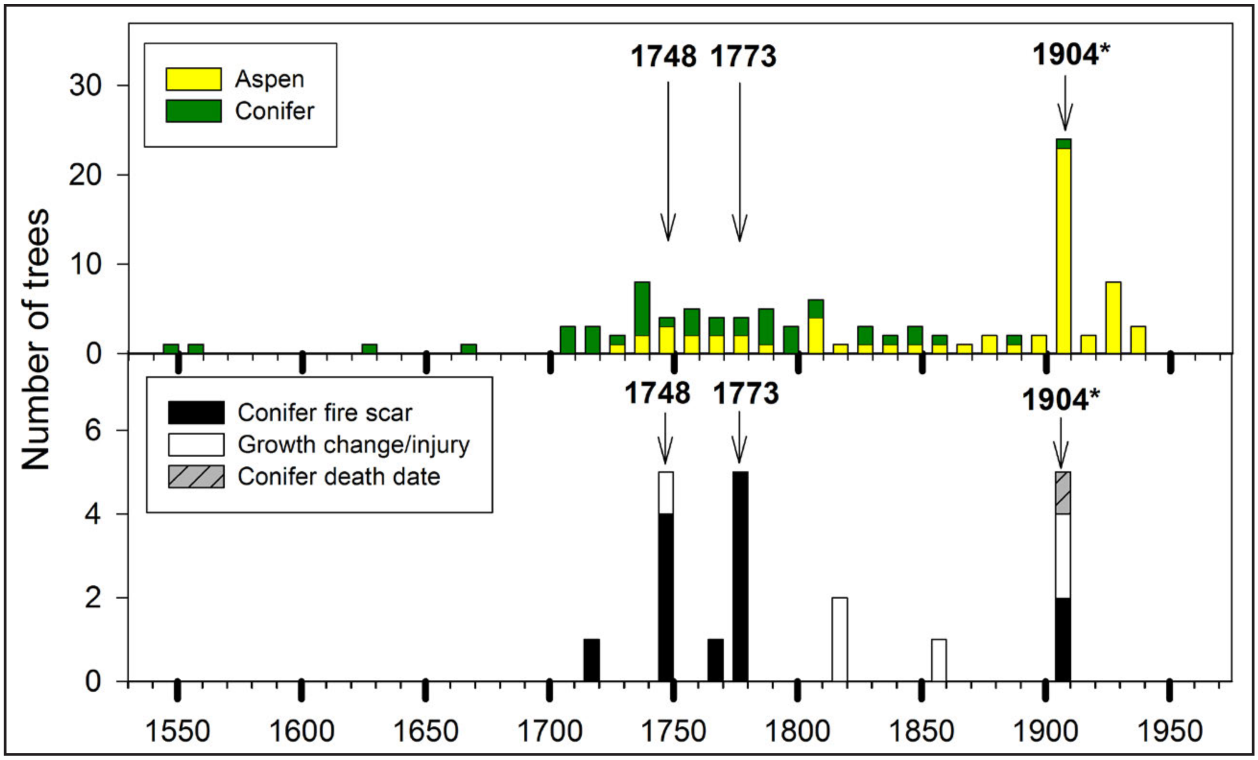

Figure 6. Mogollon Mountains (GIL) estimated pith dates (top) and direct conifer evidence of fire (bottom) in 10-year classes used to reconstruct fire history in the upper montane forests. Years (e.g., 1904) indicate annually dated fire events recorded by ≥5 trees, including fire scars. * Indicates stand-replacing fire dates.

frequent prior to *circa* 1900 (mean fire interval from 1654 to 1904 for all fires was 4.5 yr; Table 3).

The mapped aspen patches at CHI were not post-stand-replacing fire patches based on our criteria. The age structure of the dominant aspen within each patch was multi-aged, with some trees surviving (pre-dating) multiple fire events. For example, aspen from the 1886 post-fire cohort were scattered throughout multiple patches, but were often located adjacent to older aspen stems (e.g., 1851 post-fire regeneration) that survived the 1886 fire.

Pinaleño Mountains

The combined age structure of the multiple, small aspen groups (5 to 10 stems) scattered throughout the mixed conifer forest showed no evidence of a single, widespread, post-fire cohort (Figure 4). Only 27% of the dominant aspen at PIN regenerated within 5 years after a fire. Many aspen pre-dated (survived) fires recorded by multiple conifers as fire scars (e.g., 1871 fire), with the oldest living aspen dating to 1724 (estimated pith date).

Without post-fire aspen cohorts or large contiguous patches of seral, post-fire quaking aspen at PIN, the spruce-fir stand was the best potential evidence of past stand-replacing fire. The oldest tree (Engelmann spruce; 1692 estimated pith date) in the spruce-fir stand regenerated within 10 years after the 1685 fire that scarred all recording trees in the adjacent mixed conifer-aspen zone (Figure 4). The one tree that pre-dated the 1685 fire was a Douglas-fir located on the edge of the spruce-fir zone. These data met our criteria for stand-replacing fire in the spruce-fir zone at PIN in 1685.

San Francisco Peaks

Seventy-one percent of the dominant aspen at SFP regenerated within five years after a fire. Multiple lines of tree-ring evidence indicate that the 1879 fire was stand-replacing in some of the mapped patches (Figure 5). A distinct and immediate aspen recruitment pulse began in 1879, accounting for 63% of the sampled aspen. This site-level aspen age structure, dominated by a single post-fire aspen cohort, was different from the two Sky Island sites that had no dominant aspen cohort (CHI and PIN, Figures 3 and 4). The few aspen at SFP that pre-date 1879 were from the southeastern part of the site where there was no fire-scar evidence of the 1879 fire (Figures 2 and 5). In total, tree-ring evidence of the 1879 fire was present in all but one aspen patch.

The seral, post-stand-replacing fire aspen patches at SFP were located on the north-facing slopes and had the largest mean reconstructed stand-replacing fire patch size of all of the sites (145 ha). The drier, south-facing slopes contained conifers with multiple fire scars within the aspen stands. The ten fires recorded between 1836 and 1879 were all recorded by fire-scarred conifers on the south slope (Figures 2 and 5). This frequent fire regime ($MFI_{\text{All fires}}$ = 4.8 yr) that scarred, but did not kill, conifers within the south-facing aspen stands differed from the post-stand-replacing fire aspen patches, with no surviving conifers, on the north-facing slopes at SFP.

Mogollon Mountains (Gila Wilderness)

The aspen age structure at GIL was dominated by a post-1904 fire recruitment pulse (Figure 6). No sampled trees from within the mapped aspen patches survived the 1904 fire. These homogenous, even-aged aspen patches contained fire-killed Douglas-fir that died in 1904. Based on this evidence, all mapped aspen patches at GIL (totaling 744 ha) were determined to be stand-replacing fire patches from the 1904 fire. The aspen stems that predated the 1904 fire were located in the spruce-fir stands as scattered, co-dominant stems, some of which were >250 years old. A synchronous recruitment pulse was not evident from these old aspen. Overall, 42% of the sampled aspen at GIL regenerated within five years after a fire.

No direct evidence of fire (e.g., charred wood or fire scars) was observed within the spruce-fir stands at GIL. Relatively continuous conifer regeneration was recorded in the decades from 1700 to 1910, and the oldest individual (Engelmann spruce) in the spruce-fir patches dated to 1707. Multiple spruce trees were older than the oldest crossdated fire scar (1716) recorded adjacent to the spruce-fir patches (Figure 6). Therefore, the sampled age structure did not meet our criteria to be a post-fire recruitment cohort. There was no evidence that the 1904 fire, which our results suggest burned with stand-replacing severity in adjacent mixed conifer-aspen forests, burned into the spruce-fir zone.

Historical Stand-Replacing Fire Patch Size

We derived historical stand-replacing fire patch size estimates from the 10 tree-ring dated post-stand-replacing fire aspen patches (1879 to 1904) and the one post-stand-replacing fire spruce-fir patch (1685 fire; Table 4). Fires at three of the four sites (GIL, PIN, and SFP) had stand-replacing fire patches >200 ha. The maximum reconstructed historical stand-replacing fire patch size was 286 ha in the mixed conifer-aspen zone (2600 m to 3100 m) and 521 ha in the spruce-fir zone (>3100 m; Table 4).

DISCUSSION

Historical Stand-Replacing Fire

We found evidence of historical stand-replacing fire in upper elevation forests

Table 4. Historical and recent stand-replacing fire patch area statistics. Historical burn patch areas derived from combined tree-ring reconstructed aspen and spruce-fir stand-replacing fire patches. Recent burn patch area derived from fire severity maps (1984 to 2008, n = 352 fires). The conservative estimate of recent stand-replacing fire patch size includes only high-severity patches (H), and a more inclusive estimate includes high- and moderate-severity patches (H+M). Recent data only include patches >30 ha, equal to the smallest reconstructed historical stand-replacing fire patch.

	Historical burn patches		Recent burn patches					
	2600 m to 3100 m	>3100 m	2600 m to 3100 m		>3100 m		All elevations	
	Aspen	Spruce-fir	H	H+M	H	H+M	H	H+M
Count	10	1	64	85	1	2	204	675
Mean (ha)	110	521	129	206	33	110	136	233
Median (ha)	63	521	80	86	33	110	65	74
Standard deviation (ha)	89	--	134	300	--	70	204	500
Minimum (ha)	30	521	32	31	33	60	32	31
Maximum (ha)	286	521	637	1 540	33	159	1 929	5 136
Sum (ha)	1 104	521	8 251	17 507	33	219	27 810	157 482

(>2600 m) at three of the four sites. Fires with multiple large (>100 ha) stand-replacing fire patches were tree-ring dated at the two Mogollon Plateau sites using quaking aspen age structure and associated direct conifer evidence of fire (1904 in GIL and 1879 in SFP). Aldo Leopold (1922), while on a fire assignment in the Gila Wilderness, referenced a 1904 fire in the Mogollon Mountains. Abolt (1997) identified a widespread fire with a stand-replacing component in 1904 in the Mogollon Mountains from tree-rings and historical documents. In the San Francisco Peaks, Heinlein *et al.* (2005) recorded a fire in 1879 in lower elevation ponderosa pine-mixed conifer forests using fire scars, but does not report evidence of stand replacement. Historical photographs taken in 1910 at SFP show standing dead and downed trees in the spruce-fir and mixed conifer-aspen zones that likely resulted from a fire with large (estimated >500 ha) stand-replacing patches in the late nineteenth century (http://www.rmrs.nau.edu/imagedb/viewrec.shtml?id=22141&colid=fv).

We were able to associate spruce-fir age structure with direct conifer evidence of fire (i.e., fire scars) at one of the spruce-fir fire history test sites (PIN). The age structure data we collected, and prior sampling of more than 290 trees by Grissino-Mayer *et al.* (1995) from the large (521 ha) spruce-fir stand at PIN, support the hypothesis of a stand-replacing fire in 1685 (Swetnam *et al.* 2009, but see Stromberg and Patten 1991). Margolis and Balmat (2009) reconstructed a 1200 ha stand-replacing fire patch in the spruce-fir forests of the Santa Fe Watershed, New Mexico, also in 1685. This year was extremely dry (−5.0 reconstructed Palmer Drought Severity Index: Cook *et al.* 2004) and a common fire year throughout the southwestern US (Swetnam and Baisan 2003). Thus, it is plausible that the typically mesic spruce-fir zone at PIN could have been dry enough in 1685 to burn.

Frequent Fire and Quaking Aspen

We did not find evidence of past stand-replacing fire in the sampled aspen stands at CHI. Although 89 % of the aspen stems at this site regenerated within five years after reconstructed fires, the mapped aspen patches were multi-aged. This indicates that some aspen stems survived multiple fires, while other aspen in the same patch were top-killed by the same fires and then regenerated by sprouting.

We found direct evidence of repeated low-severity fire (e.g., conifers with multiple fire scars) adjacent to aspen stands at CHI and within the mixed conifer-aspen forests of PIN. Frequent fire occurred at these two upper montane Sky Island sites prior to *circa* 1900: MFI-$\text{PIN}_{\text{All fires}}$ = 10.9 yr (1685 to 1871), approximately 150 ha sample area (Grissino-Mayer *et al.* 1995), and MFI-$\text{CHI}_{\text{All fires}}$ = 4.4 yr (1654 to 1904), approximately 250 ha sample area. This history of frequent fire may have prevented sufficient fuel accumulation to sustain stand-replacing fire. This suggests that the cessation of fire for over 120 years due to late nineteenth century grazing and twentieth century fire suppression may be a cause of fuel structure changes and buildup that contributed to the recent occurrence of stand-replacing fires in the mixed conifer-aspen forests at these Sky Island sites (Swetnam *et al.* 2009).

Similar evidence of frequent low-severity fire (i.e., logs and living conifers with multiple scars) was present within and adjacent to the aspen stands on the south slope of SFP (Figures 2 and 5). The lower borders of these aspen stands are connected with ponderosa pine-mixed conifer forests that historically burned with frequent low-severity fire (e.g., Heinlein *et al.* 2005). Based on this evidence of repeated surface fire in aspen on south aspects at SFP, it is likely that the present stand structure, dominated by >20 m tall, mature aspen stems (>120 years old) may be in part an artifact of fire exclusion. These fire-sensitive aspen stems would have been historically exposed to frequent fire, thus the same stands likely looked very different in the nineteenth century. One hypothesis is that they were smaller diameter aspen "thickets" that were top-killed and regenerated after each fire (Maini 1960, Allen 1989). Alternatively, some larger diameter stems at the center of the stand may have been protected from being girdled by fire, creating a multi-cohort age and stand structure. Binkley *et al.* (2006) proposed a similar hypothesis of altered quaking aspen stand-structure in response to twentieth century fire exclusion on the Kaibab Plateau in north-central Arizona. The following hypothesis should be tested with future research: the age and stand structures of quaking aspen that historically experienced frequent fire have shifted from young or multi-aged, dense stands, to the current open structure dominated by a single mature cohort, largely due to >120 years of fire exclusion.

Spruce-Fir Fire History Challenges

The lack of burn boundaries within the spruce-fir stands at our two test sites (PIN and GIL) differs from higher latitude, Rocky Mountain landscapes where old stand-replacing fire patch boundaries are visible as obvious stand-height and structural differences that are used to map and date historical crown fires (e.g., Kipfmueller and Baker 2000, Sibold *et al.* 2006). Fulé *et al.* (2003) reported a similar lack of fire-related patch boundaries identifiable with remote sensing data in mixed conifer, aspen, and spruce-fir forests of the north rim of the Grand Canyon, Arizona. The lack of old fire boundaries within the spruce-fir zone of the current study, and on the north rim of the Grand Canyon may suggest that, in these spruce-fir forests, large crown fire patches were not as common within recent centuries as they were in the Rocky Mountains.

The inconclusive evidence of stand-replacing fire in the spruce-fir zone at GIL was possibly due to an insufficient number of tree ages to determine the complex and relatively old (>300 yr) age structure, and the relative scarcity of old (pre-1700 AD) fire scar material in this high-elevation forest type (Figure 6). Age-structure transects with a higher density of samples may be necessary to determine patch age in old (>300 years old) southwestern US spruce-fir forests. Repeated, sample-intensive age structure transects distributed throughout the mapped stands may be the best method to confidently evaluate the age structure of old spruce-fir forests in this region (e.g., Margolis and Balmat 2009). The number of trees sampled for age structure could be adjusted based on the estimated age of the stand (e.g., <150 yr old, > 250 yr old) so that only the oldest stands would require intensive sampling to overcome these challenges.

Multiple mapping and age-structure sampling methods should be tested on known and potential post-fire spruce-fir stands. The subalpine forests of the upper Rio Grande Basin or at SFP could be used to select test sites because there are large spruce-fir stands adjacent to large, post-fire aspen patches from historically documented nineteenth century fires (e. g., Santa Fe Ski Basin, New Mexico). Dating and mapping these sub-alpine conifer stands is the best available method to improve the accuracy of estimates of historical stand-replacing fire area in the highest elevations (>3100 m) in the southwestern US. These data are necessary to estimate fire frequency statistics (e.g., fire cycle or natural fire rotation) of the stand-replacing fire regimes in the upper montane mesic mixed conifer-aspen and spruce-fir forests of the region.

Historical Stand-Replacing Burn Patch Size

The occurrence of historical stand-replacing fire patches >200 ha at three of the four upper elevation sites suggest that recent large (200 ha to 500 ha) stand-replacing patches are within the historical range of variability in upper elevation forests (>2600 m) of the southwestern US outside of the southern Rocky Mountains. Based on our reconstructions, stand-replacing fire patches as large as 286 ha historically occurred in the mixed conifer-aspen zone, and patches as large as 521 ha historically occurred in the spruce-fir zone. Within these upper elevation forests, it is possible that older, larger stand-replacing fire patches were burned over by the late nineteenth century fires, or that such patches were re-colonized by mixed conifer species instead of aspen. We did not observe obvious even-aged mixed conifer stands with abundant fire-killed, remnant conifer logs or snags at our study sites that might indicate evidence of past stand-replacing fire. However, extensive (>500 ha) mixed conifer and spruce-fir patches exist in the region and could be systematically sampled to determine whether they regenerated following stand-replacing fire.

The largest historical stand-replacing fire patch we reconstructed was in the spruce-fir zone at PIN (521 ha). Historical photographs at SFP, discovered after our sampling was completed, illustrate large late-nineteenth century stand-replacing fire patches (estimated >500 ha) in the spruce-fir zone (http://www.rmrs.nau.edu/imagedb/viewrec.shtml?id=22141&colid=fv). In the southern Rocky Mountains of New Mexico, Margolis and Balmat (2009) reconstructed a 1200 ha stand-replacing fire patch in spruce-fir forest. Thus, documentary and tree-ring evidence at multiple sites in the southwestern US indicates the potential for large (500 ha to >1000 ha) stand-replacing fire patches in spruce-fir forest.

Recent Stand-Replacing Burn Patch Size

All four of our study sites have recently burned with high-severity patches. As an ancillary investigation to summarize the recent (1984 to 2008) fires, we quantified patch sizes of 352 fires >404 ha with high- or moderate-severity patches within 100 km of the four fire history study sites (Figure 1, http://www.mtbs.gov/index.html). We stratified the recent burn severity patch size data by elevation and vegetation type and fire severity to produce six subsets: 1) high severity with no elevation limit, 2) high plus moderate severity with no elevation limit, 3) high severity 2600 m to 3100 m, 4) high plus moderate severity 2600 m to 3100 m, 5) high severity >3100 m, and 6) high plus moderate severity >3100 m. The elevation ranges are the same used to categorize the upper elevation fire reconstructions. The subset with no elevation limit includes lower elevation, pine-dominant oak or shrub vegetation. Recent patch sizes were limited to ≥30 ha, equal to the minimum reconstructed stand-replacing fire patch size. Data from all sites were pooled. We were most interested in the largest patches since they arguably have the greatest ecological effects.

Significant direct and delayed mortality from crown scorch and insect attack has been documented in moderate-severity burn patches in recent fires (McHugh and Kolb 2003). Based on an assumption that a substantial percentage of the trees in moderate-severity burn patches die, high- and moderate-severity patches were combined in one subset of the data. We posit that the actual area of fire-related tree mortality (i.e., stand replacement) was probably somewhere between the "high severity" and "high plus moderate severity" patch size estimates.

The largest recent stand-replacing fire patch size with no elevation limit was 1929 ha (high severity) and 5136 ha (high plus moderate severity), with 37 patches >1000 ha (2 high severity and 35 high plus moderate severity; Table 4). In the mixed conifer-aspen zone (2600 m to 3100 m), the largest recent high-severity patch was 637 ha, and the largest high- plus moderate-severity patch was 1540 ha. Above 3100 m, in the spruce-fir zone, the largest recent high-severity patch was 33 ha, and the largest high- plus moderate-severity patch was 159 ha.

Direct comparison between recent and historical stand-replacing fire patch sizes are challenging. Due to reasons discussed above, our historical estimates are likely conservative estimates of stand-replacing patch size. Thus, we cannot confidently test whether the largest recent high- or moderate-severity patches are larger than have occurred in past fires. However, given these limitations, the data suggest that recent high- (or moderate-) severity patches that are smaller than the historical estimates (maximum reconstructed patch size, 286 ha in mixed conifer-aspen forest and 521 ha in spruce-fir forest) are likely within the historic range of variability.

In summary, historical fire regimes at multiple upper elevation (>2600 m) mixed conifer-aspen and spruce-fir sites on the Mogollon Plateau and Madrean Sky Islands included large (>200 ha) stand-replacing fire patches. Aspen recruitment was historically associated with fire, with an average of 59% of the dominant aspen stems regenerating within five years after fire (ranging from 27% to 89% among sites). In the drier portions of the mixed conifer-aspen sites, the cessation of historically frequent fires for the last 130 years has likely altered the current aspen age and stand structures. Tree-ring and photographic evidence of historical stand-replacing fire in the spruce-fir zone indicates that recent fires that burned with high severity in this forest type at the study sites (e.g., 2004 Nuttall Fire at PIN) are rare events, but not unprecedented. Based on the reconstructed estimate, recent stand-replacing fire patches as large as 286 ha in the mixed conifer-aspen zone and 521 ha in the spruce-fir zone may be within the historic range of variability and should be expected in future fires, particularly when considering predictions of a warmer and drier climate in the southwestern US (e.g., Seager *et al.* 2007).

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EVERYTHING YOU WANTED TO KNOW ABOUT WILDLAND FIRES IN FORESTS BUT WERE AFRAID TO ASK: LESSONS LEARNED, WAYS FORWARD

Salmon August fire in the Marble Mountains, California (L. Ruediger)

Dominick A. DellaSala, Ph. D

Chief Scientist, Geos Institute

Timothy Ingalsbee, Ph. D,

Fire Fighters United for Safety, Ethics, and Ecology

Chad T. Hanson, Ph. D

John Muir Project

March 30, 2018

EXECUTIVE SUMMARY

Wildfires are a fact of life for westerners. They mark the beginning of the spring season and have been a keystone architect of biodiverse ecosystems for millennia. While wildfires are not eco-catastrophes, they are a health concern, evoke public fear-of-fire exploited by decision makers seeking to push through anti-environmental policies, and generate conflicts over the best ways to coexist with this force of Nature that is not going away (nor should it), no matter how hard we try. This white paper summarizes some of the latest science around top-line wildfire issues, including areas of scientific agreement, disagreement, and ways to coexist with wildfire. It is a synopsis of current literature written for a lay audience and focused on six major fire topics:

1. Are wildfires ecological catastrophes?
2. Are acres burning increasing in forested areas?
3. Is high severity fire within large fire complexes (so called “mega-fires”) increasing?
4. What’s driving the recent increase in burned acres?
5. Does “active management” reduce wildfire occurrence or intensity?
6. Will more wildfire suppression spending make us safer?

Key findings

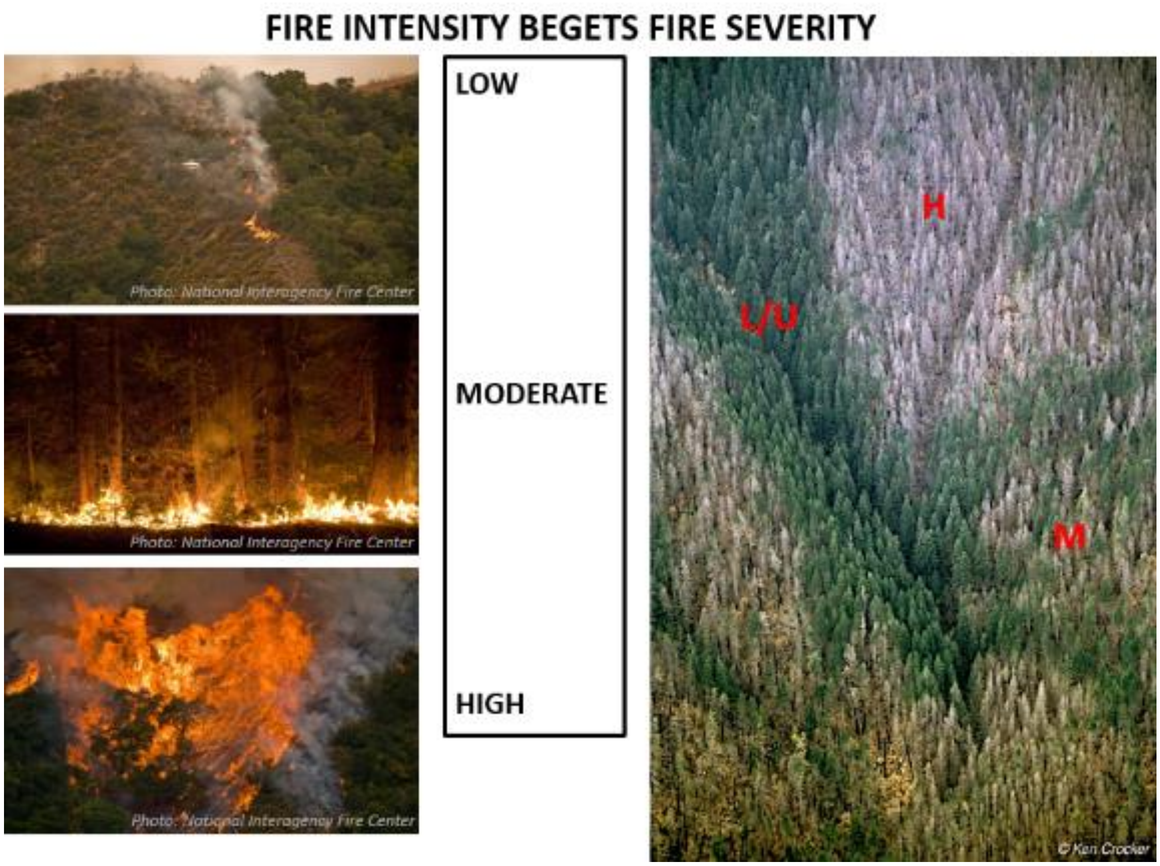
- Large wildland fire complexes, including patches of high severity fire, generate critical ecological pulses of dead trees (biological legacies) that are associated with extraordinary levels of biodiversity under-appreciated by most.
- Using long historical timelines, wildfire acres are currently at historical lows, but have been increasing in recent decades due mainly to three factors: (1) climate change; (2) human-caused fire ignitions (including suppression firing operations such as burnout and backfires); and (3) conversion of fire-resilient native forests to flammable plantations that experience relatively more high fire severity fire.
- Throwing more money at fire suppression will not abate fire concerns as more and more homes are built in indefensible places and are not designed or built with fire-resistant materials.
- Post-fire logging and associated activities (including roads) are unequivocally damaging to fire-rejuvenated forests and related aquatic ecosystems.
- Thinning small trees and prescribed burning can lower fire intensity at the stand level if done properly but this has significant limitations and ecological consequences given the scale of the perceived need and a changing climate.
- The most effective pathway to fire coexistence is to: (1) limit ex-urban sprawl through land-use zoning; (2) lower existing home ignition factors by working from the home-out with vegetation management and home retrofitting (defensible space), instead of the wildlands-in

(logging); (3) thin small trees and prescribe burn in ecologically appropriate settings (e.g., flammable plantations) while prioritizing wildland fire use in most forests away from homes; (4) store more carbon in ecosystems by protecting public forests and incentivizing carbon stewardship on non-federal lands; and (5) shift to a low-carbon economy as quickly as possible. Anything else will not achieve desired results to scale.

Issue 1: Are Wildfires "Catastrophic" or "Disastrous" Events?

Background

Large landscape wildfires are most often referred to as catastrophic "mass fires" or "megafires." Demonizing wildfires has placed this natural process in the same conversation as hurricanes and floods. Such disaster-speak and presumed logging remedies are now inculcated in the "Wildfire **Disaster** Funding Act" (emphasis added) recently passed by Congress as part of federal omnibus appropriations that also included rollbacks to forest protections. But what really goes on after a wildfire may be surprising in terms of the high biodiversity and rejuvenation capacity of forests after large fires, including severe ones.



In general, fire effects are the result of heat energy released during a fire (fire intensity – left photos) and resulting effects on ecosystems (fire severity, right). Most large fires (right) produce a mosaic of burn severity effects on vegetation (H-high severity, M-moderate, U-unburned, L-low). Fire-mediated landscape heterogeneity is habitat for a diverse assortment of species distributed across the successional gradient (new to old forests) and has been referred to as “pyrodiversity begets biodiversity” (see below)[1]. Note – in some cases a fast-moving high-intensity “running” surface fire can produce low severity effects, while a slow-moving low intensity “creeping” fire can produce high severity effects (e.g., smoldering piles of slash or logs).

Issue 2: Are Total Wildfire Acres Burning Increasing (independent of severity)?

Background

Nearly every fire season, the news media and politicians announce another “unprecedented” wildfire season. Such proclamations are incorrectly based on comparisons of contemporary wildfire acres to a recent historical timeline. This has been widely criticized in the scientific literature as the “shifting baseline perspective” (i.e., when a baseline is shifted to a more recent historical time period)[2]. Importantly, in the early part of the 20th century during a warm climatic cycle (Pacific Decadal Oscillation - PDO), wildfire acres were at least five times more abundant than today. A mid-century cool down accompanied by industrial fire suppression resulted in a substantial decline in acres burning[3]. The current warm period is associated with a recent increase in both acres burning and fire suppression (see below). In other words, wildfire activity tracks broad-scale climatic phenomenon (top-down drivers) that also influence fire suppression efficacy.

[1] DellaSala, D.A., and C.T. Hanson. 2015. The ecological importance of mixed-severity fires: nature’s phoenix. Elsevier: Boston.
[2] See Jackson, B.C., et al. 2011. Shifting baselines. Island Press: DC.
[3] For an excellent historical resource read NY Times Best Seller, Timothy Egan’s “The Big Burn.” Mariner Books: NY.

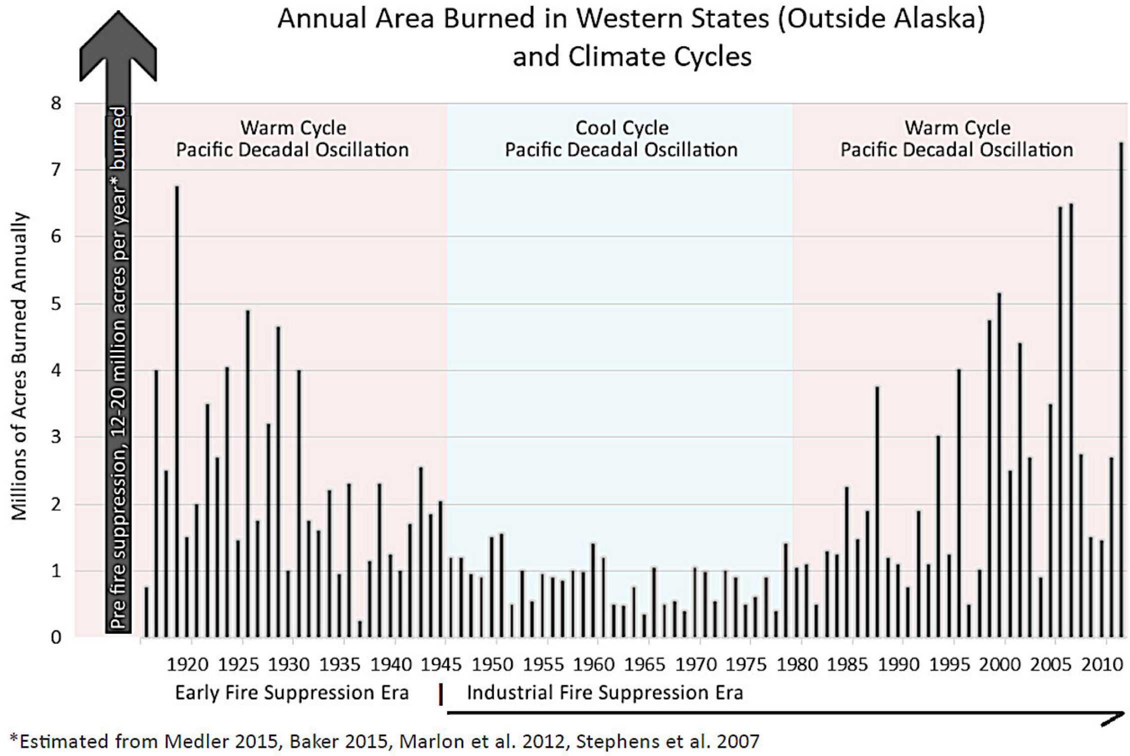


Figure interpretation caveats: prior to 1984, standardized datasets are difficult to obtain. Contemporary wildfires also have a strong back-burning influence not prevalent in historical times–i.e., errors in estimation exist on both ends of the wildfire acreage continuum. However, historical accounts (including General Land Office records and pollen-sediment core analyses) confirm very active fire seasons in the early part of the 20th century and before[4] (Figure compliments of John Muir Project).

Areas of Agreement

Fewer wildfire acres burning in forests today compared to the early 20th century has resulted in what many are calling a wildland fire deficit[5], which may seem as a surprise given fire hyperbole. The main exception to this deficit is southern California chaparral and shrub-steppe communities (too much human-caused fire is leading to ecosystem type conversions).

Areas of Disagreement

Current science debate is focused mainly on what is the best way for putting fire (i.e., "the right fire" "good fire") back on the landscape in order to restore wildland fire-forest relationships.

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[4] Whitlock, C., et al. 2008. Long-term relations among fire, fuel, and climate in the north-western US based on lake-sediment studies. Int. J. Wildland Fire 17:72-83. Baker, W.L., and M.A. Williams. 2018. Land surveys show regional variability of historical fire regimes and dry forest structure of the western United States. Ecol. Applic. 28:284-290.
[5] Parks, S.A. et al. 2015. Wildland fire deficit and surplus in the western United States, 1984-2012. Ecosphere 6:275. 13 pp.

Many claim that this cannot be done safely without massive thinning to reduce "fuels"[6], others state that we need to get to coexistence with wildland fire as the amount of thinning needed is prohibitively costly[7], and has significant consequences to ecosystems (see below). Still others want more of the "right kind" of fire in the "right places"– meaning less high severity fire, despite ecological importance of this type in low to mid elevation pine and mixed conifer forests (i.e., even predominately low severity ponderosa pine systems have a component of high severity) throughout the West.

Issue 3: Is High Severity Fire Within Wildland Fire Complexes Increasing?

Background

High severity fires that kill most of the trees in older forests are associated with extraordinary levels of biodiversity not present in low severity burns due mainly to the abundance of biological legacies (e.g., snags and down logs, shrubs)[8]. This fact is now widely accepted by the scientific community; however, the amount and spatial distribution of high severity fire patches within wildland fire complexes remains in question as to whether ecosystem thresholds are being crossed in large fires.

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[6] Hessburg, P.F., et al. 2015. Restoring fire-prone Inland Pacific landscapes: seven core principles. Landscape Ecol. 30:1805-1835.

[7] Moritz, M.A., et a. 2014. Learning to coexist with wildfire. Nature 515:58-66.

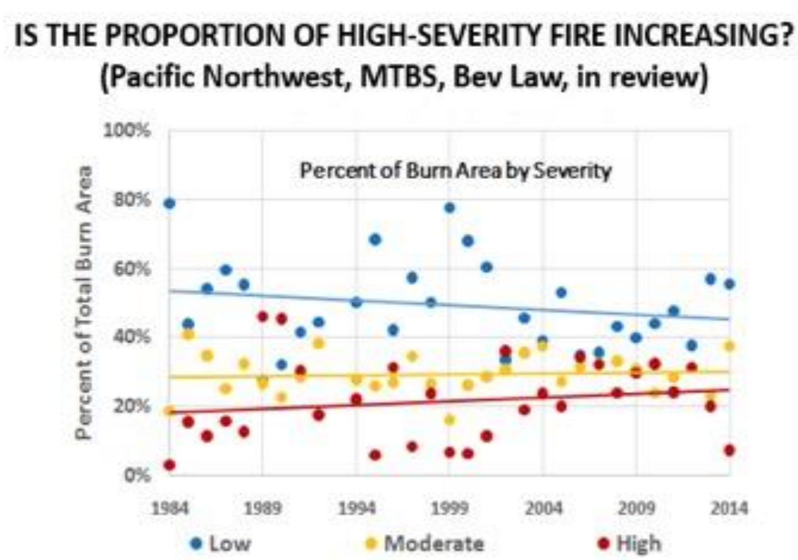
[8] Donato, D.C., J.L. Campbell, and J.F. Franklin. 2012. Multiple successional pathways and precocity in forest development: can some forests be born complex? J. Vegetation Science 23:576-585. DellaSala, D.A. and C.T. Hanson. 2015. The ecological importance of mixed-severity fires: nature's phoenix. Elsevier: Boston.

High Severity Fire Patches Become Biodiverse Snag Forests

Complex early seral forest after 12 years of natural conifer regeneration, native shrub patches, and deciduous trees (C. Hansen, Eldorado Starr Fire, Sierra).

Areas of Agreement

Nearly all studies have detected no statistically significant trend in high severity acres or proportion of high severity fire within large fire complexes (Colorado is an exception and there is debate in the Sierra)[9].



This figure shows no discernable increase in percent of various fire severities in the Pacific Northwest over a three-decade period (compliments of Bev Law, Oregon State University). Data prior to 1984 are not available for fire severity comparisons.

[9] Keyser, A., and A. LeRoy Westerling. 2017. Climate drives inter-annual variability in probability of high severity fire occurrence in the western United States.

Areas of Disagreement

Concern has now shifted to whether the size of high severity patches is increasing, believed to be a product of 21st century "mega-fires," and whether this is leading to type conversions (forests to shrubs)[10]. High severity patch data obtained from hundreds of forest fires across the West show no statistical increase in patch sizes in recent decades (DellaSala et al. in peer review). This is important as the patch size debate is used to make claims about "mega-fires" and to justify large-scale thinning, post-fire logging, and tree planting based on perceptions of inadequate tree recruitment or lack of forest resilience to fires. However, most high severity patches have high levels of internal heterogeneity that include small patches of live trees or nearby low-moderate burn areas as seed sources (in review).

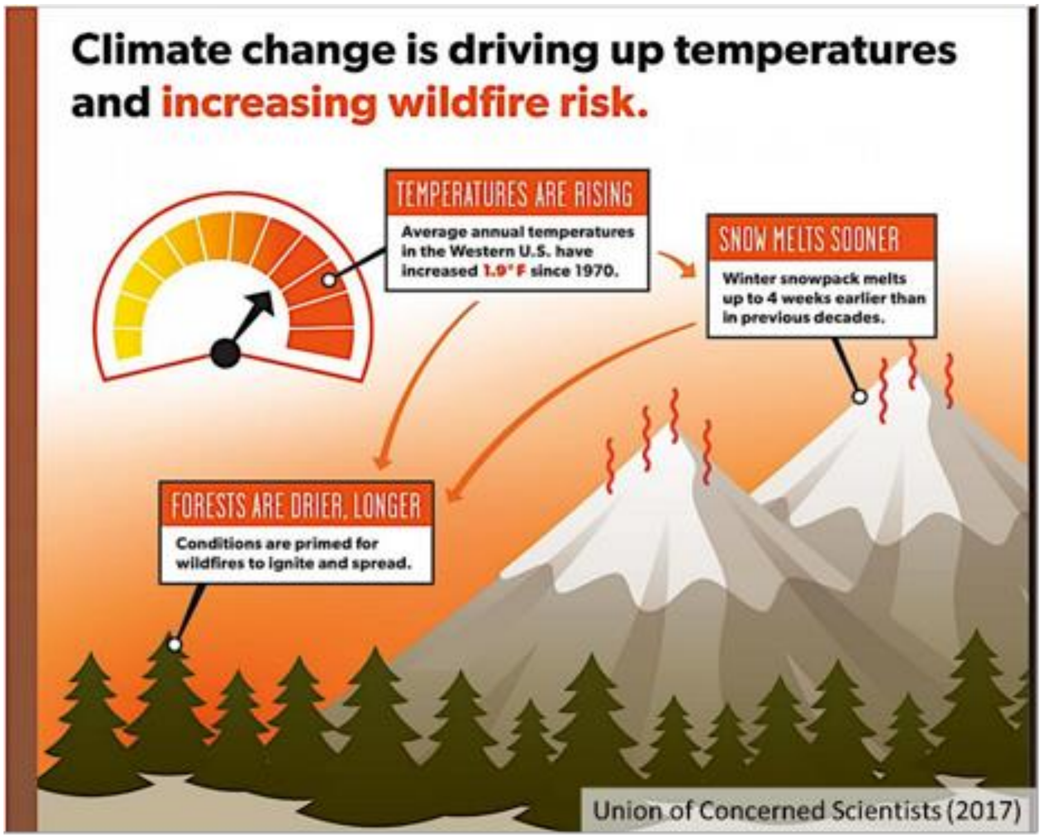
Issue 4: What's Driving Recent Increases in Wildfire Acres Burning?

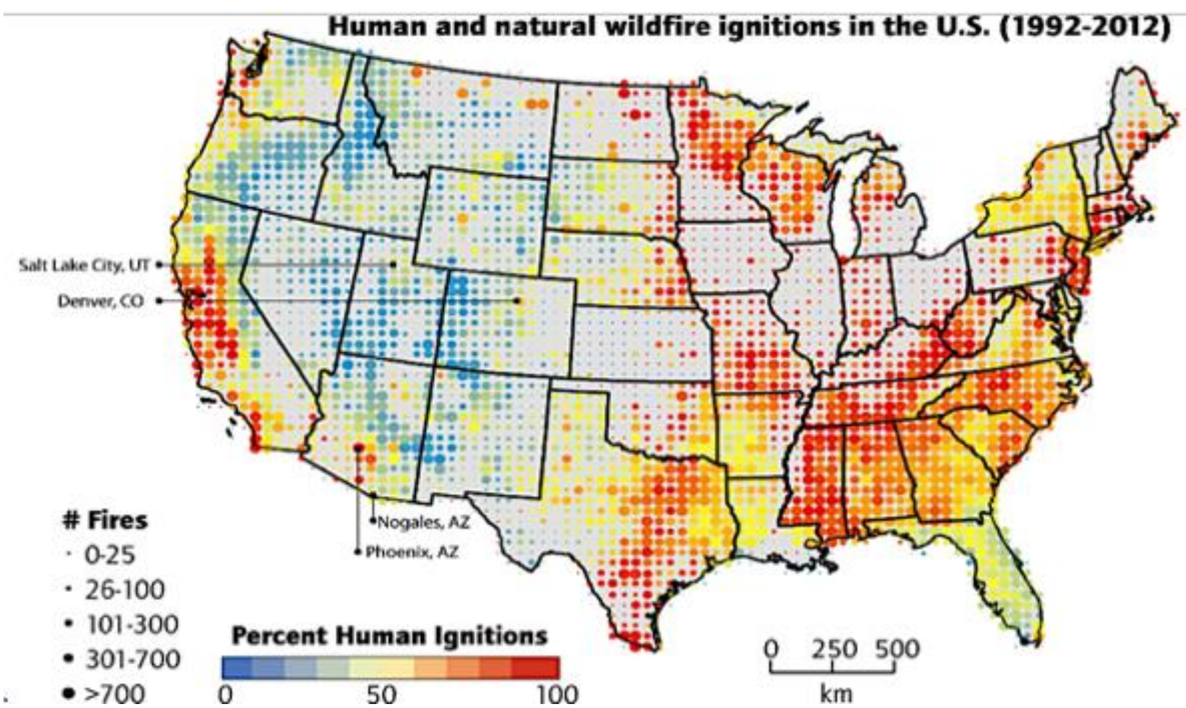
Background and Areas of Agreement

Recent increases in wildfire acres burning (see above PDO figure) can be traced to three main factors acting in concert: (1) a warming PDO from climate change; (2) increases in human-caused fire starts (accidental, arson, back burns); and (3) conversion of native forests to flammable tree plantations[11].

Over half of recent increases in wildfire acres burning has been attributed to climate change[12] (see top figure below as generalization) with 84% of all fire ignitions nationwide in recent decades caused by people (bottom figure below)[13]. Human-caused wildfire ignitions vary regionally based on population densities and remoteness.

[10] Hessburg P.F. et al. 2015. Restoring fire-prone inland Pacific landscapes: Seven core principles. Landscape Ecology 30, 1805–1835.
[11]Bradley, C., C.T. Hanson, and D.A. DellaSala. 2016. Does increased forest protection correspond to higher fire severity in frequent-fire forests of the western United States? Ecosphere 7:1-13. Odion, D.C., et al. 2004. Fire severity patterns and forest management in the Klamath National Forest, northwest California, USA. Conservation Biology 18:927-936
[12] Abatzoglou J.T., and A.P. Williams. 2016. Does Impact of anthropogenic climate change on wildfire across western US forests. PNAS 113:11770-11775
[13] Balch et al. 2017. Human-started wildfire expand the fire niche across the United States. PNAS 114:2946-2951.


Climate change is driving up temperatures and increasing wildfire risk.
TEMPERATURES ARE RISING
Average annual temperatures in the Western U.S. have increased 1.9° F since 1970.
SNOW MELTS SOONER
Winter snowpack melts up to 4 weeks earlier than in previous decades.
FORESTS ARE DRIER, LONGER
Conditions are primed for wildfires to ignite and spread.
Union of Concerned Scientists (2017)


Human and natural wildfire ignitions in the U.S. (1992-2012)
Salt Lake City, UT
Denver, CO
Nogales, AZ
Phoenix, AZ
Fires
0-25
26-100
101-300
301-700
>700
Percent Human Ignitions
0
50
100
0 250 500
km

Areas of Disagreement

While most land managers and decision makers are preoccupied with “fuels,” two of the main drivers of fire behavior (climate change, human-caused ignitions) are largely ignored (except when used to justify logging for forest resilience). Additionally, roads (a principal source of human-caused fire ignitions) are almost never addressed in fire risk reduction measures. Uncertainty exists regarding whether large-scale thinning will work in a changing climate where fire behavior will be increasingly governed by extreme fire weather (high temperatures, low soil moisture, high winds, see below)[14]. Storing more carbon in ecosystems will help mitigate climate effects, although land managers often prioritize generating revenue from commercial sales over carbon storage[15].

Issue 5: Does “Active Management” Reduce Wildfire Intensity and Lower Fire Risks?

Background

Active management encompasses a wide spectrum of actions and opinions mostly focused on pre- (thinning) and post-fire (“salvage” logging) logging widely debated by scientists, conservation groups, and decision makers. This is arguably the number one area of fire-related conflicts on public lands with the underlying assumption that forests are overstocked, they need active management to reduce fire risks, and environmental safeguards are preventing management of forests that otherwise will burn out of control.

Areas of Agreement

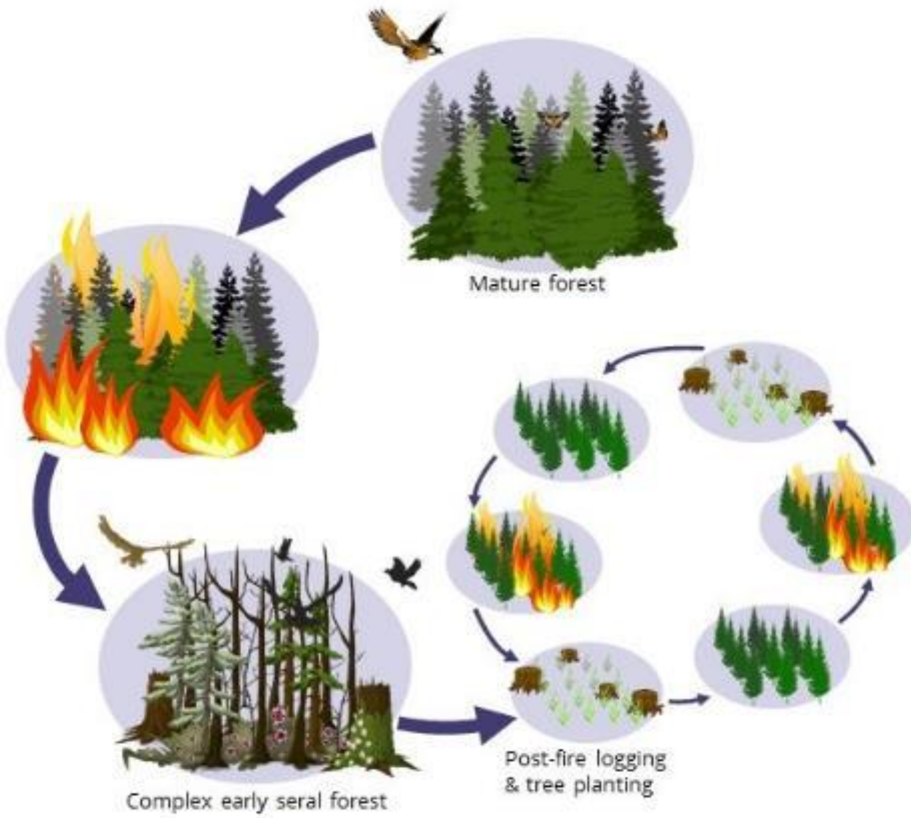
Post-fire logging is unequivocally damaging to the pyrodiverse landscapes and complex early seral forests. In general, the larger the fire, the bigger the logging project[16]. Post-fire logging involves clearcutting both live and mostly dead trees, kills naturally regenerating conifers, and often is followed by herbicides to reduce competing yet beneficial vegetation and allow for subsequent planting of artificially grown trees (from nursery stock) in dense rows. As artificial plantations increasingly replace native forests, plantations act as kindling for intense fires (i.e.,

[14] Cary, G.J. et al. 2016. Importance of fuel treatment for limiting moderate-to-high intensity fire: findings from comparative fire modeling. Landscape Ecol. 32:1473–1483. Kalies, E.L., and L.L.Y. Kent. 2016. Tamm review: are fuel treatments effective at achieving ecological and social objectives? A systematic review. Forest Ecology and Management 375:84-95.

[15] Moritz, M.A. et al. 2014 (ibid). Law, B.E et al. 2018. Land use strategies to mitigate climate change in carbon dense temperate forests. PNAS http://www.pnas.org/cgi/doi/10.1073/pnas.1720064115

[16] DellaSala, D.A., et al. 2015. In the aftermath of fire: logging and related actions degrade mixed- and high-severity burn areas. Pp. 313-347, *In* DellaSala, D.A., and C.T. Hanson (eds), The ecological importance of mixed-severity fires: nature’s phoenix. Elsevier, United Kingdom

"fire's gasoline")[17]. Post-fire logging creates a catastrophic feedback loop where fires in older forests create ecologically beneficial snag forests, those forests are then clearcut and replanted with small trees in dense rows lacking structural complexity, only to burn in higher intensities and so on (see figure below)[17,18]. Legacy trees removed by logging operations anchor soils, provide shade for developing seedlings, "nurse logs" for new growth and soil moisture retention for amphibians and invertebrates, habitat for aquatic species when snags fall into streams, and they store vast amounts of carbon as they slowly (decades to centuries) decompose. The scientific community is generally at consensus with regard to post-fire logging as damaging to ecosystems[19], particularly to spotted owl habitat[20].



Fire in a mature forest produces complex early seral (snag) forest that connects the stages of forest development through time. This cycle is interrupted by post-fire logging and tree planting leading to type conversions (native forest to flammable plantation) and unnatural fire severity.

[17] Odion, D.C., et al. 2004. Ibid. Thompson, J.R., et al. 2007. Reburn severity in managed and unmanaged vegetation in a large wildfire. PNAS 104:10743-10748.
[18] Bradley, C.M., et al. 2016. Does increased forest protection correspond to higher fire severity in frequent-fire forests of the western United States? Ecosphere 7:1-13.
[19] Lindenmayer, D.B., P.J. Burton, and J.F. Franklin. 2008. Salvage logging and its ecological consequences. Island Press: Washington, D.C.
[20] C.T. Hanson, M.L. Bond, and D.E. Lee. 2018. Effects of post-fire logging on California spotted owl occupancy. Nature Conservation 24:93-105.

Areas of Disagreement

In contrast to post-fire logging, thinning involves partial logging of trees for various purposes, including reducing competition among nearby trees, increasing tree vigor, and accelerating tree growth (e.g., in wet forests it is commonly used to accelerate development of older forest conditions as specified under the Northwest Forest Plan). Thinning also is commonly used to reduce "fuels" in dry forests and has support in the scientific community and with NGOs. When done properly, thinning of small trees followed by prescribed burning[14], or prescribed burning alone in some cases[21], can reduce fire intensity. However, it remains controversial, has significant ecological consequences (short and long-term), and substantial limitations given high costs and the massive scale believed needed to influence fire behavior especially in a changing climate (Box 1).

Large trees (dbh inches marked on trees) marked for removal on a BLM "fuels" project, southwest Oregon (L. Ruediger).

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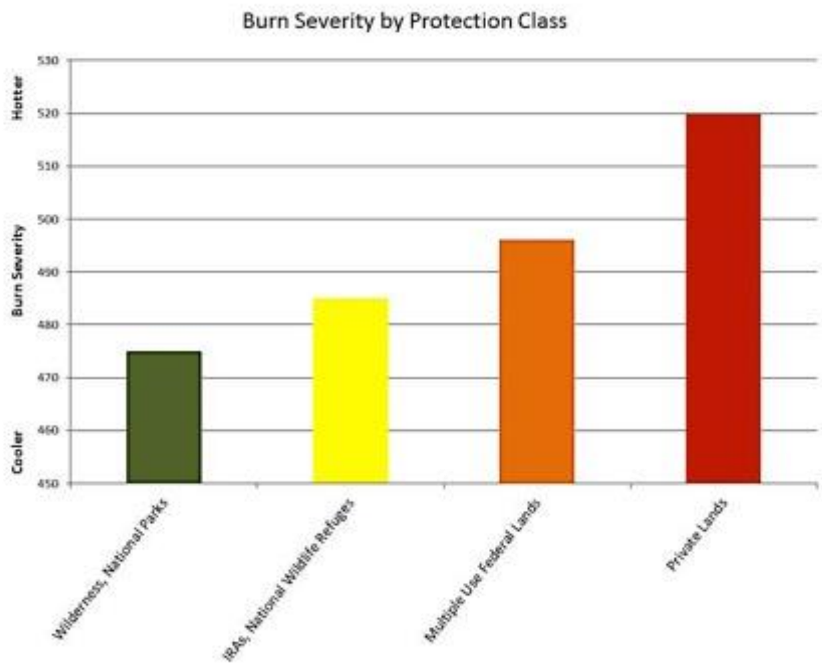
[21] Zachmann, L.J., D.W.H. Shaw, and B.G. Dickson. 2018. Prescribed fire and natural recovery produce similar long-term patterns of change in forest structure in the Lake Tahoe basin, California. Forest Ecol. & Manage. 409:276-287

Box 1. General limitations of thinning (and collateral ecosystem damages)

(1) Thinning reduces habitat for canopy dependent species, including spotted owls[22], requires an expansive road network damaging to aquatics, can spread invasive and flammable weeds, and, when implemented over large landscapes, releases more carbon emissions than fires, even severe ones[23].
(2) There is a very low probability (3-8%) that a thinned forest will encounter a fire during the narrow period (10-20 years depending on site factors) of reduced "fuels" [24], resulting in large-scale thinning proposals that alter forest conditions over large areas[6].
(3) Excessive thinning (e.g., reducing bulk crown density below 60%) can increase wind speeds and solar radiation to the ground causing increased flammable vegetation growth and fire spread.
(4) Thinning needs to be followed by prescribed fire to reduce flammable slash but this can cause soil damage especially if burning is concentrated in piles (intensifies heat effects.
(5) Thinning is seldom cost effective without public subsidies or removing large fire-resistant trees.
(6) In some regions (Sierra, Klamath-Siskiyou), time since fire is not associated with increasing fire risks (i.e., as forests mature, they become less flammable[25]).
(7) Thinning efficacy is limited under extreme fire weather (principal factor governing large fires).
(8) At regional scales, active management (unspecified forms of logging) have been associated with uncharacteristic levels of high severity fires (see figure below)[26].

[22] Odion, D.C., et al. 2014. Effects of fire and commercial thinning on future habitat of the northern spotted owl. Open Ecology Journal 7:37-51.
[23] Campbell, J.L., M.E. Harmon, and S.R. Mitchell. 2012. Can fuel-reduction treatments really increase forest carbon storage in western US by reducing future fire emissions? Frontiers in Ecol. & Environ. doi:10.1890/110057
[24] Rhodes, J.J., and W.L. Baker. 2008. Fire probability, fuel treatment effectiveness and ecological tradeoffs. The Open Forest Science Journal, 2008, 1, 1-7
[25] Odion, D.C., et al. 2004. Fire severity patterns and forest management in the Klamath National Forest, northwest California, USA. Conservation Biology 18:927-936. Zachmann, L.J., et al. 2018. Ibid.
[26] Bradley, C.M., C.T. Hanson, and D.A. DellaSala. 2016. Does increased forest protection correspond to higher fire severity in frequent-fire forests of the western United States? Ecosphere 7: Ecosphere 7:1-13.

Thinning on the Deschutes National Forest, Oregon (G. Wuerthner).



Burn severity as a function of protection levels from lower burn severity in Wilderness and National Parks (green) to greater high severity amounts in actively managed areas (red)[26]. Figure prepared by C. Bradley, CBD.

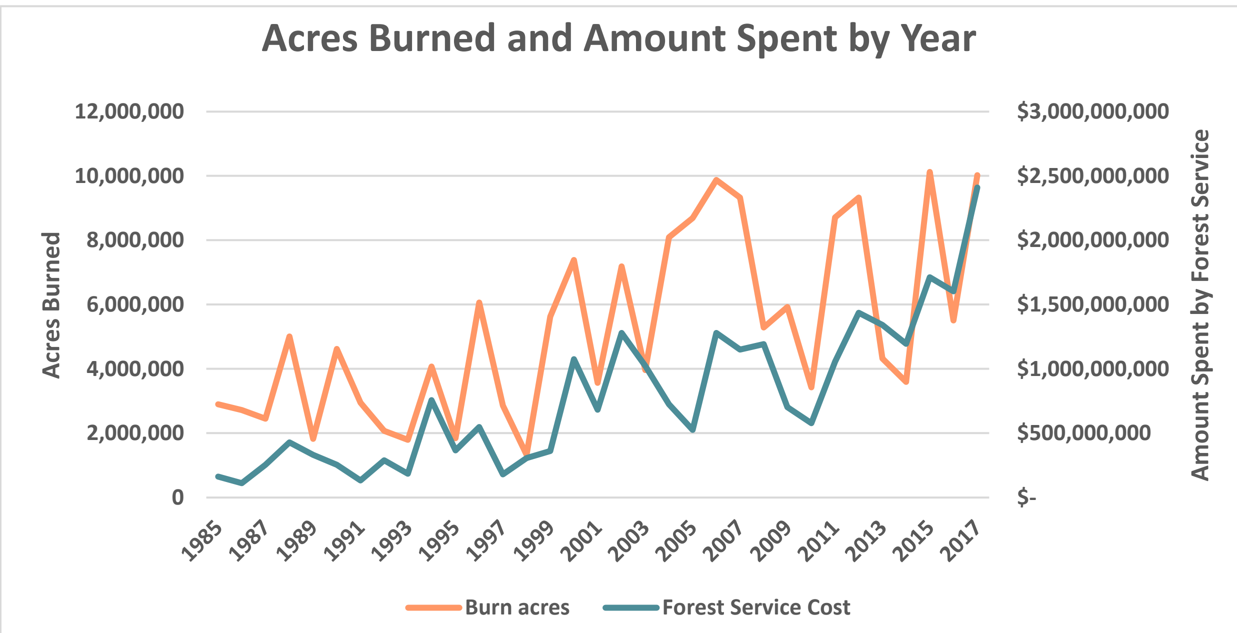
Issue 6: Will More Suppression Spending Make Us Safer?

Background

On March 21, 2018, Congress passed an omnibus spending package that established a dedicated wildfire “disaster” fund of > $2 billion per year that would increase steadily over a 10-year period. Spending measures include expanding the use of controversial categorical exclusions for logging projects up to 3,000 acres each that can conceivably be located adjacent to one another with no regard for cumulative impacts.

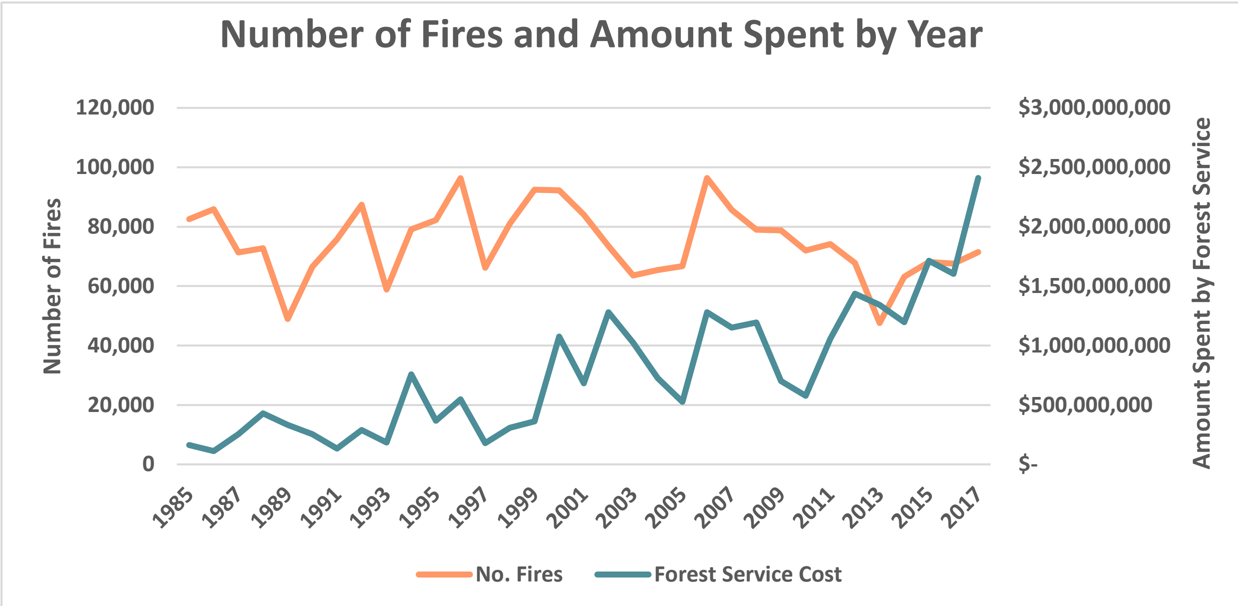
Areas of Agreement and Disagreement (combined)

While conservation groups pushed for a rider-free wildfire spending fix, throwing more money at fire suppression while expecting fewer fires is highly uncertain. In many ways, the two figures below illustrate the common definition of crazy – doing the same thing over and over again but expecting a different outcome. In sum, both acres burning and wildfire suppression costs of the Forest Service have risen dramatically over the past three decades (top figure) calling into question whether more money will achieve fewer fires or less acres burning. Interestingly, in some years (e.g., 2006-2012, bottom figure) total wildfire ignitions steadily dropped while costs generally rose presumably from fighting more fires in remote areas and few controls on spending[27] (figures prepared by J. Leonard, Geos Institute using fire data from National Interagency Fire Center[28]).



[27]Ingalsbee, T., and U. Raja. 2015. The rising cost of wildfire suppression and the case for ecological fire use. Pp. 348-317 In: D.A. DellaSala, C.T. Hanson (eds.). The ecological importance of mixed-severity fires: nature's phoenix. Elsevier: Boston.

[28] https://www.nifc.gov/fireInfo/fireInfo_statistics.html



As an example of unmitigated suppression spending, the 132,127-acre Soberanes fire in California (started by an illegal campfire) cost ~$236 million (nearly $1800 per acre) and deployed thousands of fire fighters and numerous air-tankers, making it the most expensive wildfire to fight in US history. Although the fire destroyed 57 homes (and took the life of a bulldozer operator), suppression forces were used on the fire as it burned safely in the back country far removed from homes. The fire was eventually extinguished by fall rains.

Conclusion: Moving Forward in the New Climate Wildfire Era

When it comes to fire, we each see what we want: land managers view the world as ready-to-burn ecosystems just lacking an ignition source and needing "fuels" reduction; ecologists see habitat restored by wildfires as part of the circle of life and death in a forest; the public fears fire and understandably has concerns about smoke emissions; the media portrays death and destruction during fires; conservation groups are either for or against large-scale thinning; and politicians race to sensationalize fire to justify increased commercial logging on public lands. This is no doubt the most difficult public lands issue we have ever faced as its wrapped in emotion, human health, self-interests, avarice, hyperbole, point-counter point arguments, and nearly everyone wants to do something – even if doing something is worse than the perceived problem. Moving beyond this will require communicating about fire with empathy and clear intent especially while recognizing genuine fear and health issues. It will involve a combination of science publications, public support for managing wildfires for ecosystem benefits (once safety has been addressed), tolerance for temporary smoke levels, and our own limitations in being able to influence ecological processes increasingly governed by top-down drivers (climate) rather than bottom up forest management. Based on climate change models, extreme fire conditions are predicted to be more common this century and thus the extensive thinning involved to theoretically reduce fire intensity (e.g., wide spacing among trees, open-park like conditions) would create novel or greatly engineered forest systems that impact biodiversity and ecosystem services (carbon stores, clean water) in undesirable ways.

Importantly, we need to solve for human safety with the most significant challenges coming from ex-urban sprawl (enabled by scant land-use zoning and building in the wrong places), a rapidly changing climate, an expanding logging footprint focused increasingly on extracting the "new coal" ("feed stock") for biomass burning. Rational fire approaches and communication strategies that do not sacrifice native forests for perceived fire safety are an area of much needed research and financial resources.

We know a lot more about wildfire today than in the last decade; however, much of the science is still in debate, it almost always lags behind or is ignored by decision makers, land managers, and even some scientists and conservation groups with entrenched views about fire (Box 2).

Box 2. What we know and do not know about wildfires.

- Complex early seral forests are as biodiverse as old growth, containing comparable levels of species richness (although species composition varies across seral stages).
- Wildfire effects on vegetation are highly variable (mixed)[29], calling into question fuel reduction projects (especially those that use a shifting baseline) based on restoring forests to an "historical" open park-like condition when there was a lot more variability and the climate is changing.
- It will be impossible to mechanically treat the substantial acres alleged to need fuel reduction to reduce fire intensity[7] (58 million acres according to the Forest Service), and, even if possible, this would have severe consequences to ecosystems, especially aquatics, and come with substantial taxpayer funded costs.
- Thinning under extreme fire weather ("the new norm") is highly uncertain in a changing climate.
- Additional increases in homes built within the Wildland Urban Interface (WUI) (now totaling 43.4 million)[30] will result in more human-caused fire ignitions and out of control suppression spending regardless of where the money comes from. Wildfire problems will not abate if this growth along with climate change accelerates.

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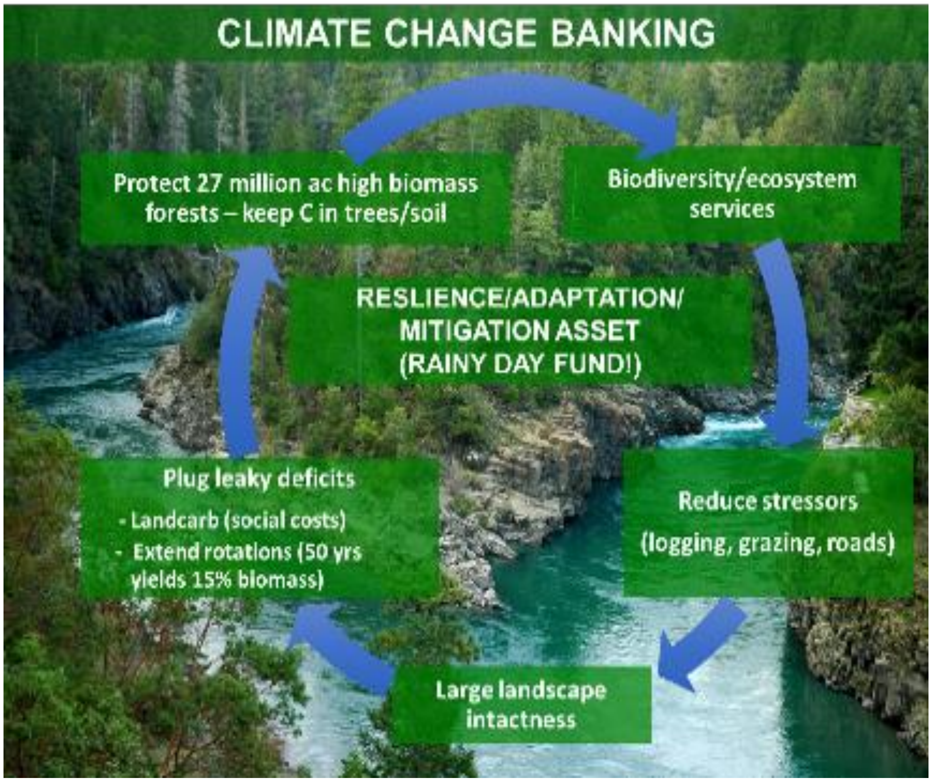
[29]Odion, D.C., et al. 2016. Areas of agreement and disagreement regarding ponderosa pine and mixed conifer forest fire regimes: a dialogue with Stevens et al. PLoSOne DOI:10.1371/journal.pone.0154579 May 19, 2016
[30]Radeloff, V., et al. 2018. Rapid growth of the US wildland -urban interface raises wildfire risk. PNAS http://www.pnas.org/cgi/doi/10.1073/pnas.1718850115

There is no “right” or “wrong” or “good” or “bad” fire. Fire is a predatory force of Nature resulting in ecological winners and losers (at least temporarily). We in the environmental community do not speak of “good” wolves or “bad” mega-wolves (that eat sheep) yet the fire debate embraces this terminology. In sum, we do not have a fire problem per se but rather a people management problem – homes built in the wrong places and with the wrong materials, fire-fighters dropped into unsafe areas, hyped-up thinning projects that may or may not work, and a rapidly changing climate that will produce surprises.

There are plenty of management options that are compatible with western forest resilience and fire-mediated biodiversity in a changing climate, including:

- Removing land-use stressors (e.g., mining, livestock, Off Highway Vehicle impacts that accumulate in space and time) so that ecosystems can adapt to climate change;
- Maintaining viable populations of imperiled species and habitats, including climate sanctuaries such as older forests, forests on north-facing slopes, and riparian areas[31];
- Curtailing the spread of invasive species;
- Managing wildfires for ecosystem benefits and prescribed fire in appropriate types;
- Thinning and girdling (killing) small trees in young plantations (along with prescribed fire) to increase structural complexity and reduce fire intensity (but see limitations discussion);
- Replacing ineffective culverts (especially important in areas where climate change will trigger more floods); restoring floodplains so they can naturally store more water (e.g., reintroducing beavers) and attenuate floods; and removing damaging roads by re-contouring the road prism to natural features (e.g., to reduce sediments to streams and improve hydrological functions);
- Managing for connectivity (up-down elevation, latitudinal-longitudinal gradients); and
- Storing more carbon in forest ecosystems (see climate robust strategies).

[31]Olson, D.M., et al. 2012. Climate change refugia for biodiversity in the Klamath-Siskiyou ecoregion. Natural Areas Journal 32:65-74.



Climate robust conservation means protecting carbon dense forests nationwide as a foundation for biodiversity and ecosystem services, reducing land-use stressors, connecting landscapes for wildlife migrations and reducing carbon emissions from logging. Fire safety measures discussed herein are compatible with this overall strategy and represent a comprehensive path forward.

Importantly, managing wildfire for ecosystem benefits is not the same as "let burn." Instead, this involves monitoring wildfire behavior initially, targeting suppression at fires likely to spread near towns, "loose-herding" and directing fire in the back-country under safe conditions, cutting fire lines nearest homes, and keeping fire fighters out of harm's way. The same fire can be compartmentalized for different treatments. The Forest Service already has existing authorities that allow them to use such approaches in deciding when to attempt to use suppression vs. managing wildfire for ecosystem benefits[32]. Implementing this policy would help keep spiraling wildfire suppression costs in check[27].

In addition, local governments need to start embracing smart growth measures to limit sprawl within the WUI. Fire safety for existing homes is about reducing risks from the home-out (defensible space), rather than from the wildlands-in (logging)[33]. Defensible space has to become as routine as changing the batteries in a home's smoke detectors and building with metal roofs the norm in home construction.

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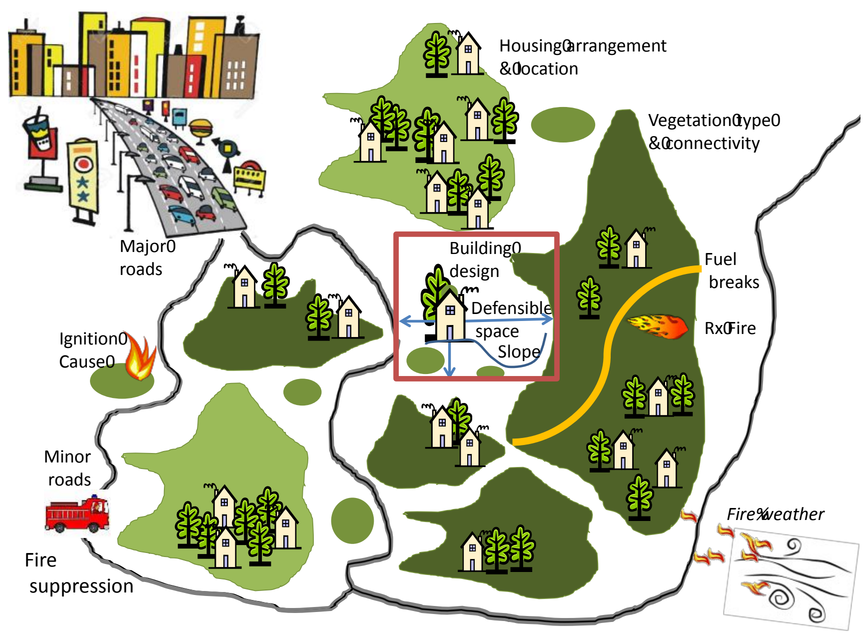
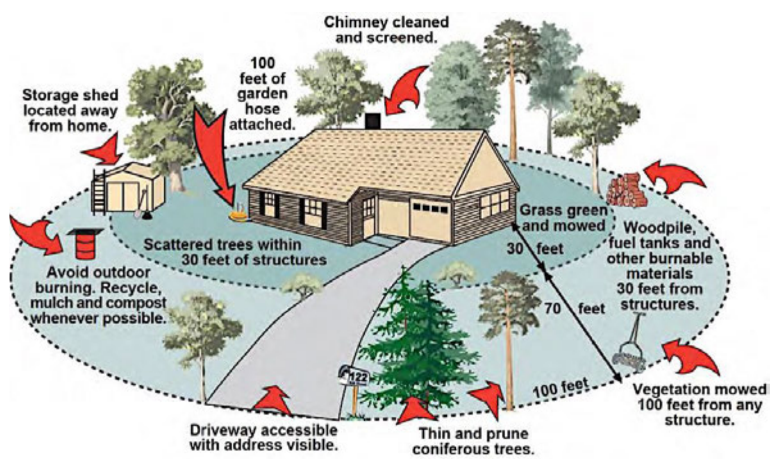
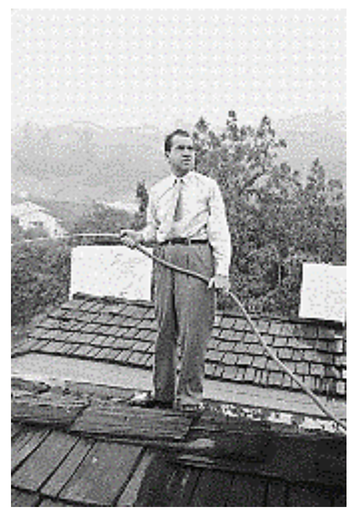
[32] https://www.nifc.gov/policies/policies_documents/GIFWFMP.pdf
[33] Syphard, A.D., T.J. Brennan, and J.E. Keeley. 2014. The role of defensible space for residential structure protection during wildfires. Int. J. Wildland Fire. http://www.publish.csiro.au/wf/WF13158



Fire prevention begins with land-use planning that limits growth in unsafe areas and includes defensible space management (figure prepared by A. Syphard, CBI; historical Nixon photo courtesy of San Francisco Chronicle[34]; lower figure – Homeowner fire safe guide for Montana).





Potential synergies and framing messages around forest issues cut across public lands campaigns that could benefit from working together, including the "keep it [carbon] in the ground," "350.org," and a much needed "keep it [carbon] in the forest" campaign. For instance, researchers at Oregon State University recently showed that the best way to increase carbon stores in Northwest forests is to reduce federal lands logging by at least 50%, increase the length of timber harvest rotations on private lands to 80 years, afforestation, and reforestation[35]. Notably, wildfires are currently not a significant contributor to greenhouse gas

[34] http://www.sfgate.com/news/article/Skirball-Fire-recalls-1961-Bel-Air-inferno-that-12410921.php
[35] Law, B.E., et al. 2018. Land use strategies to mitigate climate change in carbon dense temperate forests. PNAS www.pnas.org/cgi/doi/10.1073/pnas.1720064115

emissions, contrary to many assertions[36]. Importantly, the Northwest Forest Plan resulted in ancillary climate benefits by shifting federal forest management from a substantial source of logging emissions in the 1980s to a current "sink" (warehouse) for carbon storage due to reduced (by 80%) timber harvest on federal lands[37]. As this forested warehouse continues to accumulate carbon, it is critical to protect carbon-dense older forests on public lands and incentivize forest carbon stewardship on non-federal lands. Making the link between climate mitigation and intact forest conservation currently lacks the recognition needed to offset fossil fuel emissions and keep the planet from heating above 2° C, which cannot be accomplished without forests in the mix[38].

The long-range prognosis for public lands forests is generally favorable. On the one hand, conservation groups with significant support of the donor community have held the line on decades of hard-fought victories centered on the Northwest Forest Plan and wilderness/ roadless protections. On the other hand, the pressure to develop forests is unprecedented globally and regionally with an urgent need to solve for increasingly complex social, economic, and engrained perceptions about forest management. Conservation science continues to be a leading voice for public lands by supporting effective communications, grass-roots organizing and campaigning, and responding to maladaptive climate policies by proposing climate robust conservation strategies. When it comes to fire science, however, we have as many questions as answers, more debate than consensus, but there have been important strides forward.

In closing, we have much work to do to change public attitudes about forest fires but optimism begins when we open our hearts and minds to the intricate dance between green and burned forest orchestrated by the natural disturbance processes that have been at play since the age of dinosaurs and will continue in largely unpredictable ways in the emerging novel climate. Preparing for these changes must be comprehensive, science-based, and solve for top-down drivers of change while we hold the line and then expand on a robust conservation vision.

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[36]Law, B.E., T.W. Hudiburg, and S. Luyssaert. 2013. Thinning effects on forest productivity: consequences of preserving old forests and mitigating impacts of fire and drought. Plant Ecol & Diversity 6:73-85. Mitchell, S. 2015. Carbon dynamics of mixed- and high-severity wildfires: pyrogenic CO2 emissions, postfire carbon balance, and succession. Pp. 290-312, In D.A. DellaSala, and C.T. Hanson. The ecological importance of mixed-severity fires: nature's phoenix. Elsevier: Boston. Law et al. 2018 (ibid).
[37]Krankina, O.N., M.E. Harmon, F. Schnekenburger, and C.A. Sierra. 2012. Carbon balance on federal forest lands of Western Oregon and Washington: the impact of the Northwest Forest Plan. Forest Ecol. & Manage. 286:171-182
[38]https://primaryforest.org/

RESEARCH ARTICLE

Negative Feedbacks on Bark Beetle Outbreaks: Widespread and Severe Spruce Beetle Infestation Restricts Subsequent Infestation

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Abstract

Understanding disturbance interactions and their ecological consequences remains a major challenge for research on the response of forests to a changing climate. When, where, and how one disturbance may alter the severity, extent, or occurrence probability of a subsequent disturbance is encapsulated by the concept of *linked disturbances*. Here, we evaluated 1) how climate and forest habitat variables, including disturbance history, interact to drive 2000s spruce beetle (*Dendroctonus rufipennis*) infestation of Engelmann spruce (*Picea engelmannii*) across the Southern Rocky Mountains; and 2) how previous spruce beetle infestation affects subsequent infestation across the Flat Tops Wilderness in northwestern Colorado, which experienced a severe landscape-scale spruce beetle infestation in the 1940s. We hypothesized that drought and warm temperatures would promote infestation, whereas small diameter and non-host trees, which may reflect past disturbance by spruce beetles, would inhibit infestation. Across the Southern Rocky Mountains, we found that climate and forest structure interacted to drive the 2000s infestation. Within the Flat Tops study area we found that stands infested in the 1940s were composed of higher proportions of small diameter and non-host trees ca. 60 years later. In this area, the 2000s infestation was constrained by a paucity of large diameter host trees (> 23 cm at diameter breast height), not climate. This suggests that there has not been sufficient time for trees to grow large enough to become susceptible to infestation. Concordantly, we found no overlap between areas affected by the 1940s infestation and the current infestation. These results show a severe spruce beetle infestation, which results in the depletion of susceptible hosts, can create a landscape template reducing the potential for future infestations.

OPEN ACCESS

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www.fs.usda.gov/detail/r2/landmanagement/gis/?cid¼stelprdb519523). Vegetation data for the state of Colorado are available from the Colorado Gap Project (http://gapanalysis.usgs.gov). Additional vegetation data for the state of Colorado are available from Landfire (http://www.landfire.gov/vegetation.php).

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Introduction

In the context of a changing climate and increases in forest disturbances such as bark beetle infestations and wildfires, disturbance interactions are receiving increased attention in ecological research [1,2]. In particular, there is a need to better understand when, where and how one disturbance event may alter the severity, extent, or probability of occurrence of a subsequent disturbance, a concept known as *linked disturbances* [3]. A prior disturbance may amplify the second by increasing its likelihood or severity through positive feedbacks (e.g. blowdowns may increase the amount breeding material thereby increasing insect populations and likelihood of outbreak [4]). Or, alternatively the first disturbance may dampen the probability of occurrence or severity of the second (e.g. stand-replacing fire may decrease the probability of subsequent fire [5]).

During the late 20th and early 21st century, warm and dry conditions and suitable hosts have promoted landscape-scale (sensu [6]) and severe bark beetle outbreaks, resulting in tree mortality across 8.4 ± 2.5 Mha in the western North America (1997–2010; [7]). Given this extensive tree mortality, there is an increased need for understanding how bark beetle infestations alter subsequent disturbance dynamics. Considerable research has emphasized the potential effects of bark beetle outbreak on the fire behavior [3,8–14], occurrence [15–19], and severity [15,20–22]. Recent research has also emphasized the compound effects of bark beetle outbreak and fire on ecosystem recovery [20–23]. Far less is understood about how one bark beetle outbreak affects a subsequent outbreak.

Bark beetles of the *Dendroctonus* genus inhabit the inner bark and feed on the tree's phloem tissues. Heavy colonization and reproduction within the inner bark interrupts the flow of water and nutrients throughout the tree and usually causes tree death. When and where bark beetle outbreaks occur is constrained by both weather and forest structure conditions [6,24,25]. Warm temperatures promote the rapid growth of beetle populations by increasing the proportion of beetles that develop within one year and decreasing overwintering mortality [26–28]. Drought may stress host trees, increasing the susceptibility of trees to infestation [29–32]. Forest structure also affects the occurrence of bark beetle infestations. Bark beetles prefer large diameter trees, growing in dense stands composed predominantly of the host tree species [6,33].

In the Southern Rocky Mountains, outbreaks of spruce beetles (*Dendroctonus rufipennis*) are among the most important broad-scale disturbances in subalpine forests. Spruce beetles are found in Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) forests, where they most frequently colonize large diameter (> 23 cm diameter at breast height; DBH) spruce trees. However when beetle population levels are high and host trees are severely-drought stressed, spruce beetles may attack trees less than 10 cm DBH [34]. Like other bark beetles, heavy colonization and reproduction within the inner bark usually kills the host tree. In northwestern Colorado, severe spruce beetle infestations tend to occur at median intervals of c. 70 years for the same stand [30,35]. The return interval of spruce beetle infestations to the same stand or relatively homogeneous landscape is hypothesized to be in part a function of a negative linkage between infestations. Thus, for forest stands (100s of hectares) and forest landscapes (1000s to tens of 1000s) that are characterized by similar forest compositions and tree population age structures, forest attributes are likely to affect the probability of occurrence and severity of an outbreak [32]. For example, a severe spruce beetle outbreak, which may result in the mortality of 90% of the mature host trees (Engelmann spruce), has been hypothesized to decrease the likelihood of subsequent infestation [33]. This decrease in susceptibility to infestation is hypothesized to persist until host trees reach a suitable size for infestation. While there are studies documenting the collapse of an outbreak evidently due to host depletion [29], there is no published empirical evidence for a bark beetle infestation negatively influencing the occurrence of a subsequent bark beetle infestation.

A widespread spruce beetle outbreak affected a large part of the spruce-fir forests of western Colorado in the 1940s. This outbreak was most severe in the Flat Tops Wilderness area of White River National Forest in northwestern Colorado where 99% of the overstory spruce were killed over an area of 2,700 km^2 [33,36]. The second most severely affected area in the 1940s outbreak was Grand Mesa National Forest to the southwest of White River National Forest where mortality was estimated at over 50% [32]. There are no other known 20th century spruce beetle outbreaks in Colorado of a comparable magnitude to the 1940s outbreak that was centered on the Flat Tops area of White River National Forest. Thus, in the context of the recent spruce beetle outbreak of 1997–2012, the concentration of high tree mortality during the 1940s outbreak in one large contiguous area created the opportunity to quantitatively evaluate the potential for a landscape-scale bark beetle infestation to negatively affect the probability of a subsequent infestation ca. 60 years later. Mapping of the recent spruce beetle outbreak from Aerial Detection Surveys [37] indicate a low spruce beetle infestation in the Flat Tops area in comparison with spruce-fir forests throughout Colorado (Fig 1). Thus, the primary aim of this study is to determine if the relative lack of recent spruce beetle infestation in the Flat Tops area is due to a negative feedback from host depletion attributable to the 1940s outbreak. Because spruce beetle infestation depends on both climate and forest conditions [6], we first assess the suitability of climate and forest attributes for spruce beetle infestation during 1997–2012 in the Flat Tops study area in comparison with the entire Southern Rocky Mountain Ecoregion of the U.S. Second, we examine forest attributes across the Flat Tops study area in relation to the mapped extent of the 1940s spruce beetle infestation and compare current forest structure in field sampled stands infested and not infested in the 1940s. Thus, by documenting the climatic suitability of the Flat Tops study area for the recent infestation, we are able to associate the relative absence of infestation with host depletion from the 1940s outbreak.

Materials and Methods

Study area

The study region (Fig 1) is the spruce-fir forest type of the Southern Rocky Mountain Ecoregion. The study region is characterized by high elevations (3215 ± 205 m), cold, wet winters (mean minimum January temperature -14°C and mean total January-March precipitation 241 mm; 1981–2010) and warm, dry summers (mean maximum July temperature 20.6°C and mean total June-August precipitation 169 mm; 1981–2010) [38]. Engelmann spruce and subalpine fir co-dominate the spruce-fir forest type.

We examine the potential for spruce beetle infestation to affect the area of forest structure suitable for subsequent spruce beetle infestation within a subset of the study region comprised of the Flat Tops Wilderness and adjacent areas of White River National Forest of northwestern Colorado, USA (Fig 1). The Flat Tops study area was chosen because of the unique availability of maps of both the 1940s spruce beetle infestation derived from air photo interpretation [5] and the current (1997–2012) spruce beetle infestation produced from Aerial Detection Surveys (ADS; [37]). Historical reports document widespread spruce beetle infestation in the 1940s, when about 25% of the merchantable volume of Colorado's spruce was killed. The Flat Tops study area experienced particularly abundant mortality, characterized by more than 90% canopy mortality [33].

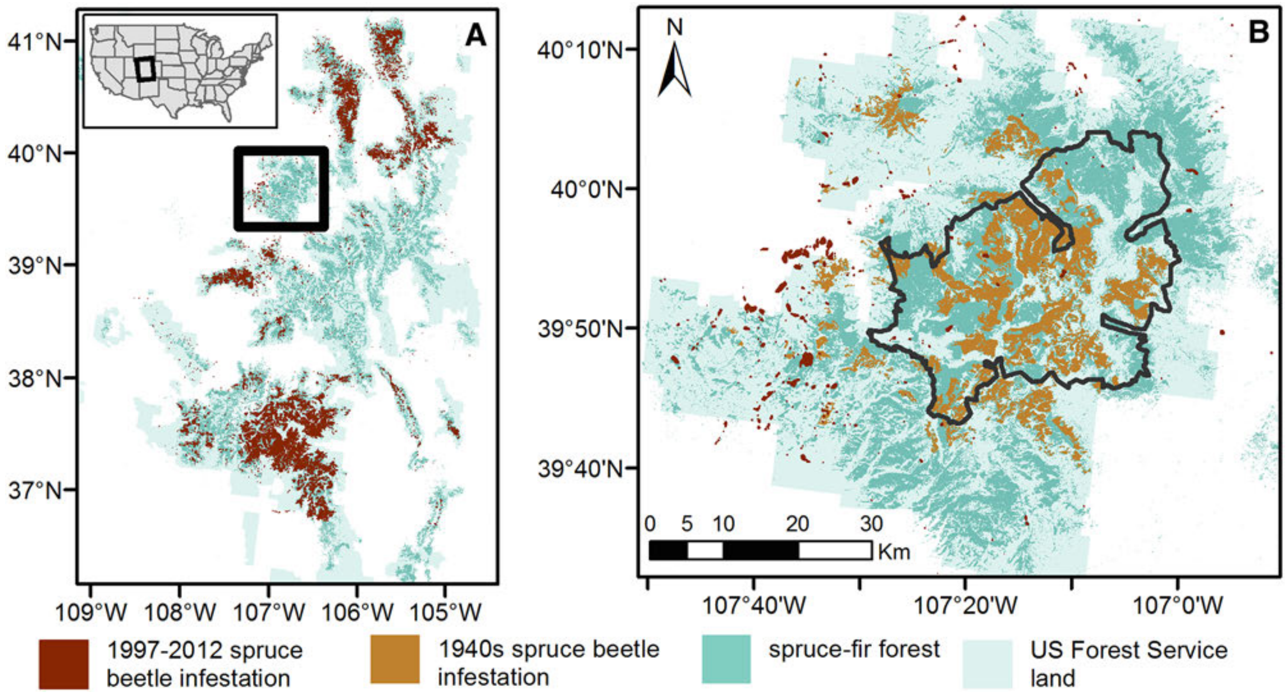

Fig 1. The larger study region and study area. (A) Map of the Southern Rocky Mountain study region displaying spruce-fir forests infested by spruce beetles during the 1997–2012 period. The upper left inset displays the location of the study region in relation to the entire United States. The black box indicates the study area displayed in B. (B) Map of the Flat Tops study area comprised of the Flat Tops Wilderness (black line) and adjacent areas of White River National Forest and areas infested by spruce beetles during the 1940s and 1997–2012 periods. Sources are given in Table 1.

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Data processing

We first obtained data on the occurrence of spruce-fir forest across the Southern Rocky Mountain study region (Table 1). Most vegetation cover-type datasets express only moderate (40–60%) overall agreement between field plot data and forest cover-type at 30 x 30 m spatial scale [39,40], thus we combined three datasets depicting the occurrence of spruce-fir forest [41]. For each vegetation dataset, we listed the presence of a spruce beetle host within a 990 x 990 m pixel, which approximates a stand scale [41]. We adopted a conservative criterion for mapping spruce-fir forest based on requiring its presence in all three datasets.

Next we obtained spatially explicit data on the presence of spruce beetle infestation over the time period from 1998–2013 from the United States Forest Service Region 2 ADS database [37]. Aerial Detection Surveys have been conducted annually in the Southern Rocky Mountains since 1994. To our knowledge robust accuracy assessments of ADS maps of spruce beetle infestation do not exist. However, accuracy assessments between ADS and ground reference data listing the presence/absence of bark beetle infestation in lodgepole pine show moderate-high agreement at coarse (500 m) spatial grains [39,40]. Thus we assumed ADS maps of spruce beetle infestation are most appropriate for assessing coarse-grain trends in presence/absence of infestation. To account for the ca. 1-year lag between initial infestation and ADS detection, we shifted the year of detection back one year to obtain year of attack [7]. Annual spatial polygon data listing the year of spruce beetle attack (1997–2012) were then converted to a 990 x 990 m grid listing the presence of spruce beetle infestation. Annual grids were then summed to obtain the cumulative area infested (1997–2012) and multiplied by a raster of spruce-fir presence to obtain a cross-validated grid of spruce beetle infestation [24].

Table 1. The GIS data layers and attributes used to examine linked spruce beetle disturbance.

Variable	Description	Data	Type	Resolution	Year
Damage casual agent	Name of forest pest or pathogen causing damage	Aerial Detection Survey Database [37]	Polygon	Compiled at 1:100,000 scale	1998–2013
1940s infestation	Presence /absence of 1940s spruce beetle infestation	Bebi et al. 2003 [5]	Polygon	Interpreted at 1:10,000 scale	Based on 1971 color & 1984 IR aerial imagery
R2VEG Cover type	Dominant life forms, based on Society of American Foresters classification	R2VEG [42]	Polygon	Interpreted at 1:24,000 scale	Based on 2002 aerial imagery
LANDFIRE EVT	Existing vegetation type, based on Nature Serve's ecological systems classification	LANDFIRE [43]	Raster	30 x 30 m	Based on 2001–2010 Landsat imagery
GAP Analysis Project Cover type	Primary cover type	GAP	Polygon	Interpreted at 1:100,000 scale	Based on 1989–1998 Landsat imagery
R2VEG Diameter at breast height	Tree DBH binned (cm): 1) <2.5, 2) 2.5–12.4, 3) 12.5–22.9, 4) 23–40.4, 5) $\geq$40.5	R2VEG [42]	Polygon	Interpreted at 1:24,000 scale	Based on 2002 aerial imagery
Southern Rocky Mountain Ecoregion	Level III Ecoregions	North America Ecoregions [44]	Polygon	Compiled at 1:250,000 scale	2013
August maximum temp	average monthly maximum temperature (°C)	PRISM [38]	Raster	4 x 4 km	1997–2012
Annual precipitation	average annual precipitation (mm)	PRISM [38]	Raster	4 x 4 km	1997–2012
March minimum temperature	average monthly minimum temperature (°C)	PRISM [38]	Raster	4 x 4 km	1997–2012
October minimum temperature	average monthly minimum temperature (°C)	PRISM [38]	Raster	4 x 4 km	1997–2012

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We also obtained a map of the presence of the 1940s infestation within the Flat Tops study area [5]. To our knowledge no other maps of the 1940s outbreak exist for the Southern Rocky Mountains. Maps of the 1940s infestation were developed from visual stereoscopic examination of 1971 color and 1984 IR aerial imagery (minimum mapping unit 5 ha). Stands mapped as infested by spruce beetles during the 1940s were defined as stands in which >30% of canopy trees were dead [5]. Spatial polygon data on the occurrence of the 1940s infestation was then converted to a 990 x 990 m grid listing the presence of spruce beetle infestation and multiplied by the raster of spruce-fir presence to obtain a cross-validated grid of spruce beetle infestation [24].

Finally we obtained spatial data on climate and forest structure variables, which were hypothesized to be important in predicting the occurrence of spruce beetle infestation. We obtained gridded monthly precipitation and temperature data from the Parameter-elevation Regressions on Independent Slopes Model (PRISM; [38]) (Table 1). To determine if warm and dry weather was associated with infestation, we calculated the 1997–2012 means of maximum August temperature and total annual precipitation, which previous research has shown to predict occurrence of spruce beetle infestations [25,29,45]. To determine if anomalously cold weather during the late autumn to early spring was associated with the presence/absence of infestation we calculated the 1997–2012 means of minimum October and March temperature, which are understood to inhibit infestation [25,45]. Next, we obtained vegetation layers depicting the mean diameter at breast height (DBH) size classes for the dominant canopy species, which were created from manual aerial photo interpretation of 1-m resolution color aerial photographs in 2002 (Table 1).

Determining the biophysical drivers of spruce beetle infestation

We used two methods to assess the biophysical variables driving the spatial variability in the occurrence of 1997–2012 spruce beetle infestation in the Southern Rocky Mountain study region. First, we used a spatial overlay approach [46,47], where spatial data on spruce beetle infestation were compared with spatially explicit climate and forest structure data (Table 1). We used spatial overlays to calculate the conditional probability of the presence/absence of spruce beetle infestation given each value of the independent variable. Conditional probability is a measure of the probability of the dependent variable (presence or absence of spruce beetle infestation) occurring given each value of the independent variable (biophysical variables). Continuous climate variables were first binned into four equal-interval classes [48]. Then we tabulated the number of 990 x 990 m pixels of all values of each independent variable that occurred in uninfested and infested areas and calculated the conditional probability of infestation. The null hypothesis is that spruce beetle infestation is independent of all values of each independent variable and thus observed conditional probabilities of infestation should equal conditional probabilities of uninfested stands. Our spatial overlays assessed entire populations and not samples. Thus all deviations between conditional probabilities are viewed as real differences between the datasets and statistical tests are not necessary. However, given that our spatial datasets exhibit classification error, we conservatively assumed that only differences greater than 10% are meaningful (e.g. [46]).

Second, to complement our conditional probability analysis of univariate relationships between biophysical predictors and the presence/absence of spruce beetle infestation, we used a Conditional Inference Framework (CIF; [49]) to assess multivariate relationships. CIF is similar to Random Forests [50] in that many classification trees are constructed by dividing the data into increasingly homogenous groups based on splits in the independent variables [49,51,52]. Classification trees are useful for detecting nonlinear relationships and interactions between variables [51]. In contrast to Random Forests where variable selection is based on the maximization of an information criterion (e.g. Gini coefficient), CIF uses conditional permutation-based significance tests to select variables [49]. This decreases selection bias in cases where independent variables have substantially different numbers of potential splits (e.g. categorical vs. continuous independent variables) [53], or where independent variables are correlated [54]. To evaluate the variables most important for predicting the presence/absence of spruce beetle infestation, we calculated conditional variable importance scores, a measure of each independent variable's contribution to overall model fit [54]. Because the calculation of conditional variable importance is computationally intensive, we randomly selected 2000 cases, stratified by spruce beetle infestation (1000 infested; 1000 uninfested). Model accuracy was asssed using overall accuracy and model sensitivity and specificity.

Effects of the 1940s spruce beetle infestation on the 1997–2012 infestation

To determine if the effects of the 1940s spruce beetle infestation on forest structure may affect the susceptibility of a stand to subsequent infestation in 1997–2012, we first used our model of the presence/absence of spruce beetle infestation to determine the relative importance of forest structure versus climate variables in constraining infestation within the Flat Tops study area. We tabulated the number of pixels within each model node and evaluated the relative importance of splits in climate vs. forest structure variables in predisposing the Flat Tops study area to infestation in 1997–2012. To this end, we calculated the percent of pixels in each model node for the entire Southern Rocky Mountain Study region and just the Flat Tops study area (Southern Rocky Mountain Study Region % | Flat Tops study area %). If the percent of pixels that met the condition were greatly different (>10%) for the Flat Tops study area than for entire Southern Rocky Mountain Study region, then that condition was interpret to be disproportionately important in constraining/promoting infestation within the Flat Tops study area.

Next, we coupled fine-scale field data with stand-level spatial data to determine if forest structure was altered by previous spruce beetle infestation. First, to test if large trees were depleted in areas of the 1940s infestation, we tabulated the number of 990 x 990 m pixels of all values of tree size that occurred in areas with and without 1940s infestation [55]. Then we calculated the conditional probability of the dominant tree size class (2.5–12.4, 12.5–22.9, 23–40.4, or ≥40.5 cm DBH) given the presence/absence of 1940s infestation.

Because the available GIS dataset depicting tree size is not species specific, we collected stand-scale (0.01 ha) field data to determine the delayed effects of a severe spruce beetle infestation on species composition. Field data were collected in the summer of 2013 at 7 sites (4 sites without evidence of 1940s infestation and 3 sites with evidence of severe spruce beetle infestation in the 1940s) across the Flat Tops study area. Plots were located using maps of the presence/absence of the 1940s infestation [5]. We field verified that our sites were located in areas affected by the 1940s infestation by locating large, dead, standing snags with spruce beetle galleries. At each site we collected data from a cluster of 10 randomly-located 100 m^2 plots. For each tree in the plot, we recorded the species, the diameter at breast height (DBH), and tree status (live, dead, or fallen). We then aggregated data for stands that experienced and did not experience severe spruce beetle infestation in the 1940s and calculated the 2000s density of live spruce and fir. We then compared 2000s stand structure and composition in stands uninfested and infested during the 1940s.

Finally, we used spatial data to assess if these structural differences between stands uninfested and infested during the 1940s affected the distribution of 1997–2012 infestation within the Flat Tops. We overlaid a 990 x 990 m grid of 1997–2012 spruce beetle infestation presence/absence with a 990 x 990 m grid of 1940s infestation presence/absence and calculated the area of overlap.

Results

Biophysical drivers of the 1997–2012 spruce beetle infestation

Across the Southern Rocky Mountain study region, spruce beetles infested approximately 15% of the spruce-fir zone over the period from 1997–2012 (areas mapped as infested in ADS surveys 1998–2013; Fig 1A). Over this time period, the Flat Tops study area has experienced very little infestation (2% of the spruce fir-zone recorded presence of infestation; Fig 1). While the annual area infested by spruce beetles across the Southern Rocky Mountain study region has been growing since 1998 [55], ADS data indicate that most spruce beetle activity in the Flat Tops study area occurred prior to 2005 (S1 Fig).

Across the Southern Rocky Mountain study region, spatial overlay analysis revealed meaningful differences between the conditional probabilities of uninfested and infested spruce-fir forest given climate and forest structure variables (Fig 2). Contrary to expectations, spruce beetle infestation was less likely in areas with high maximum August temperatures (≥19.5°C; Fig 2B). However this difference was only meaningful in areas where the average maximum temperature was greater than ≥20.5°C. Areas with cooler maximum August temperatures (<18.5°C) were more likely to be infested. Also contrary to expectation, areas with high annual precipitation (≥ 1050 mm/year) were more likely to experience spruce beetle infestation, while areas with moderately low annual precipitation (650–849 mm/year) were less likely to experience infestation (Fig 2A). There were no meaningful differences between the probabilities of uninfested and infested forest given any of the four classes of minimum March temperature or minimum October temperature (Fig 2C and 2D).

We also found that forest structure differed between forests uninfested and infested by spruce beetles in 1997–2012 (Fig 2E). Spruce beetle infestation was more likely to occur in areas with

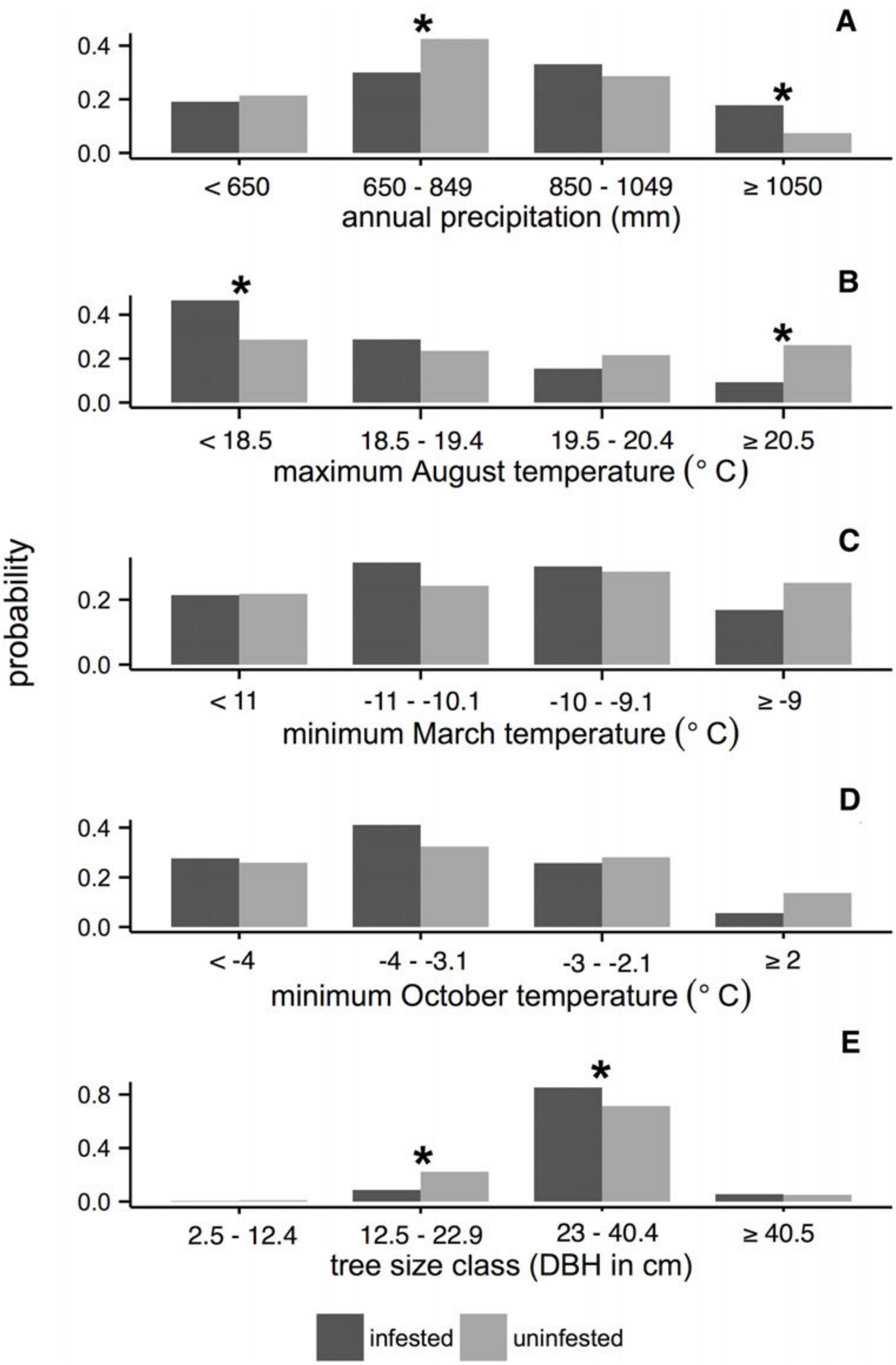

Fig 2. Conditional probabilities of the presence/absence of spruce beetle infestation (1997–2012) given selected bioclimatic variables in the Southern Rocky Mountains study region. (A) annual precipitation, (B) maximum August temperature, (C) minimum March temperature, (D) minimum October temperature, and (E) tree size class for uninfested and infested stands. Dark gray bars indicate conditional probability of spruce beetle infestation given that value of a bioclimate variable across the Southern Rocky Mountain study region. Light gray bars indicate the conditional probability of uninfested forest. The asterisk symbol (*) above a pair of bars indicates a meaningful difference between conditional probability of uninfested and infested forest (i.e. difference > 10%, see *Methods* for more description). Note y-axes extend over different ranges.

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large diameter trees (≥ 23 cm DBH; Fig 2E). For stands with smaller diameter trees (< 23 cm DBH), the probability of infestation was < 0.22 (Fig 2E).

The multivariate model of 2000s spruce beetle uninfested and infested forest performed reasonably well. The CIF model correctly predicted 809 of the 1000 pixels with spruce beetle infestation (i.e. sensitivity = 0.81), and correctly predicted 819 of the 1000 pixels without spruce beetle infestation (i.e., specificity = 0.82). Variables important in predicting 2000s spruce beetle infestation included maximum August temperature, annual precipitation, and tree size class (Fig 3A). Spruce beetle infestation was unlikely to occur in areas with maximum August temperatures above 20.3°C (probability of infestation = 0.276). Infestation was particularly unlikely when temperatures exceed 21.6°C (probability of infestation = 0.164; Fig 3B). However, more than 75% of the study area was characterized by 1997–2012 mean maximum August temperatures cooler than 20.3°C (S2 Fig). In these areas, spruce beetle infestation was particularly like to occur in areas with large trees (≥ 23 cm DBH; Fig 3A) and high precipitation (>1063 mm/year) (Fig 3B and S2 Fig).

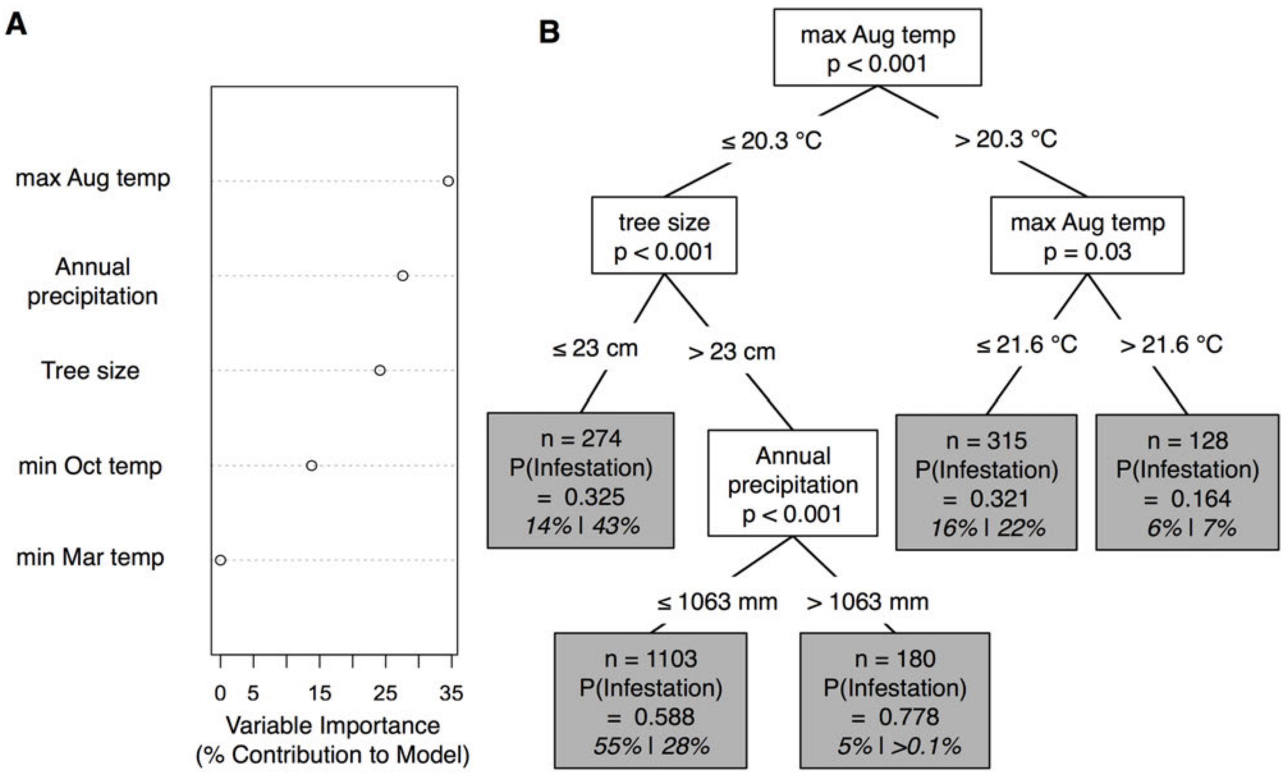

Fig 3. Results from conditional inference forest analysis of the presence/absence of spruce beetle infestation with climate and forest structure data in the Southern Rocky Mountain study region. (A) Conditional variable importance for the five biophysical variables used to model the occurrence of spruce beetle infestation across the Southern Rocky Mountain study region. Conditional variable importance scores were calculated following the Random Forest principle of mean decrease in accuracy and then transformed to express the contribution of each variable to the overall model. Higher values indicate variables are more important to the classification. Conditional variable importance scores represent 1000 model runs. All trees were built using a random sample of 2000 cases, stratified by the presence/absence of spruce beetle infestation (1000 infested and 1000 uninfested). Overall prediction accuracy is 81%. (B) A classification tree for determining the presence of spruce beetle infestation from uninfested spruce-fir stands across the Southern Rocky Mountains study region. On the tree, if condition is satisfied, proceed to the left of the tree. Tree nodes (gray boxes) describe the number of pixels across the entire Southern Rocky Mountain study region that meet the condition and the probability of spruce beetle infestation. The gray boxes also list the percent of pixels that meet the conditions for the entire Southern Rocky Mountain Study region and just the Flat Tops study area (Southern Rocky Mountain Study Region % | Flat Tops study area %). If the percent of pixels that meet the condition are greatly different (>10%) for the Flat Tops study area than for entire Southern Rocky Mountain Study region, then that condition is disproportionately important in constraining/promoting infestation within the Flat Tops study area.

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Effects of the 1940s spruce beetle infestation on the 1997–2012 infestation

Applying the decision tree to the pixels within the Flat Tops study area provided insight into the biophysical predictors important in constraining infestation in the Flat Tops study area. Across the Flat Top study area about 29% of pixels were characterized by 1997–2012 mean maximum August temperatures unsuitable for infestation (<20.3°C; Fig 3B and S2 Fig). An additional 43% of the pixels within the Flat Tops study area were characterized by small diameter trees (<23 cm DBH), which inhibit infestation (Fig 3B and S2 Fig). In comparison to the entire Southern Rocky Mountain study region, the percent of pixels with small diameter trees (<23 cm DBH) in the Flat Tops study area was three times greater (43% vs. 14%, for the Flat Tops study area and entire Southern Rocky Mountain study region, respectively; Fig 3B and S2 Fig). As a result, the percentage of pixels that were split based on annual precipitation was far lower for the Flat Tops study than the Southern Rocky Mountain study region.

Within the Flat Tops study area, comparison of forest structure of 990 x 990 m in areas uninfested and infested by the 1940s infestation indicates that infested stands are characterized by smaller tree sizes (12.5–22.9 cm DBH) 60 years following infestation (Fig 4). This coarse-scale finding based on mapping from aerial photographs (Table 1) is supported by stand-level field measurements. During the 1997–2012 period of spruce beetle infestation, field data revealed that in comparison with stands infested during the 1940s, stands not infested in the 1940s had consistently higher densities of spruce in all size classes including the largest class (i.e. $\geq$ 40.5 cm DBH; Fig 5). In contrast, 60 years following the 1940s spruce beetle outbreak

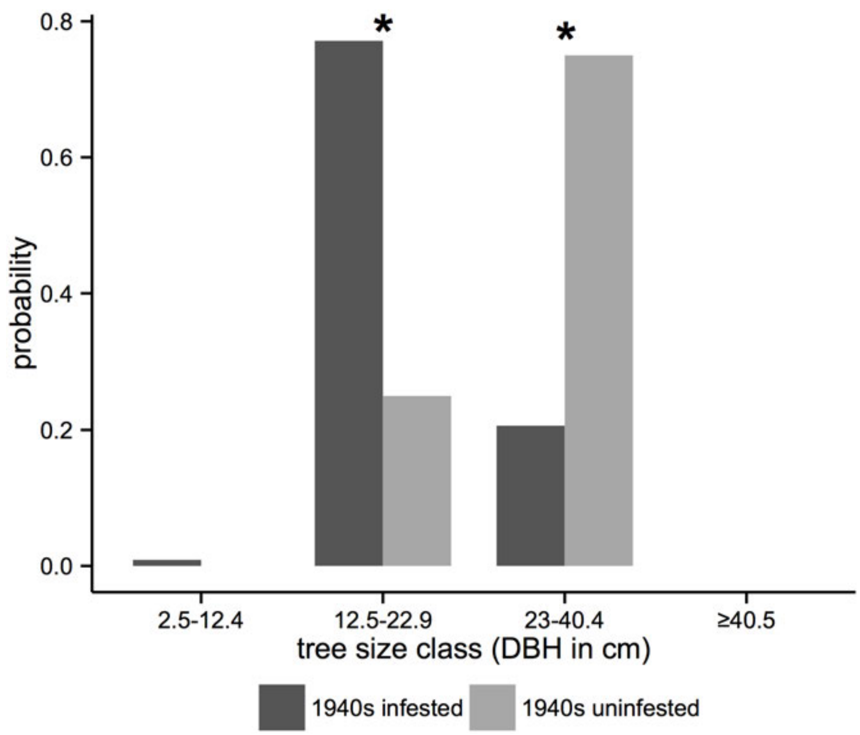

Fig 4. The conditional probability of current dominant tree size given the presence or absence of the 1940s spruce beetle infestation in the Flat Tops study area. Dark gray bars indicate the probability that a 990 x 990 m spruce-fir pixel is infested by spruce beetles; light gray bars indicate the probability a pixel is uninfested. The asterisk symbol (*) above a pair of bars indicates a meaningful difference between conditional probability of uninfested and infested forest.

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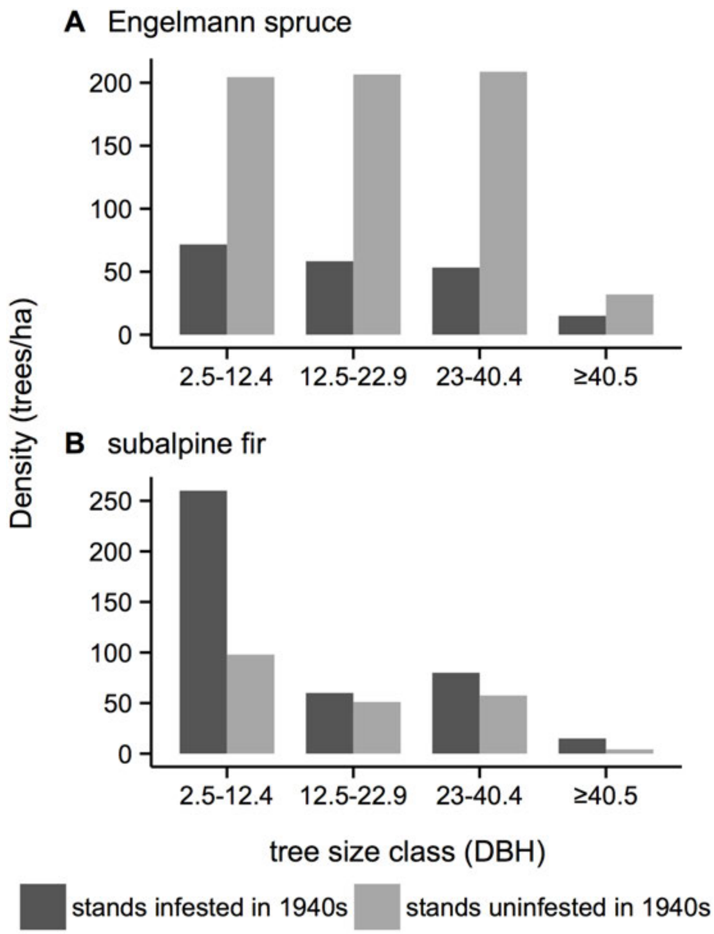

Fig 5. Current (2000s) tree size class distributions in stands uninfested and infested during the 1940s infestation within the Flat Tops and adjacent areas of White River National Forest. Data represent the aggregate of all plots (stands uninfested during the 1940s outbreak, n = 4 sites each with 10 ca. 100 m^2 plots; stands infested during 1940s outbreak, n = 3 sites each with 10 ca. 100 m^2 plots).

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subalpine fir was more abundant in all size classes in stands infested during the 1940s outbreak compared to uninfested stands. Concordantly, we found no overlap between areas infested during the 1940s and the 2000s infestation (Fig 1A). Within the Flat Tops region only three 990 x 990 m pixels were infested in 1997–2012, but none of those overlapped with the 254 pixels infested in the 1940s. Instead, all pixels infested in 1997–2012 were located in areas with large diameter trees ($\geq$ 23 cm DBH).

Discussion

Across the Southern Rocky Mountain study area, spruce beetle infestation was more likely to occur in areas with cool to moderately warm mean maximum August temperatures and higher amounts of annual precipitation. Although these results at first glance seem counter-intuitive given the importance of drought in triggering spruce beetle outbreaks [28–30,56], our results are spatial associations of infestation with mean conditions rather than temporal associations with drought events measured as departures from longer-term average conditions. While bark beetles preferentially attack drought-stressed trees [28, 29], our results describe habitat suitability for spruce beetle, which clearly is greater at the cooler and wetter sites where spruce is more common. In contrast, warmer sites are likely to be characterized by greater proportions of non-host species (e.g. lodgepole pine) and provide less potential for spruce beetle outbreak. Overall, we interpret the association of spruce beetle infestation with cooler and wetter sites as being explained primarily by the greater presence of host species at those sites.

Across the Southern Rocky Mountain study area, spruce beetle infestations have occurred overwhelmingly in spruce-fir stands dominated by large trees ($\geq$ 23 cm DBH). This corresponds with empirical results from Grand Mesa National Forest in western Colorado, which showed early 2000s spruce beetle infestation was significantly more likely in spruce larger than 24 cm DBH [57]. Spruce beetles prefer large trees, which provide both higher amounts of phloem for beetles to feed upon and thicker bark that increases overwinter survival rates [32].

While both climate and forest structure interacted to drive the occurrence of spruce beetle infestation across the Southern Rocky Mountains, our data suggest that the current infestation in the Flat Tops was severely constrained by a low proportion of large trees ($\geq$ 23 cm DBH). Our model suggests that climate variables were conducive to bark beetle infestation across most of the Flat Tops study area. Relative to the entire Southern Rocky Mountains (inclusive of the Flat Tops), infestation in the Flat Tops was severely constrained by forest structure. The paucity of large diameter trees within the Flat Tops study area was a result of a severe spruce beetle outbreak that occurred 60 years ago. Stands infested during the 1940s in Flat Tops were notably depleted of large spruce relative to uninfested stands. Given the preference of spruce beetles for large diameter spruce and relative absence of large diameter spruce in areas affected by the 1940s infestation, it is not surprising that we found no overlap between areas of current infestation and areas affected by the 1940s infestation. These results support the hypothesis that stands affected by severe spruce beetle infestation are less susceptible to infestation c. 60 years later due to a decrease in large diameter spruce.

The 1940s spruce beetle infestation in northwestern Colorado was most severe in the Flat Tops area, where three-quarters of the 1940s spruce beetle-induced tree mortality occurred [32]. Nearby spruce-fir forests in Grand Mesa National Forest also experienced 1940s spruce beetle infestation, however it was significantly less severe [32,58]. For instance, the basal area of beetle-killed spruce was ca. 4—7.5x greater in the Flat Tops than in Grand Mesa [58]. In contrast, the 1997–2012 spruce beetle infestation has affected 2% of the spruce-fir forest in Flat Tops and 19% of Grand Mesa's spruce-fir forest [34]. This suggests that the 1940s infestation in Grand Mesa was not severe enough to cause significant host depletion and thus Grand Mesa forests were much more susceptible to the 1997–2012 infestation.

Our study is notably limited by the availability of spatial datasets of both the 2000 and 1940s spruce beetle infestation. In particular we note that comparisons between these two datasets may be limited by the different methods used to map spruce beetle infestation (interpretation of aerial photography vs. aerial sketch mapping). However, our ability to accurately model the 1997–2012 infestation from a few ecologically meaningful biophysical predictors and the agreement between field data and maps of the 1940s outbreak suggest these datasets were appropriate for coarse assessment of the linkage between spruce beetle outbreaks. Subsequent analyses with datasets depicting severity of infestation at a fine spatial resolution would serve to advance our understanding of this linkage, however to our knowledge no such datasets exist for the Southern Rocky Mountains.

The findings of the current study indicate that at a broad spatial scale, severe spruce beetle outbreaks are linked disturbances (sensu [3]) at least over the 60-year period considered in our study. We suggest that the host depletion feedback not only may cause infestation collapse (sensu [32]), but may enhance ecological resistance (sensu [59]) of beetle-affected systems to spruce beetle infestation through long lasting effects of host depletion. Given that predictions of future beetle disturbance from climate-driven beetle population models do not incorporate process dynamics of disturbance-caused tree mortality and forest recovery [60], our results underscore the need for additional research on forecasting future forest dynamics, which may affect host availability for bark beetle infestations. In particular, the dampening effect of the 1940s spruce beetle infestation on the spread of the early 2000s infestation in the Southern Rocky Mountains implies that future infestations in the 21st century may be similarly restricted by disturbance-caused depletion of susceptible hosts.

Most previous studies of linked disturbances in the coniferous forests of the Rocky Mountain region have addressed how previous fire affects subsequent bark beetle outbreaks [61,62] or how previous bark beetle outbreaks alters the probability, extent or severity of subsequent fire [5,15,20,36]. To our knowledge this is the first broad-scale analysis of how prior bark beetle outbreak affects susceptibility to subsequent bark beetle outbreak. Our findings of a dampening effect of the 1940s spruce beetle outbreak on susceptibility to spruce beetle infestation 60 years later highlights the need for incorporating the process dynamics of tree growth and mortality in predictive modeling of the likelihood of bark beetle outbreaks under future climate scenarios. Simulation modeling of the probability of future insect outbreaks based on climate suitability for the growth of the insect populations has been important in identifying likely trends over relatively short time periods. However, our results show that even at a time scale of 60 years, failure to incorporate negative feedbacks into prediction of future bark beetle outbreaks is likely to over-predict the extent or severity of future outbreaks and by implication under-estimate forest resistance to altered disturbance regimes under climate change.

Supporting Information

S1 Fig. Time series displaying the percentage of spruce-fir forest infested by spruce beetles. Data is shown for the Southern Rocky Mountain Ecoregion (inclusive of the Flat Tops) and only in the Flat Tops. For each region, the percent area was calculated by the determining the number of 990 x 990 m pixels within the spruce-fir zone identified as infested by the United States Forest Service in annual Aerial Detection Surveys (ADS) and dividing it by the total number of spruce-fir pixels.
(TIFF)

S2 Fig. The importance of biophysical predictors in promoting spruce beetle infestation the Southern Rocky Mountain study region. Maps of (A) mean maximum August temperature (1997–2012), (B) mean annual precipitation (1997–2012), and (C) tree size class. The probability of infestation (derived from the classification tree in Fig 3B) is indicated by pixel color. Dark green indicates a probability of infestation <0.3, light green indicates a probability of infestation of 0.3–0.49, dark yellow indicates a probability of infestation 0.5–0.69, and dark brown indicates a probability of infestation >0.7. Sources are given in Table 1.
(TIFF)

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Author Contributions

Conceived and designed the experiments: SJH TTV NM DK. Analyzed the data: SJH. Wrote the paper: SJH TTV NM DK. Collected field data: NM.

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Conservation Planning for US National Forests: Conducting Comprehensive Biodiversity Assessments

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The US Forest Service has proposed new regulations under the National Forest Management Act that would replace a long-standing requirement that the agency manage its lands "to maintain viable populations of existing native and desired non-native vertebrate species." In its place, the Forest Service would be obligated merely to assess ecosystem and species diversity. A landscape assessment process would rely on ecosystem-level surrogate measures, such as maps of vegetation communities and soils, to estimate species diversity. Reliance on such "coarse-filter" assessment techniques is problematic because there tends to be poor concordance between species distributions predicted by vegetation models and observations from species surveys. The proposed changes would increase the likelihood of continued declines in biodiversity and fail to address the original intent of the act. We contend that responsible stewardship requires a comprehensive strategy that includes not only coarse-filter, ecosystem-level assessment but also fine-filter, species-level assessments and viability assessments for at-risk species.

Keywords: forestry, forests, management, policy, conservation

The US National Forest Management Act (NFMA) is an essential statute for maintaining biotic diversity on 192 million acres of national forests and national grasslands. It was enacted in 1976 as reform legislation in response to environmental impacts from timber harvest, grazing, and mining on national forest lands, which the public and Congress found increasingly unacceptable (Wilkinson and Anderson 1987). Among many provisions for resource protection, a primary emphasis was the protection of individual species. The statutory language of NFMA requires management of the national forests and grasslands to "provide for diversity of plant and animal communities based on the suitability and capability of the specific land area in order to meet overall multiple-use objectives" (16 US Code 1604[g][3][B]). Since 1982, the regulations governing implementation of NFMA have addressed this diversity provision by requiring that "fish and wildlife habitat shall be managed to maintain viable populations of existing native and desired non-native vertebrate species in the planning area" (36 Code of Federal Regulations, sec. 219.19, app. 13). Revisions to NFMA regulations adopted in 2000 retained the requirement for viable populations and expanded it to include all plant and animal species (Federal Register 65 [218]: 67514–67581).

Although NFMA has remained essentially unchanged since its enactment, the US Forest Service has now proposed regulations that eliminate an explicit population viability requirement and that restrict management responsibility to vertebrates and vascular plants (Federal Register 67 [235]: 72770–72816). The proposed regulations require only a "hierarchical, sequential approach to consider and assess both ecosystem diversity and species diversity" and that the Forest Service "identify species for which substantive evidence exists that continued persistence in the planning or assessment area is at risk, specific risks or threats to these species, and measures required for their conservation or restoration" (Federal Register 67 [235]: 72801). No specific language to compel species-level analyses of viability has been proposed. Moreover, the proposed regulations would subsume the existing species conservation requirement into a landscape assessment process that would use a variety of unproven

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ecosystem-level surrogates to estimate species diversity without necessarily examining the condition or status of individual species. Although not explicitly stated, the substance of these proposed regulations hinges on two underlying assumptions: (1) Land-use planning that relies solely on such "coarse-filter" (Hunter et al. 1988) approaches to assess the distributions and status of ecological communities is adequate to assess how well the needs of all their constituent species will be met, and (2) the uncertainty that accompanies indirect assessments of species status provided by coarse-filter tools is acceptable because species-level assessments are too difficult or too expensive to implement. These assumptions are not only counter to current understanding of the role and dynamics of specific species in sustaining ecosystem processes (e.g., Kinzig et al. 2002), they also negate the nature and appropriate role of population viability analyses in land-use planning.

Inadequacies of assessments employing only a coarse-filter approach

To understand the functioning of any complex system, it is necessary to identify and attempt to elucidate the parts that it comprises. For ecological systems, the most fundamental "parts" are species. Sir Arthur Tansley originally defined ecosystems as biotic communities or assemblages of species and their physical environment in specific places (Tansley 1935). Directly contradicting this view of ecosystems as collections of interacting species, the proposed regulations focus resource assessments almost entirely on vegetation types and successional stages, geology, landforms, and soils. The logic behind this coarse-filter approach is that the majority of species can be protected by conserving examples of natural vegetation communities, obviating the need to evaluate the status of each species individually (Noss 1987, Noss and Cooperrider 1994).

The original intent of coarse-filter approaches to landscape planning was to provide distribution maps of land cover that could be used to inform the conservation of entire species assemblages, including communities of interacting or potentially interacting species (Jennings 2000, Groves et al. 2002). Broadscale applications of coarse-filter methods have relied on ecoregional classifications determined by a variety of measures of climate, substrate, and plant composition. However, they commonly and often exclusively default to dominant vegetation, because vegetation types can be assessed by remote-sensing technologies and have been linked, using general habitat models, to the distributions of many vertebrate species (Scott et al. 1993). For example, recent planning efforts by the Forest Service for 4.4 million hectares of public forests and grasslands in the Sierra Nevada of California assessed the effects of various management alternatives on vertebrate species using wildlife–habitat relationship models (Mayer and Laudenslayer 1988) to classify habitats based on three attributes—dominant vegetation type, successional stage, and canopy closure. When these models were coupled with a vegetation growth and yield model (Davis and Johnson 1987), they allowed a comparison of how competing forest management scenarios would be likely to affect future wildlife populations (Forest Service 2001).

Coarse-filter approaches to assess the viability of species for land-use planning purposes can provide cost-efficient, indirect methods of assessing species distributions, but to assess the viability of species, at least three assumptions must hold true: (1) Attributes that define the coarse filter (i.e., dominant vegetation types) are sufficient and reliable surrogates for habitat and can effectively predict the occurrence of a given species; (2) managing coarse-filter attributes will address the factor(s) currently limiting abundance, density, and persistence of each species; and (3) the spatial resolution of the coarse filter matches the scale at which given species respond to environmental heterogeneity. Although these assumptions may be valid for some species in many circumstances, especially species that are small-bodied, abundant, and tightly linked to a particular vegetation community, the likelihood that the assumptions are met for all, or even most, species in an assemblage is low. For that reason, landscape planning employs "fine-filter" assessments, which are based on direct measures of the status and trends of individual species or on models of population viability to evaluate the needs of species at risk of decline.

The utility of the coarse-filter approach has been tested for many individual species with equivocal success (see Scott et al. [2002]). In general, there has been poor concordance between predicted and observed distributions. Commission errors (false positives, or predictions that a species is present when it is absent) have been shown to be more common than omission errors (false negatives, or predictions that a species is absent when it is present) at spatial scales appropriate to regional conservation planning—for example, vertebrates in the state of Maine and in national parks in Utah and breeding birds in California (Edwards et al. 1996, Boone and Krohn 1999, 2000, Garrison et al. 2000, Garrison and Lupo 2002, Robertson et al. 2002). Thus, coarse-filter assessments often overestimate the presence and, presumably, the viability of species on the planning landscape.

Only by increasing the resolution of the coarse filter (which reduces the area predicted to be suitable habitat for the species), as well as the number of land-cover types (usually by stratifying the vegetation communities more finely), can commission and omission errors be simultaneously reduced (Karl et al. 2000). Prediction errors are also related to ecological attributes of a species: Species that are rare, colonial, or habitat specialists, or that have small home ranges, are most likely to be misclassified (Karl et al. 2000, Scott et al. 2002). The misclassified groups of species usually include those most likely to be at risk of population declines or extirpation—that is, those that should be targets of conservation planning efforts (McKinney 1997). In sum, these prediction errors suggest that employing a coarse-filter approach alone is inadequate to meet NFMA requirements to provide for the diversity and viability of plant and animal communities.

Integrating the fine filter with population viability analysis

Coarse- and fine-filter approaches to conservation planning differ in both the extent and resolution of measurement employed and the targeted level of biological organization. In general, mapped coarse-filter attributes reflect higher-level processes and patterns that arise, for example, from disturbance processes that operate across entire landscapes. For pragmatic reasons, coarse-filter attributes considered during the planning process are often those that can be measured inexpensively using remote imagery. Coarse filters rarely will accurately reflect the complex and dynamic habitat requirements of any individual species. In contrast, a fine filter makes measurements directly at the species level for the subset of species whose habitat requirements were not captured by the attributes that define the coarse filter.

Neither coarse- nor fine-filter assessments alone can prescribe the extent or area of habitat necessary to maintain viable populations of plant and animal species on the landscape. Many rare and declining species are limited primarily by the availability of suitable habitat (Wilcove et al. 1998), and the viability of such species depends to a great extent on how much of their habitat is conserved. Population viability analysis (PVA) is an in-depth method of fine-filter assessment used to evaluate habitat loss or similar risk factors for specific species (Boyce 2002, Shaffer et al. 2002).

An assessment approach that includes both coarse and fine filters and PVA was recommended by the Committee of Scientists to the US Forest Service and incorporated into the 2000 NFMA regulations (COS 1999). In addition to rare and at-risk species, the committee recommended that two groups of species be evaluated using fine filters—those that provide comprehensive information on the state of a given ecosystem (indicator species) and those that play significant functional roles in ecosystems (focal species). The latter category includes species that contribute disproportionately to the transfer of matter and energy (e.g., keystone species), structure the environment and create opportunities for additional species (e.g., ecological engineers), or exercise control over competitive dominants, thereby promoting increased biotic diversity (e.g., strong interactors). Thus, fine-filter assessments might be needed for 10 to 50 of the 200 to 1100 species typically evaluated in regional planning efforts carried out by the Forest Service and may need to include select invertebrates as well as vertebrates and plants.

Formal PVAs are needed only for species in decline or at high risk or for species with such functional significance that their loss might have unacceptable ecological effects. Many methods of viability assessment exist to accommodate diverse sources and amounts of data (Beissinger and Westphal 1998, Andelman et al. 2001). All methods explicitly or implicitly require some sort of model that relates population dynamics to environmental variables, including variables affected by management. The range of available methods offers a tradeoff between complexity of analysis and generality of results.

Population viability analysis is neither inherently difficult nor expensive, but it does require thoughtful model choice and construction and good judgment in the implementation of analyses. Perhaps the most demanding aspect of building realistic PVA models for assessment of alternative management scenarios is acquisition of sufficient data to yield accurate and precise parameter estimates (Beissinger and Westphal 1998). These models then permit reliable assessments of alternative management scenarios (Noon and McKelvey 1996). The choice of models and data collection methods depends in part on the life history characteristics of the species to be assessed, the quality and quantity of existing data, the time and money available for additional data acquisition, and the resolution and extent of analysis (Beissinger and Westphal 1998, Andelman et al. 2001). A method that uses a formal mathematical model of analysis is often preferable to less quantitative methods for analyzing viability when there is sufficient knowledge of demography, dispersal, habitat use, and threats.

Currently, population viability analyses are required to address the viability requirements of NFMA. In the context of the act, viable populations consist of "self-sustaining and interacting populations that are well distributed through the species' range. Self-sustaining populations are those that are sufficiently abundant and have sufficient diversity to display the array of life history strategies and forms to provide for their long-term persistence and adaptability over time" (Federal Register 65 [218]: 67580–67581). Many population attributes included in this definition can be evaluated using population viability analyses, but they cannot be addressed solely through the application of coarse-filter analyses.

A scientifically credible approach to national forest planning

An expert panel convened by the National Center for Ecological Analysis and Synthesis, at the request of the Forest Service, concluded that "viability assessment is an essential component of ongoing forest management and forest planning processes. A variety of methods can and should be incorporated into viability assessments" (Andelman et al. 2001, p. 136). A scientifically credible approach to management of a diversity of plant and animal communities in US national forests and national grasslands combines coarse-filter and fine-filter approaches to identify conservation targets, including the judicious use of PVA for focal species and species at risk. Scientifically valid and pragmatic management does not require that the status of all species be directly assessed. But failure to detect declining species and to address the putative threats to their persistence leaves only the prohibitive provisions of the Endangered Species Act to serve as a safety net.

Although coarse-filter, fine-filter, and PVA assessment tools are imperfect, their weaknesses are sufficiently understood that the information they provide is, on balance, useful, and the Forest Service's failure to require their use is irresponsible. Insights provided by the use of these tools will inform managers about the condition of the ecosystems they are charged with protecting and the likely consequences of the management decisions they are empowered to make. Acting on these insights to change management practices when needed will aid biodiversity conservation and enable the Forest Service to meet its stewardship responsibilities.

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ECOSYSTEMS

Fire Severity in Conifer Forests of the Sierra Nevada, California

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Abstract

Natural disturbances are an important source of environmental heterogeneity that have been linked to species diversity in ecosystems. However, spatial and temporal patterns of disturbances are often evaluated separately. Consequently, rates and scales of existing disturbance processes and their effects on biodiversity are often uncertain. We have studied both spatial and temporal patterns of contemporary fires in the Sierra Nevada Mountains, California, USA. Patterns of fire severity were analyzed for conifer forests in the three largest fires since 1999. These fires account for most cumulative area that has burned in recent years. They burned relatively remote areas where there was little timber management. To better characterize high-severity fire, we analyzed its effect on the survival of pines. We evaluated temporal patterns of fire since 1950 in the larger landscapes in which the three fires occurred. Finally, we evaluated the utility of a metric for the effects of fire suppression. Known as Condition Class it is now being used throughout the United States to predict where fire will be uncharacteristically severe. Contrary to the assumptions of fire management, we found that high-severity fire was uncommon. Moreover, pines were remarkably tolerant of it. The wildfires helped to restore landscape structure and heterogeneity, as well as producing fire effects associated with natural diversity. However, even with large recent fires, rates of burning are relatively low due to modern fire management. Condition Class was not able to predict patterns of high-severity fire. Our findings underscore the need to conduct more comprehensive assessments of existing disturbance regimes and to determine whether natural disturbances are occurring at rates and scales compatible with the maintenance of biodiversity.

Key words: Condition Class; ecological restoration; Jeffrey and ponderosa pine; fire rotation interval; fire severity; fire spread; mixed conifer forests; spatial heterogeneity.

Introduction

The diversity of species in ecosystems is linked to natural disturbances and the environmental heterogeneity they create (Connell 1978; Huston 1979). However, managing the rates and scales of disturbance processes to allow for natural levels of environmental heterogeneity has its inherent risks and difficulties. This is particularly true for large disturbances that have profound influences on ecosystem structure, function, and composition (Turner and Dale 1998). Thus, although natural disturbances are vital to ecosystem integrity, maintaining their full range of variability is often at odds with management (Holling and Meffe 1996). How can disturbance-mediated environmental heterogeneity be most effectively maintained or restored where it has been suppressed over large areas? How can we recognize the levels and types of disturbance and heterogeneity that are appropriate for maintaining biodiversity? Here we explore these questions by focusing on the management of fire. Enormous resources are expended

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worldwide in efforts to manage this important disturbance or restore its effects.

To date there has been little direct assessment of how fire-mediated spatial heterogeneity might be restored or managed for in many fire-prone systems, such as the conifer forests of western North America (Rocca 2004). In many of these areas management policy is focused on the use of mechanical treatments to modify forest structure as a means of counteracting the effects of fire suppression. These efforts are controversial and are often not based on a sound understanding of the ecological role of fire as a disturbance process and the methods needed to restore its effects (Johnson 2003; DellaSala and others 2004). Perhaps nowhere in western North America has the appropriateness of structure-versus process-based forest management approaches been more controversial than in the conifer forests of the Sierra Nevada Mountains of California, USA (Stephenson 1999; Miller and Urban 2000).

Since the 1850s, grazing and fire suppression have reduced fire frequencies in the forests of the Sierra Nevada (Stephenson 1999; Miller and Urban 2000). The prevailing management view is that, because of fire exclusion, forest fires in the Sierra, which once varied considerably in severity, are now almost exclusively large, high-severity, stand-replacing events (Skinner and Chang 1996). As a consequence, an extensive program for the management of national forest lands was initiated in 2004. Its goal is to modify the structure of 283,000 ha of vegetation per decade, mainly in the dominant mixed conifer forests (USDA 2004). However, the actual severity of contemporary fire on these lands has yet to be analyzed to determine how well the prevailing view of dramatically increased fire severity and decreased heterogeneity is supported by empirical evidence.

Under the provisions of the National Forest Management Act of 1976, the national forests in the Sierra Nevada and throughout the United States are directed to ''provide for diversity of plant and animal communities.'' Natural variation and the maintenance of biodiversity in ecosystems can be assessed based on the concept of ecological integrity. ''Ecological integrity'' refers to ecosystem wholeness, including the occurrence of ecological processes such as natural disturbances at appropriate rates and scales to maintain natural levels of biodiversity (Karr 1991; Angermeier and Karr 1994). To determine the appropriateness of process-based versus structure-based management approaches for the maintenance biodiversity, we need to understand how ecological integrity is affected by contemporary fires. Thus, one of our primary objectives is to evaluate the rates and scales of contemporary fire as a disturbance process and assess their appropriateness in the context of ecological integrity.

To pursue this objective, we analyzed fire-severity data from the three largest fires that have occurred in the Sierra Nevada since 1999, accounting for most of the area burned over this time. These fires occurred in landscapes where timber harvest and silvicultural activities have been uncommon. After these burns, fire severity was classified by multi–US agency Burned Area Emergency Rehabilitation (BAER) teams. The BAER fire-severity data are derived from pre- and post-burn satellite and photo images and are used to map the effects of the fire on overstory vegetation canopy. We supplement these data with measures of ponderosa and Jeffrey pine mortality taken on the ground in areas of high-severity as defined by BAER. These pines have been harvested in many areas, and there is considerable interest in restoring their natural abundance (SNEP 1996). To gain further insight into the rates and scales of disturbance by fire under current management, we also evaluated temporal patterns of burning since 1950 in the broader landscapes in which the three fires occurred. Fire suppression has been mechanized in its current form since about 1950.

Another of our objectives was to evaluate the effectiveness of a national approach for the assessment of fire regimes and to discover how they have changed. The current basis for this approach, now used throughout the United States, is Fire Regime Condition Class (hereafter Condition Class), (Hann and Bunnell 2001); see also http://www.frcc.gov). It is an index that Estimates departure from reference conditions in vegetation, fuels, and disturbance regimes. In the national forests of the Sierra Nevada, Condition Class has been based on the number of fires estimated to have been exclude in the landscape due to fire suppression. Considerable research has revealed that historically Sierran forests were burned mostly by surface fire, but that this regime has decreased dramatically due to fire suppression (Caprio and Swetnam 1995; Skinner and Chang 1996). Condition Class predicts that these circumstances will lead to a dramatic increase in fire severity and place forest ecosystems at high risk losing key components due to fire (Hann and Strohm 2003).

A new approach to mapping departure from reference conditions, LANDFIRE, is currently under development (http: www.landfire.gov). In addition to Condition Class, it relies on the rapid assessment and mapping of wildland fuels to identify potential conditions that promote fire. The use of approaches that map departure from historic reference conditions in management is advancing rapidly. In the United States, 25 million ha have been identified for fuel treatments based on Condition Class (Brown and others 2004). Thus, it is especially timely now to evaluate the efficacy of approaches that map departure from historic reference conditions as a means of predicting fire severity.

Methods

Study Areas

The Sierra Nevada Mountains of California are a high-elevation (3000–4000 + m tall), 8-million-ha, north/south-trending mountain range (Figure 1, inset). They are forested primarily by conifer vegetation. We evaluated fire severity in the three largest burns in the Sierra since 1999—the McNally, Manter, and Storrie fires. Older fires lacked comparable fire-severity data in digital form. Smaller burns since 1999 in the main part of the Sierra occurred in areas that have been altered by past or recent timber harvesting and silvicultural activities. These effects were rare in the three burns we studied. The 2002 McNally and 2000 Manter fires occurred in close proximity in the southern Sierra (Figure 1), whereas the 2000 Storrie fire occurred in the northern Sierra near the southern Cascades (Figure 2). Together, these fires encompassed most of the area of Sierran conifer forest that has burned in the last 5 years, for a total of 49,917 ha. The McNally fire burned within the Sequoia National Forest from 22 July until 27 August 2002. The Manter and Storrie fires burned in 2000, the former from 7 July until 10 August and the latter from 17 August until 17 September. Weather initially conducive to fire spread, combined with rugged topography, enabled these fires to escape control and subsequently burn for 4–5 weeks under variable weather conditions. All three of the burns occurred in landscapes where most forests were not located within known, historic fire perimeters. In the McNally fire area, shrub ages indicate that fires had occurred there 125–150 years earlier in locations where there was no mapped record of fire (Keeley and others 2005).

Conifer forests typical of midelevations of the western Sierra (for a more detailed description of Sierran forests, see Rundel and others 1977) were abundant in the landscape that burned in the fires, particularly mixed or individually dominated forests of red and white fir (*Abies magnifica, A. concolor*); Jeffrey, ponderosa, and sugar pine (*Pinus jeffreyi, P. ponderosa, P. lambertiana*); and incense cedar (*Calocedrus decurrens*). These species are often mixed with a deciduous and an evergreen oak (*Quercus kellogii, Q. chrysolepis*). Trees in these forests are generally tall, with many overstory trees exceeding 40–50 m. Canopies are usually closed but can be open as a result of rocky substrata and other edaphic factors, particularly on granitic ridges. Open forests are mostly dominated by Jef-

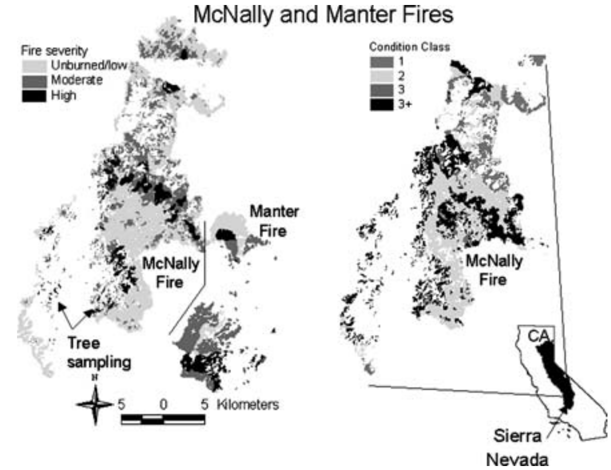

Figure 1. Patterns of burn severity in conifer-forested portions of the 2002 McNally and 2000 Manter fires in the southern Sierra Nevada, California. Preburn Condition Class is shown for the McNally fire area, not including the northernmost portion of the burn in the Inyo National Forest.

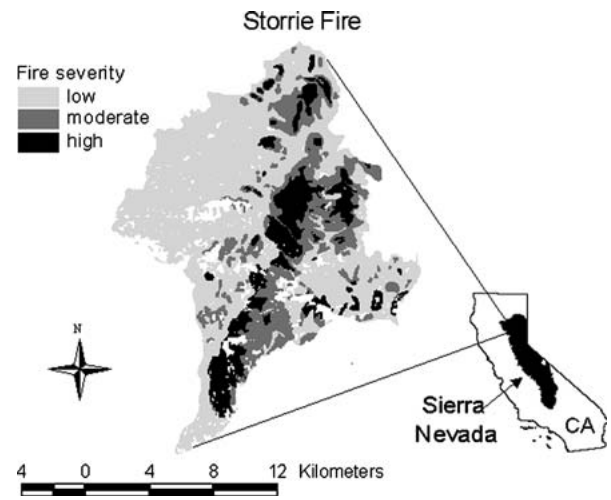

Figure 2. Patterns of burn severity in conifer-forested portions of the 2000 Storrie fire in the northern Sierra Nevada, California.

frey pine, often with shrubs in the understory. These forests are common in the Manter fire area and a portion of the McNally fire area. Closed mixed conifer forests predominated in the Storrie and McNally burn areas. One conifer, Douglas-fir (*Pseudotsuga menzesii*), is common in the Storrie fire area but absent from the southern Sierra.

Spatial Patterns of Fire Severity

BAER severity Mapping is designed to identify areas with high potential for soil erosion, which is generally based on the extent to which the fire affects the vegetation overstory canopy. The ability of remotely sensed data to identify patterns of fire severity based on the spectral response of tree canopies has been demonstrated in the Sierra (van Wagtendonk and others 2004). BAER severity in the McNally fire was mapped with Landsat 7 and SPOT multispectral satellite imagery (30-m pixel resolution) obtained immediately before and after the fire (Parsons 2002). A band ratio of mid-infrared and near-infrared reflectance was calculated from pre- and postburn image data. The band ratio data were classified and interpreted by staff at the USDA Forest Service Remote Sensing Applications Center in Salt Lake City Utah. BAER severity for the Manter and Storrie fires was mapped using aerial reconnaissance, infrared aerial photographs, and ground surveys (USDA 2000, 2002). General guidelines for severity classes are from the Forest Service Handbook (USDA 1995).

The BAER mapping identified three to four classes of fire severity based on the level of canopy effects detected. *Unburned* included areas where 0–10% canopy change was detected; this classification was distinguished only in the McNally fire. *Low severity* included areas where fire-caused crown scorch (heat-induced mortality of canopy foliage) affected less than 40% of overstory canopy foliage. The unburned and low-severity classes killed primarily conifer seedlings and saplings. *Moderate severity* included areas where fire scorched 40–89% of the forest canopy in the McNally fire and 40–80% in the other two fires. This level of severity was lethal to most conifer seedlings, saplings, and many small trees, but most overstory trees survived. *High severity* included areas where 90% or more of the canopy was scorched or affected by varying levels of incineration (direct consumption of crown foliage) in the McNally fire, whereas an excess of 80% of canopy showing these effects was considered high-severity in the Manter and Storrie fires. High-severity fire generally resulted in complete understory mortality. Overstory mortality ranged from complete to mixed depending on degree of canopy scorch and consumption (incineration), forest composition, and whether the threshold was 80% or 90% canopy mortality. Depending on imagery and other factors, different thresholds may be used for these severity levels in BAER mapping.

To characterize the spatial scales of the effects of high-severity fire in conifer forests, we describe the size of high-severity patches in each fire. To better characterize the effects, we evaluated the mortality of ponderosa and Jeffrey pine in areas of high-severity burn. Mortality assessments were restricted to a section of roadway in the McNally fire along which initial crown scorch had been assessed before there was any flushing of foliage. We identified five patches along this roadway that were dominated by trees that had no green foliage after the fire. These patches had fire effects ranging from 100% crown scorch (needles killed but not consumed) to needles consumed by crown fire. Within the patches, we chose to monitor all pines showing this range of high-severity effects that had a diameter at breast height (dbh) of more than 25 cm. These trees were generally within 50 m of the road. Our survival data are from 2 years postfire, following Stephens and Finney (2002). We did not observe any further indirect mortality caused by bark beetles over this period. Some trees were considered dead and were harvested over the course of the monitoring. We classified them as having been fire-killed, thus providing a maximum estimate of direct fire-induced mortality in the five sites.

Spatial and Temporal Patterns of Fire

To help assess the landscape-level influence of fire over time under modern fire suppression management, we calculated fire rotation intervals (amount of time needed for an area the size of the area of interest to burn one time) using fire perimeter data obtained from the US Forest Service and the California Department of Forest and Fire Protection. We used the total area of fire that has occurred from 1950 to 2005. Fire perimeters are complete and accurate over this period, and modern fire suppression was a consistent factor. Only conifer-forested areas were analyzed. The landscape we used to calculate fire rotation intervals in the McNally and Manter fire region was the southern portion of the Sequoia National Forest (210,932 ha of conifer forest), along with a smaller amount of the adjacent Inyo National Forest (10,000 ha of conifer forest), including and just beyond the northern boundary of the McNally fire (Figure 1). The landscape used to calculate fire rotation intervals in the Storrie fire region was the largest area within the Lassen and Plumas National Forests; that had the same forest vegetation types found within the Storrie fire region, which was in the center of this landscape. This landscape was more strongly dominated by conifer vegetation, which totaled 488,337 ha, than the landscape where the other two burns had occured. An estimate of rotation intervals for different severity classes in the two landscapes was calculated by assuming that all the conifer forest landscape that burned from 1950 to 2005 had the same severity proportions for the respective landscapes as either the McNally and Manter fires combined or the Storrie fire. This estimate integrates frequency and severity to help illustrate the influence of fire in the two landscapes under current management.

Fire Patterns and Condition Class

We evaluated fire patterns as a function of Condition Class in detail for the McNally fire, where preburn Condition Class data were available. These Condition Class data were mapped to the same vegetation units used in the Cal-Veg map (see Data Analysis). The Condition Class data were based on preburn Fire Return Interval Departure (FRID) and have been applied in planning efforts across the Sierra (USDA 2003, 2004). In other regions of the United States, the Condition Class approach is not necessarily based only on the estimation of FRID (http://www.frcc.gov). We obtained FRID data from the Southern Sierra Geographic Information System Cooperative, which helped to prepare them and still had a version that had not been updated after the McNally fire.

The Fire Return Interval Departure is the number of fires that, on average, may have been excluded. It is based on the time when fire last occurred in an area and the estimated historical fire frequency for the type of vegetation in that area. FRID was thus calculated as:

$$\text{FRID} = (Tsf - Fri)/Fri \qquad (1)$$

where Tsf equals time since the last fire in the landscape and Fri is the estimated fire interval for a vegetation type in the landscape. Estimated historical fire intervals for forests were developed from fire scar studies undertaken in the Sierra Nevada, southern Cascades, and the mountains of northwest and southern California, as reported by Skinner and Chang (1996). Table 1 shows estimated historic fire intervals for each forest type that burned in the McNally fire.

The FRID data we obtained identify the following categories of the number of fires that, on average, may have been excluded: *Extreme* denotes more than five (Condition Class 3 in the national three-level system), *High* is between two and five (Condition Class 3), *Moderate* is between one and two (Condition Class 2), and *Low* is less than one, or not outside the estimated historic fire return interval for a forest type (Condition Class 1) (USDA 2003). We kept the high and extreme FRID categories separate in our calculations and refer to extreme FRID as ''Condition Class 3+''.

Although preburn Condition Class data used in forest planning were not available for the same assessment in the Manter and Storrie fires, we make some inferences based on previous fire history, the Cal-Veg vegetation type within the burn perimeters, and the Condition Classes that would have been assigned based on the Condition Class criteria used in the Sequoia National Forest.

To determine how Condition Class might relate to fire spread rate—a likely predictor of fire severity that integrates weather, fuel, and topographic influences—we chose to assess BAER fire severity in relation to Condition Class in the McNally fire on days when the spread rate of fire was relatively rapid versus slow. To accomplish this, we plotted the ranked daily extent of total fire progression using data obtained from the Sequoia National Forest. This plot (Figure 3) shows that fire spread was particularly high on 2 days. Rather than analyze severity on just these 2 days, we selected additional days in which at least 2000 ha burned. On all the remaining days, an area equal to 1500 ha or less burned (Figure 3). The total areas on days where at least 2000 ha or 1500 ha or less burned were similar and constituted our relatively rapid- and slow-spread landscapes, respectively.

Data Analysis

We calculated fire-severity proportions in conifer forest vegetation types based on the primary vegetation type indicated in the vegetation map, Cal-Veg, that was used to develop Condition Class. It is a standard planning map used on national forest lands in California. Cal-Veg is a map representing current vegetation that is derived from satellite data. The map version used for the two fires in Sequoia had been updated just prior to the Manter fire, and the one for the Storrie fire had been updated the year before the fire. Updates were based

Table 1. Area of Different Conifer Forest Types Burned in the McNally Fire, Estimated Fire Interval used to Calculate Condition Class, and Percent BAER Severity for each Type

			Percent Fire Severity			
Type of Forest	Area (ha)	Fire Interval for Condition Class (y)	Unburned	Low	Moderate	High
Mixed conifer/fir	10,378	16	20.7	36.9	30.5	11.9
Red fir	10,323	50	38.6	35.1	16.3	10.0
Mixed conifer/pine	4154	16	5.5	33.5	52.1	9.0
Jeffrey pine	39,341	50	5.9	23.5	49.0	21.6
Ponderosa pine	2455	6	9.8	38.6	44.0	7.6
Lodgepole pine	1559	163	49.5	39.7	10.7	0.0
Subalpine conifers	692	163	28.9	60.8	9.9	0.4
White fir	117	16	14.9	47.0	34.4	3.6
Foxtail pine	92	163	70.5	29.5	0.0	0.0
Totals	33,704		23.4	35.1	30.5	10.9

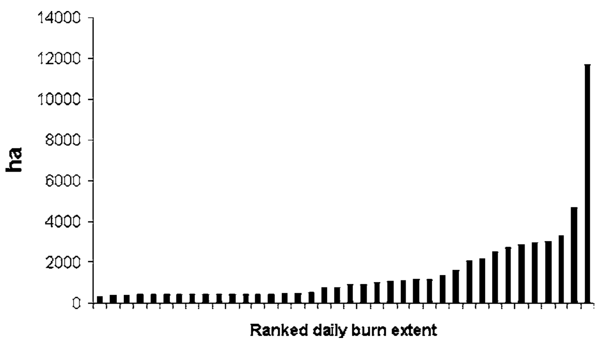

Figure 3. Ranked daily burn extent in the McNally fire as determined from the fire progression data of the Sequoia National Forest.

on accuracy assessments. A detailed description of the Cal-Veg map, and its development and accuracy for Forest Service lands, is at http://www.fs.fed.us/r5/rsl/projects/mapping. The minimum mapping unit is 1 ha. A description of the forest vegetation alliances mapped for the southern and northern Sierra and described in the results can be accessed at http://www.fs.fed.us/r5/rsl/projects/classification/zone-map.shtml.

We excluded pinyon/juniper woodlands and a small amount of open forest on the more arid east side of the Manter fire because it was not in national forest land and was subjected to different mapping protocols. Conversely, we included a small amount of area where the vegetation map indicated a hardwood conifer mix, but where the primary dominant was a conifer forest tree.

A formal statistical approach to testing for differences in severity proportions among Condition Classes by resampling independent, random point locations was not possible (for example, Odion and others 2004) because there was only enough area in some classes to locate a small number of independent points. Therefore, we present the proportions of fire severity by vegetation type and Condition Class and generally evaluate the weight of evidence provided by this information and other descriptors of the current fire regime in the context of the objectives described in the introduction.

Tree mortality was assessed for two diameter-size classes, 25–50 cm and larger than 50 cm. These two classes were compared for differences using a chi-square 2×2 independence test of the hypothesis that smaller trees would suffer greater mortality.

Results

Spatial Patterns of Fire Severity

Most of the conifer forests that burned in the McNally fire (Figure 1) showed characteristics of moderate- or lower-severity fire. High-severity fire accounted for 10.9% of all forest area (Table 1). The highest percentage of high-severity fire occurred in forests dominated by Jeffrey pine (22%), a species that is common on relatively dry and wind-exposed ridges. Most Jeffrey pine forest (83%) burned on the 3rd, 4th, and 5th most extreme-spread days of the McNally fire. Other forest types had much less high-severity fire—in particular, ponderosa pine, mixed conifer/pine, and the relatively small area of forest with long intervals of natural fire (mixed subalpine conifers, lodgepole pine, and foxtail pine). Although the McNally fire burned mostly fir and mixed conifer forests, most of the area that burned in the Manter fire was Jeffrey pine forest. The conifer forests in the Manter fire had more high-severity fire (29%) (Table 2). However, the Manter fire also had a lower

Table 2. Area of Different Conifer Forest Types Burned in the 2000 Manter and Storrie Fires, and the Percent BAER Severity for each Type

			Percent Fire Severity		
	Forest type	Area (ha)	Low	Moderate	High
Manter fire	Jeffrey pine	5,508	24.5	43.6	31.9
	Mixed conifer/fir	1,145	31.9	50.3	17.8
	Red fir	162	68.1	31.9	0.0
	Lodgepole pine	15	0.0	26.7	73.3
Totals		6,829	26.7	44.4	28.9
Storrie Fire	Mixed conifer/fir	7,583	85.8	10.0	4.2
	Mixed conifer/pine	6,577	45.6	26.3	28.1
	Douglas-fir/ponderosa pine	2,986	54.2	35.9	10.0
	White fir	1,511	72.6	5.9	21.6
	Red fir	591	95.8	2.4	1.8
	Jerey pine	128	41.7	52.8	5.6
	Lodgepole pine	7	100.0	0.0	0.0
	Ponderosa pine	2	100.0	0.0	0.0
Totals		19,384	66.3	19.2	14.5

threshold for high-severity fire than the McNally fire (80% versus 90% or more canopy foliage mortality).

For the Storrie fire, severity mapping also used the 80% threshold for high-severity fire. High-severity fire totaled 14.5% among all conifer forests, but the area incurred only about half as much moderate-severity fire as the area burned by the other two fires and consequently considerably more low-severity fire (Figure 2 and Table 2). Of the total area that did burn at high severity (2805 ha), most (1730 ha) of this fire occurred in mixed conifer/pine forests. However, forests dominated by ponderosa and Jeffrey pine had little high-severity fire. Conversely, white fir forests incurred much more high-severity fire than mixed conifer/fir, the most common forest type in the Storrie burn area. Thus, this fire had lower overall severity than the others, and even in different areas mapped with forest types that included many of the same species, the fire nonetheless burned with varying severity.

A few large high-severity patches accounted for much of the total area of high-severity fire in the conifer forests affected by the three burns (Figure 4A–C). However, all three fires produced mostly relatively small patches of high-severity fire. Patches totaling less than 5 ha accounted for 107 of the total of 157 high-severity patches in the McNally fire. They accounted for 28 of a total of 40 in the Manter fire, and 59 of 102 in the Storrie fire.

Many of the pines we monitored that incurred severe burn effects nonetheless produced new foliage from surviving terminal buds in the year after the fire. All surviving trees had either 100% crown scorch and no incineration of foliage or 100% scorch and incineration extending upward to at most 50% of total tree height. For Jeffrey pines incurring these fire effects, 22 of 44 trees survived and there was no difference between the 25–50 cm and greater than 50-cm diameter size classes in terms of the percentage of trees that survived. For the more abundant ponderosa pine, 42 of 83 and 57 of 83 trees in these two size classes survived, and diameter exerted a significant, positive effect (chi-square = 5.6, $P < 0.01$). None of the trees ($n = 90$) with higher levels of crown incineration, survived, indicating that there are significant differences between the effects of crown fire that incinerates foliage and the effects of severe surface fire, which primarily results in the death of foliage due to heat scorch.

Spatial and Temporal Patterns of Fire

For the larger landscape of the national forest in which the McNally and Manter fires occurred, the rotation interval from 1950 to the present for all fire was 185 years. The McNally and Manter fires were responsible for two-thirds of the area that was burned over this time. For both burns combined, the overall percentage proportions of high- and moderate-severity damage in conifer forests was 14% and 33%, respectively. Using these values, the rotation interval in conifer forests was estimated to be about 1330 years, for high-severity fire and about 565 years for moderate-severity fire. Fire has been less common in conifer forests of the Storrie

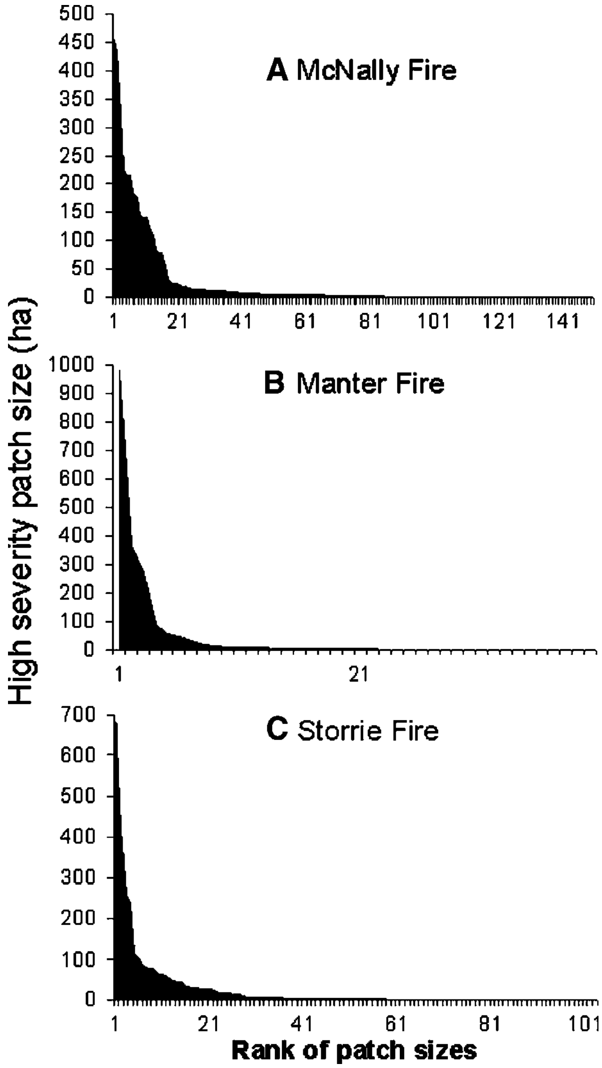

Figure 4. Ranked size of high-severity burn patches in conifer vegetation in the **A** McNally, **B** Manter, and **C** Storrie fires.

fire region. The rotation interval for all fire since 1950 was 507 years. The Storrie fire accounted for about half of all fire in conifer forests over this time period. The estimated rotation interval since 1950 was 3503 years, for high-severity fire and 2460 years for moderate-severity fire in the region in which the Storrie fire occurred.

Severity Patterns and Condition Class

Fire severity proportions by Condition Class under slow- and rapid-spread days in the Sequoia National Forest portion of the McNally fire are shown in Figure 5A–B. The 3939 ha comprising Condition Class 1 forests (2505 ha on slow-spread days plus 1424 ha on rapid-spread days) had almost no high-severity fire. These forests were predominantly comprised of subalpine and other high-elevation forests of red fir, lodgepole pine, and foxtail pine that grow on the relatively flat Kern Plateau.

For Condition Classes 2, 3, and 3+, there were distinctions in degree of severity between rapid- and slow-spread days. In particular, on rapid-spread days, moderate-severity fire was considerably more common, whereas low-severity was less common. The largest area of high-severity fire occurred on rapid-spread days in Condition Class 2 forests (Figure 5A). These forests were comprised mainly of red fir (62%) and Jeffrey pine (22%). Condition Class 3 forests consisted entirely of mixed conifer/fir or pine, whereas Condition Class 3+ forest were ponderosa pine. They had the same proportions of high-severity fire (13%) on rapid- and slow-spread days. This figure was very similar to that for conifer forests throughout the area covered by Condition Class data (Figure 1), which was 11.8%. Condition Class did not appear to have a strong effect in promoting rate of spread because a considerable area of Condition Class 3+ forest burned on slow-spread days (Figure 5).

Applying the McNally Condition Class criteria to the Manter burn area, we find that the 5400 ha of Jeffrey pine and 1145 ha of mixed conifer/fir forests that had no record of previous fire would be Condition Classes 2 and 3+, respectively. Jeffrey pine had 32% high-severity fire, and mixed conifer/fir forests had 17% high-severity. A small area of Jeffrey and lodgepole pine forest (94 ha) that would have been Condition Class 1 had 43% high-severity fire.

Applying the McNally Condition Class criteria to the Storrie fire area and presuming Douglas-fir/ponderosa pine to have an estimated past fire return interval of 16 years, like similar forests (Table 1), we find that there were 792 ha of Condition Class 2 mixed conifer forests. Most of this are burned previously in the 1970s and was primarily forested by Douglas-fir/ponderosa pine. In the Storrie fire, these forests burned with 20% high-severity and 53% moderate severity. Red fir and Jeffrey pine forests (719 ha) had no record of previous fire and would also have been Condition Class 2. They burned at much lower severity than most forests (Table 2). The rest of the forests affected by the Storrie fire had not burned for a long time and would have been condition Class 3+. Collectively they experienced the same severity proportions observed for the burn as a whole—lower than that seen in the Condition Class 2 mixed conifer forests.

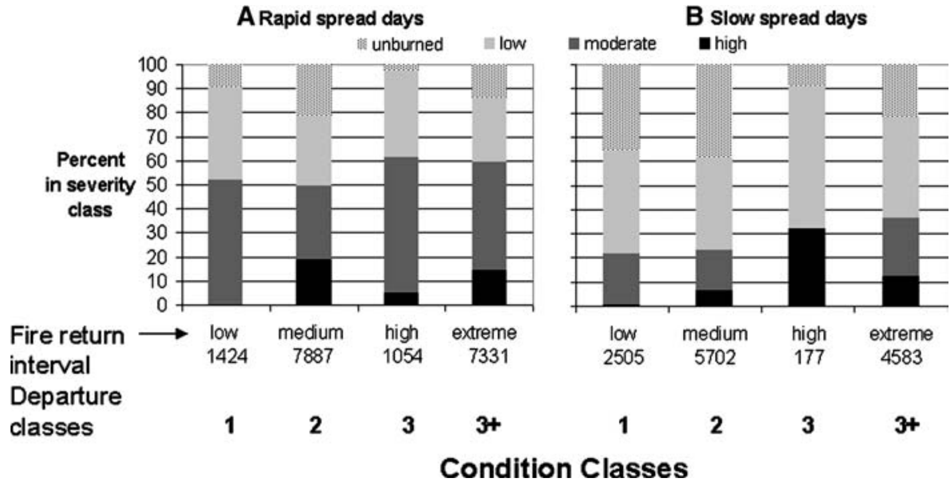

Figure 5. McNally fire severity proportions by Condition Class occurring during **A** days of relatively rapid fire spread (n = 10) and **B** days of relatively slow spread (n = 28). Numbers below columns are hectares burned.

Discussion

Contemporary fire is clearly not almost exclusively high-severity and stand-replacing in long-unburned areas of Sierran conifer forests. In the large area of burned forest that we evaluated, fire severity was highly variable and caused a relatively small amount of high-severity effects. Van Wagtendonk and others (2004) found similar levels of variation and severity proportions in another recent Sierra Nevada burn in the same forest types examined in our study. Our findings are also consistent with the result of recent modeling, which showed that long-unburned Sierran forests unaffected by silvicultural activities would not incur crown fire until temperature, relative humidity, and wind exceeded the 97.5th percentile of their summertime levels (Stephens and Moghaddas 2005).

The burn patterns we observed are also consistent with descriptions and evidence in Sierran forests not influenced by fire suppression and silviculture. There are a number of historical accounts of variability in fire ranging from light understory burning to patchy high-severity fire in Sierran mixed conifer forests, including one by the famed naturalist John Muir (reviewed by Stephenson and others 1991; Stephenson 1999), and another by a forest surveyor John Leiberg (1902). Recent studies using historic photos and field sampling have concluded that patches of high-severity fire have shaped mixed conifer forests in the Sierra Nevada and the adjacent southern Cascades (Russell and others 1998; Beaty and Taylor 2001; Taylor 2002). Show and Kotok (1924), Russell and others (1998), Beaty and Taylor (2001), and Taylor (2002) describe historic high-severity burn patches in the Sierra that are comparable in size to many of the larger patches produced by the three fires we studied. Smaller patches or gaps have also played an important role in determining forest and landscape structure and composition (Stephenson and others 1991; Keeley and Stephenson 2000) and were common in the three fires we studied. Leiberg (1902) and Beaty and Taylor (2001) have also describe the occurrence of large historic fires.

Because the fires we studied burned for 4–5 weeks, mainly in July and August, they were influenced by a range of weather conditions. This may help to explain why they were heterogeneous and qualitatively similar to descriptions of pre–suppression era fires. Most lightning ignitions occur in the Sierra during July and early August (Caprio and Swetnam 1995). Historic lightning ignitions that led to spreading fires would have been driven by the same seasonal patterns of warm, dry weather that typifies the Sierran summers. The large size of the fires we studied likely enhanced their variability by creating both fire-generated winds, which that can make combustion more active, and dense smoke, which can lower temperatures and mitigate fire behavior (Pyne 1984). Thus, it is important to stress that our results apply to fires in the Sierra that burn for long durations and spread over relatively large areas in mid- and late summer. These circumstances are representative of much of the areas burned by contemporary fire, and presumably fire in the past, given the effect of large fire on the cumulative amount of area burned. Much less heterogeneity may result from fires that burn for a shorter time and cover small areas. Our results also apply only to areas in the Sierra where timber harvesting and silvicultural activities have not been common. There are many areas of the Sierra that have been modified considerably by intensive silvicultural activities (SNEP 1996) and where severity is expected to be higher due to increases in available fuel and the loss of fire-resistant trees (Stephens and Moghaddas 2005).

After a long period of reduced fire influence, large, heterogeneous fires can hasten ecological restoration (Baker 1992; Miller and Urban 2000; Fulé and others 2004). They may affect biodiversity by thinning trees and decreasing competitive exclusion processes and by increasing structural and landscape diversity. Fire-created gaps provide opportunities for the natural regeneration of light-demanding conifers such as pines and giant Sequoia (*Sequoiadendron giganteum*) (Stephenson and others 1991; Keeley and Zedler 1998; Stephens and others 1999) whose natural abundance in the Sierra has been reduced (SNEP 1996). There are concerns about the lack of natural regeneration in these species due to the absence of fire severe enough to create openings, consume sufficient duff and litter to facilitate successful germination, and open cones in giant Sequoia (Stephenson and others 1991; Stephens and others 1999). Such fire effects may not only promote the natural reproduction of these conifers, but also favor the relative abundance of these species because they have a greater ability to survive. Large giant Sequoia may survive in areas of crown fire (Stephenson and others 1991), and we found that many medium and large ponderosa and Jeffrey pines can survive severe surface fire. There may be some additional mortality among these trees, but those that survive are likely to experience rapid growth and increased vigor, much like giant Sequoia after severe fire (Stephenson and others 1991). Mature white fir may also be more fire resistant in the Sierra than previously suspected, aided by their ability to produce epicormic branches (Hanson and North 2006). Surviving conifers may serve as sources of seed that help to ensure natural regeneration in high-severity burn patches.

Patches of habitat created by high-severity fire, with their rich array of snags, logs, and nonarborescent vegetation, are among the scarcest habitats in many forested landscapes (Lindenmayer and Franklin 2002). After 50–100 years this early successional habitat can succeed to forest (Russell and others 1998). Thus, based on estimates the area of high-severity fire predicted by our fire rotation analyses for the period since 1950 in the Sequoia and Storrie fire regions, about 4.2% and 1.5% of these landscapes, respectively, may have naturally developed early successional burned forest habitat under the current fire regimes. The maintenance of this habitat in the landscape by fire promotes biodiversity because it supports plant, insect, and wildlife assemblages not found in other Sierran habitats. In addition, there are numerous plant and animal species that have become rare due to their requirements for burned forest habitat. For example, there is some concern over the local extirpation of avian species with these habitat requirements (Kotliar and others 2002). Species such as the black-backed woodpecker (*Picoides arcticus*) may be indicators of whether sufficient, severely burned forest habitat is being maintained for biodiversity (Hutto 1995). These birds require young, severely burned patches of at least 12–25 ha (Saab and others 2002). The three fires we studied created 70 severe-burn patches larger than 12 ha where there had been none or very few due to the lack of fire.

Thus, the effects of the large fires we studied are consistent with the diversity goals of the National Forest Management Act. Elsewhere in the western United States, a number of large fires have also been found to perform the desired ecological functions of fire (for example, Turner and others 2003; Kotliar and others 2003; Fulé and others 2004; Odion and others 2004; Schoennagel and others 2004; Smucker and others 2005). These specific effects may ultimately be necessary for maintaining biodiversity that depends on fire. Prescribed burning can help, but it is limited in extent, severity, and heterogeneity (Baker 1994; Rocca 2004) and may not mimic natural fire (Moritz and Odion 2004). On National Forest Service lands, prescribed burning is often conducted outside the normal fire season, when flaming is subdued but wildlife such as herptofauna are highly vulnerable to smoldering combustion (Bury 2004). Neither these fires, nor the structural modification of forests through mechanical treatments, may provide fire-specific effects for species that require them (for example, flowering plants with fire-dependent seed germination that is sensitive to burn season, conifers with heat-opened cones, and cavity-nesting species that dependent on standing dead trees for nesting and foraging).

Fire Patterns and Condition Class

We found that the proxy for fire suppression effects, Condition Class, was not effective in identifying locations of high-severity fire. Condition

Classes 2, 3, and 3+ in the McNally fire all had similar fire severity proportions. When the same Condition Class criteria were applied to the other two fires, we found that fire severity generally decreased rather than increasing from Condition Class 2 to 3+. In short, Condition Class identified nearly all forests as being at high risk of burning with a dramatic increase in fire severity compared to past fires. Instead, we found that the forests under investigation were at low risk for burning at high-severity, especially when both spatial and temporal patterns of fire are considered.

The lack of an observed relationship between Condition Class and fire severity suggests that exogeneous forces such as weather, climate, topography, and neighboring vegetation (for example, pyrogenic shrubs) largely determine fire-severity patterns in forests. Because fire severity did not increase above Condition Class 2, the combustibility of Sierran forests may reach a maximum at the fire-free intervals indicated by this class (32–48 years for many forest types), (Table 1).

A number of interrelated factors may explain why these forests reach a maximum in combustibility. For example, the total leaf area of a forest reaches a maximum (Waring and Schlesinger 1985). Once forest overstories close in the Sierra, they may exclude pyrogenic shrubs with high light requirements (Show and Kotok 1924), greatly decreasing the potential intensity of understory combustion. The base height of the forest canopy sufficiently dense to propagate fire may also become relatively high in long-unburned forests (Stephens and Moghaddas 2005). In terms of surface fuel beds, those associated with Sierran conifers that increase in abundance with time since fire (for example, fir) are more dense than those found under pine and thus less combustible (van Wagtendonk and others 1998).

Conclusions

Our findings suggest that elevated risk of high-severity fire due to the effects of fire suppression is not the pervasive, predictable ecological problem that it has often been portrayed to be in the Sierran forests we studied. In addition, they provide evidence that fire alone can restore its past influence as a patchwise and stand-thinning disturbance agent as well as a facilitator of species diversity and fire-adapted conifers in these forests. Thus, it appears that management can shift toward process restoration by introducing more fire and increasing the use of wildland fire (Miller 2003). There may be no other effective strategy for restoring and maintaining ecological integrity and for fostering the natural diversity of species dependent on effects specific to fire. The structural modifications of forests cannot mimic the heterogeneous effects of fire. Instituting a policy that allows more fire to burn would require considerable planning and additional efforts to improve human safety, but such efforts are needed under any management scenario.

Both Condition Class and the new LANDFIRE approach are based on mapping any departure in fire regimes from reference conditions. Presuppression reference conditions for fire must be based on retrospective studies. These studies are too methodologically limited to provide a comprehensive description of the spatial extent and variation in the effects of past fires (reviewed by Veblen 2003). As a result, the importance of past surface fire may be overestimated and conversely, past heterogeneity in fire may be underestimated (for example, Minnich and others 2000). To add to the problem of uncertainty about past fire, there may be significant misconceptions about current fire severity that lead to further overestimation of the differences between past and present fire regimes.

By directly assessing existing fire regimes in the context of ecological integrity, we can avoid some of the problems that may arise when current methods for estimating departure in fire regimes are used. A general approach based on the assessment of existing rates and scales of processes in the context of ecological integrity has been recommended for the management of biodiversity as a means of overcoming problems in defining the ''natural'' range of variation in ecological systems (Parrish and others 2003). The direct assessment of fire regimes can be improved by applying more sophisticated mapping of fire severity and performing landscape analyses that provide a clearer link between pattern and process (Wagner and Fortin 2005). In the Sierra Nevada, it is important to distinguish high-severity surface fire from crown fire because the two types of behavior may have very different effects on tree mortality. There is also a need for analyses of fire behavior in areas affected by timber harvesting and silviculture. Finally, better integration of the spatial and temporal components of other forest disturbances in the Sierra Nevada in addition to fire, is needed to determine if their rates and scales are compatible with ecological integrity.

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Examining Historical and Current Mixed-Severity Fire Regimes in Ponderosa Pine and Mixed-Conifer Forests of Western North America

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Abstract

There is widespread concern that fire exclusion has led to an unprecedented threat of uncharacteristically severe fires in ponderosa pine (*Pinus ponderosa* Dougl. ex. Laws) and mixed-conifer forests of western North America. These extensive montane forests are considered to be adapted to a low/moderate-severity fire regime that maintained stands of relatively old trees. However, there is increasing recognition from landscape-scale assessments that, prior to any significant effects of fire exclusion, fires and forest structure were more variable in these forests. Biota in these forests are also dependent on the resources made available by higher-severity fire. A better understanding of historical fire regimes in the ponderosa pine and mixed-conifer forests of western North America is therefore needed to define reference conditions and help maintain characteristic ecological diversity of these systems. We compiled landscape-scale evidence of historical fire severity patterns in the ponderosa pine and mixed-conifer forests from published literature sources and stand ages available from the Forest Inventory and Analysis program in the USA. The consensus from this evidence is that the traditional reference conditions of low-severity fire regimes are inaccurate for most forests of western North America. Instead, most forests appear to have been characterized by mixed-severity fire that included ecologically significant amounts of weather-driven, high-severity fire. Diverse forests in different stages of succession, with a high proportion in relatively young stages, occurred prior to fire exclusion. Over the past century, successional diversity created by fire decreased. Our findings suggest that ecological management goals that incorporate successional diversity created by fire may support characteristic biodiversity, whereas current attempts to "restore" forests to open, low-severity fire conditions may not align with historical reference conditions in most ponderosa pine and mixed-conifer forests of western North America.

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Introduction

In just two days in 1910, 1.2 million ha of forestlands in Idaho and Montana in the western USA burned in a massive fire driven by exceptional winds [1]. In the aftermath, the United States instituted a policy of aggressive fire suppression [2]. Decades of fire suppression activities since 1910 have reduced the extent and number of wildfires in the USA, as well as parts of Canada. There is now widespread concern that fire exclusion has caused vegetation in western North America to be much more susceptible to uncharacteristically severe fire. This concern is greatest in the extensive, often drier forests of the North American Cordillera, especially those dominated by ponderosa pine (*Pinus ponderosa* Dougl. ex. Laws) and Jeffrey pine (*P. jeffreyi* Grev. & Balf.), or those mixed with ponderosa/Jeffrey-pine and other conifer species (hereafter ponderosa pine and mixed-conifer forests of western North America, defined in Table 1 and further described in Methods).

The ponderosa pine and mixed-conifer forests of western North America have traditionally been considered adapted to a low- or low/moderate-severity fire regime (see Tables 1 and 2 for definitions of fire terms) [3–8]. There have been many large mixed-severity fires in western North America in recent years [9] that have helped create widespread concern that fire exclusion has caused an unprecedented threat of uncharacteristically severe fires [6–15]. Concomitantly, however, there has been increasing recognition that fires in ponderosa pine and mixed-conifer forests of western North America were also mixed in severity prior to any significant effects of fire exclusion (Table 2) [16,17]. It has also been increasingly recognized that these forests support biota that are not adapted to low/moderate-severity fire, but rather are dependent on the high-severity fire component of mixed-severity regimes [18–22]. Thus, a better understanding of historical (i.e., generally prior to fire suppression and timber harvesting) fire regimes in these forests is needed to define reference conditions and maintain characteristic ecological diversity.

In recent decades, to address the widespread concerns about uncharacteristically severe fire in western North America, fuel reduction treatments have been implemented on millions of hectares of ponderosa pine and mixed-conifer forests at a cost of billions of dollars [23]. These treatments consist mainly of harvesting smaller trees to reduce forest density [8], but larger trees are typically harvested as well for economic reasons [24]. These treatments can negatively affect fire dependent species. For example, the Black-backed Woodpecker (*Picoides arcticus*), an imperiled fire-dependent species, largely avoids previously thinned forest areas burned at high-severity [18]. Thinning treatments also eliminate/degrade dense forest, which many species need, including the Northern Spotted Owl (*Strix occidentalis caurina*), a Threatened Species under the USA Endangered Species Act [25], and the Pacific fisher (*Pekania pennanti*), a Candidate Species under the USA Endangered Species Act [26]. In addition, forest thinning treatments often require the reopening or construction of access roads, which have many ecosystem impacts [27], and both the thinning treatments and roads promote the establishment of invasive species [27,28]. Thinning ultimately exacerbates fire suppression impacts if it facilitates fire control, or if it becomes a prerequisite for allowing wildfires to burn [13,29]. Thus, there is a need to ensure that actions are ecologically justified.

Most descriptions of the fire regimes that characterize the ponderosa pine and mixed-conifer forests of western North America (e.g., [5–7,11]) emphasize how low-severity fires maintain forests dominated by relatively old and large, fire-resistant trees, with few understory trees, dead or dying trees, or shrubs [3–7,11–13] (Table 2). Park-like conditions and low fuel loads are thought to result from effects of frequent surface fire, which kills young, fire-sensitive trees, while older, fire-resistant trees survive [4,6,7,11,12].

In contrast, mixed-severity fire regimes are characterized by more variable fire and forest structure across a wide range of spatial and temporal scales [17,21] (Tables 1 and 2). The creation of complex early seral vegetation by high-severity fire often occurs in irregular patches across the landscape and at irregular intervals [30]. Over time, the complex early successional vegetation created by fire, if not reburned, transitions to mid- and then late-successional forest, often containing pre-disturbance legacies, such as standing or fallen dead trees and often some fire resistant, large trees that survive fire crown fire (e.g., [31]). Thus, mixed-severity fire regimes create complex successional diversity high beta diversity, and diverse stand-structure across the landscape [17,21,30,32–35].

The concepts and nomenclature used to describe fire regimes in western North America can be ambiguous. Part of the problem with defining fire regimes for the drier forests of western North America is the classification of fire regimes into distinct categories of low-, mixed-, and high-severity [5], or low/moderate-severity

Table 1. Definitions of terms as used in this paper.

Term	Definition
Ponderosa pine and mixed-conifer forests of Western North America	Low- to mid-elevation, montane, non-coastal forests of western North America where a regime of low/moderate-severity fire (see Table 2 for explanation) that limit tree recruitment has traditionally been applied. These extensive forests are dominated by ponderosa pine (*Pinus ponderosa*), Douglas-fir (*Pseudotsuga menziesii*) and fir (*Abies concolor* and *A. grandis*) (see Methods). These forests are drier than coastal forests or most forests at higher elevations, though one region, the Klamath, is more mesic.
Fire dependent	Biota that occur most abundantly after high-severity fire, and which are largely or entirely absent where high-severity fire has not occurred for a long period.
Fire regime	The frequency, size, seasonality, impacts and other characteristics of naturally occurring fires that have occurred in a vegetation type over its lifespan, generally 1–3 millennia [133].
High-severity fire rotation (or moderate to high-severity fire rotation)	The length of time required for an area equal to the area of interest to burn [134]. For high-severity fire, this is calculated as the time period over which high-severity fire (or moderate- and high severity fire combined) is observed, divided by the proportion of the area of interest that burns in that time period at high- or moderate/high-severity.
High-severity fire	Fire that burns on the ground surface, and typically in the overstory canopy (crown fire) as well. Mortality of woody species as measured by basal area is generally >70%. However, sprouting canopy species, such as oaks (*Quercus* spp.) typically survive these fires. High-severity fire mainly occurs in relatively discrete patches under high winds that cause blow ups in fire behavior [108]. These patches range in size from the area occupied by a small group of trees to many thousands of ha in size, as in the case of the 1910 fires.
Low-severity fire	Fire that burns on the ground surface such that relatively little or no mortality of live, standing vegetation occurs. Mortality of woody species as measured by basal area is 0–20%, but is mostly 0–5%. See Table 2 for a detailed explanation of the effects of a regime of low-severity fire.
Moderate-severity fire	Fire that burns only on the ground surface and that has effects that are intermediate between low- and high-severity fire as defined here. Mortality of woody species as measured by basal area is generally 20–70% within a given area.
Mixed-severity fire	Fire that includes low-, moderate-, and high-severity effects. See Table 2 for a detailed explanation of the effects of a regime of mixed-severity fire.
Park-like forest	A forest of widely-spaced live, mature trees and very few, if any, dead trees (snags). The understory is open, often dominated by bunchgrasses, and is mostly lacking woody plants.
Stand age	The age within a stand of the dominant overstory canopy vegetation that recruited more or less as a cohort, typically after a previous disturbance.

These terms may have different meanings in the literature depending on the context in which they are used.
doi:10.1371/journal.pone.0087852.t001

Table 2. Characteristics of fire regimes in ponderosa pine and mixed-conifer forests of Western North America.

	Low/moderate-severity model	Mixed-severity model
Tree populations	1. Stable. Gap phase recruitment dynamics.	1. Unstable. Gap and stand-level mortality and recruitment. Stand-replacement fires at intervals often shorter than tree lifespans.
	2. Recruitment limited by frequent fire.	2. Recruitment abundant and stimulated by fire.
	3. Resistant to fire (though often described as "fire-resilient").	3. Resilient following fire.
Landscape patterns	1. Successional diversity low.	1. Successional diversity high.
	2. Gradual variation along environmental gradients.	2. Variation along environmental gradients interrupted by sharp boundaries and patchiness.
	3. Low contrast heterogeneity. Intensity/complexity of spatial pattern is low.	3. High contrast spatial heterogeneity. Intensity/complexity of spatial pattern is high.
	4. Low beta diversity.	4. High beta diversity.
Stand structure	1. Does not vary markedly over time.	1. Varies markedly as a function of time since fire disturbance.
	2. Open canopy of mature, medium and large trees; density low.	2. Variable canopy, tree size, and density variable; even-aged cohorts stimulated by fire.
	3. Understory with few trees or shrubs.	3. Understory varies.
Fire behavior	1. Typically low intensity surface fire with flame lengths <3 m; short residence time.	1. Variable intensity surface or crown fire, variable residence time.
	2. Fuel limited. Crown fire cannot initiate.	2. Not necessarily fuel limited. Crown fire can initiate under extreme conditions.
Individual fire canopy mortality	1. Mortality of canopy trees <20% by basal area.	1. Mix of low-, intermediate- and high-severity fire with (0–20%, 20–70%, >70%) mortality of canopy trees by basal area respectively.
Interactive effects of fire on fuels and forest flammability	1. Fires continuously limit fuels and fire sensitive trees.	1. Fires only temporarily lower fuels.
	2. Maintain low flammability and forest mortality over time.	2. Do not maintain low flammability and forest mortality, except initially after fires.
Evolutionary responses	1. Fire resistant trees.	1. Fire resistant and fire-dependent or specialized biota. The latter includes species with reproduction timed to coincide with fire via fire-cued germination, fire "embracer" plant species, and post-fire insect and bird specialists).
Fire exclusion leads to	1. High tree regeneration*.	1. Low tree regeneration.
	2. Greatly increased flammability.	2. Small changes in flammability (vegetation is continuously flammable except initially after fire).
	3. Increased forest susceptibility to mixed-severity fire.	3. Decreased susceptibility to mixed-severity fire.
Carbon storage[1]	1. Low-moderate; considerably lower than carrying capacity.	1. Moderate to high; Near carrying capacity.
Fuel treatments (forest thinning)	1. Restores forest tree structure and fuel loads where infill associated with fire exclusion is removed.	1. May create uncharacteristic structure and composition (reduction in small and intermediate and some overstory trees, shrubs, down wood).
	2. Increase open forest (woodland) biota.	2. Decrease in dense forest biota and post-fire habitat specialists.
	3. May create low contrast heterogeneity.	3. May reduce high-contrast heterogeneity.

[1][135–137].

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and high-severity [9], when nearly all fire regimes include a mix of all three severities. Greater clarity in terminology is needed to improve communication about fire regimes. Tables 1 and 2 document the terminology used herein.

In addition to unclear terminology, other factors create difficulties for identifying which historical (i.e. prior to fire exclusion) fire regime applies to a particular forest region. Where fire has been excluded from a mixed-severity landscape for 100 years, early- and mid-successional patches created by high-severity fire become late-successional patches, making it more likely that these patches, indicative of a mixed–severity regime, will be undetected. For example, high-severity fire patches may be detected in old but not recent aerial imagery [35]. A primary source of data on historical fires are scars in the growth rings of surviving trees damaged by fire, which can provide annually precise dates for past fires at the sampled locations [36–40]. However, these methods cannot effectively determine past occurrence of high-severity fire. Thus, additional evidence is needed to characterize historical fire regimes over more extensive areas.

The US Forest Service Inventory and Analysis (FIA) program provides an extensive dataset that is a probabilistic sample of forest structure in large landscapes. This dataset allows for landscape-scale inference and statistical analyses of forest age and structure parameters consistent with a low- or mixed-severity fire regime.

Using the FIA data, and published sources of landscape-scale (area of inference >25,000 ha) data, our objectives were to address two broad questions: (1) How prevalent were mixed-severity fire regimes historically in ponderosa pine and mixed-conifer forests of western North America; and (2) How have mixed-severity fire patterns in these forests changed with fire exclusion? Consistent with common perceptions and restoration models applied to these forests, we hypothesized that: (1) forest age-class diversity was low, reflecting long-term effects of low/moderate-severity fire regimes (Table 1); and (2) fire exclusion has led to vegetation changes that have increased the prevalence of high-severity fire.

Methods

Study Area

FIA and published sources of landscape-scale (area of inference >25,000 ha) data with inference to pre-settlement fire severity and forest structure were available from the following regions of western North America: Baja California, the Sierra Nevada, the Klamath Region, the eastern Cascades, the northern Rockies, the central Rockies, and the southwestern USA (Figure 1). We used ecoregional class III data from the US Environmental Protection Agency (http://www.epa.gov/wed/pages/ecoregions/level_iii_iv.htm) to define the Sierra Nevada, Klamath, and eastern Cascades regions. The Sierra Nevada was split along the distinct crest of the range into the east and west slopes. The portions of the northern Cascades east of the crest and the main Cascades within California were combined into the eastern Cascades. The Modoc Plateau and eastern Sierra Nevada was also combined with the eastern Cascades. The northern Rockies were in Idaho and Montana, and the central Rockies were in Colorado, Wyoming and South Dakota. The southwestern USA included Arizona and New Mexico.

The dominant conifer over most of the low- to mid-elevation, montane forests in these regions is ponderosa pine, often with lesser amounts of Douglas-fir, white fir (*Abies concolor* (Gord. and Glend.) Lindl.), and/or grand fir (*A. grandis* (Douglas ex D. Don) Lindl.). In the Sierra Nevada and Klamath regions, ponderosa pine is common and may be dominant, especially in low-elevation forests, and mixed-conifer forests generally include components of ponderosa pine, white fir, Douglas-fir, incense-cedar (*Calocedrus decurrens* (Torr.) Florin), sugar pine (*P. lambertiana* Dougl.), California black oak (*Quercus kelloggii* Newb.) and evergreen canyon live oak (*Q. chysolepsis* Liebm.). Mid-elevation forests of the Sierra Nevada and Cascades are often dominated by Jeffrey pine, ponderosa pine, white fir and sugar pine. Low- to mid-montane forests of the eastern Cascades are dominated by ponderosa pine and Douglas-fir, and can include components of white fir, grand fir (*Abies grandis* Dougl. ex D. Don.) Lindl.), and western hemlock (*Tsuga heterophylla* (Raf.) Sarg). Low- and mid-elevation forests of the Rocky Mountains are dominated by ponderosa pine and Douglas-fir. In the northern Rockies, these two dominants may co-occur with white fir and grand fir, and with western hemlock, western redcedar (*Thuja plicata* Donn. Ex D. Don.) and quaking aspen (*Populus tremuloides*). Forests of the southwestern U.S. are heavily dominated by ponderosa pine, with some white fir and Douglas-fir at middle elevations. Precipitation and temperature data for each region in this study are provided in Table 3.

Evidence for Historical Mixed-severity Fire Regimes in Ponderosa Pine and Mixed-conifer Forests

Rotations of high- and moderate-to high-severity fire. We summarized rotations for high-severity fire from published studies with inference to large landscapes (>25,000 ha) in ponderosa pine and mixed-conifer forest landscapes of western NA over a period of 70 or more years. The high-severity fire rotation is equal to the average interval between high-severity fire across the affected landscape (Table 1). Additionally, we summarized other evidence regarding the occurrence of high-severity fire where rotations could not be calculated, but where landscape-scale inference regarding the relative importance of high-severity fire was presented, or where rotations could be calculated but landscapes were <25,000 ha or the time period was <70 years.

Dominant overstory tree age distributions. To assess successional patterns indicative of mixed- vs. low/moderate-severity fire regimes, we analyzed US Forest Service Inventory and Analysis (FIA) stand ages (data available at http://www.fia.fs.fed.us/tools-data/) by region. These data capture the average age of the trees dominating the canopy layer in forest stands (stand age, Table 1) that have been sampled probabilistically, with inference to more extensive landscapes. Because the dominant trees in ponderosa pine and mixed-conifer forests may be several centuries old in the absence of disturbance [e.g., 41,42], we reasoned that the age of relatively young and intermediate-aged stands (e.g. <200 years) reflects the time since a disturbance that shifted dominance from older to younger trees. The FIA database indicated that young stands (generally 0–30 years) were initiated by fire. To determine whether disturbances in other plots were caused by fire, we evaluated the effects of fire exclusion on rates of disturbance, as described below. It is not possible to specify the level of mortality that fire or other disturbances may have caused, but it is possible to determine the extent to which forests were dominated by older age classes, which would be consistent with low−/moderate severity fire, versus stands of more diverse age classes, consistent with mixed-severity fire.

FIA is a monitoring system based on one permanent, random 1-ha plot per ~2400 ha across forested lands in the USA. For tree measurements, the plot area is sub-sampled with four circular plots of 0.1 ha for large trees and 0.017 ha for smaller trees nested within the larger tree plots. Diameter at breast height (dbh) and crown position of each tree and the ring count from cores of the dominant and co-dominant trees (i.e., the main overstory canopy layer) of each tree species are measured in each subplot [43]. The stand-age variable for a "stocked" FIA plot (i.e., one containing trees of any age) is determined from the average of all ring counts from subplot samples of dominant and codominant trees in the size class characteristic of the overstory canopy structure, weighted by cover of sampled trees, and 8 years are added for estimated time to grow to breast height (1.4 m) at which cores are sampled.

We selected FIA data from low- to mid-elevation forest types in Wilderness, Inventoried Roadless Areas, and National Parks to ensure as best we could that stand initiation was not caused by commercial harvesting of trees or other land use (Fig. 1). We had no independent way to confirm that trees were never cut at each plot location, so we interpret the results assuming only that such management was of minor importance, given that Wilderness, Roadless, and National Park designations reflect a lack of past timber harvesting. We selected lands classified as "timberlands" in Pacific states' data sets. In the Rockies and southwestern USA, where there was no such designation, we selected all areas where the potential vegetation was considered capable of >10 percent tree cover.

A small number of plots had different stand ages for different subplots due to disturbances that affected some, but not all, subplots. In FIA split-age plots where both plot ages were ≤100 years, plots were split into two stand ages by FIA if they differed by as little as 1 year. In split-age plots in which both ages were 100–199 years old, plots were split into two stand ages if they differed by as little as 2 years. In split-age plots where both ages were ≥200

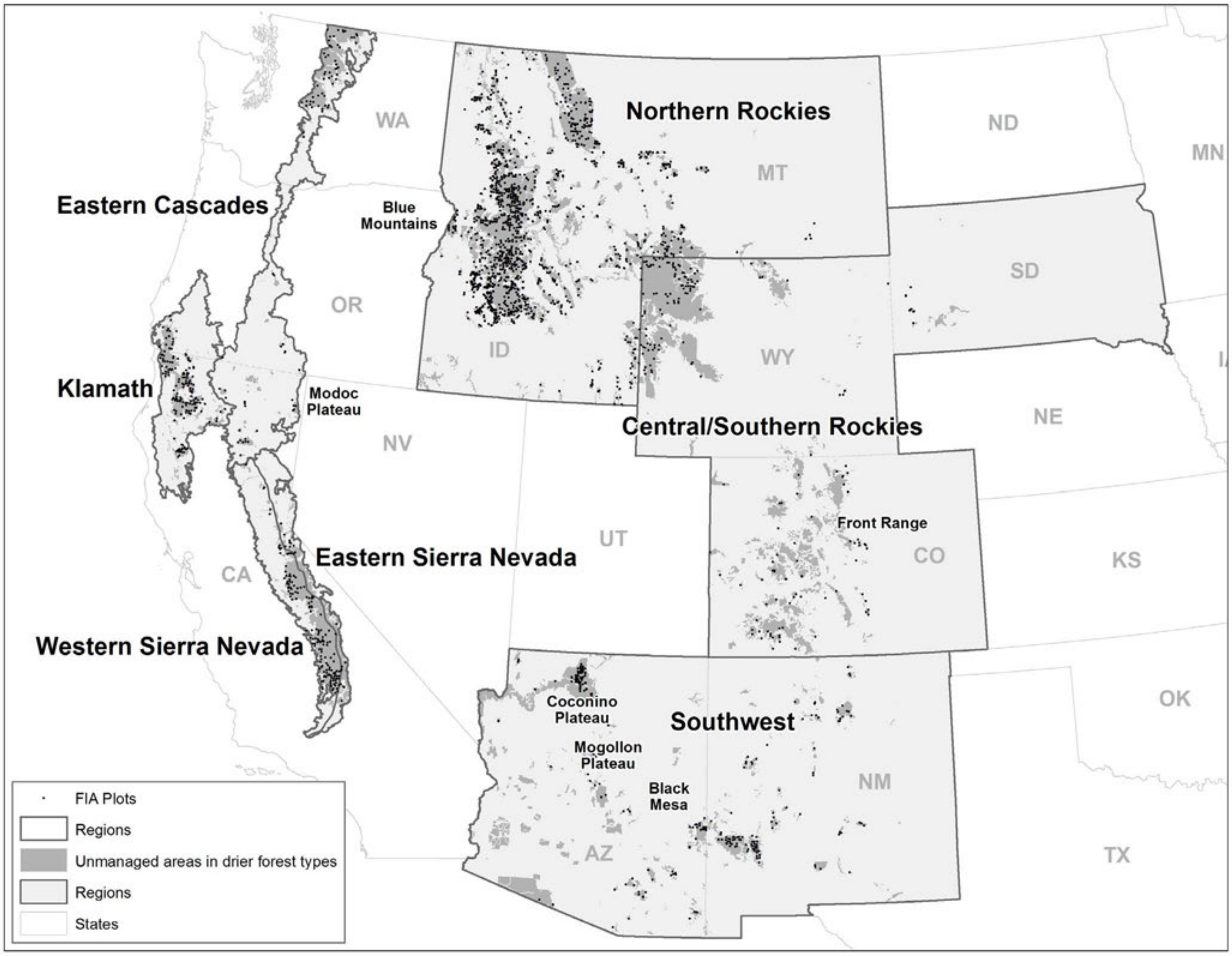

Figure 1. Study area. Dots indicate the general locations of Forest Inventory and Analysis (FIA) plots.
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years, plots were split into two stand ages if they differed by as little as 15 years. To assess the within plot variability in tree ages, we calculated the standard deviation of the trees used to age each plot. We standardized this across the range of stand ages by calculating the standard deviation of the proportional difference between stand age, and the individual trees used to determine stand age in each plot, over the range of stand ages.

We reasoned that, prior to fire suppression, under a low/moderate-severity fire regime, successional, or age-class diversity, would be low, while it would be high under a mixed-severity fire regime. With fire exclusion and greater amounts of uncharacter-

Table 3. Mean annual precipitation, and mean summer maximum and minimum temperatures, in ponderosa pine and mixed-conifer forests in each region.

Region*	Mean annual precipitation (cm)	Mean maximum temperature, June-August (degrees C)	Mean minimum temperature, June-August (degrees C)
Sierra Nevada	104	23	9
Klamath	196	26	11
Eastern Cascades and Eastern Sierra Nevada	113	21	7
Northern Rockies	88	22	6
Central and Southern Rockies	71	22	6
Southwest	58	27	11

*All values are from PRISM data (http://www.prism.oregonstate.edu/normals/) in each 2 km^2 PRISM pixel within which an FIA plot used in the study occurred.
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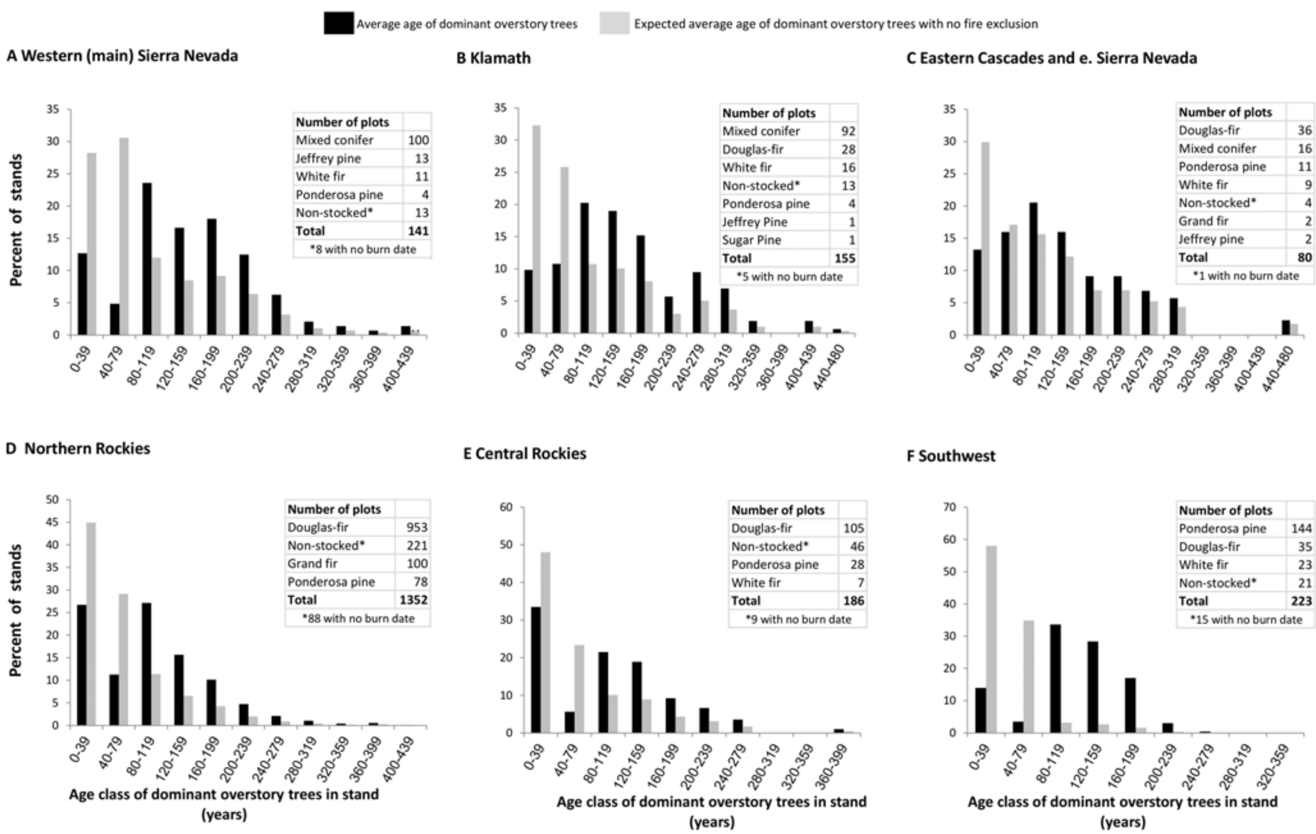
istically severe fire the pattern should reverse in both cases (i.e. increased age-class diversity in low-severity systems and decreased diversity in mixed-severity systems). We used a Chi-square test of proportions [44] to test the null hypothesis that there would be no difference between the actual distribution of stand ages and the distribution based on a hypothetical scenario of no fire exclusion. No effect of fire exclusion would indicate that fire was not a dominant influence on age class diversity. To create a distribution of average dominant stand ages by region that would exist in today's stands had fire exclusion never occurred, we used the distribution of plots with stand ages dating from 1889 or before. This time period was immediately prior to the onset of fire suppression activities by settlers and government agencies [35,45–56]. Because the average tree ages are somewhat imprecise, we binned the data into 40-year age classes for hypothesis testing. In each region, the present age structure for 80 years during effective fire suppression (1930–2009) was compared with the age structure prior to fire suppression (1810–1889). For visual analysis, we shifted the pre-fire suppression (pre-1890) tree age distributions to present (i.e., shifting 1810–1849 to 1930–1969, and shifting 1850–1889 to 1970–2009) to compare with the current age distributions (see Figure 2). This allows a clear, visual comparison of stand ages that currently exist with those that would exist had the same fire regime from 1810–1889 occurred from 1930–2009.

We included only plots where there was one stand age for the full plot because we wanted to evaluate high-severity fire occurrence in patches at least 1 ha in size, rather than include smaller torching of groups of trees. Excluding the split-age plots (27% of plots in the Sierra Nevada, 40% in eastern-Cascades/eastern-Sierra, 26% in Klamath, 14% in northern Rockies, 36% in central/southern Rockies and 14% in southwestern USA) omits some additional evidence for local high-severity fire effects; thus our results may be conservative.

We used FIA data drawn from 2001–2009, comprising 90% of available plots, in our classification of low/mid-elevation forests in the Sierra Nevada, Klamath, and eastern Cascades. In the other regions, FIA plots represented 100% of the data from low- to mid-elevation, montane forests. The number of plots in the 0–39 year age bins may be slight underestimates of the amount of high-severity fire in the last 40 years because severe fire could have occurred subsequent to the sample date (plots were sampled between 1995–2009 in the northern, central and southern Rockies, and southwestern USA and 2001–2009 in the Sierra Nevada, Klamath and eastern Cascades). To estimate the number of plots that burned severely after the sample date, we increased the 0–39 year old bin by a factor of 40/36 in the Sierra Nevada, Klamath and eastern Cascades, 40/34 in the northern Rockies and 40/32.5 in the central/southern Rockies and southwestern USA region. The denominator in these weightings is based on 40



Figures 2. Age class distributions of dominant overstory trees. Data are from US Forest Service Forest Inventory and Monitoring plots from forested areas protected from logging in A. the western (main) Sierra Nevada, B. the Klamath Region, C. the eastern Cascades and Sierra Nevada, D. the northern Rockies, E. the central/southern Rockies, and F. the southwestern USA. Shown in black bars is the current distributions of stand ages. Grey bars show an expected distribution (average age of dominant overstory trees with no fire exclusion), based on projecting the occurrence of the same age distributions that occurred from 1810–1889 into the most recent 80 year time period and rescaling these data. The number of plots by forest type are shown in the imbedded tables. Non-stocked stands are those lacking trees that grew after the fire that could be aged non-destructively.

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minus the mean amount of time in which plots in each region could have burned after being sampled.

We used year of the recent fire disturbance, captured in the disturbance data field, to define the age of very young FIA plots not containing trees that could be aged in a non-destructive way (FIA surveys do not allow trees to be killed). These ''non-stocked'' plots were relatively rare, as reported in the results, and ages, based on fire dates, all fell within the 0–39 age category. Some non-stocked stands had no disturbance coded. In California, Oregon, and Washington (Pacific states), disturbances were only coded if they were <6 years old. We placed all non-stocked plots where no disturbance was coded in the database into the 0–39 stand age bin.

Next, we considered whether the age distributions as shaped by fire were consistent with mixed- or low/moderate-severity fire regimes. We reasoned that a wide range in the plot stand ages in a landscape would be consistent with age-class diversity created by mixed-severity fire, while stand ages that were evenly distributed in predominantly older age classes would be consistent with a low/moderate-severity fire regime. To test whether stand age distributions were consistent with mixed- or low/moderate-severity fire regimes, we again used a Chi-square comparison of proportions [44]. Specifically, we tested the probability that the actual age distributions differed from an expected stand-age distribution for a low/moderate-severity fire regime. The low/moderate-severity (expected) distribution was based on 12.5% of stands falling into each 40-year age class between 80–399 years (0–319) years at the onset of fire exclusion. Our null hypothesis was that there would be no significant difference between the actual and expected (low/moderate-severity) distributions.

Third, we tested, again using a Chi-square comparison of proportions [44], the hypothesis that there would be less evidence for historical mixed-severity fire in the generally drier ponderosa and Jeffrey pine stands than in the mixed-conifer forests (i.e., the pine forests would be more frequently dominated by older stands).

Results

Evidence for Historical Mixed-severity Fire Regimes in Ponderosa Pine and Mixed-conifer Forests

Rotations of high- and moderate- to high-severity fire. The studies that allow calculation of rotations of high-severity fire over large, ponderosa pine and mixed-conifer forest landscapes of western North America over time periods of at least 70 years include areas ranging from 40,700 to 1,193,200 ha (Table 4). These large landscapes totaled 2.2 million ha in Baja California, the Sierra Nevada, eastern Cascades, northern Rockies (Blue Mountains of Oregon), the Colorado Front Range and Arizona (Black Mesa and the Mogollon Plateau). Most of the evidence presented in these studies was from ponderosa pine forests.

The high-severity fire rotations in Table 4 do not support the hypothesis that low/moderate-severity fire regimes were predominant in the majority of ponderosa pine and mixed-conifer forests of western North America. In all the large, forest landscapes for which data covering at least 70 years exist, high-severity fire rotations ranged from about 217 to 849 years [57], and were mostly ~200–500 years. This is generally less than potential tree lifespans. For combined moderate- and high-severity fires in the eastern Cascades, rotations were 115–128 years (Table 4: [35]), while they were 249 years in the Colorado Front Range (Table 4: [58]). In the Blue Mountains (northern Rockies) and on the Mogollon Plateau in Arizona, high-severity fire rotations of 849 and 828 years, and moderate/high-severity fire rotations of 235 and 319 years, respectively [57], occurred. Where high-severity rotations are relatively long, as they are in these regions, forest structure in portions of the landscape will lack evidence for high-severity fire even though it occurs often enough to create age-class diversity. Thus, while about 40% of the Blue Mountains forests and about 62% of those on the Mogollon Plateau had evidence from GLO surveys of forests shaped by low/moderate-severity fire only [57], similar to the nearby Coconino Plateau [59], structural diversity created by high-severity fire was evident on the remainder of the landscape [57,59].

Numerous other studies that describe historical patterns of fire behavior also have documented or described evidence for mixed-severity fire effects in the ponderosa pine and mixed-conifer forests of western North American, including the occurrence of large high-severity fire patches (Table S1), and high-severity fire occurring over substantial areas of smaller landscapes over a time period of only a few decades prior to fire exclusion (e.g., Klamath region and a transitional area between the Sierra Nevada and eastern Cascades [60–63]).

Previous studies (Table 4) have used evidence of past fire severity from a variety of sources: GLO and other survey data, historical aerial photos; and mapping of vegetation and burns done prior to fire exclusion. The GLO analyses have been formally assessed for accuracy [64]. The methods performed well for addressing general hypotheses about the presence or absence of vegetation shaped by low- or high-severity fire. This was tested using existing vegetation plot data with an error of 14.4–23% [64].

Plot age distributions. A total of 2119 FIA plots representing a sample population of about 5.1 million ha of unmanaged low- to mid-elevation, montane forests in six regions (Figure 1, Table 5) were included in our analysis. Stand ages from ponderosa pine and mixed-conifer forests across the western USA never managed for timber cover areas ranging from 192,200 ha in eastern Cascades-eastern Sierra Nevada to 3,244,800 ha in the northern Rockies. Average stand ages ranged from 0 to 814 years, with the oldest stand in ponderosa pine in the eastern Cascades. The within plot standard deviation of the proportional difference among individual tree ages and stand age across all plots was 0.14 (e.g., for stands 100 years old, one standard deviation would include individual trees ~86–114 years old, and two standard deviations would include trees ~72–128 years old).

The comparison of actual stand ages from 1930–2009 and the rescaled (expected) stand ages from 1810–1889 assuming no effect of fire exclusion are shown in Figs. 2A–F. In all regions, there were highly significant differences between the actual and expected stand age distributions (average ages of dominant trees with no fire exclusion) ($P<0.001$, Fig. 2A–F), indicating that fire was the predominant disturbance prior to effective fire exclusion. The FIA database also indicates that, since the onset of fire suppression, the great majority of stands were initiated by fire. As illustrated by the abundance of plots with stand ages that date to the decades prior to fire exclusion (e.g. 80–160 years old presently), much of the landscapes had young forests, but the rate of establishment decreased dramatically after 1930 (stand ages <80 years are rare). The rate of young forest establishment decreased by a factor of 4 in the Sierra Nevada and southwestern USA, by 3x in the Klamath, and 2x in the eastern Cascades and central and northern Rockies.

Chi-square comparisons between actual stand-age distributions at the onset of fire exclusion versus the expected stand-age distributions for a low/moderate-severity fire had exceptionally low probabilities in all regions ($P<<.00001$, $n=61$–877). This was because plots were mostly dominated by young and intermediate aged trees prior to fire exclusion (Figs. 2A–F). The mean stand

Table 4. Rotations for high-severity and moderate-severity fire in low/mid-elevation conifer forests of western North America.

Region	Location	Source	Analysis area (ha)	Forest types	Tree mortality	Time period	Approximate rotation (years)
Pacific states	Northern Baja California	[118][1] analysis of aerial photos	40,700	Mixed conifer and Jeffrey pine	>90% overstory mortality	1925–1991	300
	Northern Sierra Nevada	[51][2] Ground surveys and detailed maps	146,917	Mixed conifer, dominated by ponderosa pine	75% mortality by volume was mapped for patches >32.4 ha	1800–1900	488
	Eastern Cascades (Washington)	[17,35][3, 4]Analysis of historical aerial photos	175,200	Mixed conifer and ponderosa pine	>70% tree mortality[6]	~1830–1930	379–505
					>20% tree mortality[6]	~1830–1930	115–128
	Eastern Cascades (Oregon)	[56][5] Analysis of General Land Office survey data	123,500	Ponderosa pine	>70% tree mortality[7]	~1768–1882	705
			140,400	Dry mixed conifer	>70% tree mortality[7]	~1768–1882	496
Northern Rockies	Oregon (Blue Mountains)	[57][5] Analysis of General Land Office survey data	304,700	Ponderosa pine forests	>70% tree mortality[7]	~1740–1880	849
	Oregon (Blue Mountains)	[57][5] Analysis of General Land Office survey data	304,700	Ponderosa pine forests	Moderate- and high-severity fire	~1740–1880	235
Central Rockies	Central (Colorado Front Range)	[57,73][5] Analysis of General Land Office survey data	65,500	Ponderosa pine forests	>70% tree mortality[7]	~1705–1880	271
	Central (Colorado Front Range)	[58][8] Analysis of General Land Office survey data	624,156	Mostly ponderosa pine and Douglas-fir	Moderate and high-severity fire	1809–1883	249
Southwest (Arizona)	Black Mesa	[57][5] Analysis of General Land Office survey data	151,100	Ponderosa pine forests	>70% tree mortality[7]	~1760–1880	217
	Mogollon Plateau	[57][5] Analysis of General Land Office survey data	452,100	Ponderosa pine forests	>70% tree mortality[7]	~1760–1880	828
	Mogollon Plateau	[57][5] Analysis of General Land Office survey data	452,100	Ponderosa pine forests	Moderate- and high-severity fire	~1760–1880	319
	Black Mesa and Mogollon combined	[57][5] Analysis of General Land Office survey data	603,200	Ponderosa pine forests	>70% tree mortality[7]	~1760–1880	522

Data from General Land Office or mapped data over large areas (>25,000 ha) over >70 or more years prior to fire exclusion.
[1]Study area was dominated by mixed conifer and Jeffrey pine and minimally logged. Fire exclusion only began in the 1970s and has had only a modest impact [138]. Thus, historical and current rates are assumed to be comparable.
[2]Analysis of Leiberg's mapping of high-severity fire areas within unlogged mixed-conifer Sierran stands is found in [139]. According to Leiberg [51], most such fire occurred prior to 1850. In addition, he stated "If the many small lots [<32 ha] scattered throughout still growing stands were taken into account, the figure [amount of area burned severely] would be considerably increased."
[3]The numerator was estimated at 100 years based on ponderosa pine in this region [140], whose growth would surpass 30 cm dbh, rendering mixed and high-severity effects indistinguishable (see [35: Table 1]). This calculation is conservative because tree growth to 30 cm dbh in moister forests is faster than 100 years.
[4]High- and mixed-severity fire consistent with a definition of >70% and 20–70% basal area mortality, respectively, was identified from overstory canopy percentage, the overstory size class, the understory size class, and the fire tolerance of the cover type (see [35: Table 1]). Large patch sizes of historical high severity fire (100s to >5000 ha) from this work are reported in [17].
[5]The estimate is from the span of years over which fire effects were distinguishable, using forest structure evident in the Government Land Office historical survey data, divided by the fraction of the forested landscape in which those fires occurred [56]. Rotations for high-severity fire are determined by dividing the observation period (the period of time over which fire effects are distinguishable by stand structure) by the percentage of the landscape experiencing high-severity fire. The methods were found to have 14.4–23% accuracy compared to plot sampling.
[6]High- and mixed-severity fire, consistent with a definition of >70% and 20–70%, respectively, were identified from overstory canopy percentage, the overstory size class, the understory size class, and the fire tolerance of the cover type (see [35: Table 1]).
[7]High-severity consistent with a definition of >70% basal area mortality [35] was identified having a percentage of small trees >50% and a percentage of large trees <20% [56,57,73],
[8]Estimated from the length of General Land Office section lines intercepted by moderate- and high-severity fire. Accuracy tests using the length of section lines intercepted by modern moderate- and high-severity fire yielded a relative error of 15.6%.
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Table 5. Forest Inventory and Analysis (FIA) data.

Region	Number of plots (n) and forest area randomly sampled (ha)	Mean FIA stand age (yrs)		Test for difference in stand initiation since 1930 vs. 1800–1900: Chi-square, *P*
		Current	In 1930	
Sierra Nevada (main)	n = 232 338,400	148	97	86.3, <<0.001
E. Cascades and E. Sierra Nevada	n = 135 192,000	155	114	25.4, <<0.001
Klamath Mountains	n = 251 372,000	157	111	43.9, <<0.001
Central Rockies	n = 276 446,400	105	75	58.9, <<0.001
Northern Rockies	n = 1929 3,244,800	105	70	333.8<<0.001
Southwestern US	n = 319 492,000	116	59	188.2<<0.001

Area of sample population randomly sampled, mean stand age currently, and in 1930, and Chi-square test results.
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ages at the time of onset of fire exclusion were 59–114 years, depending on the region, considerably shorter than current mean ages (105–148 years: Table 5). Therefore, the FIA data were inconsistent with the hypothesis that the ponderosa pine and mixed-conifer forests of western North America, in unmanaged landscapes, were predominantly park-like with low age-class diversity due to the dominant influence of low/moderate-severity fire.

The hypothesis that mixed-severity fire prior to fire exclusion would be lower in the driest (ponderosa and Jeffrey pine) forests than other forests also was not supported. Based on stand-ages (not shown), there was as much as or more mixed-severity fire in the pine forests. In the Pacific states, we found almost identical stand-age distributions from 1800–1900 in ponderosa/Jeffrey pine stands (n = 20 plots) versus all non-ponderosa stands (n = 204 plots). Plots from the time period 1800–1900 accounted for 70% and 73%, respectively, of all plots with dominant trees that established in or before 1900. In the northern and central Rockies, 86% of ponderosa pine stands (n = 66 plots) and 81% of the non-ponderosa pine stands (n = 615 plots) that established in or before 1900 had stand-ages between 1800 and 1900 ($\chi^2 = 0.85$, n = 676, $P > 0.6$). Likewise, in the southwestern USA, 98% of ponderosa pine stands (n = 96 plots) and 92% of the non-ponderosa stands (n = 37 plots) that established in or before 1900 had stand-ages between 1800 and 1900 ($\chi^2 = 1.27$, n = 133, $P > 0.25$). However, when all plots were considered, significantly more stands established from 1800–1900 in ponderosa pine than non-ponderosa forests ($\chi^2 = 11.96$, n = 1038, $P < 0.001$), indicating higher fire disturbance in pine forests.

Comparing the Weight of Landscape-scale Evidence by Region

The consistency of multiple lines of evidence for mixed-severity fire in the ponderosa pine and mixed-conifer forests is an important finding. In all regions, there were tree-age data supporting considerable age-class diversity created by mixed-severity fire, and a paucity of undisturbed park-like forests. The full weight of landscape-scale evidence is greatest in the regions with area-specific rotations of severe fire from GLO data: the eastern Cascades, nearby Blue Mountains in the northern Rockies, central Rockies, and southwestern USA (Table 4). In the Cascades, these data are further supported by analyses of early aerial photography at a regional scale [35], and in small landscapes [61–63] and numerous historical descriptions (see [56]: Table S1). In the northern Rockies, historical documentation (e.g., [45–48,50,53,54]) of mixed-severity regimes has been summarized in regional reviews [16,65,66], and stand-age reconstructions of historical fire regimes indicate mixed-severity fire in ponderosa-pine/Douglas-fir forests [67–69]. In the Colorado Front Range, the findings based on GLO data [57,58] are remarkably consistent with earlier studies based on tree-ring stand reconstructions from broadly distributed samples [70–72]. In the

Table 6. Current high-severity fire rotations.

Region	Source	Forest Types	Time period	Rotation (yrs)
Sierra Nevada, southern Cascades	[132]	All low/mid- and mid/upper elevation conifer forests	1984–2010	645
Klamath (all)	[129]	All low/mid-elevation conifer forests	1984–2005	599
Eastern Cascades (all)	[129]	All low/mid-elevation conifer forests	1984–2005	889
Northern Rockies	[92]	Ponderosa pine forests	1980–2003	500
Central Rockies	[92]	Ponderosa pine forests	1980–2003	714
Central Rockies	[58]	Ponderosa pine forests	1984–2009	431[1]
Southwest	[92]	Ponderosa pine forests	1984–2003	625
Northwest (Eastern Cascades and Blue Mountains)	[92]	Ponderosa pine	1984–2003	1,000

Data cited are from low/mid-elevation conifer forests in western North America.
[1]Higher-severity fire: includes moderate- and high-severity.
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southwestern USA, GLO data are supportive of mixed-severity fire on most of Black Mesa and much of the Mogollon and Coconino Plateaus [57,59], while a number of other studies also describe evidence for mixed-severity fire [9,14,55,73–77].

The remaining forest regions that we assessed lack GLO analyses. However, in the Sierra Nevada and Klamath regions historical surveys and early air photo data describe mixed-severity fire regimes [20,30,49,51,60,78–87] (see Table S1 for descriptions). In all regions except the Klamath, there are multiple lines of evidence from landscape-scale studies, each supporting mixed-severity fire. In contrast, evidence supporting low/moderate-severity fire is confined to relatively small areas (e.g., [88–91]).

Historic vs. Contemporary Fire Regimes

We did not find evidence to support the hypothesis that fire exclusion has greatly increased the prevalence of severe fire in ponderosa pine and mixed-conifer forests (Tables 4–6, and Figs. 2A–F). Comparing current versus historical high-severity fire rotations, we found that current rotations were generally longer (less high-severity fire) in the Sierra Nevada and central Rockies (Tables 4 and 6, Table S1). No direct historical comparison could be made between current and historical high-severity rotations in the Klamath and northern Rockies at the spatial scale required in Table 4, but evidence presented in Table S1 suggests that current rotations of 599 years and 500 years, respectively, may be longer. The estimated rotation of 625 years for recent high-severity fire in the southwestern USA [92] was shorter than the historical estimate of 828 years for the Mogollon Plateau in Arizona. Combining the Mogollon Plateau and Black Mesa to provide a better comparison with fire across the southwestern USA produces a historical high-severity rotation of 522 years [57]. In the eastern Cascades, high-severity fire rotations since 1984 (889 years) were longer than historical rotations (Table 6 vs. Table 4).

Discussion

Historical Fire Regimes

The primary objective of this paper was to address how prevalent mixed-severity fire regimes were historically in ponderosa pine, mixed conifer, and other low- to mid- elevation, montane forests of western North America. We hypothesized that age-class diversity was low, consistent with long-term effects of low/moderate-severity fire regimes (Table 1). We reviewed evidence with inference across both large and smaller landscapes across many forest regions. The majority of the evidence did not support the low/moderate-severity fire hypothesis, but, instead, supported the alternate hypothesis that mixed-severity fire shaped these forest landscapes. This finding applies to Pacific states ponderosa pine, Jeffrey pine, and California mixed-conifer forests, as well as ponderosa pine and mixed-conifer forests in the eastern Cascades, Rockies and southwestern USA, where low/moderate-severity regimes have often been applied. In some areas (Blue Mountains, Mogollon and Coconino Plateaus) high-severity fire occurred at less frequent intervals (rotations of 828–849 years) [57,59]. Even at these rotations, high-severity fire creates considerable age-class diversity in a landscape, and moderate/high-severity fire rotations were 235–319 years, which further enhances diversity (with small groupings of high-severity fire interspersed within moderate-severity fire areas).

FIA stand ages in the unmanaged forests in all regions reflect a pattern of high age-class diversity occurring prior to federal fire suppression policies and reductions in Native American burning (by the early 20th century) with the arrival of settlers [20,46,49,51] Natural disturbances occurred at rates that led to stands numerically dominated mainly by young and intermediate-aged trees. Disturbance processes dramatically declined following the onset of fire exclusion, suggesting fire was the primary disturbance agent [35]. However, in considering the age patterns of dominant trees in the FIA plots, it is essential to also address alternative explanations for the dominance of young- and intermediate-aged stands prior to fire exclusion, such as climate variability and disturbance by insect outbreaks.

While we recognize that climate variability influences rates of tree regeneration generally [93], and may determine success or failure of tree regeneration specifically following disturbance, we believe that the broad patterns of dominant overstory tree ages in the FIA plots mainly reflect the effects of past fire for several reasons. The dominant stand ages of young and intermediate aged trees prior to fire exclusion are consistent with periodic disturbances with significant tree mortality that shifted dominance to a new generation of trees, rather than solely episodic tree establishment due to climatic variation at a multi-decadal scale. This is supported by research in the central Rockies where, at a multi-decadal time scales, large datasets of tree recruitment dates over the past c. 250 years do not correlate with moister climate at the same time scale, but instead correlate with drier climate that was conducive to high-severity fires [70–72,91]. Likewise, studies in the same area show that outbreaks of bark beetles and defoliators result in growth releases of non-host trees rather than even-aged, multi-species tree cohorts [94,95], thus facilitating discrimination from post-fire stand structures [91]. Fire exclusion was likely effective in some areas between 1900 and 1930, which could have led to understory tree recruitment in this time frame. However, research suggests that in some areas the favorable influences of timber harvesting and/or cattle grazing on tree establishment may confound the attribution of tree recruitment to fire exclusion [96]. In addition, the plot age data demonstrate that recruitment was just as common or more common in decades before 1900 as between 1890 and 1930. Lastly, while it is possible that greater mortality in older trees, from competition or insects and pathogens, might explain high levels of recruitment prior to fire exclusion, we do not see this pattern during the suppression era. Thus, higher levels of mortality in older trees seems likely to have been caused by fire.

Our findings illustrate the need for studies with a spatial scale of inference suited to describing patterns across large, heterogeneous landscapes. This is illustrated by three recent studies from old forest stands (one in the Black Hills (500 ha), one in the Sierra Nevada (3,000 ha), and one in the southwestern USA (307 ha)) that reported very little or no historical high-severity fire, and hence low-severity regimes (Table S1: [88–90]). In contrast, broader-scale analyses of historical data for the Sierra Nevada (Table S1: [78]:), Black Hills [65], and southwestern USA [57] suggest fire regimes in the broader landscape within which these three studies occurred were mixed-severity.

A fourth study [97] analyzed 1914–1922 Bureau of Indian Affairs (BIA) timber cruise plot data from within a larger area (38,651 ha), and found relatively low tree densities in ponderosa pine and mixed-conifer forests of the eastern Klamath region in Oregon, and suggested that forests were too open to support any significant crown fire. However, only a subset of the townships surveyed by BIA in these forest types were included in the analysis (Table S1), and the surveys did not include trees 10–15 cm dbh, which comprise ~20% of all trees [97], and most surveys did not include lodgepole pine, which comprise ~10% or more of these forest types in that region within unlogged areas [49]. In addition, historical data indicate that extensive timber harvesting had occurred in the areas analyzed by 1914–1922 (Table S1), and evidence of previous timber harvesting was not among the factors that BIA surveyors were required to note (Table S1). Tree densities in unlogged reference ponderosa pine and mixed-conifer forests in this landscape from the late 19th century and early 20th century indicate much denser and more variable forest conditions (Table S1). Also, USGS surveys conducted in the 1890s within unlogged ponderosa pine and mixed-conifer forests across a larger expanse (310,267 ha) map substantial high-severity fire from 1855–1900 (high-severity rotation of 352 years), suggesting a mixed-severity regime (Table S1).

The absence of evidence for mixed-severity fire in some older forests selected for study may be due to fire exclusion. If the effect of fire exclusion in reducing mixed-severity fire is not accounted for in describing reference conditions, it may lead to shifting baseline syndrome (i.e., a system is not measured against the true baseline, but against one that already has departed from the true baseline [98]). This effect may be caused or compounded by diminishing evidence of age-class diversity. For example, high-severity fire can be mapped at landscape scales from early air photos [9,17,61–63,99], but the same historic fire effects may not be visible from current imagery that can be used for assessing landscape-scale patterns.

Data with greater temporal depth than analyzed here can better capture past variability in the frequency of large fire events. Thus, it is noteworthy that paleoecological studies also support mixed-severity fire regimes for the ponderosa pine and mixed-conifer forests. These studies have found charcoal depositions from major fire episodes in ponderosa pine and interior Douglas-fir forests occurring for millennia in the northern Rockies (central Idaho: [100,101]), Klamath [102], Sierra Nevada [103], eastern Oregon Cascades [104], and southwestern USA [105–107]. These major episodes are generally interpreted as large, severe fire events [101–107].

The occurrence of mixed-severity fire prior to fire exclusion is also well supported by another line of evidence: the potential behavior of wildfire as affected by weather and climate. Based on direct observations of fire behavior, high winds (generally 10 m open wind speeds >32–35 kilometers/hr) may subject virtually any conifer forest, regardless of fuel density, to crown fire [108]. Thus, empirical data call into question a major premise of the low/moderate-severity fire regime: that ponderosa pine and mixed-conifer forests may be completely resistant to crown fire. Fire intensity increases with winds, and at winds of >30 km/hr spot fires may be ignited over 1 km ahead of the fire front [109]. The coalescing of separate spot fires with the fire front can further energize wind-driven fire [110,111]. Severe droughts also intensify fires by reducing fuel moisture to extremely low levels, allowing crown fire under less windy conditions [108,112]. Severe drought years throughout much of western North America occurred from 1856 to 1865, 1870 to 1877 and 1890 to 1896 [113]. The extensive high-severity fires of 1910 (the Big Burn in Idaho and Montana), when large areas of drier forests burned at high severity prior to fire exclusion–much of it in ponderosa pine–illustrate how fire behavior that is rare temporally due to extreme climate and weather can dominate in space [1]. Many fire episodes in the charcoal records that exceed modern fires undoubtedly involve combinations of extreme wind, drought, and mass fire.

The largest patch sizes of high-severity fire likely occurred during the most extreme conditions for fire behavior. While patch sizes of high-severity fire are difficult to document, it follows from commonly observed heavy-tailed distributions of patch sizes created by fire [114,115] that very large patches of high-severity fire (thousands of ha, e.g., [17: Fig. 1, 58]) were a primary reason why considerable area exhibited forest structure consistent with high-severity fire historically in all regions. Large patches, though numerically subordinate, are dominant in terms of total area burned, while the opposite applies to small patches [58].

There is abundant evidence that past forests may not have required extreme weather and climate for mixed-severity fire to have occurred. Younger, more flammable forests [32] appear to have been widespread in dry-forest regions based on dominant stand ages prior to fire exclusion (Fig. 2). In addition, the ranges in fire-free intervals in many low- to mid-elevation forested areas were sufficient to allow for substantial vegetation growth and recovery of fuels between fires (e.g., 20–50+ year rotations [61–63,116–118]). For example, in the Sierra Nevada, fuels may recover to pre-burn levels in nine years [119,120], so fire-free interludes (or fire rotations), more often than not, may have been sufficient to allow growth of significant amounts of high-energy shrub fuels. In describing low/mid-elevation forests throughout the northern Sierra Nevada, Leiberg [51: page 32] states: "There is a great amount of undergrowth in the forest which has attained its present proportions chiefly through the agency of fire. Most of it [undergrowth] consists of species of *Ceanothus*." For mid-elevation forests, he reports (page 37): "Nearly all the type situated at altitudes below 7,000' [2134 m] carries a vast amount of undergrowth. It consists mainly of manzanita [*Arctostaphylos* spp.], ceanothus, and scrub oak [*Quercus chrysolepis, Q.vaccinifolia*]." Similarly abundant shrub fuels were also documented historically in the westside of the central/southern Sierra Nevada [51], in the eastern Oregon Cascades [56: Appendix A] and in Oregon's Blue Mountains [57]. Flame lengths in actively burning manzanita and ceanothus are typically 4–5 times the ~1–2 m height of the shrubs, sufficient to cause ignition of forest canopy tree crowns under favorable burning conditions. Many of these shrub species recruit primarily, if not exclusively, after severe fire, and their occurrence is a further indication of the historical presence of such fire [121].

Changes in Fire Regimes and Stand Age Distributions with Fire Exclusion

We also hypothesized, consistent with existing concerns about unprecedented fire severity in western North America (e.g., [6–9,11,13,15,17,28]), that fire exclusion has greatly increased the prevalence of severe fire in ponderosa pine and mixed-conifer forests. We found little support for this hypothesis. Over the full period of effective fire exclusion in unmanaged forests, average ages of dominant overstory trees in FIA plots suggest there has been about a threefold to fourfold decrease in stand initiation due to fire in the Sierra Nevada, Klamath, and southwestern USA, and about a twofold decrease in the eastern Cascades, central and northern Rockies (Figs. 2A–F). In addition, patch sizes of high-severity fire in the central Rockies have not increased [58]. Our assessment of high-severity rotations based upon existing literature also revealed a generally lower incidence of high-severity fire in these forests in recent decades (Tables 4 and 6, and S1).

Conclusion

The importance of multiple lines of evidence has been stressed in determining whether mixed-severity fire regimes applied historically [122]. Our results illustrate broad evidence of mixed-severity fire regimes in ponderosa pine and mixed-conifer forests of western North America. Prior to settlement and fire exclusion, these forests historically exhibited much greater structural and successional diversity than implied by the low/moderate-severity model (Table 2). Lack of recognition of past variability in fire may be due, in part, to common misclassifications of fire regimes. To improve clarity in communication, we propose that "low/moderate-severity" be applied to those regimes where, as the term implies, high-severity fire is absent. These circumstances appear to be quite rare in the ponderosa pine and mixed-conifer forests of western North America. Therefore, a fire regime with a high-severity component of any amount should not be classified as low/moderate-severity [e.g., 9,17,28].

Our findings suggest a need to recognize mixed-severity fire regimes (Table 2) as the predominant fire regime for most of the ponderosa pine and mixed-conifer forests of western North America. Given societal aversion to wildfires, the threat to human assets from wildfires, and anticipated effects of climate change on future wildfires, many will question the wisdom of incorporating historical mixed-severity fire into management goals. However, focusing fire risk reduction activities adjacent to homes is needed to protect communities [123], and this may expand opportunities for managed wildland fire–away from towns–for ecological benefits of fire-dependent biota. However, a major challenge lies with the transfer of information needed to move the public and decision-makers from the current perspective–that the effects of contemporary mixed-severity fire events are unnatural, harmful, inappropriate and more extensive due to fire exclusion–to embrace a different paradigm [124]. This paradigm would not emphasize a single, appropriate condition, but would explicitly recognize the vital role of variation in fire in maintaining successional diversity and fire-dependent biota [125], and allow natural rates of ecological succession [18,19,126–128]. It would also recognize that these effects have generally diminished, and that more fire, including high-severity fire, where it is in deficit, is an ecologically desirable goal. Of course, while most current research indicates that fire severity is not increasing in ponderosa pine and mixed-conifer forests of western North America [129–132], it will be critical to continually assess fire regimes in a changing climate.

For management, perhaps the most profound implication of this study is that the need for forest "restoration" designed to reduce variation in fire behavior may be much less extensive than implied by many current forest management plans or promoted by recent legislation. Incorporating mixed-severity fire into management goals, and adapting human communities to fire by focusing fire risk reduction activities adjacent to homes [123], may help maintain characteristic biodiversity, expand opportunities to manage fire for ecological benefits, reduce management costs, and protect human communities.

Supporting Information

Table S1 Evidence of historic fire severity in ponderosa pine and mixed-conifer forests of western North America. A summary of published studies and historical documents that provide evidence regarding mixed-severity fire in the ponderosa pine and mixed-conifer forests of western North America, but do not provide sufficient information to estimate high-severity fire rotations, or were conducted in smaller landscapes. Many fire scar studies have also been done in these forests, but fire scars alone are not sufficient to distinguish low- from mixed-severity regimes.
(DOCX)

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Author Contributions

Conceived and designed the experiments: DCO CTH. Performed the experiments: DCO CTH AA WLB DAD RH WK MAM RS TTV MAW. Analyzed the data: DCO CTH AA WLB DAD RH WK MAM RS TTV MAW. Wrote the paper: DCO CTH AA WLB DAD RH WK MAM RS TTV MAW.

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FORMAL COMMENT

Areas of Agreement and Disagreement Regarding Ponderosa Pine and Mixed Conifer Forest Fire Regimes: A Dialogue with Stevens et al.

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Abstract

In a recent PLOS ONE paper, we conducted an evidence-based analysis of current versus historical fire regimes and concluded that traditionally defined reference conditions of low-severity fire regimes for ponderosa pine (*Pinus ponderosa*) and mixed-conifer forests were incomplete, missing considerable variability in forest structure and fire regimes. Stevens et al. (this issue) agree that high-severity fire was a component of these forests, but disagree that one of the several sources of evidence, stand age from a large number of forest inventory and analysis (FIA) plots across the western USA, support our findings that severe fire played more than a minor role ecologically in these forests. Here we highlight areas of agreement and disagreement about past fire, and analyze the methods Stevens et al. used to assess the FIA stand-age data. We found a major problem with a calculation they used to conclude that the FIA data were not useful for evaluating fire regimes. Their calculation, as well as a narrowing of the definition of high-severity fire from the one we used, leads to a large underestimate of conditions consistent with historical high-severity fire. The FIA stand age data do have limitations but they are consistent with other landscape-inference data sources in supporting a broader paradigm about historical variability of fire in ponderosa and mixed-conifer forests than had been traditionally recognized, as described in our previous PLOS paper.

Introduction

The accompanying paper by Stevens et al. [1] is critical of one of the several lines of evidence in Odion et al. (2014) [2] that indicate the traditional reference conditions of low-severity fire regimes are incomplete for most ponderosa pine and mixed-conifer forests of western North America. Specifically, Stevens et al. [1] believe that the stand age attribute in Forest Inventory and Analysis (FIA) data is not a useful descriptor of historical fire regimes in ponderosa pine and mixed-conifer forests.

Here, we first briefly summarize points of agreement between Stevens et al. [1] and us, and then discuss in more detail areas where we disagree, including the analysis and interpretation of FIA stand age data. Authorship of this reply is comprised by those who conducted the FIA portion of Odion et al. (2014) [2], as well as authors of Odion et al. whose contributions and backgrounds were needed to respond to FIA-critique elements by Stevens et al. [1] that went beyond the scope of the FIA analysis in Odion et al. (2014) [2].

Areas of Agreement

High-severity fire is a natural component of ponderosa pine and mixed-conifer fire regimes

In Odion et al. (2014) [2], we presented several lines of converging evidence that high-severity fire was an important part of historical fire regimes in ponderosa pine and mixed-conifer forests. Over three-quarters of our results pertained to lines of evidence other than FIA stand age data. Stevens et al. [1] reviewed this evidence, some of which was based upon studies published by co-authors of Stevens et al., and concluded the following: "High-severity fire was undoubtedly a component of fire regimes in ponderosa pine and drier mixed-conifer forests." This represents a significant shift from perspectives in much of the literature in recent decades, which often mentions only low- or low-moderate severity fire in describing historical fire regimes in ponderosa pine and mixed-conifer forests.

Significant tree recruitment occurs in the absence of fire

We did not intend to suggest that tree recruitment occurred only with fire. Stevens et al. hypothesize that pulsed recruitment in the absence of fire has shaped the age distributions in many FIA plots. We agree that this process occurred historically. There is also agreement that a dominant cohort of trees will establish after high-severity fire, but that later in stand development understory recruitment can happen with favorable climate or following insect outbreaks. This, along with the presence of some trees that pre-date the fire, will create an uneven-aged stand, but there may still be a dominant overstory size class established after fire.

FIA Stand Age Data May Provide Evidence Consistent with Past High-Severity Fire

Stevens et al. [1] report that 42% of the FIA plots used in Odion et al. (2014) [2] had demographic characteristics consistent with a mortality and recruitment event corresponding generally with the FIA stand age. These plots had an estimated 0–10% of the stand basal area in trees that were older than the stand age. The rest of the basal area (all of it in many cases) was from trees that established after (more recently than) the stand age date, even though most of the plots had stand ages < 200 years old. Despite some qualifications, Stevens et al. [1] conclude that it is plausible that these 42% of plots were visited by historical high-severity fire. However, although Stevens et al. recognize high-severity fire as a component of ponderosa pine and mixed-conifer forests, the definition (threshold of mortality) and patch size of high-severity fire remain a matter of considerable debate.

Areas of Disagreement

Appropriate threshold of mortality for high severity fire

Stevens et al. replaced the traditional 70–100% mortality definition for high-severity fire (see, e.g., [3]) that we used with a new 90–100% definition, which means their analysis does not replicate ours and does not refute our findings. Even though this replacement invalidates their analysis of our study, 42% of the FIA plots still have demographic characteristics consistent with high-severity fire with their narrowed definition. Using our original 70–100% mortality definition, there is agreement on 68% of FIA plots regarding demographic characteristics consistent with high-severity fire (and the level of agreement is even higher than this, due to a calculation error in Stevens et al., as discussed below).

Stevens et al. [1] suggest, based on findings of Miller and Quayle (2015) [4], that the high-severity fire definition used by Odion et al. (2014) [2] should be narrowed from 70–100% basal area mortality to 90–100% basal area mortality because Miller and Quayle found that high-severity fire field plots with less than 100% tree mortality were rare. However, 34% of all of their plots with ≥75% basal area mortality had live, surviving trees [4]. Thus, surviving trees in high-severity fire plots were not rare based on data that they cite. Further, Miller and Quayle [4] used plots ranging in size from 0.07 ha to 0.63 ha, while FIA plots consist of four subplots spread over an area of 1.0 ha. Thus, the plots of interest here are more likely to contain surviving trees than those of Miller and Quayle [4]. Further, Miller and Quayle (2015) [4] indicate a user and producer accuracy of 11.1 and 19.2 percent for classifying areas with 75–89% percent basal area mortality. Therefore areas with 75–89% mortality were often not identified correctly in their study.

There is also a logical problem: if high-severity fire predominantly caused 90–100% mortality historically, and 70–89% mortality was rare, then there would be very little difference between the number of FIA plots with 90–100% mortality and the number with 70–100% mortality. But, Stevens et al. found a large difference when using these basal area thresholds. Therefore, plots with 70–89% mortality were not rare, and narrowing the fire-severity definition is not supported.

Stevens et al. [1] state that the "minimum threshold of 70% mortality used by Odion et al. [2] to describe a high-severity patch (and the 75% threshold employed by Landfire) was not developed to describe mortality within a stand, but rather mortality across an entire fire." However, the two studies cited by Stevens et al. [1] to support this, Agee (1993) [3] and Barrett et al. (2010) [5], say the opposite (see page 23 of Agee 1993 [3], and page 30 of Barrett et al. (2010) [5].

Plot sizes needed to define high-severity fire

Stevens et al. [1] point out that FIA plot footprints are only 0.4 ha in size in California, Oregon, and Washington, and are only 0.067 ha in size in the other western U.S. states, and use this to suggest that the FIA plots analyzed by Odion et al. [2] were too small to capture true high-severity fire effects. However, Stevens et al. [1] recognize high-severity fire patches as small as 0.4 ha as representing high-severity fire effects. Further, although the total footprint of subplots in FIA plots may be only 0.067 or 0.4 ha, these subplots are representative of a 1.0 ha area. The FIA plots do not capture the size and shape of patches of historical fire, and do not encompass many high-severity patches, which we recognize. But, because they are probabilistic samples, the amount of high-severity fire captured by FIA is a statistical estimate of total amount of high-severity fire. It would be a problem if high-severity fire were rare, or if only a small number of FIA plots were analyzed, but evidence for high-severity fire was abundant, and we analyzed thousands of plots.

Use of diameter-age relationships for reconstructing past basal area of trees

To understand historical forest structure and fire, it is common to reconstruct the size of trees in the 1800s by subtracting tree growth since that time (e.g., [6]). Stevens et al. recognize that the "basal area of the surviving older trees would have increased in the decades between the year implied by the FIA stand age and the measurement date, thus potentially overestimating their past contribution to the stand basal area in the year implied by the FIA stand age." In other words, to the extent that the basal area of surviving trees is overestimated, this translates directly to an under-representation of the potential occurrence of historical high-severity fire. However, Stevens et al. [1] did not subtract the basal area that overestimates the past contributions of surviving trees. The effects can be seen via the following general simulation.

Suppose a plot was burned by high-severity fire 100 years ago with 6.1% basal area surviving fire consisting of 16 m^2 of dead tree basal area. There are 5 live trees of 0.5 m in diameter at breast height (dbh) in the 1-ha FIA plot for a total of 1 m^2 live, surviving basal area. The surviving trees have a higher growth increment in earlier years which decreases as they age. However, when the mean growth rate is calculated using 1594 mature ponderosa pine in dry forests in Oregon [7], the effects of the slower growth at old age is included to give a mean of 0.45 cm dbh/yr. By not considering the growth rates of surviving trees, surviving basal area at the time of the fire would be overestimated by 3.5 times 100 years later. After two hundred years, the age of some FIA plots, the overestimate would be nearly 8 times the actual plot survivorship, with nearly half the basal area incorrectly considered to have survived since prior to the stand age date. Mortality of mature trees after (more recent than) the stand age date could have occurred in some cases, reducing the overestimates by Stevens et al., but this would likely be a small amount compared to the large magnitude of the overestimates. Thus, the potential effects of high-severity fire were greatly underestimated by Stevens et al.

Evidence for historical high-severity fire patches >1,000 ha in size

Stevens et al. [1] suggest that high-severity fire patches $>$1,000 ha in size in some current fires represent a "departure" from historical conditions. However, DellaSala and Hanson (2015) ([8]: pp. 30–33) present numerous examples of historical data sources documenting high-severity fire patches $>$1,000 ha occurring before fire suppression in previously unlogged forests in both ponderosa pine and mixed-conifer forest types in every major region of the western U. S. Even though large high-severity patches may have been infrequent, they accounted for most high-severity fire [9].

Combining fire scar data and stand structure data from different plots

Stevens et al. [1] try to test the hypothesis that there would be minimal tree recruitment in the absence of high-severity fire in the FIA plots we studied. However, the locations chosen by Stevens et al. [1] to evaluate recruitment and fire in FIA plots did not actually include any FIA plots. The locations were mostly subjectively selected plots known to not have had severe fire in their long fire-scar history. The plots were up to 1 km away from any FIA plots. Therefore, they do not represent the population of FIA plots we studied.

Fire and tree recruitment

In all six regions we analyzed in Odion et al., the onset of fire suppression about a century ago coincides with a dramatic reduction in the initiation of trees that form the dominant overstory size classes. Thus, the removal of fire had a profound effect on the process of recruitment over vast areas. Recruitment following fire suppression, as hypothesized by Stevens et al., could not account for the pattern of abundant establishment of the dominant size classes of trees before fire suppression. If high-severity fire was a minor process in creating new stand ages, establishment of the dominant overstory trees would not have declined so dramatically with fire suppression.

Stevens et al. claim that "Most" ponderosa pine forests and "many" low/mid-elevation mixed-conifer forests historically were "Low-density" forests with frequent, fuel-limited low/moderate-severity fire regimes is not supported by the evidence

This suggestion by Stevens et al. [1] overstates certain evidence, and does not consider other evidence. The sources cited by Stevens et al. [1] are a biased selection of studies that were mostly conducted at relatively local spatial scales, and were often in old-growth forests that are inherently low-density and by definition have not experienced high-severity fire for centuries. The sources cited by Stevens et al. also include studies of current tree densities that try to determine past tree densities but do not have any way to measure historical trees that died, fell, and decayed, and studies where the past effects of logging or fuel wood cutting (when mining occurred and large amounts of wood fuel was needed) cannot be ruled out [2] or where incomplete historical survey data were used [10]. Additionally, Stevens et al. omit reference to dozens of scientific sources indicating more variable historical ponderosa pine and mixed-conifer forests.

In contrast, Odion et al. (2014)[2] reviewed dozens of historical data sources and reconstructions, finding that historical ponderosa pine and mixed-conifer forests: (1) were highly variable in structure/density; (2) had highly variable fire severity, and most forests were dominated by mixed- and high-severity fire; and (3) consistently had a significant component of open forests dominated by low-severity fire at any given time.

Conclusion

The concern raised by Stevens et al. [1] pertains to only one of the multiple lines of evidence in Odion et al. [2] that together strongly support the historical importance of high-severity fire in ponderosa pine and mixed-conifer forests of the western U.S. Stevens et al.'s comment, specifically on stand age analysis based on Forest Inventory and Analysis field plots, does not refute our study. This is because it is based on a different definition of high-severity fire than the classical definition used by Odion et al. (2014) [2], which is consistent with scientific literature. The new definition proposed by Stevens et al. [1] is based on errors and mischaracterizations of cited sources. Using our definition or theirs of high severity, Stevens et al. [1] found that many FIA plots had demographic structure consistent with a high-severity fire in the 200 years prior to fire suppression and the number of these plots was likely a large underestimate due to the improperly narrow definition of high-severity fire used by Stevens et al., and a major calculation error in their methods.

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Author Contributions

Analyzed the data: DCO CTH. Wrote the paper: DCO CTH WLB DAD MAW.

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Effects of Fire and Commercial Thinning on Future Habitat of the Northern Spotted Owl

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Abstract: The Northern Spotted Owl (*Strix occidentalis caurina*) is an emblematic, threatened raptor associated with dense, late-successional forests in the Pacific Northwest, USA. Concerns over high-severity fire and reduced timber harvesting have led to programs to commercially thin forests, and this may occur within habitat designated as "critical" for spotted owls. However, thinning is only allowed under the U.S. Government spotted owl guidelines if the long-term benefits clearly outweigh adverse impacts. This possibility remains uncertain. Adverse impacts from commercial thinning may be caused by removal of key habitat elements and creation of forests that are more open than those likely to be occupied by spotted owls. Benefits of thinning may accrue through reduction in high-severity fire, yet whether the fire-reduction benefits accrue faster than the adverse impacts of reduced late-successional habitat from thinning remains an untested hypothesis. We found that rotations of severe fire (the time required for high-severity fire to burn an area equal to the area of interest once) in spotted owl habitat since 1996, the earliest date we could use, were 362 and 913 years for the two regions of interest: the Klamath and dry Cascades. Using empirical data, we calculated the future amount of spotted owl habitat that may be maintained with these rates of high-severity fire and ongoing forest regrowth rates with and without commercial thinning. Over 40 years, habitat loss would be far greater than with no thinning because, under a "best case" scenario, thinning reduced 3.4 and 6.0 times more dense, late-successional forest than it prevented from burning in high-severity fire in the Klamath and dry Cascades, respectively. Even if rates of fire increase substantially, the requirement that the long-term benefits of commercial thinning clearly outweigh adverse impacts is not attainable with commercial thinning in spotted owl habitat. It is also becoming increasingly recognized that exclusion of high-severity fire may not benefit spotted owls in areas where owls evolved with reoccurring fires in the landscape.

Keywords: Fire rotation, forest regrowth rate, forest thinning, future habitat, habitat loss, late-successional forest, policy implications, severe fire, spotted owl.

INTRODUCTION

Conservation of the emblematic Northern Spotted Owl (*Strix occidentalis* ssp. *caurina*) in the Pacific Northwest of North America has become a global example of balancing conflicting land management goals (DellaSala and Williams 2006). Concern over degradation of the owl's dense, late-successional forest habitat led to the 1994 Northwest Forest Plan (NWFP). The NWFP shifted management on ~100,000 km^2 of federal USA forestlands from an emphasis on resource extraction to embrace ecosystem management and biodiversity conservation goals. Under the NWFP, ~30% of federal lands traditionally managed for timber production were placed in late-successional reserves that emphasized conservation goals and limited timber harvesting (USFS/USDI 1994).

Over the last decade, managers and policy makers have become increasingly concerned about high-severity fire and reduced timber harvesting in NWFP dry forests (e.g., Spies *et al.* 2006, Power 2006, Thomas *et al.* 2006, Ager *et al.* 2007, USFWS 2011). Forest thinning has been viewed as a solution for controlling fires in dry forests throughout western North America (Agee and Skinner 2005, Stephens and Ruth 2005) and commercial criteria have been included to pursue timber harvest goals (Johnson and Franklin 2009, Franklin and Johnson 2012). Commercial thinning prescriptions currently being implemented under these

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criteria may remove up to one-half of forest basal area, and may also include patch cutting or small clear cuts (USDI 2011). Commercial thinning is now proceeding rapidly without a full understanding of the long-term risks.

For spotted owls, thinning and associated activities often remove or reduce key habitat features in direct proportion to the intensity of the commercial prescription. Key spotted owl habitat features that may be reduced or removed directly or indirectly include high tree density and canopy cover (King 1993, Pidgeon 1995), recently killed pines (*Pinus* spp.) and abundant snags (Pidgeon 1995), multiple tree layers, with abundant medium and small white fir (*Abies concolor*) or Douglas-fir (*Pseudotsuga menziesii*) (King 1993, Pidgeon 1995, Everett *et al.* 1997, Irwin *et al.* 2012), large volume of mature-sized down logs (Pidgeon 1995), shrubs (King 1993, Pidgeon 1995, Irwin *et al.* 2012) and trees with heavy mistletoe infections (Hessburg *et al.* 2008), which are essential for spotted owl nesting (USFWS 2011). Thinning or contemporary harvest near the nest or activity center has been shown to displace Northern Spotted Owls (Forsman *et al.* 1984, King 1993, Hicks *et al.* 1999, Meiman *et al.* 2003). Telemetry studies on California Spotted Owls (*Strix occidentalis* ssp. *occidentalis*) in the Sierra Nevada found that owls avoided Defensible Fuel Profile Zones (an intensive thinning treatment) (USFS 2010). Unoccupied California Spotted Owl territories had a lower probability of re-occupancy after timber harvest, even when habitat alterations comprised <5% of a territory (Seamans and Gutiérrez 2007). In addition, Barred Owls (*S. varia*), which out-compete spotted owls (Dugger *et al.* 2011), use younger and more open forests compared to Northern Spotted Owls (Wiens 2012).

Studies also have found negative impacts of thinning to northern flying squirrels (*Glaucomys sabrinus*), the primary prey of Northern Spotted Owls in most of its range (Waters and Zabel 1995, Waters *et al.* 2000, Carey 2001, Ransome and Sullivan 2002, Gomez *et al.* 2005, Ransome *et al.* 2004, Bull *et al.* 2004, Meyer *et al.* 2007, Wilson 2008, Holloway and Smith 2011, Manning *et al.* 2012). Negative effects may persist for 15 years or longer (Wilson 2008). In addition, openings between trees from thinning may create barriers, due to predator avoidance, for flying squirrels to cross using its gliding locomotion (Manning *et al.* 2012). Thinning has also been found to have negative effects on the abundance of other main prey species for Northern Spotted Owls such as red-backed voles (*Myodes californicus*) (Suzuki and Hayes 2003) and woodrats (*Neotoma cinerea, N. fuscipes*) (Lehmkuhl *et al.* 2006).

Because of the many conflicts between thinning and spotted owl conservation, some authors have recommended that treatments aimed at controlling fire avoid spotted owl habitat and instead treat vegetation elsewhere that is the most flammable and strategic for accomplishing fuel treatment goals (Gaines *et al.* 2010). The 2011 Recovery Plan for the Northern Spotted Owl, the blueprint for management of this species on federal lands in the region (USFWS 2011), contains the proviso that long-term benefits to spotted owls of forest thinning treatments must clearly outweigh adverse impacts (USFWS 2011). The U.S. Fish and Wildlife agency that developed the plan suggested that benefits over time might accrue from a net increase in habitat because fire disturbances would be reduced (USFWS 2011). But whether the benefits would outweigh the impacts remains uncertain due to limitations of previous assessments.

Previous assessments of the efficacy of thinning treatments in reducing fire disturbances in spotted owl habitat (Wilson and Baker 1998, Lee and Irwin 2005, Roloff *et al.* 2005, 2012, Calkin *et al.* 2005, Hummel and Calkin 2005, Ager *et al.* 2007, Lehmkuhl *et al.* 2007) have not incorporated the probability of high-severity fires occurring during the treatment lifespan. The effect of this is to overestimate treatment efficacy in potentially controlling fire or fire behavior (Rhodes and Baker 2008). Nor have the effects of recruitment of dense, late-successional forest that act to offset loss from fire been included in prior assessments. In addition, impacts of the kind of commercial thinning treatments being implemented to address dry forest concerns have not been fully considered for the owl or its prey (e.g., Ager *et al.* 2007, Lehmkuhl *et al.* 2007, Roloff *et al.* 2012). Current commercial thinning prescriptions being implemented in dry forests specifically identify desired future conditions to be maintained (e.g. Johnson and Franklin 2009) that have basal area and other structural targets mostly well below the minimum levels that have been found in spotted owl nesting, roosting and foraging habitat (NRF) in dry forests. For example, basal area targets in a project in southwest Oregon designed to demonstrate the thinning prescriptions in dry forest spotted owl habitat were 13.75-27.5 m^2/ha (USDI 2011), while stands < 23 m^2/ha very rarely support spotted owl nesting territories (Buchanan and Irwin 1995). In addition, the Recovery Plan (USFWS 2011) permits thinning in core areas, but emphasizes treating areas outside of core areas, so there is a need for assessment of impacts outside core areas as well. Areas outside cores may be essential for foraging and be part of the breeding season home range. Furthermore, owls often move outside core areas (USFWS 2011). Lastly, available habitat outside existing cores may become important to owl recovery, particularly if spotted owls are displaced from higher quality habitat by Barred Owls (Dugger *et al.* 2011).

To assess whether benefits of commercial thinning outweigh adverse impacts to spotted owls in dry forests (USFWS 2011), quantitative assessments are needed that allow for direct assessment of the amounts of any dense, mature or late-successional habitat that would be reduced by both commercial prescriptions and severe fire. Accordingly, we calculated these amounts by projecting them over 40 years and incorporated into our calculations the effects of forest regrowth. For our calculations, we used empirical data on fire and forest regrowth from the potential habitat within the two dry forest regions where spotted owls occur, the Klamath and dry Cascades of California, Oregon, and Washington, that are subject to thinning. We analyzed each region separately using region-wide data. Conservation planning for spotted owls commonly occurs at the scale of these regions. For our thinning treatment, we chose a "best" scenario for minimizing the amount of dense, late-successional forest to be treated (Lehmkuhl *et al.* 2007); while we used an optimistic scenario for treatment efficacy, assuming that a 50% reduction in high-severity fire would occur (Ager *et al.* 2007). We also illustrate the effects of varying treatment amount and efficacy. To calculate rotations of severe fire in the forests of the study area, we used available fire data from a time period, 1996-2011, which includes exceptionally large, rare fire events. Our approach may be useful to managers interested in maintaining habitat for other species that rely on dense forests in fire-prone regions (Odion and Hanson 2013).

METHODS

Study Area

We analyzed fire and forest recruitment trends in 19,000 km^2 of dry forests in the Klamath and 18,400 km^2 in the Cascades provinces. As in Hanson *et al.* (2009), we analyzed only late-successional, or "older" forests present in 1995, as mapped by Moeur *et al.* (2005). This is a small fraction of the dry forest regions. Our analysis was further restricted to federal lands. Mapping by Moeur *et al.* (2005) corresponds to mid-montane forest zones where Northern Spotted Owls occur. These montane forest zones include forests dominated mainly by true firs (*A. grandis, A. concolor*), Douglas-fir (*Pseudotsuga menziesii*), and Ponderosa pine (*P. ponderosa*): Other conifers found in the central and northern Cascades in dry forests frequented by spotted owls are western hemlock (*Tsuga heterophylla*), western larch (*Larix occidentalis*), and limited amounts of western red cedar (*Thuja plicata*) and Engelmann spruce (*Picea engelmannii*). Forests in the Klamath are noted for high conifer diversity, with species such as incense cedar (*Calocedrus decurrens*) commonly found in the range of spotted owls. A variety of broad-leaved evergreen trees, such as madrone (*Arbutus menziesii*) and tanoak (*Lithocarpus densiflorus*) are also characteristic of these forests (Whittaker 1960).

Quantifying Future Habitat

We determined existing rates of dry-forest redevelopment following stand initiation in the forests of the study regions as delineated by Mouer *et al.* (2005) using the extensive U.S. Forest Service Forest Inventory and Analysis (FIA) forest monitoring data (http://www.fia.fs.fed.us/tools-data/). FIA is a monitoring system based on one permanent, random plot per ~2400 ha across forested lands. We excluded plots from forests not used by spotted owls (e.g. lodgepole pine, oak forest) and from non-conifer vegetation and non-federal lands. Most of these plots were already excluded by the mapping by Mouer *et al.* (2005) that delineated the study area.

An FIA plot consists of a 1-ha area. For tree measurements, this area is sub-sampled with four circular subplots that are 0.1 ha for large-tree sampling and 0.017 ha for smaller-tree sampling (defined by region). The diameter-at breast-height (dbh) and crown position of each tree and the ring count from two cores from dominant/codominant trees are measured in each subplot (USFS 2010). Stand age for an FIA plot is determined from the average of all ring counts from sub-plot samples, weighted by cover of sampled trees, and 8 years are added for estimated time to grow to breast height (1.4 m). We used live-tree dbh data to prepare regressions with stand age.

FIA data were available from 2001-2009, comprising 90% of the plots available within our study area. A total of 581 plots from the Klamath and 441 from the dry Cascades were considered, representing 13,944 and 10,680 km^2 in each region, respectively. The number would be higher, but we eliminated 139 plots in the Klamath and 141 in the Cascades that had different stand-initiation dates from different subplots of the main FIA plot. This situation occurs throughout the study area due to the patchy nature of mixed-severity fire. Including all the subplots as individual plots creates a larger sample size, but we chose not to do this because some individual locations would be overrepresented. Most importantly, both approaches lead to the same results.

We analyzed fire severity from 1996-2011 in late-successional, or "older" forests mapped by Moeur *et al.* (2005). For 1996-2008, we used the Monitoring Trends in Burn Severity (MTBS) (http://www.mtbs.gov/) data. We used the ordinal classification from MTBS, as MTBS analysts determine for each fire where significant thresholds exist in digital prefire and postfire images, supplemented with plot data and analyst experience with fire effects. In plot data, a composite burn index that sums mortality by vegetation stratum is used to identify high fire severity (see http://www.mtbs.gov/). For 2009-2011, we obtained U.S. Forest Service digital data (http://www.fs.fed.us/postfire-vegcondition) and classified these data following Miller and Thode (2007). We could not use pre-1996 MTBS fire severity data because the pre-burn map of spotted owl forest habitat is from 1995 (Moeur *et al.* 2005). From severity data we calculated high-severity fire rotation (FR^{hs}), the expected time to severely burn an area equivalent to the area of interest once, or the landscape mean interval for severe fire (Baker 2009).

We calculated annual high-severity fire and forest regrowth rates to future proportions for early-, mid- and mature or late-successional forests, denoted herein by "E," "M," and "L," respectively, using annual time steps. We defined late-successional forests by selecting a value, 27.5 m^2/ha. This amount corresponds with the maximum basal area that would be left according to currently implemented thinning prescriptions (USDI 2011). This is somewhat higher than the minimum basal area where spotted owls have been found to nest in dry forests. For example, the mean value minus one standard deviation in all the dry forest stands studied by Buchanan *et al.* (1995) was 23 m^2/ha. However, we did not want to identify the rate of regrowth to the very minimum basal area that constitutes habitat, but regrowth to a basal area more likely to function as habitat. Mid- and early-successional forests were defined as 13.5-27.5 and <13.5 m^2/ha tree basal area, respectively. We separated mid-successional from early-successional forest because, mid-successional forests may be included in thinning treatments, but early-successional forests may not. Thinned forest ("T") was our fourth vegetation state. The forest states are diagramed in Fig. (**1**). The proportion of each state in the landscape at time *t,* defined a vector (P_t^E, P_t^M, P_t^T, P_t^L). Transition probabilities ϕ_t^{rs} equaled the probability that any portion of state *r* at time *t* transitions to

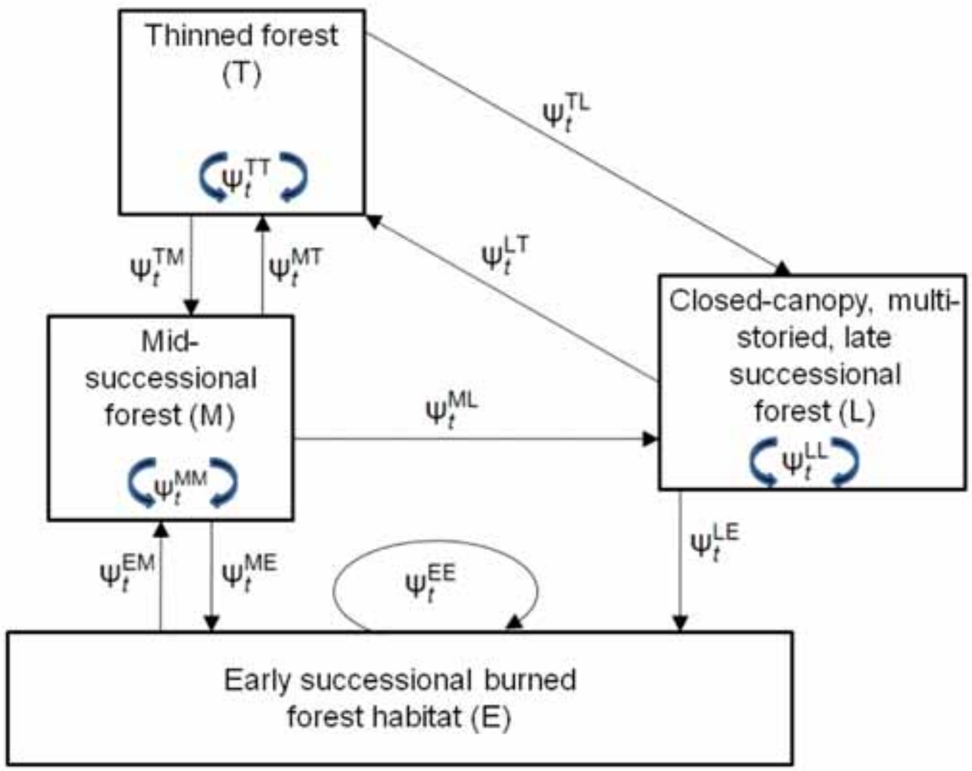

Fig. (1). State (boxes) and transition (arrows) model for dry Pacific Northwest Forest vegetation with fire disturbances and thinning. Variables are the transition rates between states indicated by the associated arrow.

state *s* at time $t + 1$, allowing calculation of future amounts of each forest type using the following equation:

$$\begin{bmatrix} \phi_t^{EE} & \phi_t^{ME} & \phi_t^{TE} & \phi_t^{LE} \\ \phi_t^{EM} & \phi_t^{MM} & \phi_t^{TM} & \phi_t^{LM} \\ \phi_t^{ET} & \phi_t^{MT} & \phi_t^{TT} & \phi_t^{LT} \\ \phi_t^{EL} & \phi_t^{ML} & \phi_t^{TL} & \phi_t^{LL} \end{bmatrix} \begin{bmatrix} \mathrm{P}_t^{E} \\ \mathrm{P}_t^{M} \\ \mathrm{P}_t^{T} \\ \mathrm{P}_t^{L} \end{bmatrix} = \begin{bmatrix} \mathrm{P}_{t+1}^{E} \\ \mathrm{P}_{t+1}^{M} \\ \mathrm{P}_{t+1}^{T} \\ \mathrm{P}_{t+1}^{L} \end{bmatrix} \qquad (1)$$

The initial proportions, $\mathrm{P}_{t=0}^{\mathrm{E\text{-}L}}$ of the three natural-forest states were from the FIA basal-area analyses, with thinned forests considered zero for simplicity and because of lack of data. The annual transition from mid- and late- to early-successional forest from high-severity fire (ϕ_t^{LE} , ϕ_t^{ME}) was $1/\mathrm{FR}^{\mathrm{hs}}$. Early-successional forests also burned at this rate (ϕ_t^{EE}). Annual rates of forest redevelopment were from the inverse of the growth period ($1/\mathrm{G}^{\mathrm{EM}}$) to reach 13.5 m^2/ha live-tree basal area, or to grow from 13.5 to 27.5 m^2/ha live-tree basal area ($1/\mathrm{G}^{\mathrm{ML}}$), calculated from the regression of live basal area on age (see results). Lower-severity fire can reduce basal area from >27.5 m^2/ha basal area to <27.5 m^2/ha. However, this transition is already considered in the regrowth rate, which also incorporates the effects of lower-severity fires that have occurred on rates of forest redevelopment. Because natural disturbances that may temporarily lower basal area are captured in the transitions from early- to late-successional forest, the transition from late to mid-successional forest was set to zero. Transition rates to thinned forest were based on treatment within 20 years, beginning in year $t + 1$, of the mid- and late-successional forests present at $t = 0$ (see Table **1** for annual rate). Based upon the empirical FIA and MTBS data described above, we used these transitions (Table **1**) and Eq. 1 to project forward 40 years (see sample calculation in the Supplementary Materials). We chose this time interval because it represents one cycle of thinning and forest recovery.

Next, we calculated the effects of varying levels of thinning, and treatment efficacy (in terms of the effect on high-severity fire rotation intervals), over the study period. According to an analysis of a spotted owl landscape by Lehmkuhl *et al.* (2007), a "best" scenario for minimizing the short-term adverse impacts of thinning while reducing fire frequency and severity was one that treated only 22% of the landscape, and limited thinning in nesting, roosting, and foraging habitat to 21% of the area of this habitat. We used this prescription in our calculations to illustrate the effects under a best-case scenario. In our calculations, the amount of mid-successional forest thinning differed between the two regions because amounts of both mid- and late-successional forests were not the same. We also considered the effects of treating from 0 to 45% of forests, holding constant the proportions of treatments that were in late-successional vs. mid-successional forests.

We assumed that there would be no high-severity fire in treated forests over the treatment lifespan. We additionally assumed that thinning 22% of the landscape would lower the amount of high-severity fire in the unthinned landscape by half. This is based on the findings of Ager *et al.* (2007) who simulated the effects of wildfire ignitions following strategic

Table 1. Annual transition probabilities used in transition matrices for each scenario analyzed for dry provinces within the range of the Northern Spotted Owl. FR^{hs} is the high-severity fire rotation. G is the time required for stands to grow from early to mid- (EM) or mid- to late-successional (ML) forest (see Table 2). K = Klamath, C = Cascades. R is the amount that high severity fire is reduced by thinning (50% reduction at 22 percent of late-successional forest thinned).

Transition Probabilities	No Treat	Treat 22% Maintain	Treat 22% Recover
ϕ_t^{LE}	$1/FR^{hs}$	$(1/FR^{hs}$-R)	$(1/FR^{hs}$-R)
ϕ_t^{EM}	$1/G^{EM}$	$1/G^{EM}$	$1/G^{EM}$
ϕ_t^{ET}	0	0	0
ϕ_t^{EL}	0	0	0
ϕ_t^{ME}	$2/FR^{hs}$	$2/FR^{hs}$	$2/FR^{hs}$
ϕ_t^{ML}	$1/G^{ML}$	$1/G^{ML}$	$1/G^{ML}$
ϕ_t^{EE}	$1-1/G^{EM}$	$1-1/G^{EM}$	$1-1/G^{EM}$
ϕ_t^{MM}	$1-1/G^{ML}-(1/FR^{hs})$	$1-1/G^{ML}-(1/FR^{hs}-R) - \phi_t^{MT*}$	$1-1/G^{ML}-(1/FR^{hs}-R) - \phi_t^{MT*}$
ϕ_t^{MT*}	0	K = 0.033 C = 0.018	K = 0.033 C = 0.018
$\phi_t^{TM\dagger}$	0	0	K = 0.033 C = 0.018
ϕ_t^{TE}	0	0	0
$\phi_t^{TT\dagger}$	0	0	$1-\phi_t^{TL} - \phi_t^{TM\dagger}$
$\phi_t^{TL\dagger}$	0	0	K = 0.0114 C = 0.0105
ϕ_t^{LM}	0	0	0
ϕ_t^{LT*}	0	K = 0.0114 C = 0.0105	K = 0.0114 C = 0.0105
ϕ_t^{LL}	$1 - 1/FR^{hs}$	$1 - 1/FR^{hs}$ -R- ϕ_t^{LT}	$1 - 1/FR^{hs}$ -R- ϕ_t^{LT}

*Only in effect for the first 20 years.
†Does not take effect until after 20 years.

thinning treatments in a spotted owl landscape. When <22% of the landscape was affected at any given time (such as any time prior to year 20 when the full treatment would be incomplete, or after one-time treatments began to recover, or for scenarios with <22% of the landscape treated) the same ratio of area treated to reduction in high-severity fire (22% treat: 50% reduction in fire) was used to reduce the area burned at high severity (see Supplementary Material for an illustration). Thus, the amount that fire was reduced by thinning increased with each year as a function of the total area thinned (all other variables were constant). Ager *et al.* (2007) found little additional effect of treatments in reducing wildfires as treatment level increased beyond 20%, so we did not calculate greater reductions in fire as treatment levels went from 22-45%. However, we additionally calculated future habitat amounts as a function of fire rotation to evaluate the effects of varying treatment efficacy, in which case we did calculate the reduced amount of habitat burned severely. This amount is the dependent variable in our summary figures. Treatment lifespan was assumed to be 20 years (Rhodes and Baker 2008) for "one-time thinning," or maintained in perpetuity over the 40 years for "maintained." A sample calculation using the model (equation 1) is presented in the Supplementary Material.

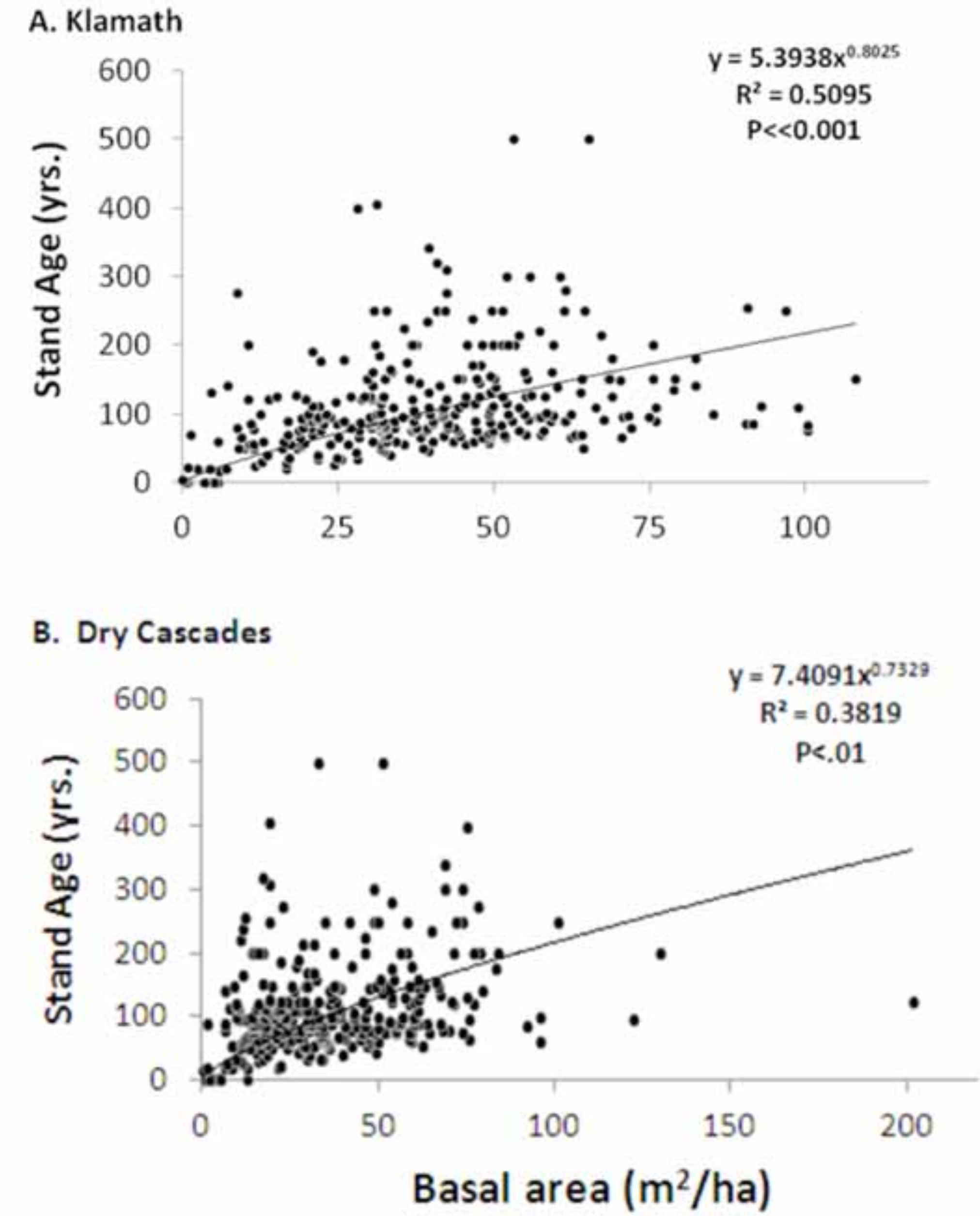

Fig. (2a-b). Scatterplots of live-tree basal area per hectare and stand age from US Forest Service FIA data for the A. Klamath region and B. dry Cascades region.

The only owl habitat we considered for impacts from thinning was suitable nesting, roosting, and foraging (so called NRF habitat). Because treatments aimed at demonstrating the type of thinning to be implemented in spotted owl habitat reduce basal area down to 13.75-27.5 m^2/ha, mostly well-below the minimum amounts for NRF habitat (Pidgeon 1995, Buchanan and Irwin 1998, LeHaye and Gutiérrez 1999), and because treated forests also have reduced amounts of key habitat features like multi-canopy structure, down wood, small firs and mistletoe infections, the area affected by these treatments will largely correspond to the amount of habitat lost. Thinning may also render adjacent, unthinned forest unsuitable or less suitable (Seamans and Gutiérrez 2007), but we did not account for this effect. The lifespan for thinning treatments that we used was 20 years for one-time thinning (Rhodes and Baker 2008), and 40 years for maintained treatments. Transition from late- to early-successional vegetation due to high-severity fire also was considered habitat loss. This may overestimate the impacts of fire on Northern Spotted Owl foraging habitat (Bond *et al.* 2009, USFWS 2011), but the assumption is largely irrelevant due to the low rates of high-severity fire in both study regions in relation to forest regrowth, as described next.

RESULTS

We found a highly significant relationship between live-tree basal area and stand age in both regions (Figs. **2a-b**, Klamath n = 442, dry Cascades n = 304). Much of the variance in the plot data was caused by a modest number of relatively old stands that had much lower basal area for their

Table 2. Forest Inventory and Analysis (FIA) plot parameters for the Klamath and dry Cascades provinces, California, Oregon, and Washington, based on most recent survey data from 2001-2009. Also shown are the amounts of time after fire that is takes forest to regrow to the specified live basal area (BA) thresholds using the regression equations shown in Figs. (2a-b).
[a]These plots have 2 or more stand ages associated with them due to different disturbance histories within the main FIA plot.

Entity	Klamath	Dry Cascades
Number of plots (total)	581	445
Number of plots excluded from analysis†	139	141
Initial (P^{E}_{t+0}) early-successional forest (%)	9	14.5
Initial (P^{M}_{t+0}) mid-successional forest (%)	14.4	26.9
Initial (P^{L}_{t+0}) late-successional forest (%)	76.6	55.6
Regrowth period, 0-13.5 m^2/ha live BA (yrs)	44	53
Regrowth period, 13.5-27.5 m^2/ha live BA (yrs)	32	36
Regrowth period, 0-27.5 m^2/ha live BA (yrs)	76	89
High-severity fire rotation	362	913

†These plots have 2 or more stand ages associated with them due to different-aged sub-plots within the main FIA plot.

age than did other plots. The amount of time following disturbance needed for regenerating forests to reach live-tree basal area >27.5 m^2/ha was 77 and 90 years, respectively, for the Klamath and dry Cascades (Table **2**).

Using the MTBS data, the rotation for high-severity fire from 1996-2011 was 362 to 913 years in the Klamath and dry Cascades, respectively (Table **2**). At these rates, a total of 1,221 and 325 km^2 of high-severity fire would occur in Klamath and dry Cascades late-successional forests, respectively, in 40 years. With annual regrowth rates of late-successional forests that were 4.5 to >10 times greater than the rates of fire disturbances (i.e. (1/77)/(1/362) for the Klamath and (1/89)/(1/913) for the dry Cascades, and no disturbances other than fire, late-successional forests would eventually come to occupy 83% of the potential forested area in the Klamath and 91% in the Cascades. Thus, over 40 years, late-successional forests in the Klamath increased slightly over their current amount of 77% of the forested landscape FIA plots to 81% or from about 10,668 km^2 to 11,335 km^2 (Fig. **3a**). In the dry Cascades, where late-successional forests were 59% of the forested landscape FIA plots, they increased relatively rapidly to 77% of the forested landscape, or from 6,253 km^2 to 8,234 km^2 in 40 years (Fig. **4a**).

Simulated thinning of 21% of dense, late-successional forest of the Klamath landscape meant that a total of 2,225 km^2 would be reduced, while treatments in mid-successional forests would cover 840 km^2 to reach a treatment level of 22% of the whole landscape. After the one-time thinning, late-successional forests returned to slightly lower amounts than occurred without thinning after 40 years (Fig. **3a**). The net effect of the one-time thinning was to reduce late-successional habitat by 10.7% over the 40-year period, or from an average of 11,086 km^2 to 9,996 km^2 over 40 years (i.e., 1,090 km^2 less each year on average, Fig **3b**). The amount of dense, late-successional forest that was prevented from burning at high severity was 16 km^2/year, resulting in 320 km^2 of dense, late-successional forest, which would otherwise have been transformed into early-successional forest, in each year on average over the 40-year period. Therefore, in this scenario, thinning reduced 3.4 times more late-successional forest than it increased. The maintained treatment reduced habitat by 15.3%, from 11,086 km^2 on average over 40 years to 9,396 km^2 (i.e., 1,690 km^2 less each year on average, Fig. **3c**). In both cases, 13% of the habitat loss was from thinning in mid-successional forest that prevented or slowed these forests from developing into dense, late-successional forest. The amount of dense, late-successional forest that was prevented from burning at high severity was 20 km^2/year, resulting in 400 km^2 of dense, late-successional forest, which would otherwise have been transformed into early-successional forest, in each year on average over the 40-year period. Therefore, the combination of thinning and maintenance reduced 4.2 times more late-successional forest than it increased.

In the Cascades, to treat 22% of the landscape, the thinning scenario targeted 1,313 km^2 of dense, late-successional forest, and 1,036 km^2 of mid-successional forest. After the one-time thinning, late-successional forests again returned to slightly lower amounts than occurred without thinning after 40 years (Fig. **4a**). The net effect of the one-time thinning treatment over 40 years was to reduce dense, late-successional forest by an average level of 11.1% (836 km^2 less each year on average, Fig. **4b**). The amount of dense, late-successional forest that was prevented from burning at high severity from the one time treatment was 3.5 km^2/year, resulting in 140 km^2 of dense, late-successional forest, which would otherwise have been transformed into

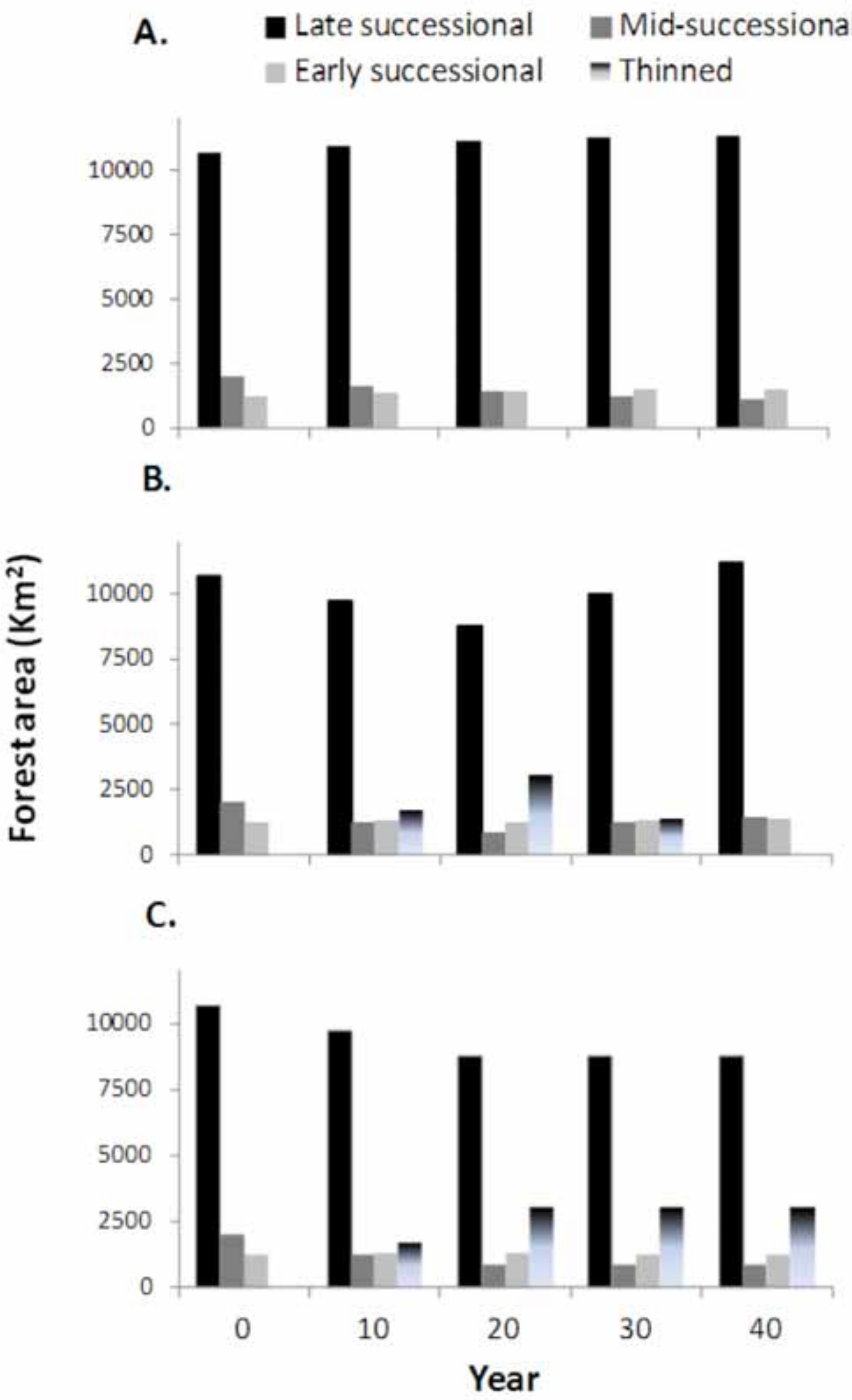

Fig. (3a-c). Amounts of the four forest types (early-, mid-, late-successional, and thinned) in the landscape over a 40-year period based on the states shown in (Fig. **1**) and transition rates (Table **2**) for the Klamath province, California, and Oregon, and the following scenarios: A) no treatment; B) one-time treatment of 21% of late-successional forests (>27.5 m^2/ha live-tree basal area) and 42% of mid-successional forests (= total of 22% of landscape treated) followed by recovery in 20 years to late-successional forest; C) treatment of 21% of late-successional forests (>27.5 m^2/ha live-tree basal area) and 42% of mid-successional (= total of 22% of landscape treated) forests with future maintenance. We converted proportions of forest types from modeling output to km^2 using the area estimate from FIA for the Klamath study region.

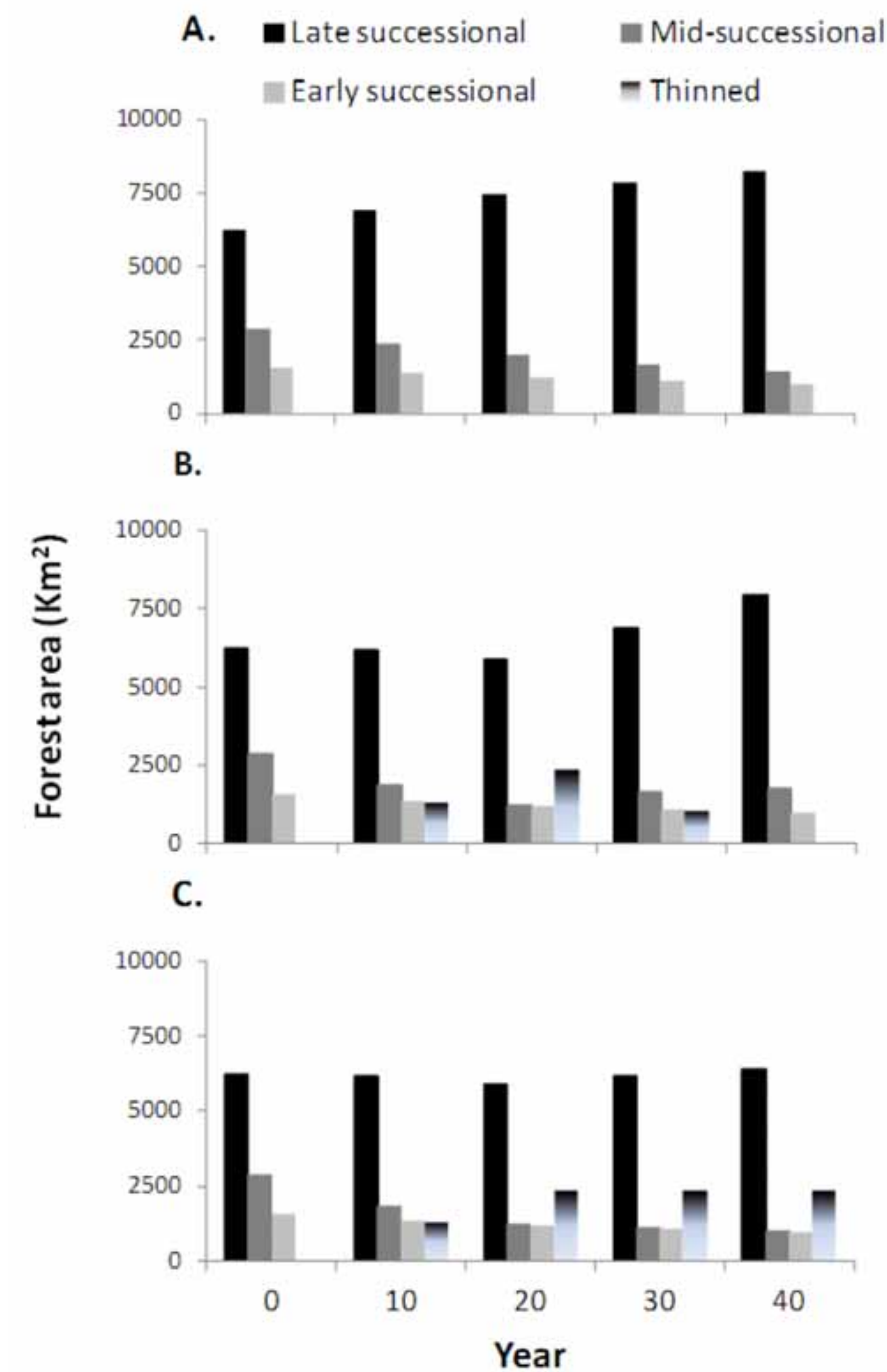

Fig. (4a-c). Amounts of the four forest types (early-, mid-, late-successional, and thinned) in the landscape over a 40-year period based on the states in (Fig. **1**) and transition rates (Table **2**) for the dry Cascades province, California, Oregon, and Washington and the following scenarios: **A**) no treatment; **B**) one time treatment of 21% of late-successional forests (>27.5 m^2/ha live tree basal area) and 36% of mid-successional forests (=22% of landscape treated) followed by recovery in 20 years to late-successional forest; **C**) treatment of 21% of late-successional forests (>27.5 m^2/ha live tree basal area) and 36% of mid-successional forests (=22% of landscape treated) in perpetuity. We converted proportions of forest types from modeling output to km^2 using the area estimate from FIA for the dry Cascades study region.

early-successional forest, in each year on average over the 40-year period. Therefore, thinning reduced 6.0 times more late-successional forest than it increased. The maintained treatment reduced dense, late-successional forest by an average of 16.4% (1,212 km^2less each year on average, Fig. **4c**). Of this reduction, 30% was from the indirect effect of thinning in mid-successional forests, more of which were treated in the Cascades scenario. The amount of dense, late-

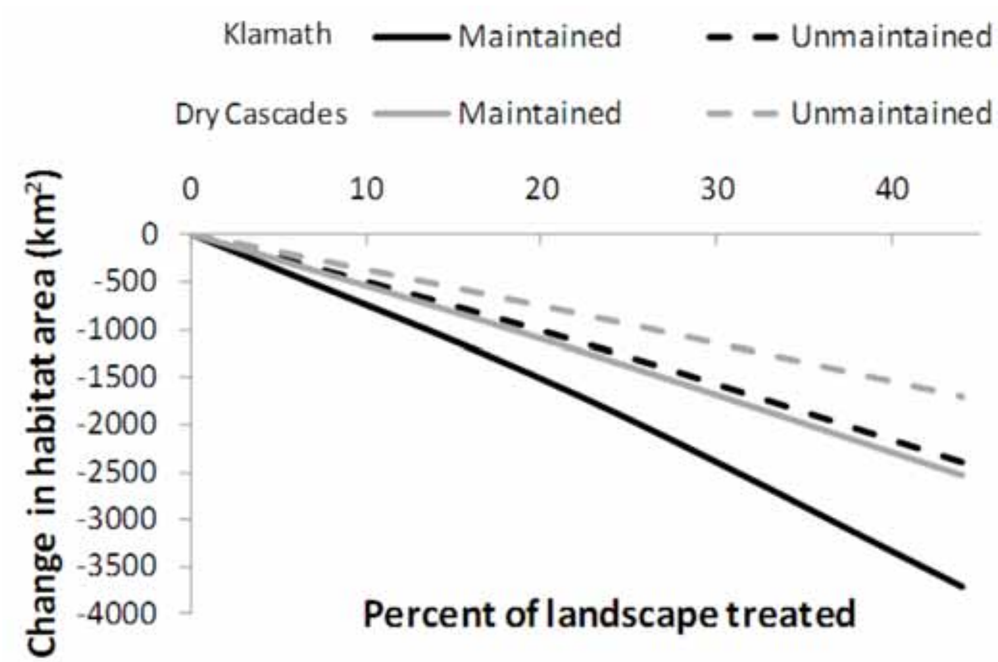

Fig. (5). Net amount of habitat lost over 40 years compared to the no-treatment scenario as a function of treatment of 0-45% of the landscape. The amount of late-successional forest treated was held constant at 21% of the area of this forest, except at very low levels of treatment. The amount of mid-successional forest treated varied from zero at very low treatment levels, to a large proportion of the mid-successional forests when 45% of the landscape was treated, particularly in the Klamath region.

successional forest that was prevented from burning at high severity from the maintained treatment scenario was 4.5 km^2/year, resulting in 180 km^2 of dense, late-successional forest, which would otherwise have been transformed into early-successional forest, in each year on average over the 40-year period. Therefore, the combination of thinning and maintenance reduced 6.7 times more late-successional forest than it increased.

As treatment level increased from 11 to 22%, habitat loss doubled (Fig. **5**). With 22% of the landscape treated, the effect of reducing fire by 50% in the rest of the landscape was reached, and there was no further reduction in fire with increasing treatment amount. With less fire prevented per km^2 treated, the rate of habitat loss increased as treatment went from 22 to 45% of the landscape.

We also assessed the effect of holding treatment level constant and varying the efficacy of treatments. Even if treatment efficacy was considerably greater than we assumed and rotations of high-severity fire substantially longer than twice their current length, the amount of dense, late-successional forest habitat that would be reduced due to thinning would only be slightly lower (Figs. **6a-b**). With complete elimination of fire over 40 years as a result of treatments, the amount of dense, late-successional forest would be 9-10% less than with no treatment. This becomes a large amount of habitat loss over time.

DISCUSSION

We found that the habitat recruitment rate exceeded the rate of severe fire by a factor of 4.5 in the Klamath and 10 in the dry Cascades, leading to a deterministic increase in dense forest habitat over time, assuming no other disturbance events. In contrast, previous published assessments of fire on spotted owls have not explicitly considered fire and forest regrowth rates (Wilson and Baker 1998, Lee and Irwin 2005, Roloff *et al.* 2005, 2012, Calkin *et al.* 2005, Hummel and Calkin 2005, Ager *et al.* 2007, Lehmkuhl *et al.* 2007). Not including the probability of high-severity fire, which is low, leads to highly inflated projections of the effects of thinning versus not thinning on high-severity fire (Rhodes and Baker 2008, Campbell *et al.* 2012).

Our calculations of thinning effects included rates of forest regrowth along with high-severity fire. The calculations illustrate how the requirement that the long-term benefits of thinning clearly outweigh adverse impacts (USFWS 2011) is not attainable as long as treatments have adverse impacts on spotted owl habitat. This is because the amount of dense, late-successional forest that might be prevented from burning severely would be a fraction of the area that would be thinned. Under our "best case" scenario, thinning reduced dense, late-successional forest by 3.4 and 6.0 times more than it prevented such forest from experiencing high-severity fire in the Klamath and dry Cascades, respectively, similar to findings in a recent unpublished report by U.S. Forest Service scientists from the Pacific Northwest Research Station (Raphael *et al.* 2013). This would not be a concern if thinning effects were neutral, but the commercial thinning prescriptions being implemented call for forests with basal area reduced by nearly half to 13.5-27.5 m^2/ha, which is mostly well below the minimum level known to function as nesting and roosting habitat (ca. 23 m^2/ha) (Buchanan *et al.* 1995, 1998). Thus, if dense forests are subjected to these treatments, much of the impacted area would no longer have minimum basal area needed to function as nesting and roosting habitat. Even an immediate doubling of fire rates due to climate change or

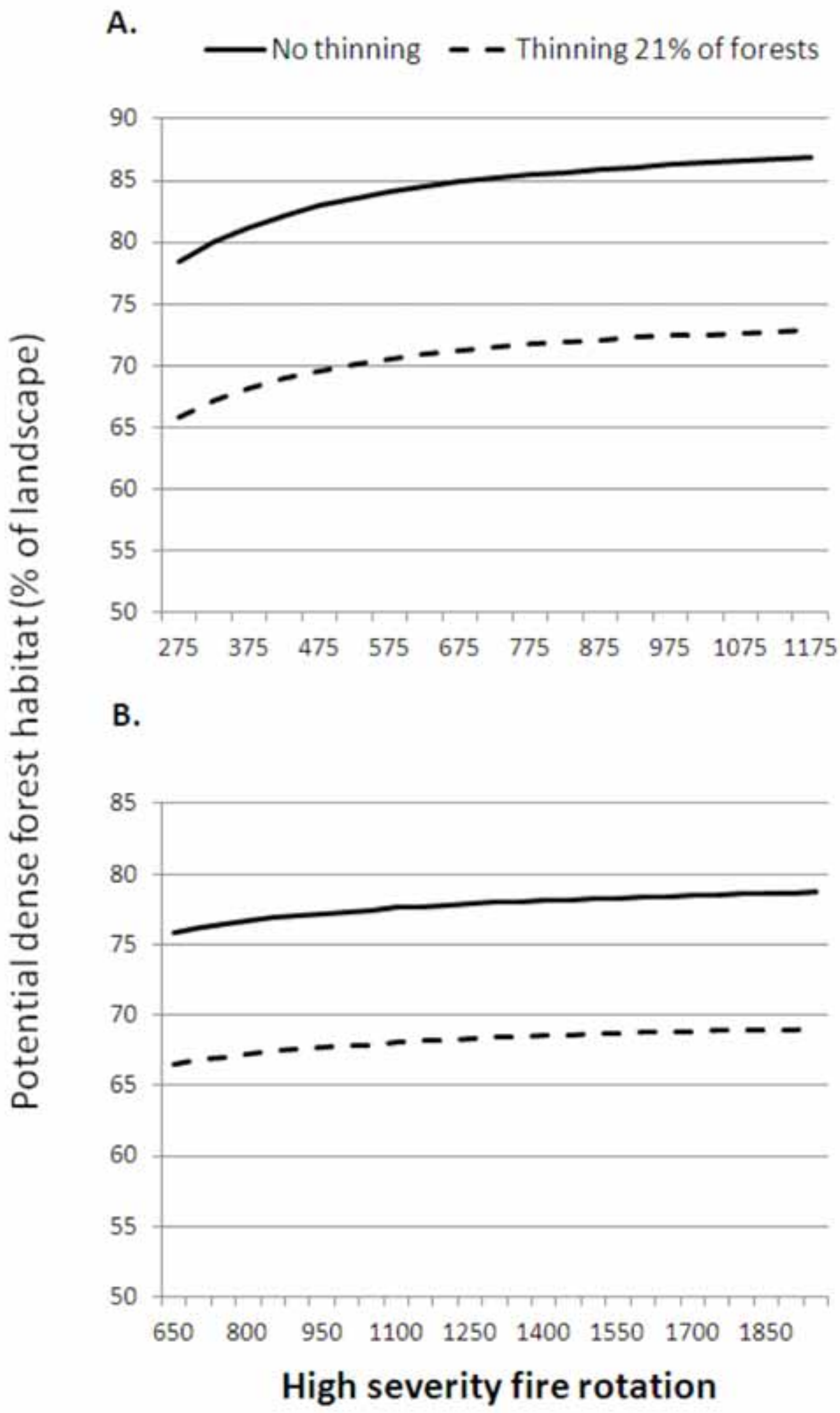

Fig. (6a-b). Amount of forest habitat in the range of the Northern Spotted Owl in the A. Klamath, and B. dry Cascades 40 years in the future as a function of the average high severity rotation over that time period, and longer rotations.

other factors would result in far less habitat affected by high-severity fire than thinning. In addition, much of the high-severity fire might occur regardless of thinning, especially if the efficacy of thinning in reducing high-severity fire is reduced as fire becomes more controlled by climate and weather (Cruz and Alexander 2010). Clearly, the strategy of trying to maintain more dense, late-successional forest habitat by reducing fire does not work if the method for reducing fire adversely affects far more of this forest habitat than would high-severity fire, and the high-severity fire might occur anyway because it is largely controlled by climate and weather.

There may be silvicultural treatments that can be done in spotted owl habitat that may reduce adverse impacts. For example, thinning that maintains at least 23-27.5 m^2 ha basal area. However, given that key habitat elements such as small trees, down wood, and likely some intermediate-sized trees are going to be targeted in any forest fuel reduction treatment, it appears unlikely that any conventional fuels reduction treatment in spotted owl habitat would not have at least some adverse impacts. This is supported by research on thinning that was often less intensive than commercial thinning prescriptions. This research showed negative impacts on spotted owls or their prey, as summarized in our introduction (Waters and Zabel 1995, Waters *et al.* 2000, Carey 2001, Ransome and Sullivan 2002, Gomez *et al.* 2003, Suzuki and Hayes 2003, Ransome *et al.* 2004, Bull *et al.* 2004, Lehmkuhl *et al.* 2006, Meyer *et al.* 2007, Wilson 2010, Holloway and Smith 2011, Manning *et al.* 2012), and how spotted owls have been displaced by even very limited amounts of thinning or contemporary harvest near the nest or activity center (Forsman *et al.* 1984, King 1993, Hicks *et al.* 1999, Meiman *et al.* 2003, Seamans and Gutiérrez 2007). Even if adverse impacts were quite modest, the amount of dense, late-successional forest that might be prevented from experiencing high-severity fire is so much smaller than the area that would be treated in an effort to accomplish this reduction in fire, that the net impact of the thinning would still be much greater. In addition, it is becoming increasingly less clear whether a reduction in high-severity fire below current rates would necessarily be beneficial to spotted owls. The dry forests in which spotted owls are found were historically characterized by mixed-severity fires (see Hessburg *et al.* (2007), Baker (2012), and Odion *et al.* (2014) for historic fire in the dry Cascades of Washington and Oregon, Beaty and Taylor (2001) and Bekker and Taylor (2001, 2010) for the California Cascades, and Wills and Stuart (1994), Taylor and Skinner (1998, 2003), and Odion *et al.* (2014) for the Klamath). Recent research suggests that this historic fire may have neutral and beneficial effects to spotted owls.

Studies on the effects of fire on spotted owls are few and often focused on other owl subspecies and some studies are confounded by post-fire logging effects (Clark *et al.* 2013). Nonetheless, it has long been known that fire in woody vegetation causes an increase in small rodent populations and consequently raptor populations (Lawrence 1966), and studies on spotted owls and fire where no logging occurred suggest that high-severity fire at current rates may confer benefits or be neutral. Bond *et al.* (2009) found that California Spotted Owls in the Sierra Nevada preferentially foraged in severely burned forests more than unburned forests within about 1.5 km of a core-use area. The percentage of high-severity fire in burned Mexican Spotted Owl (*Strix occidentalis* ssp. *lucida*) sites had no significant influence (Jenness *et al.* 2004). Roberts *et al.* (2011) found no support for an occupancy model for California Spotted Owls that distinguished between burned and unburned sites in unmanaged forests; the mean “owl survey area” that burned at high-severity was 12%, with one survey area experiencing up to 52% high-severity fire, which is almost three times the current amount of severe fire in owl habitat, according to the MTBS data. In a longer-term (1997-2007) study of California Spotted Owl site-occupancy dynamics throughout the Sierra Nevada, high-severity fire that burned on average 32% of forested vegetation around nests and core roosts had no significant effect on extinction or colonization probabilities, and overall occupancy probabilities were slightly higher in mixed-severity burned areas than in unburned forest (Lee *et al.* 2012), while other research found no significant difference in home range size between mixed-severity fire areas and unburned forest (Bond *et al.* 2013). Studies on reproduction in occupied sites of all three spotted owl subspecies indicated no difference between unburned sites and mixed-severity burned sites (excluding burn out areas created by fire suppression operations) (Jenness *et al.* 2004), or in some cases reproduction may have been greater in burned sites (Bond *et al.* 2002, Roberts 2008). The longer-term value of fire disturbances is in the creation of landscape heterogeneity with inclusions of young stands, improving habitat at the landscape scale. Fire also plays a vital role in creating snags, large down logs, and other key elements of the highest quality spotted owl habitat at the territory scale (Franklin *et al.* 2000). No assessments of fire and thinning effects on spotted owls, including this one, have accounted for any potential beneficial effects of mixed-severity fire, nor the potential negative effects of lack of mixed-severity fire in treated areas.

While much of the concern about fire and thinning in dry forests of the Pacific Northwest has focused on spotted owls, it may also apply to other biota associated with dense, old forests, including species of conservation concern, such as Pacific fisher (*Martes pennanti pacifica*), which research indicates may benefit from mixed-severity fire (Hanson 2013), the Northern Goshawk (*Accipiter gentilis*), and, following fire, the Black-backed Woodpecker (*Picoides arcticus*), which depends upon higher-severity fire in dense, older forest (Odion and Hanson 2013). Like the spotted owl, studies have documented that this woodpecker is also negatively affected by thinning (Hutto 2008). Also, like the spotted owl, the Back-backed Woodpecker, Pacific Fisher and Northern Goshawk occur in forests where the historic fire regime was not low-severity. Modeling for the fisher, similar to modeling for the spotted owl, has not used the actual rates of high-severity fire and forest regrowth to assess possible impacts of fire, and has assumed that fire represents a loss of fisher habitat (Scheller *et al.* 2011), contrary to more recent empirical findings (Hanson 2013). Not including the actual probability of fire leads to considerably inflated projections of the effects of thinning vs. not thinning in reducing high-severity fire (Rhodes and Baker 2008, Campbell *et al*. 2012). Our findings highlight the need to be cautious about conclusions that thinning treatments are needed for species found in dense forest and that they will not have unintended consequences (e.g., Stephens *et al.* 2012) until long-term, cumulative impacts are better understood. As we found with spotted owls, long-term and unintended consequences may be substantial for species that rely on dense, late-successional forests, especially when these species are sensitive to small amounts of thinning in their territory.

CONCLUSION

We used a quantitative approach that, unlike others, accounted for rates of high-severity fire and forest recruitment, allowing assessment of future amounts of spotted owl habitat at current rates of fire, with and without thinning. We found that the long-term benefits of commercial thinning would not clearly outweigh adverse impacts, even if much more fire occurs in the future. This conclusion applies even if adverse impacts of treatments are quite modest because of the vastly larger area that would need to be treated compared to area of high-severity fire that might be reduced by thinning. Moreover, our results indicate that, even if a longer time interval is analyzed (e.g., 100 years), the declines in dense, late-successional habitat due to thinning would not flatten, as long as thinning is reoccurring. Thus, where spotted owl management goals take precedence, the best strategy for maintaining habitat will be to avoid thinning treatments that have adverse impacts in spotted owl habitat or potential habitat (Gaines *et al.* 2010). There is ample area outside of existing or potential spotted owl habitat where managers wishing to suppress fire behavior or extent may focus their efforts without directly impacting spotted owls (Gaines *et al.* 2010), such as in areas adjacent to homes or in dense conifer plantations with high fuel hazards (Odion *et al.* 2004). In addition, there are management approaches that may be more effective than thinning in helping accomplish these fire prevention goals, such as controlling human-caused fire ignitions (Cary *et al.* 2009). Lastly, emerging research suggests that fire is not the threat it has been assumed to be for spotted owls, suggesting that, rather than management that focuses on suppressing fire behavior, other, no regrets active management may be more appropriate (Hanson *et al.* 2010). Research is needed to determine if these findings might apply to other species that are characteristic of dense forests, particularly given the widespread and growing emphasis on thinning as a management tool for suppressing wildland fires.

CONFLICT OF INTEREST

The authors confirm that this article content has no conflict of interest.

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SUPPORTIVE/SUPPLEMENTARY MATERIAL

Supplementary material is available on the publishers Web site along with the published article.

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Land use strategies to mitigate climate change in carbon dense temperate forests

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Strategies to mitigate carbon dioxide emissions through forestry activities have been proposed, but ecosystem process-based integration of climate change, enhanced CO_2, disturbance from fire, and management actions at regional scales are extremely limited. Here, we examine the relative merits of afforestation, reforestation, management changes, and harvest residue bioenergy use in the Pacific Northwest. This region represents some of the highest carbon density forests in the world, which can store carbon in trees for 800 y or more. Oregon's net ecosystem carbon balance (NECB) was equivalent to 72% of total emissions in 2011–2015. By 2100, simulations show increased net carbon uptake with little change in wildfires. Reforestation, afforestation, lengthened harvest cycles on private lands, and restricting harvest on public lands increase NECB 56% by 2100, with the latter two actions contributing the most. Resultant cobenefits included water availability and biodiversity, primarily from increased forest area, age, and species diversity. Converting 127,000 ha of irrigated grass crops to native forests could decrease irrigation demand by 233 billion $m^3 \cdot y^{-1}$. Utilizing harvest residues for bioenergy production instead of leaving them in forests to decompose increased emissions in the short-term (50 y), reducing mitigation effectiveness. Increasing forest carbon on public lands reduced emissions compared with storage in wood products because the residence time is more than twice that of wood products. Hence, temperate forests with high carbon densities and lower vulnerability to mortality have substantial potential for reducing forest sector emissions. Our analysis framework provides a template for assessments in other temperate regions.

forests | carbon balance | greenhouse gas emissions | climate mitigation

Strategies to mitigate carbon dioxide emissions through forestry activities have been proposed, but regional assessments to determine feasibility, timeliness, and effectiveness are limited and rarely account for the interactive effects of future climate, atmospheric CO_2 enrichment, nitrogen deposition, disturbance from wildfires, and management actions on forest processes. We examine the net effect of all of these factors and a suite of mitigation strategies at fine resolution (4-km grid). Proven strategies immediately available to mitigate carbon emissions from forest activities include the following: (*i*) reforestation (growing forests where they recently existed), (*ii*) afforestation (growing forests where they did not recently exist), (*iii*) increasing carbon density of existing forests, and (*iv*) reducing emissions from deforestation and degradation (1). Other proposed strategies include wood bioenergy production (2–4), bioenergy combined with carbon capture and storage (BECCS), and increasing wood product use in buildings. However, examples of commercial-scale BECCS are still scarce, and sustainability of wood sources remains controversial because of forgone ecosystem carbon storage and low environmental cobenefits (5, 6). Carbon stored in buildings generally outlives its usefulness or is replaced within decades (7) rather than the centuries possible in forests, and the factors influencing product substitution have yet to be fully explored (8). Our analysis of mitigation strategies focuses on the first four strategies, as well as bioenergy production, utilizing harvest residues only and without carbon capture and storage.

The appropriateness and effectiveness of mitigation strategies within regions vary depending on the current forest sink, competition with land-use and watershed protection, and environmental conditions affecting forest sustainability and resilience. Few process-based regional studies have quantified strategies that could actually be implemented, are low-risk, and do not depend on developing technologies. Our previous studies focused on regional modeling of the effects of forest thinning on net ecosystem carbon balance (NECB) and net emissions, as well as improving modeled drought sensitivity (9, 10), while this study focuses mainly on strategies to enhance forest carbon.

Our study region is Oregon in the Pacific Northwest, where coastal and montane forests have high biomass and carbon sequestration potential. They represent coastal forests from northern California to southeast Alaska, where trees live 800 y or more and biomass can exceed that of tropical forests (11) (Fig. S1). The semiarid ecoregions consist of woodlands that experience frequent fires (12). Land-use history is a major determinant of forest carbon balance. Harvest was the dominant cause of tree mortality (2003–2012) and accounted for fivefold as much mortality as that from fire and beetles combined (13). Forest land ownership is predominantly public (64%), and 76% of the biomass harvested is on private lands.

Significance

Regional quantification of feasibility and effectiveness of forest strategies to mitigate climate change should integrate observations and mechanistic ecosystem process models with future climate, CO_2, disturbances from fire, and management. Here, we demonstrate this approach in a high biomass region, and found that reforestation, afforestation, lengthened harvest cycles on private lands, and restricting harvest on public lands increased net ecosystem carbon balance by 56% by 2100, with the latter two actions contributing the most. Forest sector emissions tracked with our life cycle assessment model decreased by 17%, partially meeting emissions reduction goals. Harvest residue bioenergy use did not reduce short-term emissions. Cobenefits include increased water availability and biodiversity of forest species. Our improved analysis framework can be used in other temperate regions.

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Data deposition: The CLM4.5 model data are available at Oregon State University (terraweb.forestry.oregonstate.edu/FMEC). Data from the >200 intensive plots on forest carbon are available at Oak Ridge National Laboratory (https://daac.ornl.gov/NACP/guides/NACP_TERRA-PNW.html), and FIA data are available at the USDA Forest Service (https://www.fia.fs.fed.us/tools-data/).

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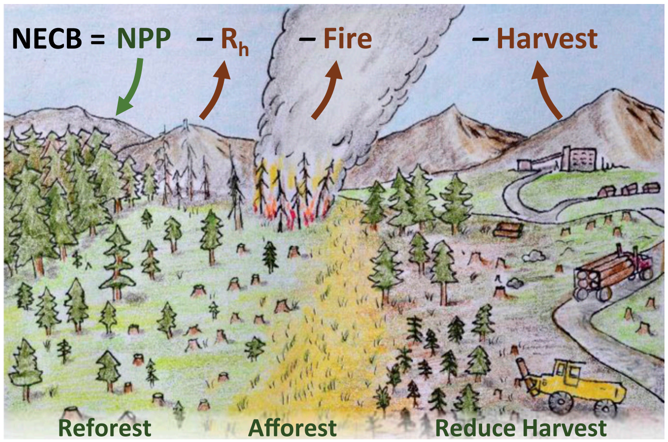

Fig. 1. Approach to assessing effects of mitigation strategies on forest carbon and forest sector emissions. NECB is productivity (NPP) minus Rh and losses from fire and harvest (red arrows). Harvest emissions include those associated with wood products and bioenergy.

Many US states, including Oregon (14), plan to reduce their greenhouse gas (GHG) emissions in accordance with the Paris Agreement. We evaluated strategies to address this question: How much carbon can the region's forests realistically remove from the atmosphere in the future, and which forest carbon strategies can reduce regional emissions by 2025, 2050, and 2100? We propose an integrated approach that combines observations with models and a life cycle assessment (LCA) to evaluate current and future effects of mitigation actions on forest carbon and forest sector emissions in temperate regions (Fig. 1). We estimated the recent carbon budget of Oregon's forests, and simulated the potential to increase the forest sink and decrease forest sector emissions under current and future climate conditions. We provide recommendations for regional assessments of mitigation strategies.

Results

Carbon stocks and fluxes are summarized for the observation cycles of 2001–2005, 2006–2010, and 2011–2015 (Table 1 and Tables S1 and S2). In 2011–2015, state-level forest carbon stocks totaled 3,036 Tg C (3 billion metric tons), with the coastal and montane ecoregions accounting for 57% of the live tree carbon (Tables S1 and S2). Net ecosystem production [NEP; net primary production (NPP) minus heterotrophic respiration (Rh)] averaged 28 teragrams carbon per year (Tg C y^{-1}) over all three periods. Fire emissions were unusually high at 8.69 million metric tons carbon dioxide equivalent (tCO_2e y^{-1}, i.e., 2.37 Tg C y^{-1}) in 2001–2005 due to the historic Biscuit Fire, but decreased to 3.56 million tCO_2e y^{-1} (0.97 Tg C y^{-1}) in 2011–2015 (Table S4). Note that 1 million tCO_2e equals 3.667 Tg C.

Our LCA showed that in 2001–2005, Oregon's net wood product emissions were 32.61 million tCO_2e (Table S3), and 3.7-fold wildfire emissions in the period that included the record fire year (15) (Fig. 2). In 2011–2015, net wood product emissions were 34.45 million tCO_2e and almost 10-fold fire emissions, mostly due to lower fire emissions. The net wood product emissions are higher than fire emissions despite carbon benefits of storage in wood products and substitution for more fossil fuel-intensive products. Hence, combining fire and net wood product emissions, the forest sector emissions averaged 40 million tCO_2e y^{-1} and accounted for about 39% of total emissions across all sectors (Fig. 2 and Table S4). NECB was calculated from NEP minus losses from fire emissions and harvest (Fig. 1). State NECB was equivalent to 60% and 70% of total emissions for 2001–2005 and 2011–2015, respectively (Fig. 2, Table 1, and Table S4). Fire emissions were only between 4% and 8% of total emissions from all sources (2011–2015 and 2001–2004, respectively). Oregon's forests play a larger role in meeting its GHG targets than US forests have in meeting the nation's targets (16, 17).

Historical disturbance regimes were simulated using stand age and disturbance history from remote sensing products. Comparisons of Community Land Model (CLM4.5) output with Forest Inventory and Analysis (FIA) aboveground tree biomass (>6,000 plots) were within 1 SD of the ecoregion means (Fig. S2). CLM4.5 estimates of cumulative burn area and emissions from 1990 to 2014 were 14% and 25% less than observed, respectively. The discrepancy was mostly due to the model missing an anomalously large fire in 2002 (Fig. S3*A*). When excluded, modeled versus observed fire emissions were in good agreement ($r^2 = 0.62$; Fig. S3*B*). A sensitivity test of a 14% underestimate of burn area did not affect our final results because predicted emissions would increase almost equally for business as usual (BAU) management and our scenarios, resulting in no proportional change in NECB. However, the ratio of harvest to fire emissions would be lower.

Projections show that under future climate, atmospheric carbon dioxide, and BAU management, an increase in net carbon uptake due to CO_2 fertilization and climate in the mesic ecoregions far outweighs losses from fire and drought in the semiarid ecoregions. There was not an increasing trend in fire. Carbon stocks increased by 2% and 7% and NEP increased by 12% and 40% by 2050 and 2100, respectively.

We evaluated emission reduction strategies in the forest sector: protecting existing forest carbon, lengthening harvest cycles, reforestation, afforestation, and bioenergy production with product substitution. The largest potential increase in forest carbon is in the mesic Coast Range and West Cascade ecoregions. These forests are buffered by the ocean, have high soil water-holding capacity, low risk of wildfire [fire intervals average 260–400 y (18)], long carbon residence time, and potential for high carbon density. They can attain biomass up to 520 Mg C ha^{-1} (12). Although Oregon has several protected areas, they account for only 9–15% of the total forest area, so we expect it may be feasible to add carbon-protected lands with cobenefits of water protection and biodiversity.

Reforestation of recently forested areas include those areas impacted by fire and beetles. Our simulations to 2100 assume regrowth of the same species and incorporate future fire responses to climate and cyclical beetle outbreaks [70–80 y (13)]. Reforestation has the potential to increase stocks by 315 Tg C by 2100, reducing forest sector net emissions by 5% by 2100 relative to BAU management (Fig. 3). The East and West Cascades ecoregions had the highest reforestation potential, accounting for 90% of the increase (Table S5).

Afforestation of old fields within forest boundaries and nonfood/nonforage grass crops, hereafter referred to as "grass crops," had to meet minimum conditions for tree growth, and crop grid cells had to be partially forested (*SI Methods* and Table S6). These crops are not grazed or used for animal feed. Competing land uses may decrease the actual amount of area that can be afforested. We calculated the amount of irrigated grass crops (127,000 ha) that could be converted to forest, assuming success of carbon offset programs (19). By 2100, afforestation increased stocks by

Table 1. Forest carbon budget components used to compute NECB

Flux, Tg C·y^{-1}	2001–2005		2006–2010		2011–2015		2001–2015
NPP	73.64	7.59	73.57	7.58	73.57	7.58	73.60
Rh	45.67	5.11	45.38	5.07	45.19	5.05	45.41
NEP	27.97	9.15	28.19	9.12	28.39	9.11	28.18
Harvest removals	8.58	0.60	7.77	0.54	8.61	0.6	8.32
Fire emissions	2.37	0.27	1.79	0.2	0.97	0.11	1.71
NECB	17.02	9.17	18.63	9.14	18.81	9.13	18.15

Average annual values for each period, including uncertainty (95% confidence interval) in Tg C y^{-1} (multiply by 3.667 to get million tCO_2e).

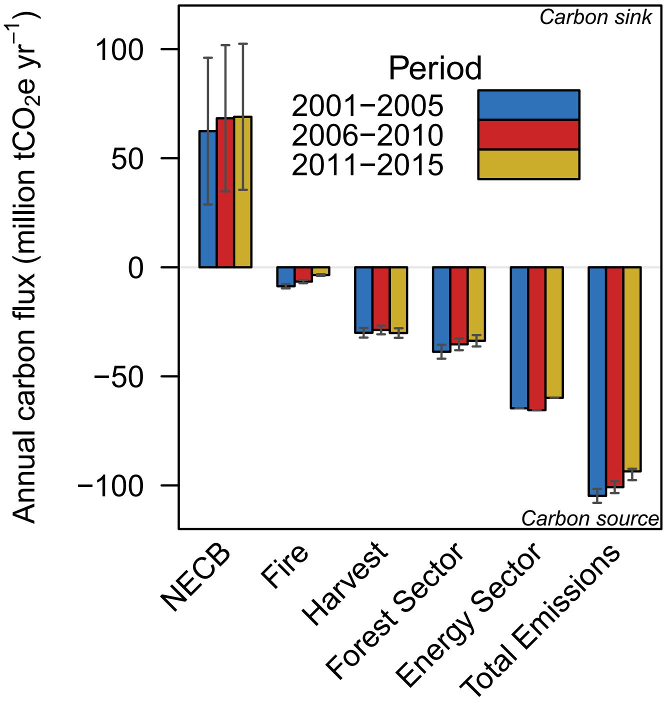

Fig. 2. Oregon's forest carbon sink and emissions from forest and energy sectors. Harvest emissions are computed by LCA. Fire and harvest emissions sum to forest sector emissions. Energy sector emissions are from the Oregon Global Warming Commission (14), minus forest-related emissions. Error bars are 95% confidence intervals (Monte Carlo analysis).

94 Tg C and cumulative NECB by 14 Tg C, and afforestation reduced forest sector GHG emissions by 1.3–1.4% in 2025, 2050, and 2100 (Fig. 3).

We quantified cobenefits of afforestation of irrigated grass crops on water availability based on data from hydrology and agricultural simulations of future grass crop area and related irrigation demand (20). Afforestation of 127,000 ha of grass cropland with Douglas fir could decrease irrigation demand by 222 and 233 billion $m^3 \cdot y^{-1}$ by 2050 and 2100, respectively. An independent estimate from measured precipitation and evapotranspiration (ET) at our mature Douglas fir and grass crop flux sites in the Willamette Valley shows the ET/precipitation fraction averaged 33% and 52%, respectively, and water balance (precipitation minus ET) averaged 910 $mm \cdot y^{-1}$ and 516 $mm \cdot y^{-1}$. Under current climate conditions, the observations suggest an increase in annual water availability of 260 billion $m^3 \cdot y^{-1}$ if 127,000 ha of the irrigated grass crops were converted to forest.

Harvest cycles in the mesic and montane forests have declined from over 120 y to 45 y despite the fact that these trees can live 500–1,000 y and net primary productivity peaks at 80–125 y (21). If harvest cycles were lengthened to 80 y on private lands and harvested area was reduced 50% on public lands, state-level stocks would increase by 17% to a total of ~3,600 Tg C and NECB would increase 2–3 Tg C y^{-1} by 2100. The lengthened harvest cycles reduced harvest by 2 Tg C y^{-1}, which contributed to higher NECB. Leakage (more harvest elsewhere) is difficult to quantify and could counter these carbon gains. However, because harvest on federal lands was reduced significantly since 1992 (NW Forest Plan), leakage has probably already occurred.

The four strategies together increased NECB by 64%, 82%, and 56% by 2025, 2050, and 2100, respectively. This reduced forest sector net emissions by 11%, 10%, and 17% over the same periods (Fig. 3). By 2050, potential increases in NECB were largest in the Coast Range (Table S5), East Cascades, and Klamath Mountains, accounting for 19%, 25%, and 42% of the total increase, whereas by 2100, they were most evident in the West Cascades, East Cascades, and Klamath Mountains.

We examined the potential for using existing harvest residue for electricity generation, where burning the harvest residue for energy emits carbon immediately (3) versus the BAU practice of leaving residues in forests to slowly decompose. Assuming half of forest residues from harvest practices could be used to replace natural gas or coal in distributed facilities across the state, they would provide an average supply of 0.75–1 Tg C y^{-1} to the year 2100 in the reduced harvest and BAU scenarios, respectively. Compared with BAU harvest practices, where residues are left to decompose, proposed bioenergy production would increase cumulative net emissions by up to 45 Tg C by 2100. Even at 50% use, residue collection and transport are not likely to be economically viable, given the distances (>200 km) to Oregon's facilities.

Discussion

Earth system models have the potential to bring terrestrial observations related to climate, vulnerability, impacts, adaptation,

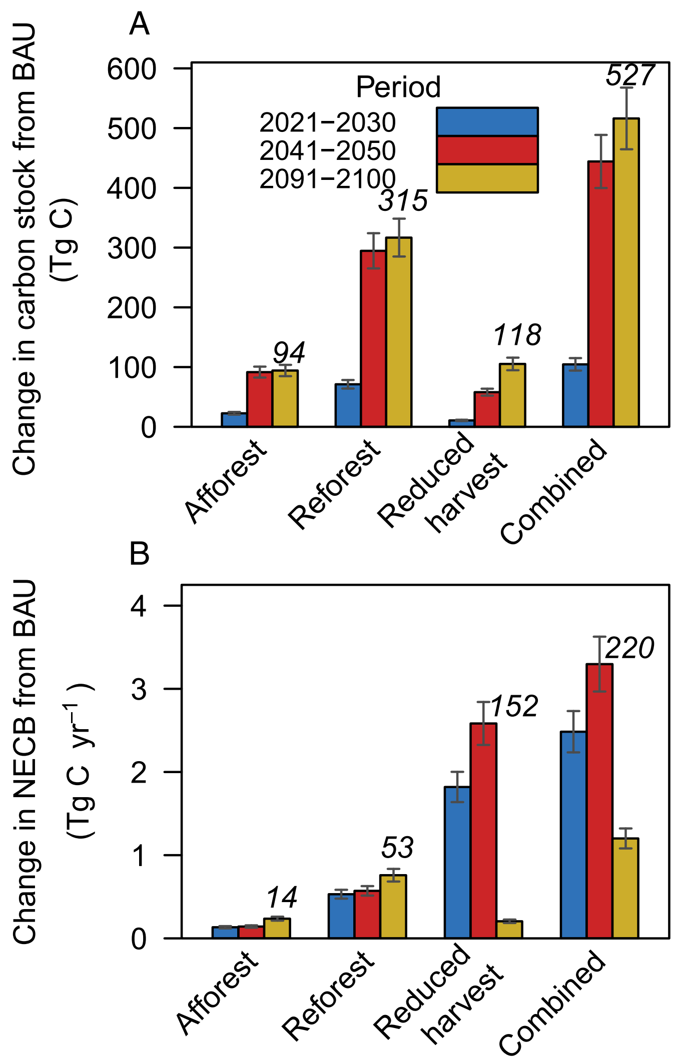

Fig. 3. Future change in carbon stocks and NECB with mitigation strategies relative to BAU management. The decadal average change in forest carbon stocks (*A*) and NECB relative to BAU (*B*) are shown. Italicized numbers over bars indicate mean forest carbon stocks in 2091–2100 (*A*) and cumulative change in NECB for 2015–2100 (*B*). Error bars are ±10%.

and mitigation into a common framework, melding biophysical with social components (22). We developed a framework to examine a suite of mitigation actions to increase forest carbon sequestration and reduce forest sector emissions under current and future environmental conditions.

Harvest-related emissions had a large impact on recent forest NECB, reducing it by an average of 34% from 2001 to 2015. By comparison, fire emissions were relatively small and reduced NECB by 12% in the Biscuit Fire year, but only reduced NECB 5–9% from 2006 to 2015. Thus, altered forest management has the potential to enhance the forest carbon balance and reduce emissions.

Future NEP increased because enhancement from atmospheric carbon dioxide outweighed the losses from fire. Lengthened harvest cycles on private lands to 80 y and restricting harvest to 50% of current rates on public lands increased NECB the most by 2100, accounting for 90% of total emissions reduction (Fig. 3 and Tables S5 and S6). Reduced harvest led to NECB increasing earlier than the other strategies (by 2050), suggesting this could be a priority for implementation.

Our afforestation estimates may be too conservative by limiting them to nonforest areas within current forest boundaries and 127,000 ha of irrigated grass cropland. There was a net loss of 367,000 ha of forest area in Oregon and Washington combined from 2001 to 2006 (23), and less than 1% of native habitat remains in the Willamette Valley due to urbanization and agriculture (24). Perhaps more of this area could be afforested.

The spatial variation in the potential for each mitigation option to improve carbon stocks and fluxes shows that the reforestation potential is highest in the Cascade Mountains, where fire and insects occur (Fig. 4). The potential to reduce harvest on public land is highest in the Cascade Mountains, and that to lengthen harvest cycles on private lands is highest in the Coast Range.

Although western Oregon is mesic with little expected change in precipitation, the afforestation cobenefits of increased water availability will be important. Urban demand for water is projected to increase, but agricultural irrigation will continue to consume much more water than urban use (25). Converting 127,000 ha of irrigated grass crops to native forests appears to be a win–win strategy, returning some of the area to forest land, providing habitat and connectivity for forest species, and easing irrigation demand. Because the afforested grass crop represents only 11% of the available grass cropland (1.18 million ha), it is not likely to result in leakage or indirect land use change. The two forest strategies combined are likely to be important contributors to water security.

Cobenefits with biodiversity were not assessed in our study. However, a recent study showed that in the mesic forests, cobenefits with biodiversity of forest species are largest on lands with harvest cycles longer than 80 y, and thus would be most pronounced on private lands (26). We selected 80 y for the harvest cycle mitigation strategy because productivity peaks at 80–125 y in this region, which coincides with the point at which cobenefits with wildlife habitat are substantial.

Habitat loss and climate change are the two greatest threats to biodiversity. Afforestation of areas that are currently grass crops would likely improve the habitat of forest species (27), as about 90% of the forests in these areas were replaced by agriculture. About 45 mammal species are at risk because of range contraction (28). Forests are more efficient at dissipating heat than grass and crop lands, and forest cover gains lead to net surface cooling in all regions south of about 45° latitude in North American and Europe (29). The cooler conditions can buffer climate-sensitive bird populations from approaching their thermal limits and provide more food and nest sites (30). Thus, the mitigation strategies of afforestation, protecting forests on public lands and lengthening harvest cycles to 80–125 y, would likely benefit forest-dependent species.

Oregon has a legislated mandate to reduce emissions, and is considering an offsets program that limits use of offsets to 8% of

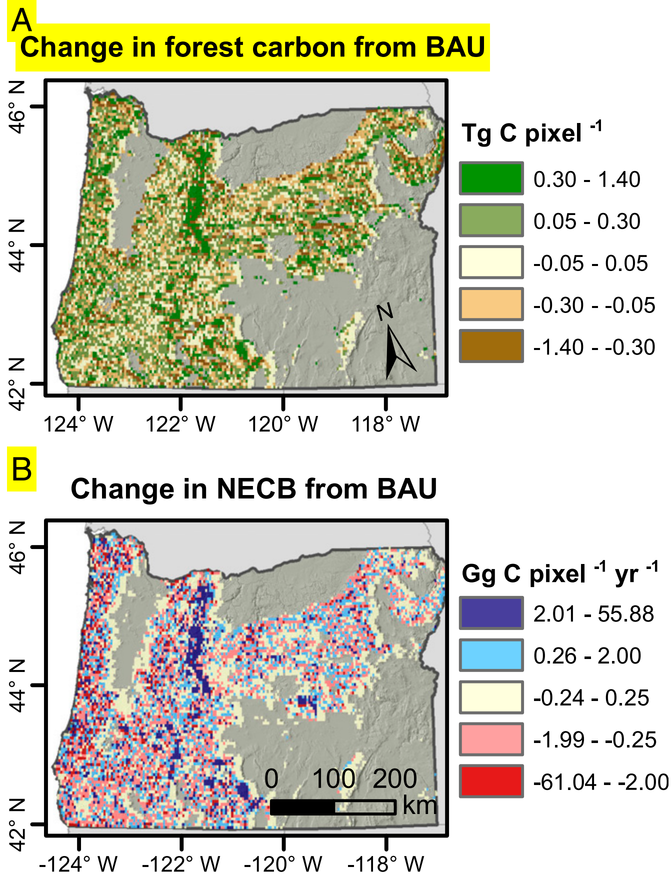

Fig. 4. Spatial patterns of forest carbon stocks and NECB by 2091–2100. The decadal average changes in forest carbon stocks (*A*) and NECB (*B*) due to afforestation, reforestation, protected areas, and lengthened harvest cycles relative to continued BAU forest management (red is increase in NECB) are shown.

the total emissions reduction to ensure that regulated entities substantially reduce their own emissions, similar to California's program (19). An offset becomes a net emissions reduction by increasing the forest carbon sink (NECB). If only 8% of the GHG reduction is allowed for forest offsets, the limits for forest offsets would be 2.1 and 8.4 million metric tCO_2e of total emissions by 2025 and 2050, respectively (Table S6). The combination of afforestation, reforestation, and reduced harvest would provide 13 million metric tCO_2e emissions reductions, and any one of the strategies or a portion of each could be applied. Thus, additionality beyond what would happen without the program is possible.

State-level reporting of GHG emissions includes the agriculture sector, but does not appear to include forest sector emissions, except for industrial fuel (i.e., utility fuel in Table S3) and, potentially, fire emissions. Harvest-related emissions should be quantified, as they are much larger than fire emissions in the western United States. Full accounting of forest sector emissions is necessary to meet climate mitigation goals.

Increased long-term storage in buildings and via product substitution has been suggested as a potential climate mitigation option. Pacific temperate forests can store carbon for many hundreds of years, which is much longer than is expected for buildings that are generally assumed to outlive their usefulness or be replaced within several decades (7). By 2035, about 75% of buildings in the United States will be replaced or renovated, based on new construction, demolition, and renovation trends (31, 32). Recent analysis suggests substitution benefits of using wood versus more fossil fuel-intensive materials have been overestimated by at least an order of magnitude (33). Our LCA accounts for losses in product substitution stores (PSSs) associated with building life span, and thus are considerably lower than when no losses are assumed (4, 34). While product substitution reduces the overall forest sector emissions, it cannot offset the losses incurred by frequent harvest and losses associated with product transportation, manufacturing, use, disposal, and decay. Methods for calculating substitution benefits should be improved in other regional assessments.

Wood bioenergy production is interpreted as being carbon-neutral by assuming that trees regrow to replace those that burned. However, this does not account for reduced forest carbon stocks that took decades to centuries to sequester, degraded productive capacity, emissions from transportation and the production process, and biogenic/direct emissions at the facility (35). Increased harvest through proposed thinning practices in the region has been shown to elevate emissions for decades to centuries regardless of product end use (36). It is therefore unlikely that increased wood bioenergy production in this region would decrease overall forest sector emissions.

Conclusions

GHG reduction must happen quickly to avoid surpassing a 2 °C increase in temperature since preindustrial times. Alterations in forest management can contribute to increasing the land sink and decreasing emissions by keeping carbon in high biomass forests, extending harvest cycles, reforestation, and afforestation. Forests are carbon-ready and do not require new technologies or infrastructure for immediate mitigation of climate change. Growing forests for bioenergy production competes with forest carbon sequestration and does not reduce emissions in the next decades (10). BECCS requires new technology, and few locations have sufficient geological storage for CO_2 at power facilities with high-productivity forests nearby. Accurate accounting of forest carbon in trees and soils, NECB, and historic harvest rates, combined with transparent quantification of emissions from the wood product process, can ensure realistic reductions in forest sector emissions.

As states and regions take a larger role in implementing climate mitigation steps, robust forest sector assessments are urgently needed. Our integrated approach of combining observations, an LCA, and high-resolution process modeling (4-km grid vs. typical 200-km grid) of a suite of potential mitigation actions and their effects on forest carbon sequestration and emissions under changing climate and CO_2 provides an analysis framework that can be applied in other temperate regions.

Materials and Methods

Current Stocks and Fluxes. We quantified recent forest carbon stocks and fluxes using a combination of observations from FIA; Landsat products on forest type, land cover, and fire risk; 200 intensive plots in Oregon (37); and a wood decomposition database. Tree biomass was calculated from species-specific allometric equations and ecoregion-specific wood density. We estimated ecosystem carbon stocks, NEP (photosynthesis minus respiration), and NECB (NEP minus losses due to fire or harvest) using a mass-balance approach (36, 38) (Table 1 and *SI Materials and Methods*). Fire emissions were computed from the Monitoring Trends in Burn Severity database, biomass data, and region-specific combustion factors (15, 39) (*SI Materials and Methods*).

Future Projections and Model Description. Carbon stocks and NEP were quantified to the years 2025, 2050, and 2100 using CLM4.5 with physiological parameters for 10 major forest species, initial forest biomass (36), and future climate and atmospheric carbon dioxide as input (Institut Pierre Simon Laplace climate system model downscaled to 4 km × 4 km, representative concentration pathway 8.5). CLM4.5 uses 3-h climate data, ecophysiological characteristics, site physical characteristics, and site history to estimate the daily fluxes of carbon, nitrogen, and water between the atmosphere, plant state variables, and litter and soil state variables. Model components are biogeophysics, hydrological cycle, and biogeochemistry. This model version does not include a dynamic vegetation model to simulate resilience and establishment following disturbance. However, the effect of regeneration lags on forest carbon is not particularly strong for the long disturbance intervals in this study (40). Our plant functional type (PFT) parameterization for 10 major forest species rather than one significantly improves carbon modeling in the region (41).

Forest Management and Land Use Change Scenarios. Harvest cycles, reforestation, and afforestation were simulated to the year 2100. Carbon stocks and NEP were predicted for the current harvest cycle of 45 y compared with simulations extending it to 80 y. Reforestation potential was simulated over areas that recently suffered mortality from harvest, fire, and 12 species of beetles (13). We assumed the same vegetation regrew to the maximum potential, which is expected with the combination of natural regeneration and planting that commonly occurs after these events. Future BAU harvest files were constructed using current harvest rates, where county-specific average harvest and the actual amounts per ownership were used to guide grid cell selection. This resulted in the majority of harvest occurring on private land (70%) and in the mesic ecoregions. Beetle outbreaks were implemented using a modified mortality rate of the lodgepole pine PFT with 0.1% y^{-1} biomass mortality by 2100.

For afforestation potential, we identified areas that are within forest boundaries that are not currently forest and areas that are currently grass crops. We assumed no competition with conversion of irrigated grass crops to urban growth, given Oregon's land use laws for developing within urban growth boundaries. A separate study suggested that, on average, about 17% of all irrigated agricultural crops in the Willamette Valley could be converted to urban area under future climate; however, because 20% of total cropland is grass seed, it suggests little competition with urban growth (25).

Landsat observations (12,500 scenes) were processed to map changes in land cover from 1984 to 2012. Land cover types were separated with an unsupervised K-means clustering approach. Land cover classes were assigned to an existing forest type map (42). The CropScape Cropland Data Layer (CDL 2015, https://nassgeodata.gmu.edu/CropScape/) was used to distinguish nonforage grass crops from other grasses. For afforestation, we selected grass cropland with a minimum soil water-holding capacity of 150 mm and minimum precipitation of 500 mm that can support trees (43).

Afforestation Cobenefits. Modeled irrigation demand of grass seed crops under future climate conditions was previously conducted with hydrology and agricultural models, where ET is a function of climate, crop type, crop growth state, and soil-holding capacity (20) (Table S7). The simulations produced total land area, ET, and irrigation demand for each cover type. Current grass seed crop irrigation in the Willamette Valley is 413 billion $m^3 \cdot y^{-1}$ for 238,679 ha and is projected to be 412 and 405 billion m^3 in 2050 and 2100 (20) (Table S7). We used annual output from the simulations to estimate irrigation demand per unit area of grass seed crops (1.73, 1.75, and 1.84 million $m^3 \cdot ha^{-1}$ in 2015, 2050, and 2100, respectively), and applied it to the mapped irrigated crop area that met conditions necessary to support forests (Table S7).

LCA. Decomposition of wood through the product cycle was computed using an LCA (8, 10). Carbon emissions to the atmosphere from harvest were calculated annually over the time frame of the analysis (2001–2015). The net carbon emissions equal NECB plus total harvest minus wood lost during manufacturing and wood decomposed over time from product use. Wood industry fossil fuel emissions were computed for harvest, transportation, and manufacturing processes. Carbon credit was calculated for wood product storage, substitution, and internal mill recycling of wood losses for bioenergy.

Products were divided into sawtimber, pulpwood, and wood and paper products using published coefficients (44). Long-term and short-term products were assumed to decay at 2% and 10% per year, respectively (45). For product substitution, we focused on manufacturing for long-term structures (building life span >30 y). Because it is not clear when product substitution started in the Pacific Northwest, we evaluated it starting in 1970 since use of concrete and steel for housing was uncommon before 1965. The displacement value for product substitution was assumed to be 2.1 Mg fossil C/Mg C wood use in long-term structures (46), and although it likely fluctuates over time, we assumed it was constant. We accounted for losses in product substitution associated with building replacement (33) using a loss rate of 2% per year (33), but ignored leakage related to fossil C use by other sectors, which may result in more substitution benefit than will actually occur.

The general assumption for modern buildings, including cross-laminate timber, is they will outlive their usefulness and be replaced in about 30 y (7). By 2035, ~75% of buildings in the United States will be replaced or renovated, based on new construction, demolition, and renovation trends, resulting in threefold as many buildings as there are now [2005 baseline (31, 32)]. The loss of

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the PSS is therefore PSS multiplied by the proportion of buildings lost per year (2% per year).

To compare the NECB equivalence to emissions, we calculated forest sector and energy sector emissions separately. Energy sector emissions ["in-boundary" state-quantified emissions by the Oregon Global Warming Commission (14)] include those from transportation, residential and commercial buildings, industry, and agriculture. The forest sector emissions are cradle-to-grave annual carbon emissions from harvest and product emissions, transportation, and utility fuels (Table S3). Forest sector utility fuels were subtracted from energy sector emissions to avoid double counting.

Uncertainty Estimates. For the observation-based analysis, Monte Carlo simulations were used to conduct an uncertainty analysis with the mean and SDs for NPP and Rh calculated using several approaches (36) (*SI Materials and Methods*). Uncertainty in NECB was calculated as the combined uncertainty of NEP, fire emissions (10%), harvest emissions (7%), and land cover estimates (10%) using the propagation of error approach. Uncertainty in CLM4.5 model simulations and LCA were quantified by combining the uncertainty in the observations used to evaluate the model, the uncertainty in input datasets (e.g., remote sensing), and the uncertainty in the LCA coefficients (41).

Model input data for physiological parameters and model evaluation data on stocks and fluxes are available online (37).

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Do insect outbreaks reduce the severity of subsequent forest fires?

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Abstract

Understanding the causes and consequences of rapid environmental change is an essential scientific frontier, particularly given the threat of climate- and land use-induced changes in disturbance regimes. In western North America, recent widespread insect outbreaks and wildfires have sparked acute concerns about potential insect–fire interactions. Although previous research shows that insect activity typically does not increase wildfire likelihood, key uncertainties remain regarding insect effects on wildfire severity (i.e., ecological impact). Recent assessments indicate that outbreak severity and burn severity are not strongly associated, but these studies have been limited to specific insect or fire events. Here, we present a regional census of large wildfire severity following outbreaks of two prevalent bark beetle and defoliator species, mountain pine beetle (*Dendroctonus ponderosae*) and western spruce budworm (*Choristoneura freemani*), across the US Pacific Northwest. We first quantify insect effects on burn severity with spatial modeling at the fire event scale and then evaluate how these effects vary across the full population of insect–fire events ($n = 81$ spanning 1987–2011). In contrast to common assumptions of positive feedbacks, we find that insects generally reduce the severity of subsequent wildfires. Specific effects vary with insect type and timing, but both insects decrease the abundance of live vegetation susceptible to wildfire at multiple time lags. By dampening subsequent burn severity, native insects could buffer rather than exacerbate fire regime changes expected due to land use and climate change. In light of these findings, we recommend a precautionary approach when designing and implementing forest management policies intended to reduce wildfire hazard and increase resilience to global change.

1. Introduction

Forest ecosystems play a vital role in the biosphere, but anthropogenic climate change and shifting disturbance regimes threaten to destabilize the ecosystem services that forests provide from local to global scales (Kurz *et al* 2008, Littell *et al* 2010, Seidl *et al* 2011, Turner *et al* 2013). Indeed, the indirect effects of climate change on forests via disturbances (including wildfires, insect outbreaks, introduced species, and pathogens) are expected to exceed the direct but more gradual effects of warmer temperatures (Ayres *et al* 2014, Hart *et al* 2015). In an era of rapid, nonlinear changes in the Earth system, understanding the causes, consequences, and feedbacks of forest disturbances is a crucial scientific and policy frontier.

Disturbance interactions—when one disturbance influences the likelihood, extent, or severity of another (Paine *et al* 1998, Simard *et al* 2011, Buma 2015, Meigs *et al* 2015a)—are a particularly important example of feedbacks that could be reinforced under novel climatic conditions (e.g., persistent drought (Turner *et al* 2013, Harvey *et al* 2014b, Hart *et al* 2015)). In western North America, insect outbreaks and wildfires are the two most ecologically and economically significant natural forest disturbances (Westerling *et al* 2006, Kurz *et al* 2008, Hicke *et al* 2013). Both disturbances have been widespread in recent decades and are projected to increase in response to climate and land use change (Hessburg *et al* 2000, Westerling *et al* 2006, Raffa *et al* 2008, Bentz *et al* 2010, Littell *et al* 2010, Ayres *et al* 2014). By killing trees and redistributing forest fuels, insect outbreaks influence fire regimes in many parts of the world, and recent large outbreaks have sparked acute societal concerns about potential insect–fire interactions and impaired ecosystem resilience (Hicke *et al* 2012, Harvey *et al* 2014b, Jenkins *et al* 2014). For example, based on the implicit assumption that insect outbreaks increase wildfire hazard by generating abundant dead fuels, the 2014 US Farm Bill designated $200 million annually to support fuel reduction activities across 18 M ha of US National Forest lands affected by diseases and insects (Agricultural Act of 2014, Hart *et al* 2015).

Despite concerns about altered fire regimes and insect–fire interactions, recent studies indicate that insect outbreaks generally do not increase wildfire likelihood (Lynch and Moorcroft 2008, Kulakowski and Jarvis 2011, Flower *et al* 2014, Hart *et al* 2015, Meigs *et al* 2015a). When they do overlap, however, key uncertainties remain regarding the influence of insect outbreaks on subsequent wildfire severity (Hicke *et al* 2012, Harvey *et al* 2014b, Hart *et al* 2015). Specifically, although insect-caused tree mortality may increase the flammability of canopy fuels at fine scales in time and space (Jolly *et al* 2012), a pivotal question in contemporary environmental management is whether these insect-altered fuels increase burn severity (i.e., ecological impact; a major fire regime component) at broader spatiotemporal scales. If insect outbreaks do amplify subsequent fire effects, the resultant compound impacts may hasten climate-induced shifts in disturbance regimes toward more severe fire and altered ecosystem structure and function. Conversely, if insects buffer subsequent fire effects by redistributing fuel density and/or availability, recent widespread outbreaks may bolster ecosystem resistance to shifting fire regimes. Empirical studies that identify particular time lags and locations where insect-altered fuels either exacerbate or dampen fire effects on surviving trees are directly applicable to time-sensitive management activities (e.g., post-insect salvage logging, fuel reduction at the wildland–urban interface) as well as broader policy discussions of forest health in a time of shifting disturbance regimes.

Due in part to data paucity, computational limitations, and the relative rarity of insect–fire co-occurrence, recent empirical assessments of insect effects on burn severity have been limited to specific insect outbreaks, fire events, or insect–fire time lags (e.g., Crickmore 2011, Harvey *et al* 2013, Harvey *et al* 2014b, Prichard and Kennedy 2014). These studies suggest that burn severity is either unaffected by or weakly positively associated with outbreak severity, that insect effects are context-dependent, or that factors like fuel treatments, topography, and weather are stronger predictors of fire effects. To further elucidate general system behavior and inform regional management strategies, it is essential to investigate numerous fire events spanning multiple insect types (e.g., bark beetle versus defoliator), insect and burn severities, and time lags. Here, we leverage recent advances in remote sensing of forest disturbance dynamics (Kennedy *et al* 2010, Meigs *et al* 2015b) to conduct a burn severity census of large wildfires following recent outbreaks of the two most prevalent native forest insects across a large forested region, the US Pacific Northwest (PNW; 40 M ha; Oregon and Washington; figure 1). We focus on all large fire events (⩾400 ha) with substantial overlap of fire perimeters with prior outbreaks of either mountain pine beetle (MPB) [*Dendroctonus ponderosae* Hopkins (Coleoptera: Curculionidae: Scolytinae); a bark beetle] or western spruce budworm (WSB) [*Choristoneura freemani* Razowski (Lepidoptera: Tortricidae); a defoliator] (total $n = 81$; table S1). Our specific objectives are (1) to quantify the fine-scale (30 m) effects of recent insect outbreaks on subsequent burn severity with spatial modeling at the fire event scale and (2) to evaluate the role of insect type, time since outbreak, insect and fire extent, fire season, and interannual drought across the full population of insect–fire events.

2. Methods

2.1. Study area and recent insect dynamics

Conifer forests of the PNW vary across gradients of climate, topography, soil, and management history (Franklin and Dyrness 1973, Hessburg *et al* 2000, Meigs *et al* 2015a). Despite climatic variability, a common feature is that low precipitation during summer months (Franklin and Dyrness 1973) yields conditions conducive to periodic insect and wildfire disturbances, particularly in mixed-species conifer forests east of the crest of the Cascade Range (Meigs *et al* 2015a). In general, these forests occur in remote, mountainous terrain and are managed by US federal agencies for multiple resource objectives. Given the extent of similar geographic conditions, vegetation types, and anthropogenic pressures, recent PNW insect and wildfire patterns are broadly representative of contemporary disturbance dynamics in conifer forests of western North America.

Bark beetles, especially MPB outbreaks, have altered forest composition and structure across tens of millions of hectares of North American forests in recent decades (Raffa *et al* 2008, Bentz *et al* 2010). MPB adults attack pine tree stems [*Pinus spp.*, particularly mature lodgepole pine (*Pinus contorta* Douglas ex Louden)], inducing variable but relatively rapid tree mortality during major outbreaks (Raffa *et al* 2008, Meigs *et al* 2011). In contrast, WSB larvae typically consume the current year's foliage of host trees {particularly true firs [*Abies spp.*], spruces [*Picea spp.*], and Douglas-fir [*Pseudotsuga menziesii* (Mirb.) Franco]},

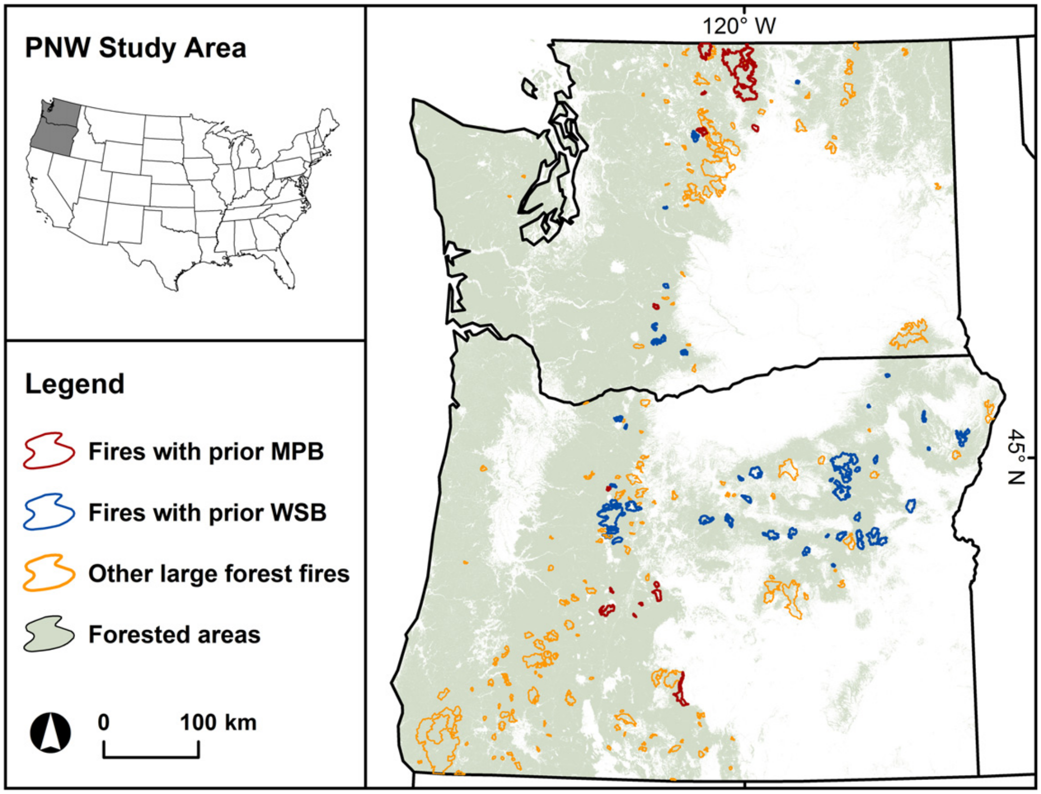

Figure 1. Distribution of large forest fires affected by prior outbreaks of mountain pine beetle (MPB; red; $n = 19$) and western spruce budworm (WSB; blue; $n = 62$) within the US Pacific Northwest study area (Oregon and Washington; inset). Note that 29% of large wildfires during the study period ($\geqslant$400 ha; 1987–2011; $n = 277$) had $\geqslant$10% prior insect damage, highlighting that these potential compound disturbance events are limited to drier, interior conifer forests. We include only large wildfire perimeters with $\geqslant$50% forest cover (see methods). Oregon and Washington encompass ca. 40 M ha, half of which is forested (forest cover from Ohmann *et al* 2012).

and multiple years of WSB defoliation can result in tree mortality, often in conjunction with secondary bark beetles (Hummel and Agee 2003, Meigs *et al* 2011). Across the PNW, both insects have erupted in multiple outbreaks since 1970, with WSB exceeding MPB in cumulative extent and tree mortality (Meigs *et al* 2015b). Importantly, WSB host forests are more widespread and occur in relatively warmer, more productive locations than MPB host forests in the study area.

2.2. Insect and fire census data

Recent advances in remote sensing of forest dynamics across the PNW (Kennedy *et al* 2010, Meigs *et al* 2015b) provide an unprecedented opportunity to investigate relationships between insect outbreaks and wildfire severity in a retrospective, empirical, census-based framework. We used regional maps of insect and fire effects developed with LandTrendr time series analysis, which is described in detail by Kennedy *et al* (2010). Briefly, we acquired georectified images from the USGS Landsat archive and applied a series of steps—pre-processing (atmospheric correction, cloud masking), processing (temporal segmentation), and analysis (disturbance attribution, regional mosaicking)—to reduce multiple sources of uncertainty and assess trajectories of vegetation change (Kennedy *et al* 2010, Meigs *et al* 2015b).

We accounted for insect activity with LandTrendr-based maps of the cumulative magnitude, cumulative duration (count of years), and time since onset of MPB and WSB outbreaks developed by Meigs *et al* (2015b). These insect maps improve on regional aerial surveys by capturing fine-scale variation of insect impacts (30 m) and constraining maps to locations with durable vegetation change in known insect host forests from 1985 to 2012. The maps also quantify the impacts of MPB and WSB in consistent units of spectral change as seamless mosaics across the PNW study area (including all or part of 35 Landsat satellite scenes (Meigs *et al* 2015b)).

We accounted for burn severity by combining LandTrendr-based regional mosaics of spectral change (Kennedy *et al* 2010) with fire perimeters from a database of large wildland fires in the western US ($\geqslant$400 ha; 1985–2012 (available online: http://mtbs.gov)). We first compiled annual time series (temporally stabilized at the pixel scale) of the normalized burn ratio (NBR; which combines near-infrared and mid-infrared wavelengths of the Landsat TM/

ETM + sensor (Miller and Thode 2007)). Importantly, the Landsat time series are anchored in time near the median date of each scene (generally 1 August), which reduces seasonal variability associated with phenology and sun angles. We then computed the relative differenced normalized burn ratio (RdNBR (Miller and Thode 2007)) in two-year intervals to ensure pre- and post-fire coverage for all pixels within a given fire event. By capturing the relative change in dominant forest vegetation, RdNBR enables the assessment of burn severity across numerous fire events spanning heterogeneous vegetation (Miller and Thode 2007, Cansler and McKenzie 2014) or variable prefire disturbances (including insect outbreaks (Harvey *et al* 2013, Prichard and Kennedy 2014)). Although remotely sensed spectral change indices such as RdNBR have inherent limitations and do not measure very fine-scale fire effects and responses (e.g., tree charring, forest floor combustion, or postfire regeneration (Harvey *et al* 2014b)), they provide the only spatially and temporally consistent metric of burn severity encompassing all fires since 1985. Furthermore, because NBR is at the core of many current fire monitoring protocols (e.g., Key and Benson 2006), our RdNBR-based analysis is directly applicable to contemporary fire research and management.

We conducted a regional insect–fire severity census by focusing on large fire events with the following characteristics: total fire extent $\geqslant$400 ha; $\geqslant$10% of fire extent affected by prefire insect outbreaks (either MPB or WSB); $\geqslant$50% forest cover (30 m resolution (Ohmann *et al* 2012)). Because this forest cover map targets conditions in the year 2000 and classifies some previously burned areas as non-forest, we manually included several fires ($n = 8$) with mapped forest cover <50%. To avoid potential confounding effects, we excluded fire polygons with prior outbreaks of both MPB and WSB ($n = 8$), fires in 1986 with only one full year of prefire insect data ($n = 5$), fires in 2012 without postfire imagery for RdNBR calculations, and one fire classified as a prescribed fire. With these criteria, we refined the total population of forest fires ($n = 425$ spanning 1985–2012) to our final census of large wildfires with prefire insect activity ($n = 81$ spanning 1987–2011; figure 1).

2.3. Statistical analysis

We developed a hierarchical framework to investigate insect effects on burn severity within and among all wildfires in our census (i.e., at the individual and population level). Within each large insect–fire event, we assessed fine-scale (30 m) insect effects on burn severity with sequential autoregression (SAR), a powerful spatial modeling approach advanced recently for wildfire analysis (e.g., Wimberly *et al* 2009, Prichard and Kennedy 2014). SAR incorporates the inherent spatial autocorrelation in dependent and independent variables with a spatial error term (Haining 1993, Wimberly *et al* 2009). This spatial error term also accounts for spatially autocorrelated variables not included explicitly, resulting in more robust inferences than traditional approaches like ordinary least squares regression (Wimberly *et al* 2009, Prichard and Kennedy 2014).

We conducted all analyses in the R statistical environment (R Core Team 2015), constructing SAR models with the spautolm function in the spdep package (Bivand *et al* 2013) in the form:

$$Y = X\beta + \lambda W(Y - X\beta) + \varepsilon,$$

where Y is the vector of the dependent variable, X is the matrix of independent variables, β is the vector of parameters, λ is the autoregressive coefficient, W is the spatial weights matrix, and ε is the uncorrelated error term. W is based on the spatial structure of the dependent and independent variables and is defined by an inverse distance rule that assigns a weight of zero to all pixels outside the focus pixel neighborhood and weights equal to the inverse of the distance within the focus pixel neighborhood. We determined the most parsimonious inverse distance rule of W by selecting the neighborhood that minimized both the Akaike information criterion (AIC) and residual spatial autocorrelation of the SAR model (Moran's I) (Kissling and Carl 2008, de Knegt *et al* 2010). Specifically, we ran SAR models with all dependent and independent variables (described below) across seven neighborhood distances (30–210 m in 30 m increments) for a subset of fires ($n = 15$) spanning the range of conditions in the large fire census. We then calculated AIC and Moran's I of the SAR residual values (moran.test function in Bivand *et al* 2013), which indicated an optimal neighborhood distance of 30 m, consistent with previous SAR burn severity modeling in the study area (Prichard and Kennedy 2014) and typical for a spreading disturbance phenomenon such as fire.

Following these initial steps, we quantified insect effects on subsequent burn severity at the individual fire level (Objective 1) by running a SAR model for each large insect–fire event in the regional census ($n = 81$). We used all 30 m pixels within each fire perimeter to predict burn severity (RdNBR) with the same set of independent variables related to forest fuels and topography (table 1). We included the insect damage and duration variables described above as well as pixel-level estimates of prefire biomass from annual Landsat time series and nearest neighbor imputation with forest inventory data derived for a regional analysis of carbon trajectories (http://lemma.forestry.oregonstate.edu/projects/cmonster).

Although our primary focus was insect effects on burn severity via their impacts on vegetation/fuels, we recognize that topography and weather are fundamental drivers of fire behavior and effects. We thus included a set of five topographic variables (aspect, elevation, slope, and topographic position index at 150 and 450 m; derived from a 30 m digital elevation model) associated with burn severity in the region

Table 1. List of variables used in sequential autoregression modeling of burn severity (RdNBR spectral index) of all fire events affected by prior mountain pine beetle or western spruce budworm. All data were compiled as regional mosaics encompassing the Pacific Northwest study area (figure 1) and processed at 30 m resolution.

Variable	Description	Source
Burn severity (response)	Relative differenced normalized burn ratio (RdNBR, two year interval)	(Miller and Thode 2007)
Prefire insect damage	Cumulative prefire vegetation change due to insect activity from Landsat time series (NBR)	(Meigs *et al* 2015b)
Prefire insect duration	Count of years with prefire insect activity from Landsat time series (y)	(Meigs *et al* 2015b)
Prefire biomass[a]	Prefire tree biomass from imputation mapping (kg ha^{-1})	
Aspect[b]	Cosine transformed aspect (°)	
Elevation[b]	Elevation (m)	
Slope[b]	Slope steepness (%)	
Topographic position index (150 m)[b]	Difference between a pixel's elevation and the mean elevation of pixels within 150 m	
Topographic position index (450 m)[b]	Difference between a pixel's elevation and the mean elevation of pixels within 450 m	

[a] Annual biomass maps were derived from Landsat time series and nearest neighbor imputation with forest inventory data as part of a regional analysis of carbon trajectories (http://lemma.forestry.oregonstate.edu/projects/cmonster).

[b] Topographic variables derived from 30 m digital elevation model.

Table 2. List of population-level predictor variables used to assess drivers of insect effects on burn severity across all fire events affected by prior mountain pine beetle or western spruce budworm.

Variable	Description	Source
Insect type	Mountain pine beetle (bark beetle) or western spruce budworm (defoliator)	(Meigs *et al* 2015b)
Time since outbreak	Time since onset of insect outbreak according to Landsat time series (y)	(Meigs *et al* 2015b)
Area affected by insect	Area of fire extent affected by prior mountain pine beetle or western spruce budworm according to Landsat time series (cumulative %)	(Meigs *et al* 2015b)
Fire size	Extent of fire event (ha)	http://mtbs.gov
Fire season	Day of year of fire ignition	http://mtbs.gov
Interannual drought	Palmer drought severity index (mean June–August PDSI) by fire year and state	http://www.wrcc.dri.edu/wwdt/time/

(Thompson *et al* 2007, Dillon *et al* 2011, Prichard and Kennedy 2014). Unlike these spatially static covariates, fire weather is a dynamic variable that needs to match SAR model resolution in both space and time. Recent advances in the development of gridded meteorological data (e.g., Abatzoglou 2013) have great potential for such analysis but must be combined with accurate fire progression maps to assign fire weather conditions to each pixel for the day it burned. Because consistent fire progression maps are a recent development in North American wildfire monitoring, they are not available for most fires in our census, precluding the use of fire weather covariates in our SAR analyses. Nevertheless, a major strength of SAR is that the spatial error term captures unmeasured but spatially structured variables at the pixel scale (Haining 1993, Wimberly *et al* 2009), including fire weather.

To evaluate key drivers of insect–fire effects at the population level (Objective 2), we assessed the distribution of SAR regression coefficients derived for each fire event with a set of predictor variables not included in the SAR models (table 2). Because the large variability and range of the independent SAR variables precluded direct comparison across model coefficients, we first standardized the coefficients by calculating *z*-scores based on the standard deviation of the mean across all SAR models. We then investigated whether insect effects on burn severity (*z*-scores of prefire insect damage coefficients) varied with insect type (MPB versus WSB), time since outbreak, total area affected by prior insect outbreaks (%), fire size (total extent), fire season (inferred from fire ignition date), or drought condition of each fire year (Palmer drought severity index; PDSI). We derived these predictor variables from the insect and fire census data described above, with the exception of state-level PDSI values (available online: http://www.wrcc.dri.edu/wwdt/time/), which we assigned to each fire, averaging June–August after Heyerdahl *et al* (2008). We estimated time since onset of insect outbreak at the fire event scale as the majority year of first detection in the Landsat-based insect atlas (Meigs *et al* 2015b), recognizing that actual insect activity begins one year before vegetation changes are detected (Meigs *et al* 2015a) and that outbreak initiation varies within a given fire perimeter, depending on outbreak and fire extent. Finally, we computed linear models to assess univariate relationships between these population-level predictors and the insect–fire coefficients.

We evaluated uncertainty in the SAR models for each fire event as well as the distribution of model accuracy across all fire events (table S1). Specifically, we

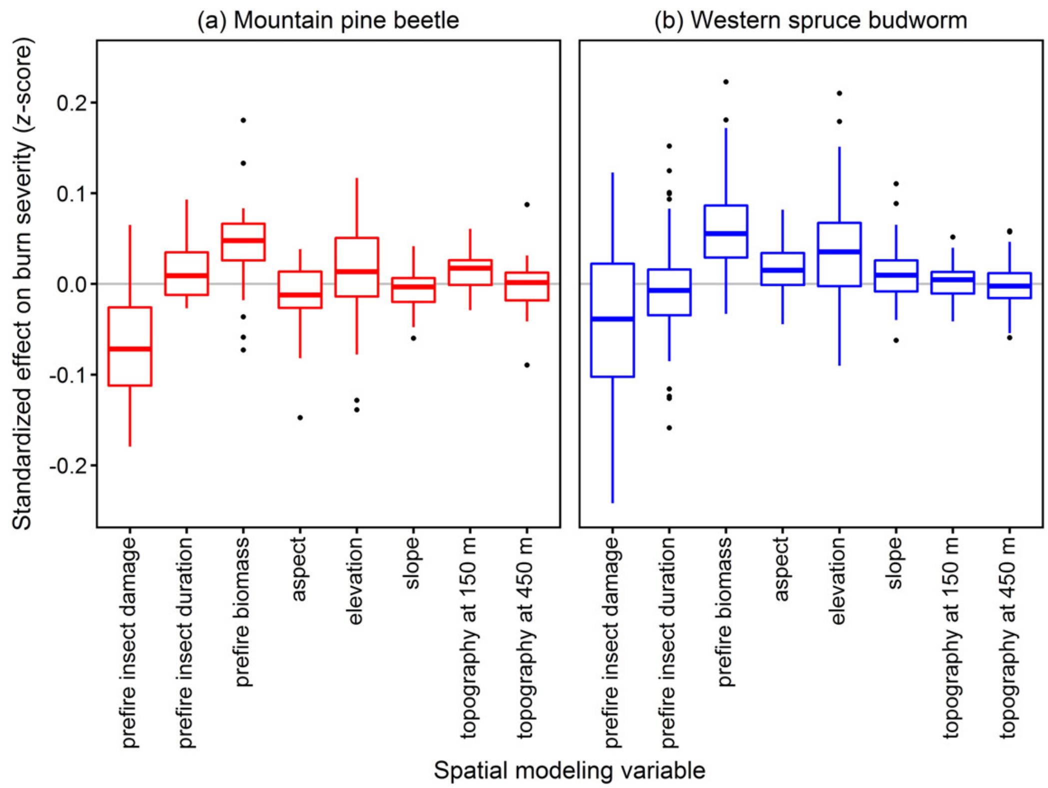

Figure 2. Effect of insect, vegetation, and topography drivers on remotely sensed burn severity across fire events with prior mountain pine beetle (a, $n = 19$) or western spruce budworm (b, $n = 62$). The standardized z-scores from spatial modeling (sequential autoregression; SAR) of each large fire event indicate that burn severity (RdNBR spectral index) is lower in forests with higher cumulative prefire insect damage and lower prefire biomass. The SAR models also account for prefire insect outbreak duration and five potential topographic drivers of severity (table 1). The box and whisker plots show the median as a horizontal line, 25th and 75th percentiles as box edges, 1.5 times the interquartile range as whiskers, and outliers as points.

graphed SAR model coefficients of determination (R^2) by insect type and across the same key predictor variables used in the population-level analysis. Recognizing additional uncertainties inherent to these spatial datasets, we emphasize general patterns across the regional census and the relative effects of insect outbreaks. For example, because the insect outbreak year is offset by one year and uncertain for any given pixel within a fire perimeter, we focus on the relative time since insect outbreak across all fires rather than the specific time lag for a given fire event.

3. Results

Our census of recent insect–fire events across Pacific Northwest forests reveals that, after accounting for prefire biomass and topography, burn severity is generally lower in forests with higher cumulative prefire insect damage (figure 2). Notably, this negative effect of prior insect damage on burn severity is strong enough to emerge without directly accounting for weather conditions at the time of burning.

Following both MPB and WSB outbreaks, burn severity is lower across most time lags (figure 3). The two insects exhibit divergent temporal trajectories, however, revealing differential insect effects on tree mortality, vegetation response, and associated fuel dynamics. Specifically, whereas burn severity decreases with time following MPB outbreaks (figure 3(a)), severity increases with time following WSB outbreaks, eventually recovering to a neutral effect within 20 years (figure 3(b)).

In addition, insect effects on burn severity do not depend on the other population-level predictor variables. Specifically, the insect–fire coefficients are not associated with the proportion of fire extent affected by insects (%), total fire extent (ha), fire season (ignition date), or interannual drought condition (PDSI) for either insect species (figures S1–S4). This lack of association underscores the importance of time-since-outbreak as an emergent predictor of fine-scale insect effects on burn severity (figure 3).

In general, SAR model accuracy is high (MPB mean $R^2 = 0.64$; WSB mean $R^2 = 0.72$; table S1), indicating that the insect, vegetation, and topography variables—as well as the inherent spatial patterning represented by the spatial error term—explain a large proportion of variation in estimated burn severity. In addition, the coefficients of determination are generally evenly distributed across the regional predictor

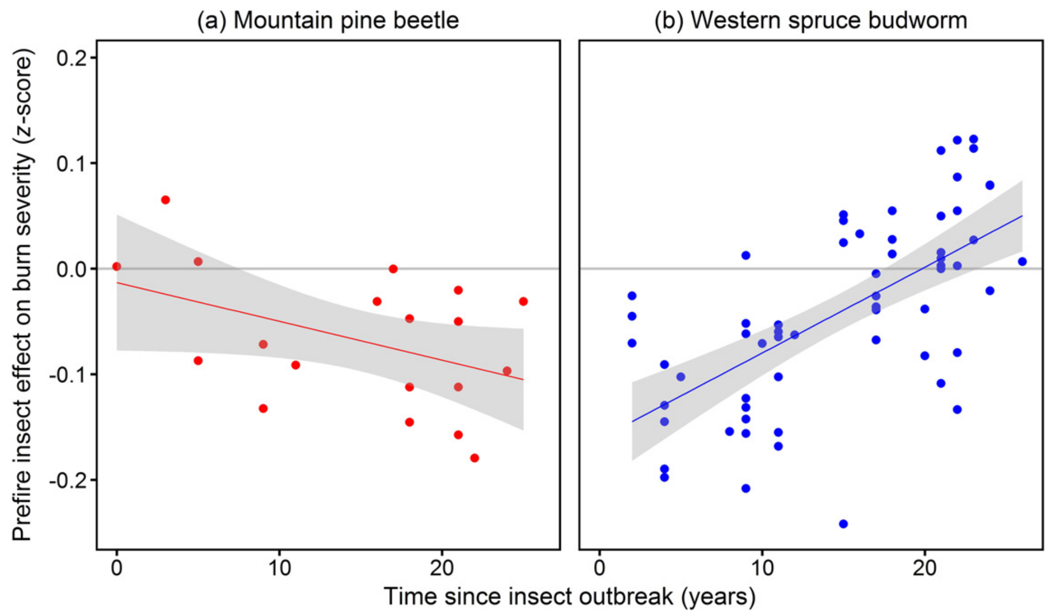

Figure 3. Relationship between prefire insect effect on burn severity (RdNBR spectral index) and years since onset of insect outbreak across fire events (individual points) with prior mountain pine beetle (a) or western spruce budworm (b). For both insects, burn severity is lower in forests with higher prefire insect damage across most time lags, although the two insects exhibit divergent trajectories. Whereas burn severity decreases with time following mountain pine beetle outbreaks ($R^2 = 0.19, P = 0.06, n = 19$), severity increases with time following western spruce budworm outbreaks ($R^2 = 0.40, P < 0.0001, n = 62$), recovering to a neutral effect within 20 years.

variables, which encompass a broad range of insect, fire, and drought conditions (figures S5–S9). Finally, other recognized drivers of burn severity that we did not model explicitly, particularly fire weather and fire-fighting response at the event scale, contribute to the spatially autocorrelated variance captured indirectly by the SAR spatial error term associated with each fire.

4. Discussion

By quantifying the fine-scale effects of insect outbreaks on burn severity within all large insect–fire events across a heterogeneous forest region, this study demonstrates a general pattern of lower burn severity following outbreaks of both bark beetles and defoliators, in contrast to recent findings that burn severity is either unaffected by or weakly positively associated with outbreak severity (e.g., Crickmore 2011, Harvey *et al* 2013, 2014a, 2014b, Prichard and Kennedy 2014). We suggest that higher severity insect outbreaks reduce the abundance of live vegetation susceptible to wildfire while altering vertical and horizontal fuel distributions, particularly as trees defoliate, die, and transition from canopy to surface fuels (Hummel and Agee 2003, Simard *et al* 2011, Hicke *et al* 2012, Cohn *et al* 2014, Harvey *et al* 2014a).

In the case of MPB, this forest thinning effect results in a lasting reduction of fire impacts on residual vegetation (figure 3(a)). Moreover, the continuing decline in post-beetle burn severity indicates that the thinning effect may persist until vegetation and fuel distributions recover to pre-insect conditions. Because there were relatively few fire events within the first few years following MPB outbreak in our census (figure 3(a)), future studies should continue to investigate the transient yet highly flammable red stage of outbreak (Jolly *et al* 2012). Nevertheless, our finding of generally lower burn severity in forests affected by MPB outbreaks—as well as the relative rarity of red-stage fire events in recent decades despite major beetle outbreaks in the study region (Meigs *et al* 2015b)—highlights the need for discretion in forest and fuel management following beetle outbreaks.

In the case of WSB defoliation, lower initial burn severity is consistent with reduced potential fire behavior and effects due to fine-scale canopy thinning and mortality dynamics (Cohn *et al* 2014). The relatively rapid increase of the budworm-fire coefficient with time (figure 3(b)) indicates that the thinning effect on fuel profiles is less persistent for the defoliator (WSB) than for the bark beetle (MPB). In addition to relatively lower per-unit-area tree mortality impacts (Meigs *et al* 2011), WSB affects host forests that are more productive than those affected by MPB in the study region (Meigs *et al* 2015b), leading to more rapid accumulation of live overstory and understory vegetation. Thus, as time elapses following WSB outbreaks, fuel density and connectivity likely increase in multiple strata, including dead surface fuels (Hummel and Agee 2003) and total live biomass, the latter of which is associated with higher burn severity (figure 2). The potentially synergistic budworm-fire effects in older outbreaks have important implications for current forest management in the US Pacific Northwest, where regional WSB outbreaks peaked 25–30 years ago, exceeding recent MPB outbreaks in cumulative extent and impacts (Meigs *et al* 2015b).

Very few studies to date have assessed post-insect burn severity in an empirical, spatially explicit manner, and our census of numerous large fire events occurring up to 26 years following bark beetle and defoliator outbreaks provides a broader context for assessments of specific insect outbreaks, wildfires, locations, and time lags. In so doing, our analysis demonstrates generally negative feedbacks, in comparison with the neutral or relatively transient positive effects quantified with field observations in wildfires occurring up to 15 years following MPB outbreaks in Northern Rocky Mountain forests (Harvey *et al* 2014a, 2014b). In addition, our results differ from the positive MPB-fire feedbacks identified via SAR for the 2006 Tripod Fire Complex in northern Washington (Prichard and Kennedy 2014). Finally, analyses of fire effects following WSB defoliation have been especially rare. The post-budworm temporal trend suggests a neutral effect ca. 18–23 years post-outbreak (figure 3(b)), consistent with the lack of association between budworm damage and the severity of the 2003 B&B Fire Complex in central Oregon (18 years post-outbreak (Crickmore 2011)).

Our core finding that insect outbreaks actually dampen wildfire severity across numerous large insect–fire events has direct applications to natural resources management. Specifically, policies based on the assumption that recent insect outbreaks increase the hazard of subsequent wildfires might be unjustified (Hart *et al* 2015). Furthermore, given that insects also can reduce wildfire likelihood (Lynch and Moorcroft 2008, Meigs *et al* 2015a), these findings illustrate the role that a biotic disturbance (i.e., insect outbreak) can play in limiting both the occurrence and impacts of an abiotic disturbance (i.e., wildfire). Because bark beetle and defoliator effects on burn severity appear to diverge over time, however, forest management strategies should recognize the differential and dynamic effects of each insect on fuel conditions and associated fire potential.

Although our regional census reveals negative insect effects on burn severity across a range of conditions that has not been assessed to date, numerous uncertainties and research questions remain, particularly regarding the mechanistic linkages among insects, fuels, and other known drivers of fire behavior and effects. Specifically, our inference is limited to the locations and years captured by the available spatial datasets, and future studies could investigate insect–fire severity relationships over broader spatiotemporal scales. Future studies also could combine our spatially extensive methods with the temporally rich insights provided by tree ring analysis (e.g., Flower *et al* 2014). Such a fusion approach would enable forest researchers and managers to determine whether recent insect and fire patterns represent a departure from historic disturbance regimes. In addition, because our census uses remotely sensed relative spectral change (RdNBR) as a proxy for fire effects, we cannot directly address causal relationships, fine-scale ecological impacts and responses (e.g., soil heating, tree regeneration), fire behavior (e.g., fire intensity, crowning), or operational fire management (e.g., firefighter safety, suppression tactics) (Thompson *et al* 2007, Harvey *et al* 2014b, Jenkins *et al* 2014, Hart *et al* 2015). Moreover, although the SAR spatial error term indirectly captures the effects of missing variables (Haining 1993, Wimberly *et al* 2009), future studies could explicitly address the effects of other key drivers like fire weather on a subset of events where fine-scale, consistent, and accurate weather and fire progression data are available (e.g., Harvey *et al* 2014b, Prichard and Kennedy 2014). Similarly, topography and climate are known drivers of burn severity in the western US (Dillon *et al* 2011), and future research could further investigate the generally positive association between elevation and burn severity in our SAR modeling (figure 2) and lack of association between drought and insect–fire effects across this census, which spans a range of drought conditions (figure S4). Finally, our analysis is limited to the relatively rare events where wildfires occur within the initial decades following insect outbreaks, and future studies should continue to evaluate the pervasive ecological and economic impacts of these and other disturbance agents separately (e.g., Westerling *et al* 2006, Kurz *et al* 2008, Hicke *et al* 2013).

5. Conclusion

Contrary to common assumptions of positive feedbacks, recent forest insect outbreaks actually dampen subsequent burn severity at multiple time lags across the US Pacific Northwest. Indeed, by altering forest structure and composition from forest stand to regional scales (Raffa *et al* 2008, Flower *et al* 2014, Meigs *et al* 2015b), these native insects contribute to landscape-scale heterogeneity, potentially enhancing forest resistance and resilience to wildfire. Because insect outbreaks do not necessarily increase the severity of subsequent wildfires, we suggest a precautionary approach when designing and implementing forest management policies aimed at reducing wildfire hazard in insect-altered forests.

In addition, by dampening subsequent burn severity, insect outbreaks could buffer rather than exacerbate some fire regime changes expected due to global change (e.g., climate warming, drought, invasive species (Littell *et al* 2010, Ayres *et al* 2014)) and forest response to land use (e.g., fire exclusion, timber harvest, livestock grazing (Hessburg *et al* 2000)). However, each of the disturbances assessed here (bark beetle, defoliator, wildfire) influences more forest area separately than in combination (Meigs *et al* 2015a), and it will remain a high priority to monitor and adaptively manage their individual impacts on forest health and ecosystem services. Given projected increases in the activity of both wildfires and insects (Raffa *et al* 2008, Bentz *et al* 2010, Littell *et al* 2010), the potential for disturbance interactions will continue to increase, as will the potential for ecological surprises like the negative feedbacks apparent in this census.

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Learning to coexist with wildfire

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The impacts of escalating wildfire in many regions — the lives and homes lost, the expense of suppression and the damage to ecosystem services — necessitate a more sustainable coexistence with wildfire. Climate change and continued development on fire-prone landscapes will only compound current problems. Emerging strategies for managing ecosystems and mitigating risks to human communities provide some hope, although greater recognition of their inherent variation and links is crucial. Without a more integrated framework, fire will never operate as a natural ecosystem process, and the impact on society will continue to grow. A more coordinated approach to risk management and land-use planning in these coupled systems is needed.

Fire is unique among the natural hazards that affect human communities and the ecosystems on which we depend[1]. Although humans sometimes intentionally ignite and manage fires, our main focus is on fighting them. For other natural hazards, such as earthquakes, hurricanes and floods, there is much more emphasis on identifying vulnerabilities and adaptations. The 'command and control' approach[2] typically used in fire management neglects the fundamental role that fire regimes have in sustaining biodiversity and key ecosystem services[3–6]. Unless people view and plan for fire as an inevitable and natural process, it will continue to have serious consequences for both social and ecological systems.

Over the past two decades, wildfires around the world have increasingly affected human values (for example, lives, views or sacred environments) and assets (for example, damage to homes or public infrastructure) and ecosystem services (for example, air quality and long-term carbon storage). The growing list of negative outcomes and their financial effects have complex causes and consequences[7]. The natural range of fire sizes and resultant frequencies, timings and intensities — the 'fire regime' — varies greatly among ecosystems, as do the ways in which human activities have altered them (for example, through timber harvesting, fire suppression, urban or agricultural encroachment, novel ignition patterns and invasive species). Not surprisingly, policy strategies to address wildfires often emphasize fuel reduction[8,9]. However, even where strategies recognize interacting cultural, environmental and economic dimensions of wildfire[10–12], few tackle the difficult land-use issue of where and how humans choose to build their communities in the first place. The prospect of widely increasing fire activity with climate change[13] intensifies the need for a new path forward.

Viewing fire-related problems in the context of coupled socioecological systems (SESs)[14], which explicitly recognize links between humans and their natural environments, provides insights into achieving a more sustainable coexistence with wildfire. We have learned a great deal about fire as an essential ecosystem process and the human dimensions of living on fire-prone landscapes. Synthesis of this knowledge through a coupled systems approach can highlight specific vulnerabilities and trade-offs, and facilitate adaptation strategies across widely varying public and private landscapes (Fig. 1). In this Review, we summarize research on fire-prone ecosystems and fire effects on human communities through the lens of SESs, identify links in these coupled systems, and discuss recommendations for greater resilience. We emphasize insights from three regions (Fig. 2) where major fire-related losses have occurred in recent decades: the Mediterranean basin, the western United States and Australia.

Socioecological systems and fire

Sustainable solutions to most environmental problems will be impossible if the links and interdependencies between humans and ecosystems are ignored[14]. In the context of wildfire, the most well-developed SES research that incorporates this coupling concerns climate-change effects on Alaskan boreal forest ecosystems and rural indigenous communities[15,16]. Case studies in rural communities of New Zealand[17] and California[18] also exist. Remarkably, a coupled wildfire SES framework has yet to be adopted for the more densely developed wildland–urban interface (WUI; area in which communities intermix with or abut natural vegetation), where most of the human fatalities, home losses and fire-suppression expenditures occur.

The complexity of how wildfire operates in different ecosystems and how humans interact with it indicates that place-based hazards and risks should be addressed as a coupled SES[16,19]. Reframing the problem to minimize harmful effects as the climate changes and humans increasingly inhabit fire-prone landscapes identifies an integrated set of coupled SES linkages (Fig. 1). Importantly, this allows us to recognize how the geographic context of the coupling itself contributes to impacts and losses of assets throughout the wildfire SES. Local characteristics of the WUI, and the components on either side of it, will largely determine the degree to which fire may be accommodated and how communities will be affected. The spatial scale of the coupling may also be broad in some cases, such as when fires compromise recreation values (for example, trail access, camping facilities or fishing habitat) and water supplies of distant urbanized areas, or when concerns over human exposure to drifting smoke influence management decisions about fires that are burning relatively far away. Although this framing does not intrinsically address connections between fire and global-scale climate change mitigation[13,15,20], it helps to

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reveal geographically relevant solutions for decreasing harmful effects and increasing the positive benefits of fire on the landscape. The institutional complexity that underlies many aspects of this coupled SES framework — agency mandates, property rights, building ordinances, indigenous governance, economic subsidies and political pressures — will also feed into a particular set of solutions, often creating challenging constraints.

Sustainable coexistence with wildfire is both a process and a long-term goal, such that policy, planning and management are adapted and refined through time (Fig. 1). Responsibility must be shared between governments and the people at risk, and the approach integrates building, planning, fuel management, suppression capability, and knowledge of fire and ecosystem dynamics at different scales. Coexistence with wildfire should ultimately allow ecologically appropriate fire regimes to operate on landscapes near and far from the WUI, with relatively low risks to people, property and resources, while also allowing us to enjoy ecosystem services enhanced by fire (for example, habitat maintenance, potential hazard reduction, natural hydrologic functioning, and carbon and nutrient cycling). This outcome should also reduce the costs of fire suppression and the need to put firefighters at risk.

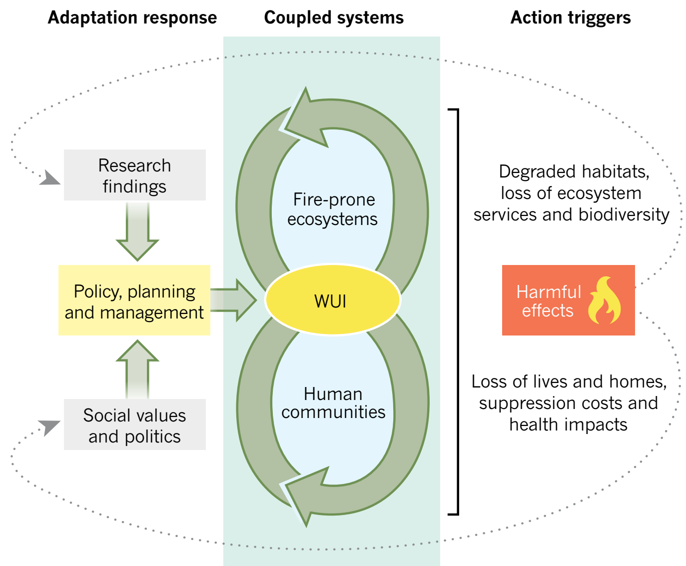


Figure 1 | Links and pathways to resilience in coupled socioecological systems affected by fire. Coexistence with wildfire is strongly influenced by the type of natural fire regimes that operate on a given landscape, and the degree to which communities can reduce exposure and vulnerabilities there. The wildland–urban interface (WUI) is the spatial manifestation of the coupling, and the most proximate scale of exposure and risk mitigation. To learn from and minimize the harmful effects of fire in both the ecosystem and the community, links between systems and scales of interactions must be recognized. Doing so will trigger, through research and in response to changing social values and political context, further adaptation and change in policy, planning and management.

Fire and ecosystems

The role of fire in different ecosystems varies by the degree of current landscape modification, relative to natural or historical patterns and processes. Some regions have large expanses of semi-wilderness where maintenance or restoration of certain fire regimes is crucial to ongoing habitat characteristics or ecosystem services (for example, the western United States and Australia). Here the links between fire characteristics and ensuing ecological effects, or fire 'severity', are often emphasized. Other regions have been so completely altered for various human needs that what is 'natural' is no longer a clear consideration (for example, the Mediterranean basin). Furthermore, climatic controls on fire regimes (for example, frequency of droughts or high-wind events, or length of fire season) tend to dominate in some ecosystems, whereas local controls (for example, topography, fuel loads and ignitions) strongly influence others. Fire resilience is thus context-dependent, varying with the biophysical environment and desired future conditions. Accordingly, our capacity to avoid ecosystem degradation and catastrophic shifts[21] (Fig. 1) depends on the ecosystem in question and how climate change will manifest there.

Mediterranean basin

Mediterranean landscapes are mosaics of various shrublands and oak- and pine-dominated woodlands intermixed with extensive pastures, cultivated lands and abandoned agricultural fields[22]. Despite fire's ecological influence there[4], no reference conditions exist for fire management or restoration, and traditional use of fire for rangeland and game management has strongly influenced historical landscape dynamics[23]. Pronounced biophysical and land-use gradients have recently resulted in contrasting fire and vegetation dynamics. The southern and eastern regions are subject to land over-exploitation and reduction in vegetation cover that increases the risk of desertification and loss of ecosystem services. By contrast, socioeconomic drivers are increasing fire hazards and losses over Mediterranean Europe (northern region) owing to rural depopulation, increased WUI exposure and land-cover changes that are sometimes promoted through afforestation policies[24]. Most shrublands and woodlands in the northern region are becoming dense enough to support climate-driven high-intensity 'crown' fires[22,25].

Wildfire in European Union countries is addressed in national and regional forest policy plans, but consensus on fire and ecosystem management is lacking. In spite of large expenditures, increased preparedness and greater firefighting abilities, extreme fire-weather conditions have caused devastating fires in several Mediterranean countries[26]. A new framework to regulate and promote traditional fire practices, accommodating diverse territorial contexts and operational use of fire, has thus been advocated[27]. Currently limited to local management, prescribed burning is increasing across Europe as a tool that aims to reduce fuel loads and diminish the risk of high-intensity fires[28]. Modest changes to regional and national wildfire policies have therefore included long-term preventive actions, but fire management is still primarily centred on short-term fuel- and suppression-oriented measures[8]. There are concerns over the ecological consequences of recent fire patterns[29], but human-centred fire exclusion generally prevails on most Mediterranean-basin landscapes.

Western United States

Fire management in many western US ecosystems is informed by research on the historical role of fire[30], especially through dendrochronology[31] and landscape reconstructions[32]. Before modern management, different types of fire occurred among vegetation types and maintained important natural structures and functions, with great variation geographically[5,32–35].

In western US forests, high-severity fires that kill overstory trees are typical of cool, high-elevation, subalpine environments[36,37]. Although severe fires may seem catastrophic from a human perspective, in these forests they stimulate vegetation regeneration, promote landscape diversity in terms of vegetation types, provide habitat for many species and sustain other ecosystem services[5]. The many organisms and propagules that may survive the fire, combined with heterogeneity in age, structure and species composition across landscapes, confer resilience against shifts to non-forest types. High-severity fires predominate across about 30% of western US forests, naturally mixing with low-severity fires through time and space across another ~45%[36]. Key regional controls of high-severity fire regimes are extreme drought and high winds[37], and local (for example, topographic) influences on severity patterns can emerge during less dry conditions[38]. Fuels tend to be naturally abundant in these ecosystems, so modern fire suppression may have decreased historical levels of landscape fragmentation, but it has not increased fuel loads[5,39].

By contrast, many dry and mesic, low-elevation and mid-montane forests historically experienced more frequent low-severity fires that maintained relatively open forest structures of fire-resistant

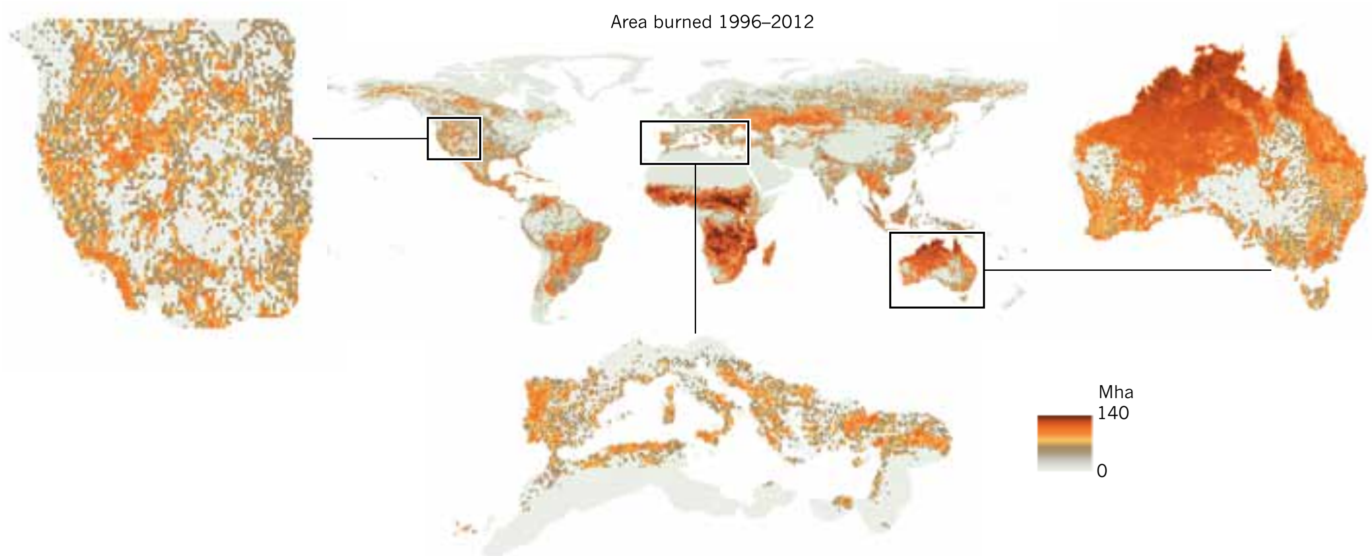


Figure 2 | Area burned patterns and locations of fire-prone regions. The cumulative area burned between 1996 and 2012 in millions of hectares (Mha) per mapped cell. The western US region consists of the 11 western states in the conterminous United States (left), the Mediterranean basin (middle) contains the Mediterranean-climate biomes and the Australian region (right) encompasses the entire continent (see Supplementary Information).

trees[33,34,40], across about 25% of western US forests[36]. Ignition patterns, vegetation structure and fuel amount exert a strong control on regimes of frequent low-severity fire, making them more sensitive to modern human perturbations and also more amenable to fuel-management techniques[33,39–41]. Unlike high-severity fire regimes, timber harvesting and decades of fire suppression in drier forests have lengthened intervals, increased densities of smaller trees and shifted regimes of mostly low-severity fires to include more high-severity, stand-replacing fires. The extent to which this has happened is a topic of debate, raising questions about how widespread 'mixed severity' fire regimes were prehistorically[32,35,42]. Regardless, reducing accumulated fuels in these forests is often a high management priority. Only where such departures from natural fire regimes have led to denser, multilayered, fire-intolerant forests, however, may fuel-reduction treatments restore more characteristic forest structure and function (Box 1).

There is a general consensus regarding the importance of fire, including the need for prescribed burning, to maintain native grasslands and open woodlands. Woody plant encroachment in many ecosystems with sparse tree cover, driven by a lack of fire and replacement of native herbivores, has reduced plant biodiversity, altered vegetation structure and threatened the fauna that depend on those habitats[43,44]. Fire also plays a crucial part in regeneration for some of the vast shrublands of the western United States, especially California's densely urbanized chaparral ecosystems. Similar to high-elevation forests, fire in chaparral is stand-replacing and under strong climatic control (patterns of drought and extreme fire weather)[45], meaning that fuel-reduction efforts have limited effect except in strategic locations[46,47]. Increased fire frequencies, due to abundant human ignitions and non-native grasses that support rapid reburning, threaten to convert many native shrublands to degraded habitats[48]. Invasive grasses also cause very frequent and often large fires across parts of the Great Basin in the western United States[34,49], driven by the 'grass-fire cycle' positive feedback[50] and bringing serious management challenges even to fire-sensitive desert ecosystems[51].

Australia

Fire is ubiquitous in Australian ecosystems, including deserts and tropical forests, and a wide range of fire regimes have been mapped using remote sensing[52]. Annual pulses of relatively intense fire dominate the extensive savannahs of northern Australia, with less frequent, massive fires in the arid zone occurring after above-average rainfall[53]. By contrast, large fires in the temperate forests of the south, although intense, are less extensive and also less regular (decadal occurrence). Biophysical models of fire-regime controls[54] and analysis of trade-offs in fuel characteristics and fire types[52] confirm the primary role of climate, especially the gradient in summer monsoonal precipitation. Thus, fire frequencies tend to vary with latitude, decreasing towards the south and especially the arid interior. Most fire activity on the Australian continent is in grass fuels and of relatively low intensity.

Although palaeo-charcoal deposits document fire's very long history in Australia[55], fine-scale understanding of fire-regime variability through dendrochronology is generally lacking, hindering detailed perspectives on long-term variations in fire regimes. Comprehensive fire management initiatives focus on key environmental objectives, such as biodiversity conservation[20] and emissions reduction[56], as a function of local context. Maintenance of contemporary fire regimes for biodiversity conservation is a priority in most regions, as opposed to the emphasis on restoration that dominates western US approaches.

Australia's productive eucalyptus forests, which can burn at very high intensities and low–moderate frequencies, are largely restricted to southern and eastern edges of the continent. Although these forests are characteristically Australian, their proximity to urbanized areas has probably fed the continent's reputation for high-intensity fire events (see 'Where do people live?'). Debates over the degree to which fuel reduction, whether by mechanical or prescribed fire treatment, can alter the probabilities of high-intensity events[57,58] are similar to those that occur for western US forests.

Prescribed burning in Australia is extensive, but controversial. Fuel reduction burning can partially reduce risk to human life and economic assets, although trade-offs with risks to environmental assets such as biodiversity and ecosystem services are not well understood[3,59]. However, functional responses of species to fire frequencies, sizes, timings and intensities provide a measurable basis for predicting how ecological diversity will respond to management and climate change[60,61].

Resilience and climate change

Ecosystem managers in the three regions covered here (Fig. 2) may have limited ability to alter the numbers, sizes and characteristics of fires occurring in different ecosystems[5,34,39,59]. As already discussed, this is because coarse-scale climatic influences tend to control fire regimes in many ecosystems, especially those that are naturally prone to large and high-severity fires. Except under the most extreme conditions, fire regimes typically constrained by more local-scale controls, such as ignition frequencies and biomass accumulation rates, may respond more strongly to prescribed fire and mechanical fuel reductions. This characterization of two opposing types of fire regimes is, however, a vast over-simplification — idealized end points along a spectrum of variation within and between fire-prone ecosystems[62] — and management prescriptions need to somehow accommodate such complexity. Furthermore, fire-related sensitivities and responses vary among plant and animal species, so fire management for the persistence of one important group of organisms may not favour that of the others.

The potential for climate change to cause 'novel' or 'no analogue' environmental conditions in some ecosystems presents new challenges for management, policy and planning. An obvious goal is to have ongoing fire regimes that minimize the risk of biodiversity loss[59]. Yet, what adaptation responses are appropriate (Fig. 1) if we do not know how future climates and related biophysical processes will differ from the recent past? These uncertainties have resulted in somewhat similar recommendations about fire and ecosystem resilience[63–65]. Heterogeneity in vegetation types, stand structures and successional age classes at all spatial scales and environmental settings is emerging as a strategy for enhancing ecosystem resilience to climate change. This essentially facilitates diverse initial conditions for multiple future ecological trajectories, the most likely and successful of which will not be known for decades. The role of diverse topography in creating microclimate refugia, or 'holdouts'[66], as well as in influencing fire sizes and severity characteristics within large fires[38,67], comprises the physical template for resilience in more mountainous regions. In ecosystems with a recent paucity of burning, fire management that fosters burning under diverse conditions may be useful for achieving this desired heterogeneity and reducing fuel accumulations[41]. Not all fire-generated heterogeneity is ecologically significant, however, so understanding the effects of specific types of 'pyrodiversity' is important[68].

Where do people live?

The WUI is the most proximate spatial manifestation of the coupling in a wildfire SES (Fig. 1). Understanding and addressing vulnerabilities related to the WUI in fire-prone areas is therefore crucial to long-term solutions. As distances between urbanized areas and those protected from development decrease globally[69], a growing WUI will expand the scope of coupling in wildfire SESs worldwide. Negative fire effects that were once due to 'distant' fires (for example, the impacts of smoke on human health) will be increasingly common, making coexistence with wildfire much more challenging.

The current WUI of the western United States is relatively well characterized, with over 60% expansion since 1970 (ref. 70) and about 70% in private ownership[71]. The WUI in this region also predominantly occurs where fire severities are high[70]. Only 14% of private land in the western US WUI is developed, so substantial increases in human exposure to fire may occur as the remaining portions become populated[72]. Although less well characterized, there is growing awareness of expanding WUI in Mediterranean Europe[24,73,74] and Australia[19,75].

Global systematic analyses of human settlement in fire-prone environments is important, but lacking[76]. Coarse-scale characterization of how population densities relate to various fire-prone environments (Fig. 3) provides some insight. Although often characterized as a 'forest fire' problem, western US patterns indicate that highly fire-prone locations with large numbers of people tend to be associated with sparse or no tree cover (for example, the chaparral shrublands of southern California); locations with both high population densities and denser forests exhibit the least area burned (Fig. 3, left). Australia exhibits greater area burned over a broader range of environments, with intermediate population densities being more fire-prone regardless of the amount of forest cover (Fig. 3, middle). The Mediterranean basin is unique because the greatest area burned coincides with the highest population densities (Fig. 3, right), although this too occurs in locations with relatively low forest cover (for example, abandoned agricultural lands[26]).

Acknowledging the diversity of the fire-prone environments and vegetation types where people live is important, because it has implications for the types of fuel treatments that may or may not work to mitigate fire hazards within or near the WUI, and it could help to guide future resource allocation decisions (for example, among vegetation removal, evacuation planning and home vulnerability retrofits)[77]. Awareness of the institutional and social diversity of different human communities is also important, as we discuss in the next section, because it influences their capacity for preparation and mitigation of hazards such as wildfires[18].

Fire and human communities

This section reviews research on how fires affect human communities and is organized by the scale of coupling in a wildfire SES (Fig. 1), ranging from individuals to landscapes. Social science research on wildfire, primarily undertaken in Australia and the United States,

BOX 1

What can 'thinning' of fuels achieve?

There is intense pressure on land-management agencies to reduce fire hazards (for example, rates of spread or flame lengths if a fire occurs). Treatments should be prioritized, however, where they may help to protect communities or reduce fuel loads in the areas that are most likely to experience uncharacteristically severe burns[36,71]. Mechanical fuel-reduction treatments are most suited to certain dry and fire-prone mesic forests[34,39–41,77], where thinning the density of smaller understory trees and removing surface fuel residues (non-merchantable tree tops and limbs) created by these treatments can reduce fire intensities and rates of spread[40]. Not treating the additional surface-fuel by-products can actually increase fire intensity and severity when a wildfire does occur[41].

Some of the most basic trade-offs that limit the widespread use of mechanical fuel reductions involve their economic viability. Often, larger commercial trees will be harvested to help offset operational costs, but this typically generates more surface-fuel residues. Moreover, opening up the overstory canopy and increasing sunlight penetration can increase growth of highly flammable understory vegetation. Controlling this growth response is an ongoing endeavour, the economic feasibility of which is unknown.

Uncertainty about when and where treatments might actually perform as desired must also be considered. Although there are many examples of fuel treatments reducing fire behaviour when conditions are not extreme, recently treated forests can experience a stand-replacing crown fire when wind speeds exceed 30 km h^{-1} and when fuel moisture is low[102]. When the probability of fire occurring in a particular area is relatively low, the odds of a fuel treatment influencing the behaviour of a wildfire there, within the time frame that treatments are effective, is also low[103]. The degree of protection provided by a particular mechanical treatment may thus depend on uncertain parameters (for example, ignition patterns and extreme wind frequencies).

In many areas, ecological restoration and fuel-management goals may be best balanced and accomplished through fire[4,41], which creates natural heterogeneity and provides for fire-dependent species.

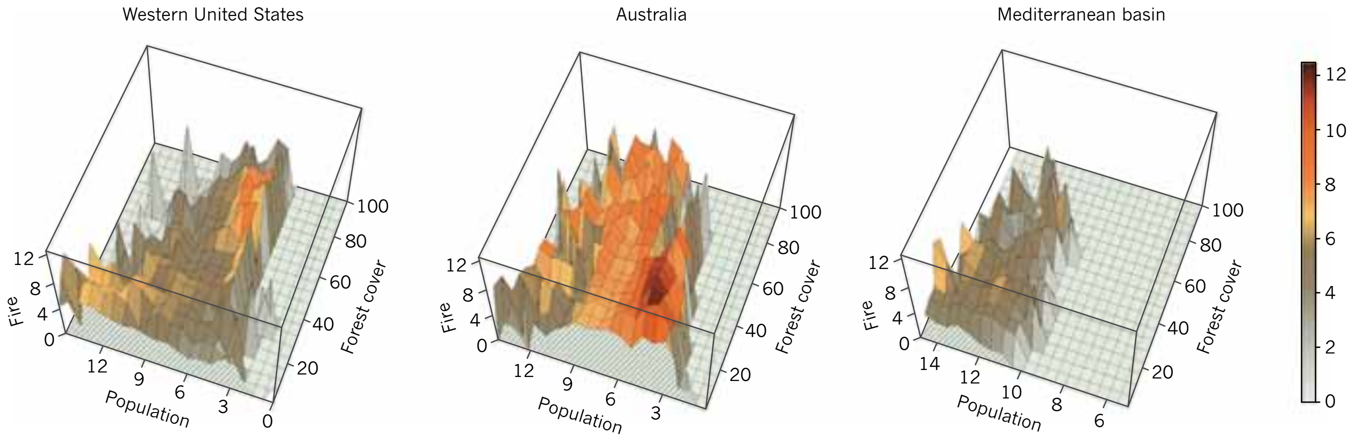


Figure 3 | Relationship between forest cover, population density and area burned in fire-prone regions. Locations with both higher human populations and greater amounts of burning tend not to be consistently characterized by high forest cover. Patterns vary greatly among regions, reflecting the different contexts in which each side of the wildfire socioecological system have intersected. (Data were aggregated from original sources (see Supplementary Information) to 0.25° resolution cells and plotted as density surfaces.) Forest cover is the percentage area covered by trees (>5 m height) per cell in 2000; population is number of people per cell (log transformed) in 2000; and fire is total area burned in hectares per cell (log transformed) between 1996 and 2012. The colour scale for fire is to help differentiate higher peaks in area burned.

is relatively sparse and not easily generalized. Work in the United States emphasizes social acceptance of techniques to mitigate fire risk (for example, fuel reduction on public and private lands) and, more recently, public response during and after fires[78]. In Australia, where many people do not evacuate during fires, risk perception, homeowner preparedness and response during fires, and community safety[79] are key areas of research. We also include studies outside the social sciences that have examined the role of vegetation and fuel treatments linked with losses and the built environment itself.

Risk perception and public response

Public response to wildfire is shaped by numerous factors, such as local context and individual personality and experience, so simple explanations for action or inaction do not exist. For instance, many researchers and managers assume that individuals do not understand fire risk. But US studies show that most people living in high-fire-risk areas understand their exposure, but there is a tenuous link between understanding risk and taking action to mitigate it; whereas recognizing risk might be necessary to consider mitigation, perceived efficacy of mitigation and resource constraints can be more influential[80]. Similarly, whereas around 80% of people in the fire risk areas of Victoria, Australia, know they are in a hazardous area[81], this does not necessarily translate to safer actions. After the devastating 2009 Black Saturday fires in Victoria, most people in high-fire-risk areas were aware of what new fire warnings meant and how to ensure their safety, but few acted on the knowledge when the highest-level warning was issued[81]. A deeper understanding of the influences on preparedness, evacuation decisions and support for hazard mitigation is needed.

Specific cultural and institutional systems affect public response to wildfire, as do psychological and social dynamics. For example, institutional structures in the United States and Australia are quite different, but key social dynamics have many similarities. In both countries, trust is a key factor shaping public support for agencies, whether they provide information or engage in fire-management activities[82]. US studies of public acceptance of prescribed fire reveal that trust in the personnel implementing the burn, along with familiarity with the practice, are associated with higher acceptance levels[83]. In terms of the US public response during fires, evacuating has long been the norm, often with mandatory evacuation orders; until Black Saturday, Australians were urged to either prepare to stay and protect their properties, or to leave early, on the basis that either option was safer than leaving late[79]. Despite this difference, the range of public behaviours in both countries is similar, with some residents leaving early, some staying to defend and a substantial number waiting to see how the situation develops. Furthermore, individual actions do not necessarily reflect a consistent response, as some household members may leave and some stay, while others go back and forth to check on property, animals or those who stay[84]. Although historically 'stay or go' seems to have worked reasonably well in Australia[79], the approach was questioned after the Black Saturday fires, as it was widely seen to have contributed to many of the 173 deaths. However, roughly half the people (around 3,000 households) in the burnt areas seemed to have stayed and defended their properties successfully and about half left, almost as the fire front was approaching. Most were satisfied with their decision and said they would do the same thing again[84]. Most also stated that they would like to be better prepared. The post-fire effort naturally concentrated on fatalities, with official advice after Black Saturday inquiries shifting to leaving early.

When the public response is to evacuate, key elements to success include environmental conditions (especially fire-weather severity), patterns of roads, neighbourhoods and topography. In Australia, public warnings have been based on a fire-weather danger scale, which was revised after Black Saturday to capture the most extreme conditions, along with altered warning messages and advice for these extremes. There is some public understanding of the reclassification, but little evidence of altered behaviour[81] or understanding that weather conditions well below the extreme level are still dangerous. Analogous fire-weather warnings are issued regularly in other parts of the world, but are not standardized and rarely trigger evacuation orders. Similar to many regions, fatalities during evacuations in the Mediterranean basin tend to occur during the most severe weather conditions, when fires have already begun and people choose to evacuate too late[85]; in addition, such extreme events seem to be on the rise[26]. A growing public safety challenge associated with evacuating people from fire-prone communities in mountainous terrain is limited road access. For example, housing densities are increasing in many WUI regions of the western United States without commensurate increases in the road network to support their evacuation[86]. Emergency planning, including preparation of structures and training for those who choose to stay or simply cannot evacuate safely[87], is thus increasingly important to the resilience of many communities in the regions reviewed here.

Structures and surrounding vegetation

To mitigate the risk of structure losses during wildfires, there is increasing evidence from many regions that it is best to focus on the house first and move outward from there[77]. Most structure losses are due to ember attack[88,89], when flaming or smoldering plant material is lofted by winds and blown inside or against the building or adjacent elements, often long before the flaming front arrives. Embers can cause structure ignition by entering through gaps as small as

2 mm[90] or accumulating outside against flammable building (or surrounding) features. Once ember ignition is addressed through structural design or retrofitting, less prevalent modes of structure loss are important, such as radiant heat and flame exposure. To address these, both building design and surrounding vegetation management are normally considered in unison[19], with the balance of these treatments being site specific. Similar to evacuation success, an understanding of the local fire-weather conditions and expected types of fires is required[91]. Hence, the building design strategy is to either consider all possible extremes and the weakest link in the system[88] or to pick a threshold level beyond which the structure may not survive. By relating these to a corresponding fire-weather severity, the occupant has the information for deciding when it is necessary to leave early. As a contingency, egress paths from the building interior to another building or area of minimal fuel could improve safety, but preparation for such a fallback is needed long before a wildfire arrives.

Vegetation reduction is most effective immediately adjacent to structures[88,92–94], as it can eliminate the most immediate sources of combustible material. Vegetation overhanging the structure[91] and ornamental plants[95] have been strongly associated with structure loss. Vegetation clearances more than about 30 m away, however, seem to provide no significant additional benefit in shrubland environments of southern California, even on steep slopes[94], reflecting an important trade-off between hazard reduction and habitat values (for organisms dependent on the vegetation removed). Although these findings may only apply to similar shrubland environments, a similar distance to heavily vegetated areas has also been identified for some forested environments, based on radiant heat exposure to structures[77,96]. In Australia, however, a distance from forest edges of more than 30 m was found to influence home losses[93], indicating that this buffer distance may vary substantially (for example, with fuels, weather and construction types). Another key reason to reduce vegetation near the home is to provide a relatively safe place to engage in structure protection, in case home owners or firefighters are present. It is notable, however, that some species of well-maintained trees (litter removed and high foliar moisture) near the home can actually provide protection, screening embers[19] and acting as a heat sink[96] for an approaching wildfire.

Landscape-scale patterns

Although fuel treatments seem to provide the greatest protection when located near human communities[19,88,93,94,97], landscape-scale characteristics of the WUI itself are important. For this reason, a long-term

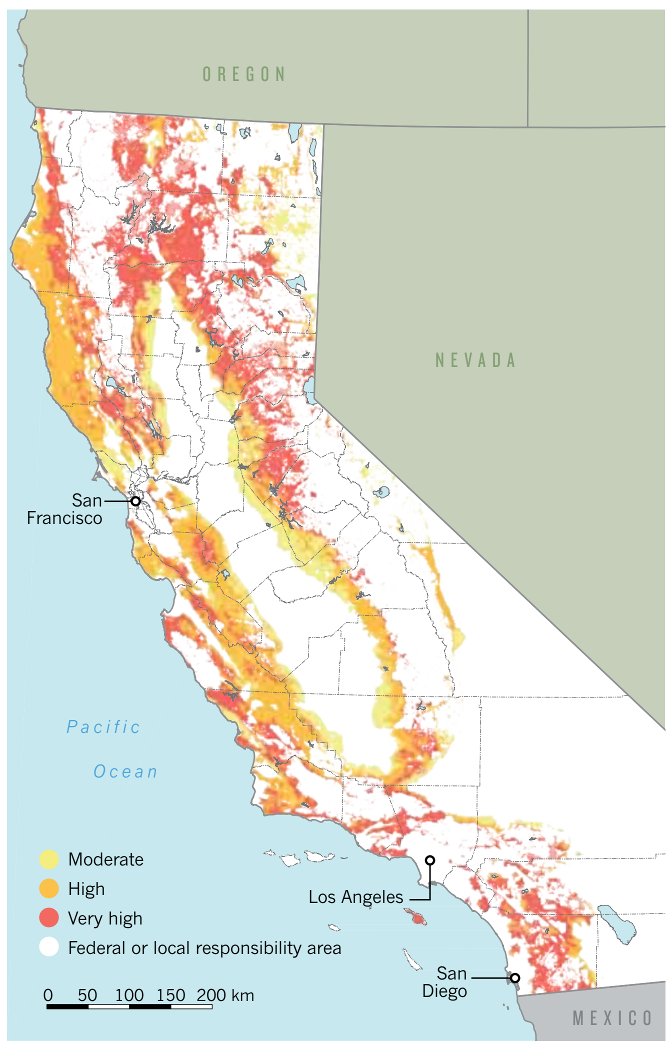
BOX 2

Adaptation measures and fire-hazard mapping

Regardless of the surrounding ecosystem conditions, all communities can better coexist with fire by taking several steps: retrofitting homes against ember attack, effectively managing fuels around homes, developing household and community plans for evacuation compared with stay-and-defend decisions, and participating in risk awareness continuing education. For existing high-hazard wildland–urban interface (WUI) areas, landowners may need to take primary responsibility for pursuing the optimal combination of adaptation measures, based on their local vulnerabilities and wildfire exposure. For development of new communities in high-hazard WUI areas, governments need to take a leadership role in planning. Regardless of responsibility, however, all of these efforts will be guided by better mapping of the fire hazard itself.

The fire hazard severity zone (FHSZ) maps (Box Fig.) of California are an official product of the state Department of Forestry and Fire Protection based on a consistent statewide methodology for estimating potential fire behaviour under a set of relatively dry and high wind conditions. Variables that affect modelled fire behaviour include local topography and potential fuel loads, although weather conditions in the current iteration of maps are not tailored to local extremes. Future updates to the FHSZ methodology will incorporate locally varying wind patterns, better reflecting conditions that cause the worst fire-related losses of lives and homes[45,98].

Fire-resistant residential construction standards are determined by the FHSZ rating of the location in question. In addition, FHSZ classifications must be disclosed at the time of home sales; although this may not deter a sale, it can affect the cost of insuring the home against fire losses. FHSZ maps are thus an incremental but important step towards treating fire like other natural hazards (for example, land-use restrictions associated with flood-plain and earthquake fault maps). Similar mapping methods and codes are produced in Victoria, Australia. Such maps do not explicitly restrict development from occurring — a constraint that should be considered in extremely hazardous locations. Comprehensive approaches should, however, help to better design communities within a complex matrix of both risk and resilience that such maps could reflect spatially. (See Supplementary Information).



approach involving land-use planning offers great potential for reducing wildfire impacts in human communities. A greater understanding is needed concerning building configuration in the WUI and how it relates to risk of losses and fatalities in various environments[73,74]. In some shrubland-dominated landscapes, the arrangement and location of homes have been the most important factors for explaining structure loss: landscape factors such as low housing density, isolated clusters of residential development and long distances to major roads are better predictors of house loss than local factors such as defensible space, fuel or terrain[94,98]. Whether these findings apply to fire-prone landscapes in general or whether there are variations between development patterns and fire regimes needs further research. Although isolated clusters of development and low housing density mean that homes are embedded within, and more exposed to, a matrix of wildland vegetation[19], ignition-prone homes that are closely spaced in neighbourhoods can also facilitate the spread of house-to-house fire, especially during extreme fire weather.

Achieving a sustainable coexistence with wildfire

A coupled SES view of wildfire highlights the variation in each half of the SES, as well as how they come together at the WUI, to create many permutations of hazards and vulnerabilities for both human and natural systems. As such, there will be different thresholds for how harmful effects trigger action before, during and after wildfires, and competing societal pressures will influence the degree to which scientific findings are able to guide adaptive responses (Fig. 1). Despite such complexity, some priorities for future work emerge from the extensive research reviewed here.

Context-specific and place-based approaches will be needed to address many existing and future coupled wildfire SES problems. This is because certain fire regimes are inherently more amenable to management activities than others, and also due to the institutional and social diversity that influences human capacity for mitigating risks to individuals and their communities. It is possible, however, that the permutations mentioned above collapse into characteristic typologies that could inform more systematic analyses. If so, are there mutually resilient combinations that are well matched or somehow compatible? Some fire regimes might dictate the degree to which evacuations should be mandatory or how resources might be allocated (for example, training homeowners to protect homes compared with fuel reduction or structure retrofits). A deeper understanding of the variation, links and scales of causes and effects in coupled wildfire SESs is therefore vital.

Governments have a primary responsibility in the long-term evolution of the WUI and the degree to which it limits or amplifies trans-boundary threats in coupled wildfire SESs, so much greater attention to land-use planning is warranted. Land-use regulations to guide fire-related building codes (Box 2) or restrict development in the most fire-prone locations[2,26,99,100] are clearly important steps that government agencies could take to manage the coupling in a wildfire SES. Agencies have a deeper role, however, in the growth of these trans-boundary threats. For example, the 'safe development paradox' applied to flood and hurricane protection demonstrates that making hazardous areas safer for human habitation in the short term actually increases the potential for severe losses over longer time scales[101]. Given that government agencies around the world have focused on reducing fire hazards (for example, through subsidized fire suppression and/or fuel reduction), much less attention has been paid to the ways in which vulnerable WUI development might have been designed from the start. As further development occurs and the WUI expands, so does the need for increased hazard reduction. A perverse consequence of the typical human reaction to fire — to fight it instead of accommodate it — thus contributes to a deepening of coupled wildfire SES problems.

Strategically addressing threats at the WUI maximizes the potential for both effective risk mitigation within developments and management for sustainable fire regimes over the broader sweep of landscapes. Ultimately, trade-offs and sacrifices must be made to balance these competing demands, but concentration of management effort for risk mitigation in the WUI minimizes the area where adverse effects on environmental assets are likely. Better maps of fire hazards, ecosystem services and climate change effects are thus important for assessing these and other related trade-offs. Addressing all social, economic and environmental assets at risk will necessarily focus on separating those that require exclusion of fire from those where fires of some sort are desirable or inevitable. However, it is unlikely that any planning or management regime will completely exclude fires from vulnerable developments on many landscapes (considerable residual risk to people and property will endure). The capacity for communities to cope with the inevitability of fire, as well as its effects at multiple scales, will therefore be essential.

There is a great deal of research to support better policy, planning and management in all aspects of the coupled wildfire SES problem. Viewing fire as a natural and inevitable hazard should be central to most solutions, so we can anticipate its important positive and negative effects on both human and natural systems. Given that combustion is one of the most basic and ongoing natural processes on Earth, we must continue to learn from our experiences to achieve a sustainable coexistence with wildfire. ■

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Supplementary Information is linked to the online version of the paper at www.nature.com/nature.

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The Holocene record of fire and erosion in the southern Sacramento Mountains and its relation to climate

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Introduction

As highlighted in this issue's Gallery of Geology on page 24, large, severe wildfires have become part of the New Mexico late spring and early summer experience in the last few decades. Such fires have considerable relevance to geomorphologists, as erosion rates in mountainous landscapes are often dramatically increased in the several years after severe fires. Erosion and sediment transport often take place during major debris flows and flash floods (Fig. 1). These events are most commonly triggered by intense thunderstorm rainfall, as in New Mexico's summer monsoon, and very rapid runoff from slopes devoid of vegetation or forest litter. Although water-repellent soils formed by fire effects are often cited as the primary cause of increased runoff, the creation of extensive bare, smooth soil surfaces alone is more than sufficient—for example, consider the erosion that would occur on a plowed, smooth farm field at slope angles of 15–30° or more! The extreme flows that are generated can entrain huge volumes of sediment as they course down slopes and channels. Events of this nature affected a number of small, steep drainages in the Sacramento Mountains southeast of Cloudcroft after the 2002 Peñasco fire. Large quantities of mud- to boulder-sized sediment may be deposited on alluvial fans along the valley margins, and in some cases deep gullies are also cut in the fans. Major damage to roads, buildings, and property resulted in several locations in the Peñasco fire area, as valley-side alluvial fans are common sites for residential and other development.

Along with their importance in understanding geologic hazards and watershed impacts, sediments deposited on alluvial fans by postfire debris flows and floods also provide a means of assessing the timing and spatial distribution of past forest fires, and relations between fire and climate, in particular episodes of severe drought. These sediments are often rich in charcoal fragments from the burned area, which allows radiocarbon dating of fire-related sedimentation events thousands of years into the past, providing an important supplement to the more commonly available tree-ring fire histories. Tree-ring dating has provided a wealth of information on low-severity surface fires that scar trees, but leave them living. Such fires swept through the understory of many southwestern forests every few years to a few decades before European settlement and intensified grazing, logging, and fire suppression in the late 1800s (e.g., Brown et al. 2001). However, tree-ring fire-scar records extend back about 500 yrs at most, and do not provide data on severe fires that kill large stands of trees. Stand-destroying fires can be dated via the ages of living trees that germinated after fire, but this reveals the last such fire only, and again is limited to about the last 500 yrs. Therefore, alluvial sediments can greatly extend fire histories, albeit with greater uncertainty in ages. Climatic change on time scales of a thousand years or more has strongly affected Earth environments over the Holocene Epoch, the ~12,000 yrs since the last episode of continental glaciation. Thus, the sensitivity of fire activity to climate change over such time scales is of great interest. It is also critical to understanding the potential impacts of future droughts on New Mexico's mountain forests, given that predicted warming over the next century has no precedent on the short time scales covered by tree-ring fire chronologies.

FIGURE 1—Deposits of a debris flow from a tributary basin of lower Cox Canyon in the Sacramento Mountains. The tributary basin was severely burned in the 2002 Peñasco fire. Debris-flow deposits partially filled and dammed the main valley arroyo at this location (the arroyo wall is visible across the center of the photo). The surface deposits lack mud because of reworking by later flood flows.

Fire and alluvial history in the Sacramento Mountains

South of Cloudcroft, the Sacramento Mountains are essentially a broad eastward-dipping cuesta, where the range crest and ridges on the eastern slope are capped by resistant limestone and dolomite of the Permian San Andres Formation. Below the San Andres Formation, the highly erodible Permian Yeso Formation forms slopes that have contributed large volumes of fine-grained Holocene alluvium to valleys. Abundant exposures of these alluvial sediments are present in both deep main valley arroyos that predate the Peñasco fire (Fig. 2), and in gullies cut in alluvial fans by recent postfire debris flows and floods (Fig. 1). Deposits of modern postfire events helped us to define criteria for recognition of fire-related sediments in the Holocene alluvium. Conifer forests of the Sacramentos range from spruce and fir near the range crest, through ponderosa pine and mixed conifers at middle elevations, to piñon-juniper stands near the lower forest border.

We focused our investigations along the valleys of the Rio Peñasco and lower Cox Canyon in the eastern Sacramentos (Frechette and Meyer 2009), and in Caballero Canyon west of the range crest (New 2007). Fire-related deposits dating to as much as 8000 yrs ago were discovered, interbedded with deposits with no clear evidence of an origin in a burned area. The most active period of fire-related sedimentation occurred from about 6000 to 4000 yrs ago, as is highlighted in Figure 3. Although it does not stand out markedly as a dominant peak in the fire-related sedimentation curve, this interval saw fans build rapidly with several meters of accumulated sediment including thick, charcoal-rich debris-flow deposits (Fig. 2, inset). This evidence for severe fires is consistent with a generally warmer middle Holocene climate, characterized by widespread and persistent drought conditions in the Southwest and in the interior western United States in general (Buck and Monger 1999; Shuman et al. 2009). However, it may also reflect a higher variability in precipitation and (or)

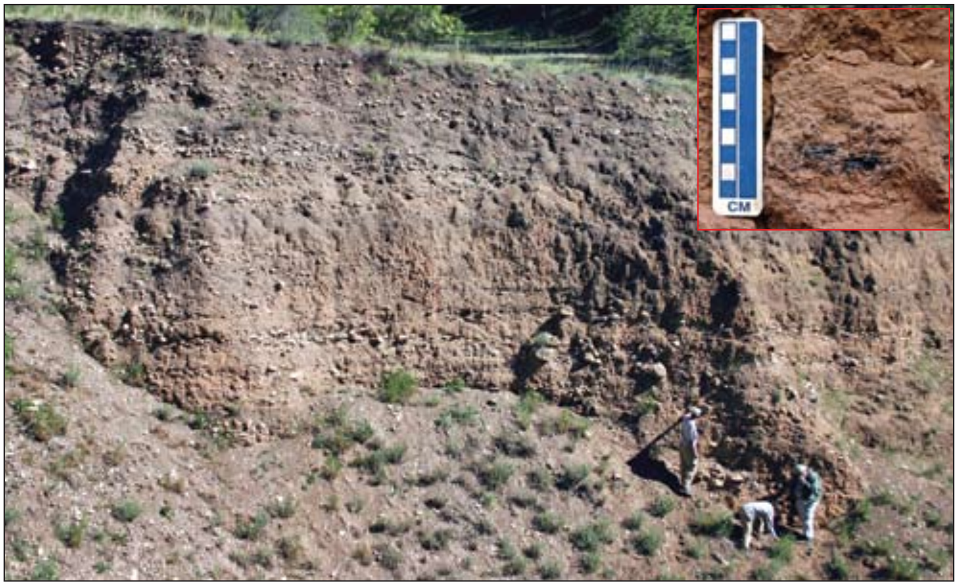

FIGURE 2—More than 10 m (32 ft) of sediment in the toe of an alluvial fan is exposed in this main valley arroyo along the Rio Peñasco. The fan deposits range in age from about 8000 to 650 yrs before present. Darker layers represent organic-enriched soils that developed during times of slow sediment accumulation. The middle part of the section with little soil development starting above the persons' heads dates to about 6000–4000 yrs ago, when rapid fire-related aggradation occurred. Inset shows abundant charcoal fragments in a muddy, poorly sorted fire-related debris-flow deposit from this section.

temperatures at this time (e.g., Asmerom et al. 2007), which would allow forests to grow more dense during wetter intervals that minimize the occurrence of surface fires. Severe crown fires would then be more likely during major droughts.

Evidence for severe fires in charcoal-rich debris flows in the Sacramento Mountains became substantially less common after 4,000 yrs ago, probably because of the climatic cooling associated with declining summer solar radiation in the Northern Hemisphere, and advances of mountain glaciers in the western USA and elsewhere known as the Neoglacial. However, episodic fire-related sedimentation punctuated this time interval, most notably around 650 yrs ago (Fig. 3). This peak in fire-related sedimentation is associated with the "Great Pueblo Drought" of AD 1276–1297, a period of persistent, severely dry conditions that was noted in some of the earliest climate reconstructions using tree rings (Douglass 1929). This megadrought centered on the Four Corners area but affected a much larger region of the Southwest (Cook et al. 2007). It came at the end of a period of generally warmer climate in the western USA from about AD 900–1300 known as the Medieval Climatic Anomaly (Fig. 3). There is also substantial evidence from tree rings, lake levels, and other paleoclimatic records of unusually large fluctuations between wet and dry conditions during this time, which could again promote dense growth of conifer stands followed by extensive severe fires. Prior work in Yellowstone National Park and in central Idaho has also shown the Medieval Climatic Anomaly to be a time of major fire-related debris-flow activity and building of alluvial fans (Meyer et al. 1995; Pierce et al. 2004), illustrating the widespread effects of drought in this interval. The most prominent episode of postfire debris flows in these areas, however, is correlated with the AD 1140–1162 megadrought, considered to be the worst in North America in the last two millennia (Cook et al. 2007). Fire-related deposits dating to the time of the mid-1100s megadrought are found in the Sacramento Mountains (Fig. 3), but are less common than those emplaced after fires in the late AD 1200s. Overall, the occurrence of major droughts, severe fires, and debris flows across the interior western USA is consistent with an inference of generally higher temperatures during the Medieval Climatic Anomaly.

The effects of warmer climate and severe drought notwithstanding, fire-related sedimentation in the Sacramentos over most of the Holocene is characterized by relatively small and sporadic events, consistent with a regime of low-severity surface fires, with the occasional patch of higher-severity crown fires. Likewise, tree-ring fire-scar records in the Sacramentos show that frequent, low-severity surface burns dominated fire activity throughout the range of forest types, from the beginning of the record about AD 1580 to the late 1800s (Brown et al. 2001). This period falls within the generally cooler and effectively moister climate of the Little Ice Age (e.g., Cook et al. 2007) (Fig. 3). A regime of low-severity fires makes sense during the Little Ice Age, as reduced temperatures and evapotranspiration would promote grass growth to fuel surface fires, as well as limit the potential for reducing moisture levels in the forest canopy to the point where extensive, high-severity crown fires could occur. Since the late 1800s, fire suppression and other land uses greatly limited surface fire activity, and the resulting denser forest stands—along with a warming climate, especially in the last several decades—have created conditions that are ripe for extensive stand-destroying crown fires.

An interesting aspect of Sacramento Mountains forests are the large, dense stands of Gambel oak that are especially prominent on the upper western slopes of the range. Professor Thomas Swetnam of the University of Arizona Laboratory of Tree-ring Research has hypothesized that these brushy patches represent areas where severe crown fires destroyed conifer stands and Gambel oak recolonized. Identification of charcoal fragments found in soils under oak brush in Caballero Canyon suggest that some past fires at these sites were indeed in conifer stands (New 2007), but further work is necessary to test this hypothesis.

Another question we considered in our Sacramento Mountains work is whether the deep arroyos found along many valley reaches may, at least in part, stem from land uses such as railroad logging and intensive grazing, especially along the eastern flank of the range. The main valley arroyos predate modern severe wildfires, but the timing of their initiation is largely unknown. At lower elevations in the Southwest and on the Colorado Plateau, most valley-filling alluvial sequences show clear evidence of alternating episodes of arroyo cutting and valley filling, especially in the last 4000 yrs (e.g., Waters and Haynes 2001; Love 1983). We found no clear evidence of past arroyo incision in the exposures examined for fire-related deposits in the eastern Sacramento Mountains. We also found no place where gravelly deposits filled a deep paleochannel in finer-grained valley fill, as occurred in modern times after the Peñasco fire (Fig. 1). Such a relationship should have been obvious despite imperfect exposures along many present arroyos. Some evidence exists for paleo-arroyo cutting in Caballero Canyon on the west slope, however. The lack of clear precedence for modern arroyos in the eastern Sacramentos suggests that 19–20th century land use may have been an important factor. However, we again need to investigate this question further, including dating the initiation of the present episode of arroyo cutting, and conducting a focused search for paleochannels.

Implications of long-term fire-climate relations

As in other studies of Holocene fire-climate relations (e.g., Pierce et al. 2004), our work in the Sacramento Mountains shows that fire behavior is highly sensitive to relatively modest climatic change. With the high probability of increased temperatures and episodes of severe drought over the next century, catastrophic wildfires and their accompanying debris flows and flash floods will become even more likely in New Mexico's mountain forests. Thinning of over-dense stands that have resulted from fire suppression, especially in ponderosa pine forests, can reduce

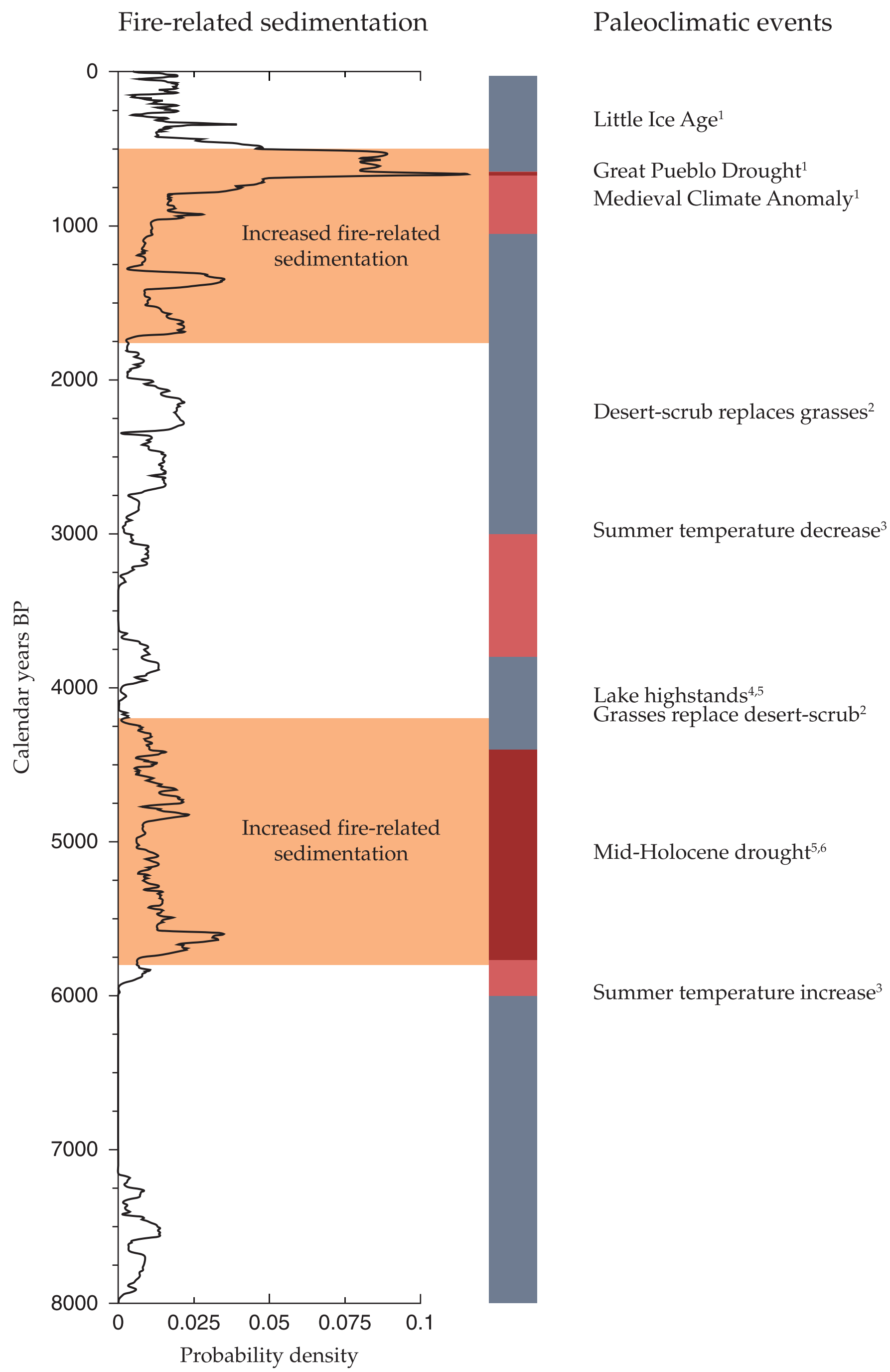

FIGURE 3—Record of fire-related sedimentation in the Sacramento Mountains compared to simplified Holocene paleoclimatic events from the western USA. The curve at left shows the relative probability of fire-related sedimentation, and was constructed by summing probability distributions for calibrated radiocarbon ages on individual fire-related sedimentation events. Periods of increased fire-related sedimentation are inferred based on both radiocarbon age distributions and the nature of stratigraphic evidence for fires. References for paleoclimatic information: [1]Cook et al. 2007, [2]Buck and Monger 1999, [3]Viau et al. 2006, [4]Castiglia and Fawcett 2006, [5]Shuman et al. 2009, and [6]Davis and Shafer 1992.

the impact of wildfires, but this is a very large and expensive task that can have impacts of its own, for example, in roadless areas, and there is limited commercial value to the small-diameter trees that must be removed. Public awareness of the hazards that stem directly from development in fire-prone forests, as well as those from postfire debris flows and flash floods on alluvial fans, is key to reducing risks to life and property.

Acknowledgments

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Recent Forest Insect Outbreaks and Fire Risk in Colorado Forests: A Brief Synthesis of Relevant Research

Photo: Jeff Hicke

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Introduction

Extensive outbreaks of tree-killing insects are occurring in many parts of the West, including Colorado. In combination with recent high-intensity forest fires, these insect outbreaks are raising concerns about the health of our forests and our ability to deal with these issues. The visual impact of a high-severity bark beetle outbreak or fire may give the impression that we are in a crisis situation and that we must take dramatic steps to deal with this "emergency". However, recent scientific research on the ecology of forest disturbances, by scientists in Colorado and elsewhere, leads us to interpret these recent events in a much more nuanced manner.

We believe that the responses to insect outbreaks and fires will not produce beneficial results unless those responses are consistent with the basic ecology of the affected forest ecosystems. Hence, we have written this brief synthesis of the current state of knowledge about forest insects and fires in Colorado to help inform effective management options. Our emphasis is on the ecological aspects of the insect outbreaks now affecting thousands of acres in the state. We do not deal extensively with other dimensions of insect outbreaks and fires, although we acknowledge that aesthetics, economics, wildlife management, recreation, watersheds, and fuels are all important considerations in making decisions about forest policy and management.

This report is organized into two sections. The first section addresses nine key questions about the basic ecology of insect outbreaks in Colorado forests; the second section evaluates six possible treatment options. We do not advocate any particular policy or management treatment, but instead describe the likely ecological effects of each potential option. We also provide a very brief synopsis of each answer or treatment option in italics at the beginning of each section. Our hope is that the information summarized here will aid managers and policy-makers in making decisions about how to deal (or not deal) with different kinds of insect outbreaks occurring in different contexts. As will become clear below, not all forests and not all insects are alike. The authors all have training and research experience in forest ecology or hydrology, both in Colorado and elsewhere.

Questions about the Basic Ecology of Forest Insects

Question #1: Which insects are killing trees across large areas in Colorado?

Summary: *The major insects killing trees in Colorado today include bark beetles (mountain pine beetles, spruce beetles, and piñon ips beetles) and defoliators (notably western spruce budworm). All of these insects are native to Colorado and have co-existed with their host tree species for thousands of years (Figure 1).*

Two major groups of insects have been responsible for killing large numbers of trees over extensive areas under outbreak conditions in Colorado: bark beetles and defoliators (Schmid and Mata 1996). Adult bark beetles bore through a tree trunk and lay eggs within the inner bark. The eggs hatch and the beetle larvae eat the inner bark, killing the tree. After the larvae mature, the new adults fly to new trees, bore through the bark, and continue the cycle. There are several species of bark beetles, each of which feeds on one or several species of trees. For example, the mountain pine beetle feeds on ponderosa, lodgepole, and limber pine; the spruce beetle feeds on Engelmann spruce; and the piñon Ips beetle feeds on piñon pine.

Defoliators are a group of insects having a life cycle very different from the bark beetles. The adult defoliators are tiny moths that lay their eggs in the buds of trees. The eggs hatch into caterpillars that feed on the emerging new leaves in spring and early summer. When numerous, the caterpillars may eliminate essentially all of a tree's annual production of leaves or needles. Small trees, or trees that are stressed by other factors, may die after a few years of defoliation, though usually most of the trees in a stand survive the outbreak of defoliators. The most important defoliator in Colorado forests is the western spruce budworm which feeds on Douglas-fir, white fir, subalpine fir, and spruce. Douglas-fir tussock moth is a less frequent but locally significant defoliator of Douglas-fir, white fir, and spruce. Aspen trees may be defoliated by tent caterpillars and large aspen tortrix.

These insects are usually present in a forest in very low numbers, killing only the occasional weak tree. Such low numbers are referred to as

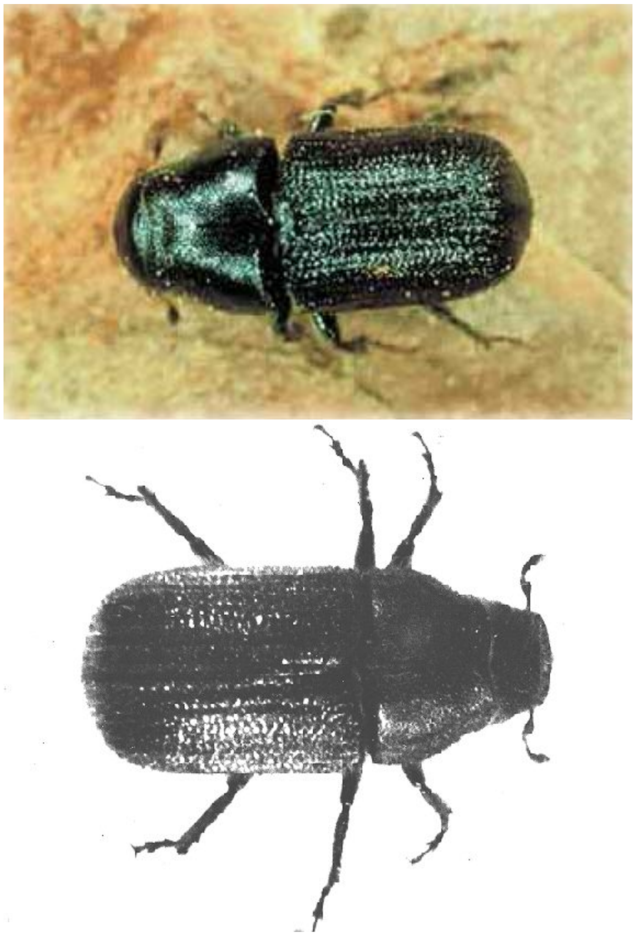

Figure 1. The major insects killing trees in Colorado today include several species of bark beetles (such as the mountain pine beetle above and the spruce beetle below) as well as various species of defoliators. All are native to Colorado and have co-existed with their host tree species for thousands of years. Mountain pine beetle photo from Colorado State Educational Extension Service. Spruce beetle photo from USDA National Agricultural Library.

"endemic" populations. Periodically, however, insect populations grow rapidly and kill large numbers of trees over large areas. This is referred to as an "outbreak" or "epidemic" population. Outbreaks of all of the insect species described above have occurred recently, and have caused extensive mortality events in their respective tree hosts. It is important to note, however, that the trees of Colorado and the Rocky Mountains have coexisted with these native bark beetles and defoliators for thousands of years.

Question #2: Are the insect outbreaks now occurring in Colorado unprecedented in the ecological history of this region, or are they "natural" events similar to outbreaks that occurred in the past?

Summary: *There is no evidence to support the idea that current levels of bark beetle or defoliator activity are unnaturally high. Similar outbreaks have occurred in the past (Figure 2).*

There is no evidence to support the idea that current levels of bark beetle or defoliator activity in Colorado's lodgepole pine and spruce-fir forests are unnaturally high. The outbreaks now taking place in Colorado are similar in intensity and ecological effects to previously documented outbreaks in the Rocky Mountains. For example, mountain pine beetle outbreaks killed millions of lodgepole pine trees over thousands of square miles in the Cascade and Rocky Mountains during the 1960s, 1970s, and early 1980s (Lynch 2006; chapter 4); and a spruce beetle outbreak in the 1940s killed spruce trees over much of the White River Plateau in western Colorado. Historic photos and tree-ring evidence also document extensive insect outbreaks prior to the 20th century (Baker and Veblen 1990, Veblen et al. 1991, Veblen et al. 1994, Swetnam and Lynch 1998, Eisenhart and Veblen 2000, Veblen and Donnegan 2006). Thus, insect outbreaks are a natural occurrence in almost all of the different kinds of forests in Colorado. Outbreaks do not occur very frequently; the time interval between successive outbreaks in any given area is usually measured in decades. Nevertheless, outbreaks can be expected periodically in almost any place in the state where forests are found.

It is true that bark beetle outbreaks are now

Figure 2. The insect outbreaks now occurring in Colorado are similar in extent and severity to outbreaks of the past. For example, spruce beetles killed millions of trees over thousands of acres in the White River National Forest in the late 1940s and early 1950s. The dead trees (above) are still visible. (Photo by T. T. Veblen).

occurring in parts of Colorado where such extensive insect activity had not been seen at any time during the previous hundred years (e.g., in the Fraser Valley). However, in the absence of tree-ring reconstructions or other spatially detailed information on historical mountain pine beetle outbreaks in Colorado, it is not known if similar outbreaks occurred in the same locations or habitats in the past several centuries. Given the naturally long intervals between recurrent bark beetle outbreaks in Rocky Mountain forests, there is nothing unusual about a hundred-year period of low activity followed by an extensive outbreak. It also is true that mountain pine beetles now are killing trees at unusually high elevations (Wayne Shepperd, personal communication). This may be a significant departure from previous outbreaks. However, it is difficult to know if the current insect activity at high elevations is truly unprecedented, given the lack of data on precise spatial patterns of prehistoric outbreaks. The occurrence of outbreaks today at high elevations, where the insects ordinarily are limited by cold temperatures, is not surprising considering the warm temperatures we have experienced during the past decade, as we discuss in the next question.

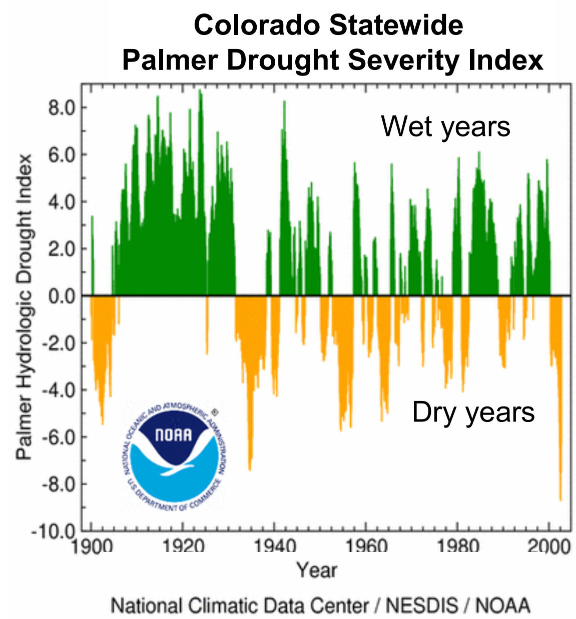

Figure 3. The reason why bark beetle outbreaks are so extensive and severe in Colorado today is because of four interacting ecological factors: (i) long-term drought, as shown above, that stresses trees; (ii) warm summers and (iii) warm winters, which enhance beetle growth and survival; and (iv) abundant food sources (trees) for beetles.

Question #3: Why are the insect outbreaks so severe and so widespread at this time?

<u>Summary</u>: *The ecological factors that control insect populations are complex. Recent bark beetle outbreaks in Colorado probably are a result of four interacting factors: (i) long-term drought, which stresses trees and makes them more vulnerable to insects, (ii) warm summers, which further stress the trees and may accelerate growth of the insects, (iii) warm winters, which enhance survival of insect larvae, and (iv) abundant food (trees) for the insects in Colorado's extensive and often dense forests (Figure 3).*

The factors that control the initiation, spread, and termination of insect outbreaks are complex, and involve a combination of climatic conditions and characteristics of forest stand structure. The relative importance of climate vs. stand structure in any given outbreak is not fully worked out, and in fact may vary from place to place and among the various insect and tree species. Nevertheless, the following is what we know about the interacting influences of drought, temperature, and stand conditions on insect outbreaks.

Evidence from observational, laboratory, and modeling studies indicates that climate is a major controlling factor of bark beetle outbreaks (Bentz et al. 1991, Logan et al. 2003, Carroll et al. 2004, Breshears et al. 2005). The initiation of a bark beetle outbreak is often associated with drought. It is thought that the dry conditions stress the trees and make them less able to defend themselves against the beetles (Carroll et al. 2004). For some insects, the end of the drought usually means the end of the outbreak. However, with mountain pine beetles and spruce beetles, once the beetles have killed a large number of trees and produced abundant offspring, their numbers may become so great that they can overwhelm even healthy trees. If this point is reached, continued drought is not so important: the beetle population continues to grow until it is checked either by a prolonged period of bitter cold weather or until they exhaust their food supply. Low temperatures (around –40 degrees F for about a week), especially in late fall or early spring, may kill the beetle larvae in the trunks of the trees, and thereby terminate the outbreak at any stage in its development.

A warming climate during the last 100 years, particularly in the last few decades, also appears to have played a role in driving recent insect outbreaks. Higher temperatures and a longer frost-free period subject the trees to additional water stress, and may accelerate the growth and development of the beetle larvae. The warming trend of the past few decades (Westerling et al. 2006) may have contributed to the current outbreak of mountain pine beetle in Colorado, as well as recent outbreaks that have occurred outside of Colorado in historically marginal environments for bark beetles, such as at the northern extent of their range in Canada (Carroll et al. 2004) or in high elevations of the northern Rockies (Logan and Powell 2001, Hicke et al. 2006). Furthermore, changing climate conditions are thought to have been responsible for a very severe mortality event in the piñon trees of southern Colorado and adjacent states. Between 2002 and 2004, extensive piñon mortality occurred during a severe drought and an accompanying outbreak of Ips bark beetle (Breshears et al. 2005). Although a more intense drought actually occurred in the 1950s, piñon mortality was far more severe and widespread in 2002 - 2004, apparently because the unusually warm conditions that accompanied the recent drought put additional stress on the trees and allowed more extensive outbreaks of the piñon Ips beetle. Breshears et al. (2005) documented elevated maximum and minimum temperatures at numerous weather stations throughout the Four Corners region during the past decade.

Stand structure also is important in bark beetle outbreaks. The inner bark of very small trees usually is not thick enough to support beetle larvae, and consequently the adult beetles tend to select larger trees to lay their eggs. The minimum tree size for the mountain pine beetle is around four to five inches diameter, but is different for other beetle species (Furniss and Carolin 1977). Thus, stands with large trees are more susceptible to bark beetle outbreaks than are stands with smaller trees. In addition, trees in old or dense stands may be less vigorous and therefore more susceptible to beetles than trees in young or less dense stands, because of competition among trees for limited water and nutrients (Shore and Safranyik 1992). At the landscape scale, if most of the forest is of similar age and has a structure conducive to bark beetle outbreaks, it is likely that outbreaks will be widespread -- if climate conditions are also appropriate. Although fire suppression in the lodgepole pine zone probably reduced opportunities for establishment of young stands since about 1940, young stands have established after timber harvests during this period. The main influence on lodgepole pine age structure in Colorado, however, is widespread burning in the late 1800s that resulted in extensive cohorts of relatively similar age that now are entering a stage that is susceptible to bark beetle outbreaks.

So, why have recent insect outbreaks been so extensive and severe in Colorado? We believe the answer is as follows. The past decade has brought severe drought to many parts of the state (Pielke et al. 2005, Figure 3), accompanied by relatively warm temperatures in both summer and winter (Westerling et al. 2006). The combination of drought and hot summers probably stressed the trees and made them more susceptible to bark beetles; the warm summers may have accelerated the growth and reproduction of some bark beetle species (e.g., spruce beetles and piñon Ips); and the mild winters produced very little mortality of beetle larvae. These climatic conditions probably are the major reason why insect outbreaks have gotten started in many different regions of the state. Once the outbreaks began, the beetles found an abundant food supply (trees) in most of Colorado's forests. Many stands are densely stocked with trees because they have not been disturbed for a very long time by fire, insects, or harvest. All of these factors have combined to create a "perfect storm" of bark beetle outbreaks across much of Colorado.

Question #4: Are the dense forest stands that we see in Colorado today the unnatural consequence of past fire suppression and lack of timber harvesting?

Summary: *The answer to this question depends on the type of forest and its geographic location, as explained below. For example, high density in lodgepole pine and spruce-fir forests is not related to fire suppression; it is simply a natural* *ecological feature of these subalpine forests. It is important to note that not all forests have been affected in the same way by past fire suppression and other human activities (Figure 4).*

Many Colorado forests are very dense, but not all dense forest stands are the unnatural consequence of past fire suppression and lack of timber harvesting. For example, high tree density is a natural condition of most high-elevation forests, including lodgepole pine and spruce-fir. On the other hand, some ponderosa pine forests (but not all) do have unnaturally high tree densities -- higher than would have been seen prior to Euro-American settlement of the region. Thus, it is necessary to distinguish among different forest types in Colorado and elsewhere in the West when considering the effects of past fire suppression and timber harvest (or lack thereof) on current stand density.

Ponderosa Pine Forests Summary: *Tree densities have increased significantly in dry ponderosa pine forests in parts of Arizona, New Mexico, and southern Colorado, largely as a result of fire suppression and other human activities. Ponderosa pine in northern Colorado has been affected to a lesser extent, because fires were historically less frequent in this region than farther south, and the historical landscape was a mosaic of dense and open stands. The proportion of dense vs. open stands is greater in some areas of the Front Range today than historically, in part because of fire suppression, but also because of recovery from 19th century disturbances and because 20th century climate was generally favorable for tree growth.*

Dry ponderosa pine forests in the Southwest were formerly characterized by frequent, low-intensity surface fires, and it is primarily in these forests where fire suppression has contributed to unnaturally dense stands and increased fire severity today (Covington and Moore 1994, Mast et al. 1999, Moore et al. 1999, Allen et al. 2002). Although fire suppression is part of the reason for very dense stands of ponderosa pine in the Southwest, previous grazing, logging, and climate have also contributed to this change in forest structure (Allen et al. 2002). For example, abundant recruitment of pine seedlings typically occurs during moist climatic periods, and the twentieth century has been characterized by several such periods (Savage et al. 1996, Brown and Wu 2005). In the Colorado Rockies, a

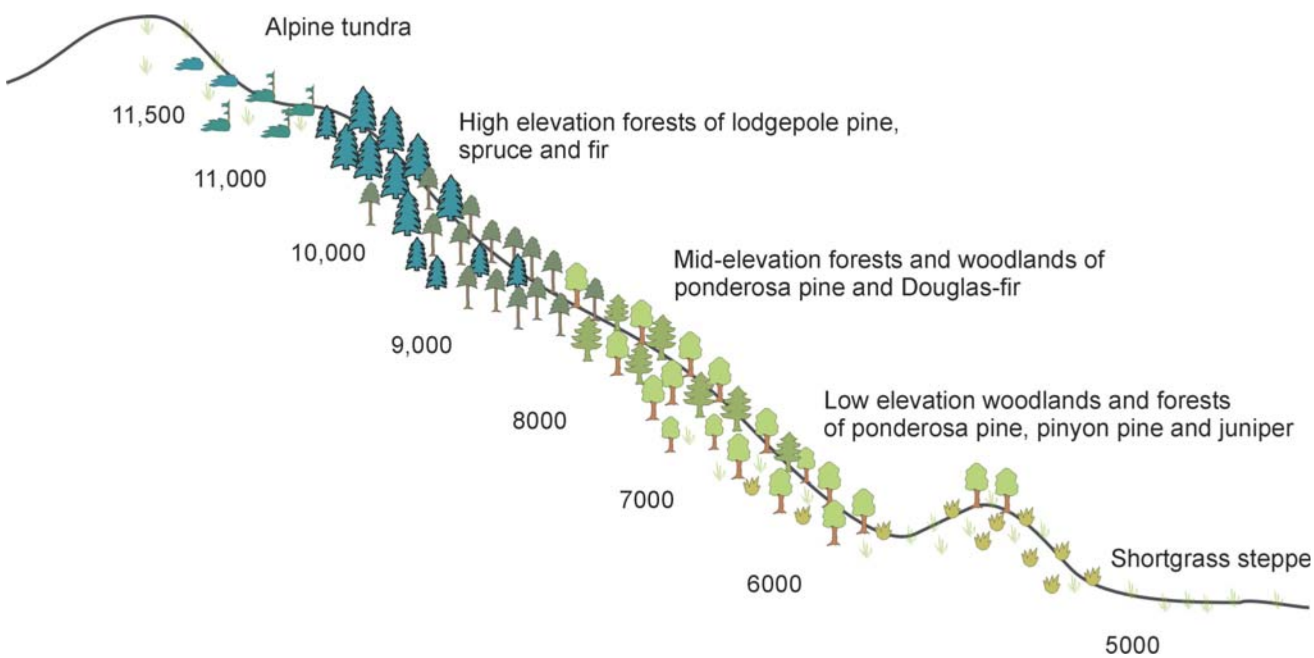

Figure 4. Colorado's forests and woodlands are diverse, ranging from piñon-juniper woodlands in the foothills and basins, to ponderosa pine and Douglas-fir forests at middle elevations, to lodgepole pine and spruce-fir forests at the highest elevations. The natural frequency and effects of forest fires are equally diverse. Tree density in some ponderosa pine forests is greater today than historically because of fire suppression, grazing, and logging during the past century. In contrast, dense stands in high-elevation forests are not related to 20th century fire suppression or land use history; they are simply natural features of these forests where fires have always occurred infrequently. (Figure prepared by L. Huckaby)

model similar to that in the Southwest -- suppression of formerly frequent low-severity fires followed by increased tree density -- applies to some but not all ponderosa pine forests. This “Southwestern ponderosa pine” model appears to be most applicable towards lower elevations and more southerly portions of Colorado.

The Southwestern ponderosa pine model generally does not apply throughout the moister, cooler forests in northern Colorado and at higher elevations, even though ponderosa pine may still dominate (Kaufmann et al. 2006, Baker et al. 2006). For example, in ponderosa pine forests of the Colorado Front Range, tree-ring and other evidence demonstrates that the historical fire regime included both low-severity fire (i.e., surface fires that thin the forests) and high-severity fires (i.e., fires that kill canopy trees and often result in dense regeneration) (Mast et al. 1998, Brown et al. 1999, Kaufmann et al. 2000, Veblen et al. 2000, Huckaby et al. 2001, Ehle and Baker 2003, Sherriff 2004, Kaufmann et al. 2006). In fact, less than 20% of the ponderosa pine zone in the northern Colorado Front Range appears to have been characterized mainly by frequent, low-severity fires. Instead, most of the ponderosa pine zone was characterized by a variable–severity fire regime that included a significant component of high-severity fires (Sherriff, 2004).

The high-severity fires of Front Range ponderosa pine forests tend to occur less frequently than the low-severity fires, and forests naturally grow dense during the long intervals between successive fires. These dense stands are interspersed with more open stands, creating a complex mix of forests. Thus, we conclude that the dense ponderosa pine forests seen in some parts of Colorado’s northern Front Range are only partly due to 20th century fire suppression and low rates of timber harvest in recent decades. In contrast to some forests in the Southwest, dense stands of ponderosa pine have always been a component of the Front Range landscape. The proportion of dense vs. more open pine stands has shifted towards more dense stands during the past half-century in many areas, in part because of fire suppression, but also because of climatic conditions conducive to tree growth and natural recovery of forests that were burned or logged in the late 19th century.

Lodgepole Pine Forests <u>Summary</u>: *Dense lodgepole pine stands are not an artifact of fire suppression. These forests have always burned infrequently (intervals of many decades or centuries between fires) and at high intensity, and these fires are naturally followed by development of a dense young stand. Fire suppression has not significantly altered the natural frequency or ecological effects of fire in most lodgepole pine forests.*

Dense stands historically were the norm in lodgepole pine and other high-elevation forests throughout the Rocky Mountain region (Parker and Parker 1994, Kashian et al. 2005, Schoennagel et al. 2004). In these forests in Colorado, fires occur infrequently (on the order of many decades or a century or more between successive fires in any given stand) and naturally tend to be high-intensity fires, usually crown fires, that kill the majority of the trees (Buechling and Baker 2004, Sibold et al. 2006, Veblen and Donnegan 2006). This type of natural fire behavior contrasts strikingly with the frequent surface fires of dry, low-elevation ponderosa pine forests: rather than thinning forests by killing primarily small, fire-intolerant individuals, the naturally severe fires of high-elevation forests typically kill all of the forest canopy and stimulate regeneration of the stand. Post-fire regeneration of lodgepole pine often results in a dense stand, especially where a large proportion of the trees have serotinous cones. Serotinous cones remain sealed by resin until the heat of a fire melts the resin and releases the seeds; thus, even though the adult trees are killed by the fire, they have stored huge numbers of seeds in their cones and those seeds are released into an optimal seed bed created by the fire.

The effect of fire suppression on the structure of individual stands and on the characteristics of stands across the landscape has been relatively minimal in lodgepole pine and other high-elevation forests in Colorado and throughout the Rocky Mountains (Schoennagel et al. 2004). The remote mountainous areas where these forests grow were generally difficult to access for fire-fighting, especially prior to the 1950s. Furthermore, the length of time that fire has been effectively excluded (~50 to 80 years) is short relative to the natural fire return interval (measured in centuries). As a consequence, fire exclusion has not significantly lengthened fire intervals in lodgepole pine forests. Note that this is in marked contrast to frequent, low-severity fire regimes such as Southwestern ponderosa pine.

It is true that a large proportion of the lodgepole pine stands in Colorado are more than 100 years old today (e.g. as reflected in stand age data from USDA Forest Service). However, this pulse of tree establishment was mainly due to widespread severe fires during the second half of the 19th century when climate was conducive to fires in the subalpine zone (Sibold and Veblen 2006). Tree-ring data show that similar pulses of establishment of lodgepole pine followed similar episodes of widespread fire in the 17th and 18th centuries across the subalpine zone of northern Colorado (Kulakowski and Veblen 2002, Kulakowski et al. 2003, Sibold et al. 2006). Thus, the tree-ring record of fire and tree establishment in subalpine forests indicates a high degree of variability in fire extent and stand initiation at time scales of 100 years. This variability included periods of extremely rare fires over 100-year periods of climate unfavorable to fire spread, so that long fire-free intervals such as in the 20th century are not outside the historical range of variability for these forests. Thus, age structures similar to the current dominance of the 100+ year old age class are typical of the historical conditions of lodgepole pine forests.

Because of the natural disturbance regime in lodgepole pine forests, characterized by infrequent but periodically large severe fires and insect outbreaks, these high-elevation forests do not exhibit a static or consistent average age class over time. We know that fires before 1900 in this forest type were infrequent but could grow to very large size under very dry weather conditions (Schoennagel et al. 2004, Sibold et al. 2006). It follows that we should expect large fires in lodgepole pine in the future, and that these future large fires should not be viewed as abnormal from an ecological standpoint. The key point about lodgepole pine forests is that they were dense and burned infrequently historically, and they are dense and burn infrequently today. High density in lodgepole pine forests is not related to fire suppression in any way; on the contrary, it is a natural feature of their ecology.

Spruce-Fir Forests Summary: *Dense spruce-fir stands are not artifacts of fire suppression either. Spruce-fir forests have always burned infrequently (intervals of centuries between fires) and at high intensity, and these fires are naturally followed by development of a dense young stand. Fire suppression has not significantly altered the natural frequency or ecological effects of fire in most spruce-fir forests.*

As in lodgepole pine forests, dense stands are also normal in spruce-fir forests (Veblen and Donnegan 2006). Prior to the beginning of fire suppression efforts in the 20th century, these forests were primarily shaped by large and severe fires that occurred in a given stand, on average, only once per several hundred years (Kulakowski et al. 2003, Buechling and Baker 2004, Sibold et al. 2006). Natural patterns of post-fire stand development resulted in high tree densities. Since long fire-free periods were normal in these forests prior to fire suppression efforts, it is very unlikely that several decades of fire suppression have fundamentally changed the natural fire regime or have resulted in forest structures that could be considered unnaturally dense. Instead, the dense spruce-fir forests today are very much like they have been in past centuries.

Question #5: Are recent wildfires in some of Colorado's dense forest stands unusually severe compared to pre-20th century fire severity?

Summary: *Recent fires have been more severe than historically in some forests, notably dry ponderosa pine forests in parts of Arizona, New Mexico, and southern Colorado. However, recent fires have behaved just as they did historically in most of Colorado's high-elevation forests, such as lodgepole pine and spruce-fir. Large intense fires are the normal fire behavior in these latter kinds of forests, and 20th century fire suppression has not caused them to be unnaturally severe (Figure 5).*

Again we stress the importance of distinguishing among forest types. Recent fires have been more severe, for example, in dry ponderosa pine forests in the Southwest, including some of the forests in southwestern Colorado. However, recent fires clearly are not

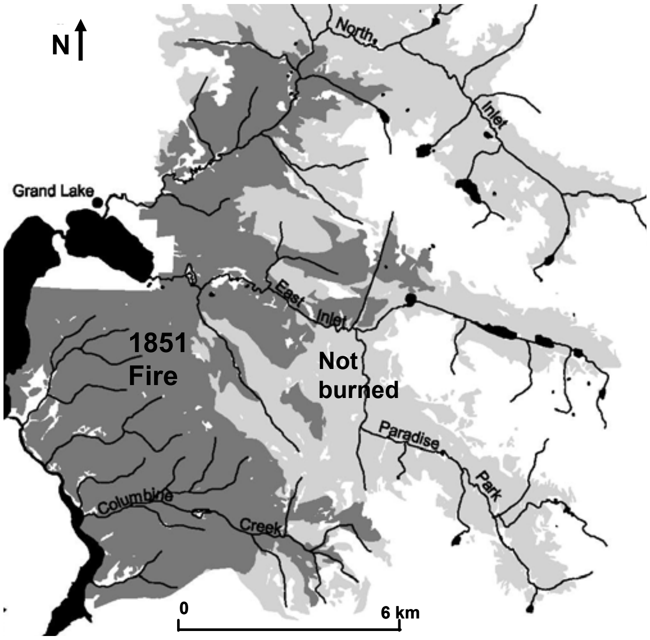

Figure 5. Large, intense forest fires are a natural feature of high-elevation forests in Colorado. For example, much of the country around Grand Lake, Colorado, burned in 1851. Most of the burned area now is covered by 150-year old lodgepole pine forests. (Figure from J. Sibold, 2005 Ph.D. Dissertation, CU Boulder). Some recent fires in ponderosa pine forests have been more severe than would have occurred historically, because of fuels changes associated with fire suppression, grazing, and logging during the past century. However, recent fires in lodgepole pine and spruce-fir forests also have been intense -- but no more intense than occurred historically.

more severe in lodgepole pine or in spruce-fir than fires that occurred in previous centuries. Even in the case of ponderosa pine forests in Colorado, not all areas follow the Southwestern pattern of increased stand densities following the near elimination of fires by grazing and fire suppression in the late 19^{th} and 20^{th} centuries. As noted above, in the Colorado Front Range the ponderosa pine zone was characterized by an historical mixed-severity fire regime in which some areas burned at low severity (as in the Southwest) but other areas, often large, were burned severely and regenerated to dense stands (Mast et al. 1998, Brown et al. 1999, Kaufmann et al. 2000, Veblen et al. 2000, Huckaby et al. 2001, Ehle and Baker 2003, Sherriff 2004, Kaufmann et al. 2006).

Question #6: Do outbreaks of mountain pine beetles and other forest insects increase the risk of severe wildfires?

<u>Summary</u>: *Although it is widely believed that insect outbreaks set the stage for severe forest fires, the few scientific studies that support this idea report a very small effect, and other studies have found no relationship between insect outbreaks and subsequent fire activity. Theoretical considerations suggest that bark beetle outbreaks actually may reduce fire risk in some lodgepole pine forests once the dead needles fall from the trees. It is true that severe fires have occurred recently in some forests following insect outbreaks (e.g., in spruce-fir forests of western Colorado). However, these fires burned under very dry weather conditions, and severe fires are the norm for these kinds of forests even without insect activity. Based on current knowledge, the assumed link between insect outbreaks and subsequent forest fire is not well supported, and may in fact be incorrect or so small an effect as to be inconsequential for many or most of the forests in Colorado (Figure 6).*

Our focus here is on active crown fires, i.e., fires that move from tree crown to tree crown under dry windy conditions. Surface fires also are significant; they can affect soils and understory plants, cause major damage to homes and other structures, and can be difficult to control, especially when burning in heavy fuels. However, in this discussion we emphasize crown fires because these often are the most fast-moving fires, they are the fires that typically cause the most damage to homes and other vulnerable structures, and they are almost impossible to control even with modern fire-fighting technology. It is important to realize that active crown fires do not burn only the dead fuels. On the contrary, crown fires are propagated through both live fuels (needles and small twigs) and dead fuels. Tree-killing insects do not really increase the amount of fuels in a forest stand; what they do is shift some of the live fuels into the dead fuel category. Both live and dead fuels can carry fire under very dry weather conditions.

Although more research is needed to confidently predict the effects of insect outbreaks on subsequent fires in Colorado forests, we offer the following interpretation based on theoretical considerations. Whether beetle-caused mortality enhances fire risk and severity compared to an

Figure 6. Lodgepole pine and spruce-fir forests typically burn at high intensity even without previous insect activity. It is widely believed that insect outbreaks set the stage for intense forest fires, but there is little scientific evidence for such a connection. Some recent Colorado fires have burned intensely in lodgepole pine and spruce-fir forests where insect outbreaks had occurred from a few to 50 years previously (e.g., in the Routt and White River National Forests). However, these fires occurred during extremely dry weather conditions, and forests unaffected by bark beetle outbreaks burned in similar fashion. (Photo by W. H. Romme)

unaffected stand very likely depends on time since outbreak. Post-outbreak stand development and associated fire risk may proceed through three stages. (i) Immediately following an outbreak, when trees are dead and dry needles remain on the trees, the chance of a crown fire getting started may be greater than for live trees. However, the dead needles may not significantly change the likelihood of a crown fire spreading from tree to tree, because crown fire spread is controlled not just by dead fuel quantity, but also by live fuel moisture, wind speed, and canopy bulk density (total amount of live and dead fuels in the canopy). This first stage lasts a relatively short time, because the dead needles usually fall within about two years of a tree's death. (ii) Once the needles fall off the dead trees, the likelihood of both crown fire initiation and spread actually may be reduced in comparison to an unaffected stand, since the dead trees create gaps in the canopy and reduce canopy bulk density. It is known that reducing canopy continuity and bulk density through mechanical thinning or harvesting can reduce crown fire risk (Graham et al. 2004), and it is likely that reductions in canopy continuity and bulk density resulting from insect-caused mortality would have a similar effect. (iii) After the dead snags fall, typically one to several decades after the insect outbreak, it is expected that the risk of crown fire initiation and spread may increase once again through two mechanisms. First, the fallen snags may fuel an intense surface fire, with heat and flame lengths that reach into the crowns of the trees. Second, small trees, which generally survived the outbreak and grew more rapidly in the more open conditions resulting from death of canopy trees, create “ladder fuels” that can carry a surface fire into the canopy. In sum, crown fire risk may be elevated for a brief time during and immediately after the peak of the outbreak, while the trees retain their dead needles; then fall to lower levels for the next few decades while the bare snags remain standing; and finally return to pre-outbreak levels some 20 – 50 years after the outbreak when the snags have fallen and a fast-growing understory has created ladder fuels between the heavy surface fuels and the canopy.

We emphasize again that the interpretation just presented is primarily theoretical and requires further study before definitive conclusions can be drawn. We also stress that this analysis focuses on effects of insect-caused mortality within a single stand. The impact on subsequent fire behavior will be different depending on the proportion of the trees killed in the stand. Moreover, it is important to recognize that a large forest landscape is composed of many individual stands. Substantial changes in stand structure and fire behavior within just one or a few stands may have little influence on fire spread and fire severity across the entire landscape.

A few empirical studies have evaluated subsequent fire activity in areas across the West that have been affected by major insect outbreaks, as summarized below. The general conclusion of these studies has been that the outbreak had no effect or only a small effect on subsequent fire occurrence or severity. However, more research of this kind is needed before we can make definitive statements about insects and fire.

Spruce beetle in subalpine spruce-fir forests*.* It is well established that in spruce-fir forests, extensive fires are highly dependent on infrequent, severe droughts (Buechling and Baker 2004, Sibold and Veblen 2006). Under those extreme drought conditions, dead fuels from insect outbreaks or other causes appear to play only a minor role, if any, in increasing fire risk. For example, following the 1940s spruce beetle outbreak that resulted in dead-standing trees over most of the subalpine zone of White River National Forest of western Colorado, there was no increase in the numbers of fires compared to unaffected subalpine forests (Bebi et al. 2003). Large fires did not occur in these forests until the drought of 1980, when 10,000 acres burned in the Emerald Lake Fire, and in the very severe drought year of 2002 (Pielke et al. 2005) when 31,000 acres burned in the Big Fish and Spring Creek fires. The 2002 fires in western Colorado affected extensive areas of spruce-fir and lodgepole pine forests that were previously affected by outbreaks of spruce beetle and of mountain pine beetle. Yet despite the expectation that these outbreaks (both the 1940s and an ongoing post-1998 outbreak) would have led to an increased risk of severe fires, the forests that were affected by the outbreaks generally did not burn more extensively or more severely than forests that were not affected (Bigler et al. 2005; Kulakowski and Veblen 2006).

Mountain pine beetle outbreaks in lodgepole pine forests. Turner et al. (1999) evaluated the influence of beetle outbreaks that had occurred 5-15 years previously on the behavior of the 1988 Yellowstone fires in lodgepole pine forests. They found that the likelihood of crown fire was increased somewhat where beetle-caused tree mortality had been high (perhaps because the fallen trees created heavy fuel loads), but was reduced where beetle-caused mortality was only moderate (perhaps because the dead trees interrupted the horizontal continuity of the canopy). Lynch (2006; chapter 3) also examined the influence of previous beetle activity on the 1988 Yellowstone fires by testing whether fire was more likely where the beetles had killed trees than in areas unaffected by the beetles. She found that beetle-affected areas did have a higher probability of burning, but that the increase was only about 11% compared with areas unaffected by beetles.

Spruce budworm defoliation*.* Massive outbreaks of western spruce budworm affected the Douglas-fir forests of the northern Colorado Front Range in the late 1970s and 1980s, but there is no evidence that they resulted in increased fire occurrence. Widespread fires have occurred recently in these forests, but these fires were associated with the extreme drought of 1998-2002. Therefore, if there was any potential increase in fire risk associated with the spruce budworm outbreaks, that potential was not realized until at least 25 years later when weather conditions were conducive to extreme fire behavior even in the absence of insect effects. In Ontario, Canada, Fleming et al. (2002) found a significant increase in probability of fire 3-9 years after an outbreak (perhaps because of increased vertical fuel continuity between fuels on the forest floor and fuels in the canopy), but probability of fire was not continuously elevated after the outbreak. However, in British Columbia, Canada, Lynch (2006; chapter 2) reported a significant decrease in risk of forest fire for nine years following a spruce budworm outbreak.

The upshot of these few studies of insect effects on subsequent fire risk is that the relationships are complex, and that no simple statements can be made about how outbreaks do or do not increase the risk of fire. One reason for the lack of clear-cut patterns is that spruce-fir and lodgepole pine forests naturally burn very infrequently, and only under very dry weather conditions. When the weather conditions are right for a big fire in spruce-fir or lodgepole pine, fire behavior is naturally intense, whether affected by previous insect activity or not. If insect outbreaks do in fact increase the likelihood of fires getting started or burning intensely through these kinds of forests, the magnitude of increase probably is small and difficult to detect, because fire is so strongly controlled by weather in these forests, and because they naturally burn at high intensity.

Question #7: Are forests with large amounts of insects and dead trees “unhealthy?”

Summary: *"Forest health" is an ambiguous concept, one that is not well defined scientifically. The presence of dead or dying trees does not necessarily mean that the forest ecosystem as a whole is not functioning appropriately, even when such trees are numerous. In fact, dead trees and fallen logs perform some important ecological functions in forests, such as providing wildlife habitat and returning nutrients and organic matter to the soil. Nevertheless, dead trees are unattractive and unappealing to many people, and it can be quite painful to lose trees that have special meaning to an individual, such as large pines surrounding one's home (Figure 7).*

Although it may be relatively easy to ascertain whether an individual tree is healthy or not, the concept of “forest health" is very ambiguous. The presence of unhealthy trees does not necessarily imply that the forest as a whole is unhealthy. On the contrary, standing dead trees and fallen logs (coarse wood) play important roles in wildlife habitat, soil development, and nutrient cycling, and are a defining characteristic of old-growth forests. Bark beetle outbreaks rarely kill all of the trees in a stand, because they preferentially attack the larger trees and generally ignore the smaller trees. These smaller trees may be hidden by the red needles of the large killed trees during the peak of the outbreak, such that one often has an impression of total tree mortality. However, once those needles fall it usually becomes apparent that many small and moderate sized trees survived the outbreak. These smaller trees may grow two to four times more rapidly after the outbreak than they did before, because they are no longer competing with the big trees for light, water, and nutrients (Romme et al. 1986). In mixed forests of lodgepole pine and aspen, the aspen may grow more vigorously after beetles kill the dominant pine trees. Even when all of the trees are killed, as in a severe forest fire, the result usually is stand regeneration, as described above for lodgepole pine. Thus, from a purely ecological standpoint, dead and dying trees do not necessarily represent poor “forest health." They may instead reflect a natural process of forest renewal.

Figure 7. "Forest health" is an ambiguous concept. The presence of dead and dying trees does not necessarily mean that the forest ecosystem as a whole is not functioning appropriately. Dead trees and fallen logs perform important ecological functions, such as providing wildlife habitat and returning nutrients and organic matter to the soil. (photo by W. H. Romme)

Nevertheless, dead trees are unattractive and unappealing to many people, especially when those dead trees are abundant, and it can be quite painful to lose trees that have special meaning to an individual, such as large pines surrounding one's home. The change in the appearance of the forest after an insect outbreak also can have negative economic consequences for a community. Over time, the visual impacts are lessened as aspen and small pines grow larger and more abundant, and the gray trunks of the beetle-killed trees gradually fall to the ground. Nevertheless, the visual evidence of an insect outbreak may persist for a decade or more after the outbreak subsides.

Question #8: Does a large insect outbreak constitute an "emergency?"

Summary: *Forests naturally change slowly, almost imperceptibly, over long periods of time. But periodically this slow process of change is punctuated by rapid change via insect outbreak, fire, or other natural disturbance. The sudden death of thousands of trees may be an emergency for people and communities whose amenities, economic activities, and management plans were based on the slowly changing forest that used to occupy the area. From an ecological perspective, however, insect outbreaks are part of the natural rhythm of change in forest ecosystems, and are followed by a gradual re-development of the forest through natural ecological processes. Where aspen was present before the outbreak, the death of the pines may lead to an increase in the aspen component of the forest (Figure 8).*

The normal development of forests involves very slow changes that continue over decades or centuries. A large-scale insect outbreak or forest fire changes a forest rapidly, over a period of a few weeks or years. Such a rapid change often generates great concern about the health and future of the forest and landscape. Is this an emergency? The sudden death of thousands of trees may be an emergency for people and communities that are accustomed to the slowly changing forest that used to occupy the area. Recreational opportunities and values suddenly change, and long-term plans that relied on only slow changes in the forest (such as estimations of annual wood yield) no longer apply. Thus, these may be emergencies from certain standpoints.

From an ecological perspective, we recognize that the forest will slowly re-develop through natural processes. Many montane landscapes in central Colorado are well suited for both conifers (lodgepole pine, spruce, and fir) and aspen, and several of these species commonly occur in the same forest. A century of forest development without any major disturbance typically leads to decreasing abundance of aspen as the conifers increase in dominance. A bark beetle outbreak that kills many of the conifers may be beneficial to the aspen. Old aspen trees will likely grow faster, and new aspen will become established. An increase in aspen will occur only where aspen clones were present before the beetle outbreak. If there was not aspen already present, then composition of the forest will not change; the surviving conifers (mostly smaller individuals and non-susceptible species) will increase their growth rates and replace the large conifer trees that were killed by beetles.

The terms "ecological emergency" and "insect emergency" suggest that insect outbreaks are unforeseen events. However, insect outbreaks, even extensive ones that kill canopy trees over hundreds of thousands of acres, are natural events in forest ecosystems throughout the Rocky Mountains, and have been occurring for thousands of years (e.g., Swetnam and Lynch 1998, Lavoie 2001). The insects have long been natural components of these forest ecosystems. Therefore, from a purely ecological perspective, an insect outbreak generally would not be regarded as an "emergency," but as an infrequent but normal episode of rapid change within an ecosystem that most of the time is changing only slowly.

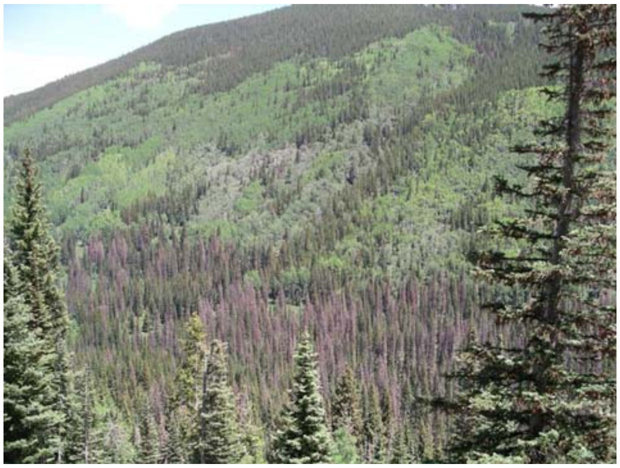

Figure 8. Forests naturally change slowly, almost imperceptibly, over long periods of time. But period-ically this slow process of change is punctuated by rapid change via insect outbreak, fire, or other natural disturbance. From an ecological perspective, insect outbreaks are part of the natural rhythm of change in forest ecosystems, and are followed by a gradual re-development of the forest through natural ecological processes. (photo by Dominik Kulakowski)

Question #9: How do insect outbreaks affect streamflow and water quality?

Summary: *An insect outbreak, or any disturbance that reduces the total area of leaf surface in a forest, can potentially increase streamflow by reducing the amount of interception and transpiration. No increase in streamflow is likely when the total annual precipitation is less than 18-20 inches. In areas with more than 18-20 inches of annual precipitation, an increase in streamflow generally will not be detectable unless at least 15-20% of the forest canopy is killed. By themselves, insect outbreaks are unlikely to cause erosion or degrade water quality because they do not disturb the forest soil. Unpaved roads and high-severity wildfires can cause much greater effects on runoff, erosion, and water quality (Figure 9).*

The hydrologic effects of insect infestations vary with the type of forest, the number and size of trees that are killed, and the amount and type of precipitation. The likely effects of a given change in forest density and structure can be predicted with a relatively high degree of confidence because of the long history of plot, process, and watershed scale studies in Colorado and elsewhere (MacDonald and Stednick, 2003). Over the last decade there has been a sharp increase in our understanding of how wildfires, prescribed fires, and thinning affect runoff and erosion rates in Colorado (e.g., Moody and Martin, 2001; Benavides-Solorio and MacDonad, 2005; Kunze and Stednick, 2006).

Removal of all or a part of the forest canopy may potentially increase streamflow via two mechanisms. First, the forest canopy intercepts a portion of incoming precipitation, and this intercepted rain or snow simply evaporates or sublimates back into the atmosphere without ever reaching the soil. A reduction in the forest canopy generally reduces the amount of water that is intercepted and thereby increases net precipitation (but see below for other complicating factors). Second, live trees take up water from the soil and transpire that water into the atmosphere.

Several principles determine whether a particular insect infestation or management action will significantly alter the amount and timing of runoff. First, removing the forest cover from areas that receive less than about 18-20 inches of annual precipitation will have little effect on the amount and timing of runoff as long as there are no significant changes to the infiltration rate of the soil. The primary reason for this lack of change is that any reductions in interception and transpiration are negated by an increase in soil evaporation and transpiration by any remaining vegetation (Bosch and Hewlett, 1982). Once annual precipitation exceeds about 18-20 inches, the reduction in interception and transpiration due to forest harvest or dieback will increase annual runoff, and this increase generally will be proportional to the amount of annual precipitation. Second, at least 15-20% of

Figure 9. An insect outbreak can potentially increase streamflow by reducing the amount of water transpired by trees. However, the increase probably will not be detectable unless total annual precipitation is greater than 18-20 inches and at least 15-20% of the forest canopy is killed. By themselves, insect outbreaks generally do not cause erosion or degrade water quality, because they usually do not disturb the soil. (photo by J. A. Hicke)

the forest canopy has to be killed or removed before there will be any measurable increase in annual runoff. Removing a smaller proportion of the forest cover may still increase the amount of runoff, but this increase probably will not be statistically detectable. Third, the increase in annual runoff due to forest harvest or tree death is roughly proportional to the amount of the forest canopy that is removed or killed. Fourth, the absolute changes in streamflow will be much smaller in dry years than wet years, and become harder to detect as spatial scale increases (MacDonald and Stednick, 2003).

Extrapolation of paired-watershed studies in snow-dominated areas of Colorado and Wyoming indicates that removing the forest canopy from 100% of a watershed will increase mean annual water yields as follows: by a little over 1 inch or about 18% of the mean annual runoff when the mean annual precipitation is 21 inches (Bates and Henry, 1928); by 8 inches or roughly 90% when the mean annual precipitation is 30 inches (Troendle and King, 1985); and by over 12 inches or about 70% when the mean annual precipitation is 34 inches (Troendle et al., 2001). Nearly all of this increase in water yield will come on the rising limb of the snowmelt hydrograph in May-June. Complete removal of the forest canopy can be expected to increase the size of the mean annual peak daily flow by about 40% while having minimal effect on the timing of the annual peak flow (MacDonald and Stednick, 2003).

The hydrologic effects of insect outbreaks are similar in many respects to the effects of forest harvest, but there also are some important differences (MacDonald and Stednick, 2003; Uunila et al., 2006). One difference is that under natural conditions the insect-killed trees remain in place, and this residual canopy will still intercept a portion of the incoming rain and snow, especially while the needles and fine twigs are still in place. This means that the water yield increase due to bug-killed trees will be smaller than the water yield increase due to a comparable amount of forest harvest. A second important difference is that although the insects may kill most or all of the trees within small patches of a few acres, outbreaks never kill all of the trees across a large watershed or landscape; thus, the increases in water yield following insect outbreaks will be smaller than the values listed in the previous paragraph for complete tree harvest (Schmid et al., 1991). Finally, any increase in runoff will decay over time with forest re-growth, and the time to hydrologic recovery may be shorter for an insect outbreak as compared to forest harvest. Studies in Colorado indicate that the time needed for hydrologic recovery after a clearcut varies from about 60 years in the spruce-fir and lodgepole pine zones to around half this time in aspen stands (MacDonald and Stednick, 2003). Insect outbreaks usually kill a portion of the trees, and the surviving trees may grow two to four times faster than they did before the outbreak. Therefore, canopy basal area may return to pre-outbreak levels within a shorter period of time, and this will reduce the potential increase in water yields relative to timber harvest.

Several studies have attempted to evaluate or predict the hydrologic effects of insect outbreaks in Colorado and elsewhere, but most of these studies were been hampered by the lack of a well-controlled design and the available statistical tools. After the 1939-1946 spruce beetle epidemic in the White and Yampa River basins, Love (1955) claimed that annual streamflow in the White River increased by about 2.3 inches or 22%, but this was refuted by Bue et al. (1955). Bethlahmy (1974, 1975) conducted more extensive analyses using different techniques and claimed that the beetle epidemic increased annual water yields by up to 2.0 inches in the White River basin and 2.4 inches in the Yampa River basin, and that the water yield increases were still present after 25 years. A more recent modeling study predicted that water yields would increase in the North Platte River basin by 2.2 inches if 30-50% of the trees were killed by insects (Troendle and Nankervis, 2000). While none of these studies can be considered definitive, the general results are consistent with the principles and values outlined in this section.

In terms of water quality, forested areas typically have very high infiltration rates and rarely generate surface runoff. The death of trees by insects should not compact the soil or cause a loss of the protective litter layer. In the absence of any compaction or ground disturbance, there should be minimal change in soil infiltration rates or the soil moisture storage capacity. Hence an insect outbreak should not induce overland flow or increase erosion rates, even on steep slopes. On the other hand, the increased duration of high flows due to forest harvest or dieback can increase watershed-scale sediment yields by increasing the stream's sediment transport capacity (Troendle and Olsen, 1994). In practical terms this is of little significance because the sediment yields from forested areas are typically very low (MacDonald and Stednick, 2003). In many forested areas, unpaved roads are a primary source of sediment (Libohova, 2004), and the number, location, and design of forest roads is a key control on whether thinning or harvest activities will affect water quality and aquatic ecosystems (MacDonald and Stednick, 2003; Libohova, 2004). Forest harvest and bug kill can reduce slope stability as a result of the decay in root strength (Sidle et al., 1985), but the increased susceptibility to landslides and debris flows is rarely an issue in Colorado.

Although insect outbreaks usually produce little or no soil erosion, and may have minimal impact on runoff, other disturbances may have significant impacts on soils and runoff. The effects of wild and prescribed fires on runoff and erosion depend primarily on fire severity as well as the timing and cause of peak flows. Low severity fires have minimal effects on runoff and erosion rates because these do not remove the protective litter layer and generally do not kill the larger and more mature trees. In contrast, high severity fires consume all of the protective organic layer, kill most or all of the vegetation, and can induce a water repellent layer at or near the soil surface (Huffman et al., 2001; Benavides-Solorio and MacDonald, 2005; Pietraszek, 2006). In areas with summer convective storms, peak flows and erosion rates can increase by several orders of magnitude after a high-severity fire (Moody and Martin, 2001; Libohova, 2004; Benavides-Solorio and MacDonald, 2005), and the combination of ash and sediment can severely degrade water quality (Moody and Martin, 2001; Kunze and Stednick, 2006). A series of studies in the ponderosa pine zone in the Colorado Front Range suggests that long-term sediment delivery rates from unpaved roads may be similar in magnitude to rates from periodic high-severity fires, while forest thinning has no detectable effect on runoff or erosion rates (MacDonald and Larsen, in press). In snowmelt-dominated areas high-severity fires may have a much smaller effect because soils are not water repellent under wet conditions (MacDonald and Huffman, 2004), and the number and intensity of summer thunderstorms may be lower than in mid-elevation forests. Hence the hydrologic effects of fires in the higher-elevation forests may be more similar to the effects of forest harvest, but there are few data from these higher-elevation sites.

Potential Treatment Options

Even though the insect outbreaks now occurring in Colorado generally cannot be regarded as *ecological* emergencies, there is no denying that the extensive stands of dead and dying trees do affect the aesthetic and economic attributes of many forests. Moreover, forest fires may cause serious damage to property and may even threaten human lives – whether or not previous insect activity has caused those fires to be more severe than they would be otherwise. Therefore, efforts to reduce the impacts of insects and fires are warranted in many areas. The following sections describe and evaluate the likely effects of a range of treatments that have been used or proposed to ameliorate the effects of insect outbreaks and fires.

Option #1: Spraying with Insecticide

<u>Summary</u>: *This can be an effective means of saving high-value trees in localized areas, but is not feasible over large landscapes (Figure 10).*

Spraying trees with an appropriate insecticide can be an effective means of preventing bark beetle attack or reducing defoliator damage. County extension agents and personnel of the Colorado State Forest Service and USDA Forest Service can recommend the best products to use against a particular insect in a particular area.

This may be the best means available for protecting high-value trees around homes, in town parks, or other localized places. However, there are limits to what can be accomplished by spraying insecticides. Annual spraying, or even spraying several times in a single year, is required to prevent attacks by each successive

Figure 10. Spraying with insecticide can be an effective way to preserve high-value trees, such as around a home. However, spraying is not feasible or effective in stopping insect outbreaks over large landscapes. (photo by W. H. Romme)

generation of insects. Spraying is not feasible at the scale of an entire forest landscape because of cost and difficulty of hitting all of the places where insects may be present. In addition, insecticides are not entirely species-specific: a broad-scale spraying of insecticides will kill many harmless and beneficial insects, such as pollinators and butterflies, in addition to the target bark beetles and defoliators. In general, bark beetle preventive sprays have less impact on non-target insects than do insecticide sprays used to control defoliatiors, because the former sprays are targeted to the trunk of the tree whereas the latter sprays need to cover entire tree canopies.

Option #2: Preventing or controlling outbreaks through forest management

Summary: *Removing stressed or unhealthy trees, and thinning to prevent crowding and competition among trees, can effectively reduce the risk of an insect outbreak getting started in a forest stand. Forest management is unlikely to prevent all outbreaks, however, because (i) it will never be feasible to intensively manage all of the forests of Colorado, and (ii) drought and warm temperatures are also important causes of outbreaks. Once an outbreak has begun, management generally cannot stop it, because the insects are numerous enough to overcome even healthy trees (Figure 11).*

Because outbreaks may initiate in stressed or unhealthy trees, intensive forest management focused on regular removal of old or unhealthy trees may reduce the likelihood of an insect outbreak getting started in a stand. Thinning may reduce tree-to-tree competition, increase tree vigor, and thus provide an enhanced ability of trees to defend against an attack (Amman and Logan 1998, Schmid and Mata 2005). If periodic harvest removes large trees and maintains a preponderance of small-diameter trees, this too may help prevent the start of a bark beetle outbreak, since bark beetles (but not defoliators) prefer larger trees. Thus, careful forest management, including appropriate timber harvest, may help locally to prevent the onset of an outbreak (Cole et al. 1976).

By itself, however, forest management probably cannot prevent all insect outbreaks -- for two reasons. First, it is unlikely that all stands in Colorado landscapes will be managed intensively enough to remove all of the stressed trees in which an outbreak can get started; in fact, the public values “unmanaged” forests that contain large and old trees. Second, drought and warm temperatures are major causes of bark beetle outbreaks, and forest management by itself cannot entirely overcome these climatic effects. And it is important to recognize that once an extensive bark beetle outbreak has started, it is unlikely that timber management can stop it. Under outbreak conditions, the beetles can overwhelm even the healthiest trees, so selective removal of weak or stressed trees will

Figure 11. Removing stressed or unhealthy trees, and thinning to prevent crowding and competition among trees, can effectively reduce the risk of an insect outbreak getting started in a forest stand. Forest management is unlikely to prevent all outbreaks, however, because it will never be feasible to intensively manage all of the forests of Colorado, and drought and warm temperatures are also important causes of outbreaks. (photo by W. H. Romme)

likely have little impact. Most entomological evidence indicates that once an outbreak has started, there is nothing that can be done to stop it. The outbreak ends when there are no more suitable trees for the beetles, or when unusually cold conditions kill beetle populations. Intensive even-aged management was applied to lodgepole pine forests in the Targhee National Forest, along the western boundary of Yellowstone National Park, from the 1960s through 1980s; yet a mountain pine beetle outbreak that swept through the region in the 1970s and early 1980s appeared to affect the managed Targhee stands as severely as the unmanaged stands in Yellowstone Park (Romme et al. 1986). Similarly, the lodgepole pine forests of British Columbia, Canada, are now being affected by a very extensive and severe mountain pine beetle outbreak, despite a long history of intensive forest management in this province.


Figure 12. Removing dead trees and other fuels can effectively reduce the risk of fire damage at a local scale, e.g., in the immediate vicinity of a home or community. However, the effectiveness of harvest in reducing fire risk to homes and communities over larger areas, e.g., a forest landscape, is less clear. (photo by W. H. Romme)

Option #3: Harvesting insect-killed trees to reduce wildfire risk

Summary: *Removing dead trees and other fuels can effectively reduce the risk of fire damage at a local scale, e.g., in the immediate vicinity of a home or community. However, the effectiveness of harvest in reducing fire risk over larger areas, e.g., a forest landscape, is less clear. Conventional timber harvest may do little to reduce fire risk at any scale if it removes primarily large trees, because smaller trees, brush, and dead fuels often are the major carriers of a spreading fire. Harvesting smaller trees and removing small fuels may more effectively reduce fire risk (Figure 12).*

As with the spraying and forest management options, the effectiveness of this option varies with the scale at which it is applied. Removing dead trees – plus other flammable material (including wood roofs and decks, woodpiles and burnable vegetation) from the immediate vicinity of a home or other vulnerable structure -- has been shown to be effective in protecting the structure from wildfire (Cohen 2000). The local characteristics of a home's external materials and adjacent fuels are the primary determinant of home ignitability -- not spatially extensive wildland fuel conditions. For example, the heat released even from intense crown fires will not ignite wooden walls at distances greater than 40 meters away (Cohen 2000). Fuel reduction around a home needs to focus not just on the dead fuels (e.g., the insect-killed trees), but often needs to include some of the live fuels (living trees and shrubs) which also carry fire under severe fire weather conditions. Specific guidelines for reducing fire risk around a home can be found at the Firewise website (Firewise.org) or from extension agents or the Colorado State Forest Service.

Moving up to a broader scale, however, the effectiveness of harvesting insect-killed trees to reduce fire risk across an entire forest landscape is far less certain than the effectiveness of Firewise techniques to protect an individual home. This is especially true in high-elevation forests such as lodgepole pine and spruce-fir. Commercial tree harvest typically involves the removal of large fuels (tree trunks) rather than smaller fuels (branches and needles) due to economic and logistical constraints. These smaller fuels contribute to ignition and spread of fire (e.g., to start a campfire one begins with tinder and kindling). Smaller surface and ladder fuels are important precursors to crown fire initiation (Agee and Skinner 2005). Hence, harvesting tree trunks has little effect on the risk of fire ignition or spread. It is true that if tree harvest also results in reduced canopy bulk density, this may make it more difficult for crown fires to spread. Nevertheless, it is the fine fuels (on the ground or in the canopy) that have the greatest influence on fire initiation and spread, not the large pieces of wood. Thus, management of fine surface or ladder fuels (which is usually time-consuming and expensive) would have the greatest impact on fire spread and potential high-severity crown fire.

It is important to acknowledge that traditional timber management usually is not designed or intended to reduce crown fire risk, but to produce wood fiber in an economically sustainable manner. Although anything that thins the canopy without greatly increasing the amount of fine fuels can reduce fire spread and intensity during moderate weather conditions (Graham et al. 2004), the most damaging wildfires typically occur under extreme conditions of wind and drought. Most traditional harvesting techniques (including overstory removal and individual tree selection) do not effectively reduce fire severity under extreme fire weather conditions (Stephens and Moghaddas 2005). In the 2002 Hayman fire, pre-fire harvesting where residual fuels (small, non-merchantable material) had not yet been removed, actually contributed to higher severity fire compared to unmodified areas (Omi and Martinson, 2002). If the goal is to reduce fire risk, removal of small trees either via mechanical thinning or prescribed fire (or a combination of both), plus retention of large, old-growth trees, can lower expected fire severity (Stephens and Moghaddas 2005, Agee and Skinner 2005). For example, portions of the 2002 Rodeo-Chediski fire in Arizona experienced lower fire severity where prescribed burning and other management activities during the previous decade had reduced fine fuels and small trees, but had left larger trees intact (Finney et al. 2005). Much of the research on thinning and underburning effects on subsequent wildfire severity has primarily been conducted in low-elevation, dry-forest types: similar effects cannot be assumed in high-elevation forests.

A single thinning treatment cannot maintain lowered wildfire risk over the long-term, because thinning typically stimulates rapid growth of the vegetation that is not taken (Graham et al. 2004). Research shows, for example, that past timber harvesting in ponderosa pine forests is responsible in part for the high densities we witness today (Kaufmann et al. 2000, Gruell et al. 1982, Baker et al. 2006). Although low-intensity prescribed burns reduce fine fuels in the short-term, they also contribute to subsequent dead fuels by killing understory trees, which can result in fuel levels that exceed pre-burn levels within a decade (Agee 2003). Therefore, repeated or staged prescribed fire or mechanical thinning treatments are essential for maintaining lower forest densities; otherwise, a one-time thinning may facilitate dense tree establishment.

Thus, it may be possible to reduce fire intensity and to obtain some control of fire spread patterns across a forest landscape by strategic placement of appropriate timber harvest activities, which may need to focus more on removal of small trees than of commercially valuable sawtimber (Finney 2001, Stratton 2004, Graham et al. 2004). Research is underway to develop specific prescriptions for effective use of vegetation management to alter wildfire intensity and spread at the scale of an entire forest landscape, e.g., at the U.S. Forest Service's fire laboratory in Missoula, MT (Mark Finney, personal communication). Another recently developed tool is the Fuel Treatment Evaluator, a web-based program that uses standard U.S. Forest Service inventory data to identify locations offering the greatest opportunities for hazardous fuel reduction activities (Wayne Shepperd, personal communication). However, this research is still in the early stages, and most has been conducted in only a few forest types (notably drier, lower-elevation forests like ponderosa pine). Thus, it is difficult at this time to make confident predictions of how a specific forest treatment will affect fire behavior under a range of forest types and fire weather conditions.

A major source of uncertainty about the effectiveness of landscape-level fuel treatments in altering fire behavior, is the fact that extreme fire weather can over-ride fuel effects (as seen, for example, in Hayman 2002, Routt National Forest 2002, and Yellowstone 1988). In the Hayman fire, most of the vegetation treatments that had been implemented prior to the fire had very little impact on the severity or direction of the fire during the extreme weather conditions of June 9th and 18th, which were the two days when the majority of the area burned (Finney et al. 2003). It should be noted that not all previous vegetation treatments in the Hayman area had been designed to mitigate fire behavior, but were implemented for other objectives such as timber stand improvement -- further illustrating the point that not all timber harvest activities can be assumed to reduce fire hazard. In the 1988 Yellowstone fires, once fuels reached critical moisture levels, the spatial pattern of burning was largely controlled by weather (wind direction and velocity), rather than by fuels (Minshall et al. 1989, Turner et al. 1994). A study of the 2002 fires in Routt National Forest in Colorado found that previous salvage logging had no detectable influence on fire extent or severity during the extreme drought conditions (Kulakowski and Veblen 2006).

In sum, there is no doubt that Firewise activities in the immediate vicinity of vulnerable structures can increase their survivability in a forest fire (though it must be recognized that the risk of fire damage can never be reduced to zero). However, it is far less certain how effective fuel reduction treatments at greater distances from homes will be in protecting those homes. We also note that timber harvest may be conducted for more purely ecological objectives rather than or in addition to protection of homes. In some types of forests, notably Southwestern ponderosa pine, thinning of overly dense small trees can reduce the risk of stand-replacing wildfire and also contributes to a larger goal of forest restoration (Friederici 2003, Schoennagel et al. 2004). But in other forest types, notably lodgepole pine and spruce-fir, thinning of small trees does not represent restoration of more natural conditions, because these kinds of forests are naturally dense and naturally burn at high intensities; fuel management also has less influence on fire behavior in these ecosystems where climate so strongly controls fire occurrence and severity (Schoennagel et al. 2004). We emphasize again the importance of distinguishing among forest types in evaluating the opportunities and impacts of forest management for wildfire mitigation and ecological restoration.

Option #4: Salvaging insect-killed trees to improve overall forest health

Summary: *From a purely ecological standpoint there usually is little or no need to remove insect-killed trees. However, many people do not like to see great numbers of dead trees surrounding their communities or places they like to visit. If the dead trees have a negative impact on aesthetic preferences or local economics, then it may be desirable to remove them (Figure 13).*

As discussed above, “forest health” is an ambiguous concept. From a purely ecological standpoint there usually is little or no need to remove insect-killed trees. In fact, standing snags and fallen logs actually contribute to a number of ecological and aesthetic values in forests, including maintenance of "natural" forest structures and processes, protection of soils and water quality, and preservation of species at risk from the effects of roads, exotic species, and habitat alteration. For example, the three-toed woodpecker feeds on bark beetles in dead and dying trees, and nests most successfully in areas of recent fire or beetle outbreak. Withdrawing all or most of the large dead trees after a fire or insect outbreak will reduce habitat quality for this and other species.

At the same time, there is a widespread public perception that a forest filled with dead or

Figure 13. From a purely ecological standpoint there usually is little or no need to salvage insect-killed trees in the interest of improving forest health. However, if the dead trees have a negative impact on aesthetic preferences or local economics, or if timber production is an important goal in an area, then it may be desirable to remove the dead trees. (photo by J. A. Hicke)

Figure 14. Salvage of insect-killed trees may be a preferred option in some areas because of the economic value of the timber product that can be obtained. In these situations, especially where lodgepole pine trees have been killed by mountain pine beetles, the dead trees must be harvested as soon as possible, because the wood quality deteriorates rapidly after the trees die. (photo by D. Binkley)

dying trees is “unhealthy,” and many people do not like to see great numbers of dead trees surrounding their communities or in places that they like to visit. Whether or not this perception is consistent with what we know about forest ecology, it nevertheless has an impact on aesthetic preferences and local economics. Visitors may choose not to come to a resort surrounded by dead trees; home buyers may avoid locations where the view is one of sick and dying trees. For these and other reasons, efforts to reduce tree mortality (options 1 and 2) and to remove the unsightly results of that mortality (this option), will be the preferred response to insect outbreaks in some locations.

Option #5: Salvaging insect-killed trees for economically valuable products

Summary: *Salvage of insect-killed trees may be a preferred option in some areas because of the economic value of the timber product that can be obtained. In these situations, the trees usually must be harvested as soon as possible, because the wood deteriorates rapidly after the trees die (Figure 14).*

Although salvage of insect-killed trees usually is not necessary for the normal development of the forest, it may be a preferred option in some areas because of the economic value of the timber product that can be obtained. Harvest of large trees for economic reasons can be done in ways that minimize adverse ecological impacts, e.g., by laying out harvest units in spatial patterns that mimic the patterns created by natural disturbances such as fire (e.g., Kohm and Franklin 1997, Friederici 2003, Romme et al. 2003, Perera et al. 2004). If ponderosa pine or lodgepole pine killed by mountain pine beetles are to be salvaged for their timber value, they must be harvested as soon as possible, because the wood deteriorates rapidly after the trees die. However, spruce trees killed by spruce beetles may remain merchantable for decades (Wayne Shepperd, personal communication).

Option #6 -- No treatment

Summary: *Natural ecological processes generally lead to the development of new forests after insect outbreaks, so a "no treatment" option can be a form of responsible forest management (Figure 15).*

Natural ecological processes generally lead to the development of new forests after insect outbreaks and fires, without salvage logging or other operations, so post-outbreak or post-fire treatment usually is unnecessary from a purely ecological perspective. Other choices may be made for other reasons, such as including a

Figure 15. Natural ecological processes generally lead to the development of new forests after insect outbreaks, as in this lodgepole pine forest 30 years after a bark beetle outbreak killed more than 50% of the canopy. Thus, a "no treatment" option can be a form of responsible forest management. (photo by W. H. Romme)

logging program to salvage economic value from dead trees or to create more desirable visual conditions (options 4 and 5 above).

Nevertheless, a "no treatment" option can be a form of responsible forest management.

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forest policy

Implementing the 2012 Forest Planning Rule: Best Available Scientific Information in Forest Planning Assessments

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National forests and grasslands in the United States are governed by land and resource management plans that should be updated every 15 years to reflect changing social, economic, and environmental conditions and to address new priorities. A new forest planning rule finalized in 2012 introduces new planning approaches and requirements, and several forests have completed the forest assessment phase of their planning process. Using document analysis and interview data, we analyzed four completed forest assessments to gain insights into early forest planning efforts under the 2012 rule. We found that forest assessments address the required topics, although the organization and depth of treatment varies across cases; government sources and academic publishers are relied on most often as sources of scientific information; and approaches to best available scientific information rely on peer-reviewed information, agency technical reports and syntheses, and personal expertise and judgement.

Keywords: early adopter, expertise, US Forest Service

Management of the 154 national forests and 20 grasslands in the United States is governed by land and resource management plans (also called forest plans), as required by the National Forest Management Act of 1976 (NFMA; 16 U.S.C. 1604). The forest plan functions as a guiding document that outlines goals, objectives, and strategies for management of the unit. Periodically, the rule related to forest planning is revised to reflect societal changes, new approaches and technologies, and scientific discoveries. For many years the US Forest Service (USFS), which manages the system of national forests and grasslands, has operated under a planning rule finalized in 1982 (47 FR 43026) despite several efforts (2000, 2005, and 2008) to revise and improve the rule (Schultz et al. 2013). A new planning rule issued in April 2012 (77 FR 21161) introduces several significant changes, including a renewed emphasis on collaboration, improved transparency, and a strengthened role for public involvement throughout the planning process. Of interest for our study is the requirement to use the best available scientific information (BASI) to inform the assessment, plan revision decisions, and monitoring program.

To date, little research has addressed implementation of the 2012 planning rule. Schultz et al. (2013) examined approaches to wildlife conservation planning under the new rule, raising concerns regarding potential extirpation of species. Another study analyzed public participation processes in 12 national forests (University of Montana 2015), and Schembra (2013) explored the role of standards and guidelines and how they are used in planning activities. Forest planning under the 2012 rule consists of three phases (assessment, plan development, and monitoring). The assessment phase is important, as it assembles relevant scientific information that planners will rely on to make decisions on forest management in the plan development phase. Our study contributes to this growing body of knowledge by examining the assessment phase of the forest planning process.

Eight "early adopter" national forests, along with several other forests, are currently developing their forest plans using the 2012

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rule. These forests were designated as early adopters because they provide important benefits, had strong existing collaborative networks in place, and needed to revise their forest plans (USDA Forest Service 2012a). The eight early adopter forests are: Cibola (NM), Chugach (AK), El Yunque (PR), Nez Perce and Clearwater (ID), and three forests that are coordinating planning on a regional basis: Inyo, Sequoia, and Sierra (CA).

Although implementation is still in early stages, several of the early adopter forests have completed their forest assessments and draft forest plans, which presents an opportunity to study implementation of the planning process under the new rule. One forest (the Francis Marion in SC) has completed the full plan revision process as of this writing. We examined four forests that have completed their assessments, including three forests identified by the agency as early adopters and one forest that is keeping pace with this group. The study explored three questions: 1) What does the 2012 planning rule require regarding the structure, content, and process for forest assessments? 2) How have forests implemented the directives related to forest assessments under the 2012 planning rule? 3) How are forests approaching the requirement for the use of best available scientific information in their assessments?

Forest Planning under the 2012 Rule

The 2012 planning rule suggests an adaptive approach to forest planning, instructing managers to 1) *assess* forest conditions; 2) *revise or amend* plans if the assessment indicates a need for change; and 3) *monitor* plan implementation (36 CFR 219.5). The process is cyclical, with monitoring data feeding back into the assessment of conditions in the management unit (USDA Forest Service 2012b). During the assessment phase, planners are expected to "rapidly evaluate existing information about relevant ecological, economic, and social conditions, trends, and sustainability, and their relationship to the land management plan within the context of the broader landscape" (36 CFR 219.5(a)(1)). The second phase of the planning process is plan development, amendment, or revision, where planners use the results of the assessment to establish a need for change and generate planning alternatives (36 CFR 219.5(a)(2)), and the public has the greatest opportunity for input. The plan development phase includes environmental impact assessment, public input, and plan publication (36 CFR 219.5(a)(2)). The third phase (monitoring) is an opportunity to track and measure management effectiveness over time (36 CFR 219.5(a)(3)). The planning *process* under the 2012 rule is similar to the process specified under the 1982 rule, but differs in terms of the specific elements required for the *assessment* (2012 rule) and the *analysis of the management situation* (the assessment's counterpart in the 1982 rule).

We focused our study on the assessment phase of the planning process. The assessment phase is important because it requires the forest to assemble and synthesize the most recent, relevant, and highest-quality science on social, ecological, and economic conditions to inform the plan development. Not only does this provide planners an opportunity to evaluate changes in biophysical and socio-economic conditions based on the latest monitoring data, it also represents a chance to reflect on new concepts, models, and methods that result in new scientific information about the local forest environment. Under the 2012 planning rule, the assessment phase identifies existing conditions, trends, risks, uncertainties, and information gaps that are relevant to land and resource management issues in the unit (36 CFR 219.5–219.6). In the assessment phase, the planning unit is not required to generate new studies or information, but is expected to obtain pre-existing information that is publicly available or voluntarily provided (36 CFR 219.6). Information can come from government and nongovernment sources, and the rule instructs the Forest Supervisor to provide opportunities for stakeholders to provide information for the assessment (36 CFR 219.6). The primary product of the assessment phase is an assessment document that evaluates existing information for 15 specific topic areas (Figure 1). Although the general topic areas are mandated by the 2012 rule, the Forest Supervisor has discretion to determine the scope, scale, and timing of the assessment, assuming the other requirements in the planning rule are followed (36 CFR 219.6).

Role of Science in Natural Resource Management

Historically, natural resource management in the United States was guided by the idea of scientific management and Progressive-era approaches (Taylor 1896). In particular, Samuel Hays's "gospel of efficiency" relied on a rational and scientific method of making decisions through a single, central authority. The thought was to avoid conflict via a scientific approach to social and economic issues (Hays 1959, p. 267). The US Forest Service exemplifies the approach of technical rationality and empirical science as the basis for sound resource management practices (Wellman 1987; Kaufman 1960). Foresters and natural resource managers

Management and Policy Implications

Although implementation of the US Forest Service's 2012 planning rule is still in the early stages, several national forests have completed the assessment phase and moved on to the next phase of forest planning. Our analysis of forest assessments from several "early adopter" forests illustrates that forest planners are making serious efforts to address required topics and rely on the best available scientific information. Assessment reports were disproportionately heavy in science related to terrestrial and aquatic ecosystems, and more limited in treatment of infrastructure, land ownership and access patterns, cultural heritage, and areas of tribal importance. Ensuring that assessment teams include broad and diverse disciplinary experts will help address this challenge, recognizing that some forests may not have access to necessary disciplinary specialists. It is also possible that some of the topics (e.g., ecosystem services, tribal and cultural resources, land status and use patterns) simply do not have as much relevant and available information as other topics. Assessment teams may want to consider additional ways to interact with scientists and others to create functioning communities of practice related to science exchange for forest planning. In the same way, agency scientists may consider forging new and enduring relationships with planners and managers that could generate new science that is of immediate relevance. We found similarities across all forests in the most common approaches to identifying BASI in addition to other approaches such as data sharing meetings, a wiki review site, and requests for a science synthesis. Information from non-peer-reviewed sources was more difficult for planners to assess and evaluate. Sharing best practices, along with revised guidance for planning rule implementation, may help national forest planners improve the utility, efficiency, and quality of forest assessments.

- Terrestrial ecosystems, aquatic ecosystems, and watersheds
- Air, soil, and water resources and quality
- System drivers, including ecological processes, disturbance regimes, and stressors
- Baseline carbon stocks
- Threatened, endangered, proposed and candidate species; potential species of concern
- Social, cultural, and economic conditions
- Benefits people obtain from the planning area (ecosystem services)
- Multiple uses and their contributions to economies
- Recreation settings, opportunities, and access, and scenic character
- Renewable and nonrenewable energy and mineral resources
- Infrastructure (recreational facilities, transportation and utility corridors)
- Areas of tribal importance
- Cultural and historic resources and uses
- Land status, ownership, use, and access patterns
- Existing designated areas including wilderness and wild and scenic rivers; need and opportunity for new designations

Figure 1. Topics for forest plan assessments (36 CFR 219.6)

are expected to incorporate state-of-the-art scientific knowledge to manage public lands (Lachapelle et al. 2003). However, the role of science in natural resource decision-making has become much more complex (Mills and Clark 2001). Recent literature acknowledges that no important policy issue or decision is purely technical, that established practices are problematic, and that politics are unavoidable (Brunner et al. 2005). In spite of this, numerous policies reflect the scientific management paradigm in their calls for best available science.

In the United States, many policies and statutes contain references to best available science, including the Marine Mammal Protection Act of 1972, the Endangered Species Act of 1973, and the Safe Drinking Water Act of 1974. Despite references to the concept of best available science, these policies do not include specific definitions of its properties, standards, or practical application in the decision-making process (Doremus 2004; Smallwood et al. 1999), leading to different definitions of what it means. Ryder et al. (2010) identify attributes of best available science from published literature that span topics such as endangered species legislation, protection of conservation areas, forest management, water resource management, and ocean fisheries. The paper highlights the diversity of attributes assigned to best available science, and demonstrates that no single attribute is common to all studies, suggesting that best available science is context specific (Ryder et al. 2010). Moreover, as Lowell and Kelly (2016) observe, the ability to use best available science may be inhibited by institutional constraints within particular agencies limited by time or organizational capacity. Other literature has attempted to assign descriptors to the concept. For example, "best" often connotes scientific information with the greatest degree of excellence and authenticity based on sound logic (Moghissi et al. 2010), or that there is no better scientific information, and suggests the use of the most relevant and contemporary data and methods (National Research Council 2004). "Available" connotes scientific information that is accessible and attainable (Moghissi et al. 2010), or that decisions can be consistent with the scientific information that is available even though data gaps exist (National Research Council 2004). "Science or Scientific information" is defined as knowledge that emerges from a process of observation, identification, description, and testing of explanatory hypotheses about fundamental principles that govern cause-and-effect (National Research Council 2004). The National Research Council report includes guidelines for effectively using best available science, including concepts of relevance, inclusiveness, objectivity, transparency and openness, timeliness, and peer review. Finally, Charnley et al. (2017) analyzed a science synthesis for three national forests and suggest criteria for evaluating "best available *social* science," which may be different from the criteria used to evaluate best available biophysical science.

A key aspect of the 2012 planning rule is that it requires the planning process to draw on the best available scientific information (36 CFR 219.3). The preamble to the planning rule notes that there is a range of information that can be considered BASI, stating:

> In some circumstances, the BASI would be that which is developed using the scientific method, which includes clearly stated questions, well-designed investigations and logically analyzed results, documented clearly and subjected to peer review. However, in other circumstances the BASI for the matter under consideration may be information from analyses of data obtained from a local area, or studies to address a specific question in one area. In other circumstances, the BASI also could be the result of expert opinion, panel consensus, or observations, as long as the responsible official has a reasonable basis for relying on that scientific information as the best available. (77 FR 21192 [April 9, 2012])

Planning Directives are agency guidance documents that direct implementation of rules such as the 2012 planning rule, and directives for assessments are in Chapter 10 of the Land Management Planning Handbook (USDA Forest Service 2015a). The definition of BASI is contained in the "zero code" chapter of the handbook and specifies three primary criteria for determining BASI: accuracy, reliability, and relevance (FSH 1909.12.07.12), in addition to referencing the Data Quality Act (PL 106–554) for guidance on evaluating available information (Figure 2). Available is defined as information that currently exists in a form useful for the planning process without further data collection, modification, or validation (FSH 1909.07.01).

The directives also provide guidance regarding sources of scientific information. The sources mentioned in the guidance include peer-reviewed articles, scientific assessments, other scientific information (expert opinion, panel consensus, inventories, or observational data), data prepared and managed by the Forest Service

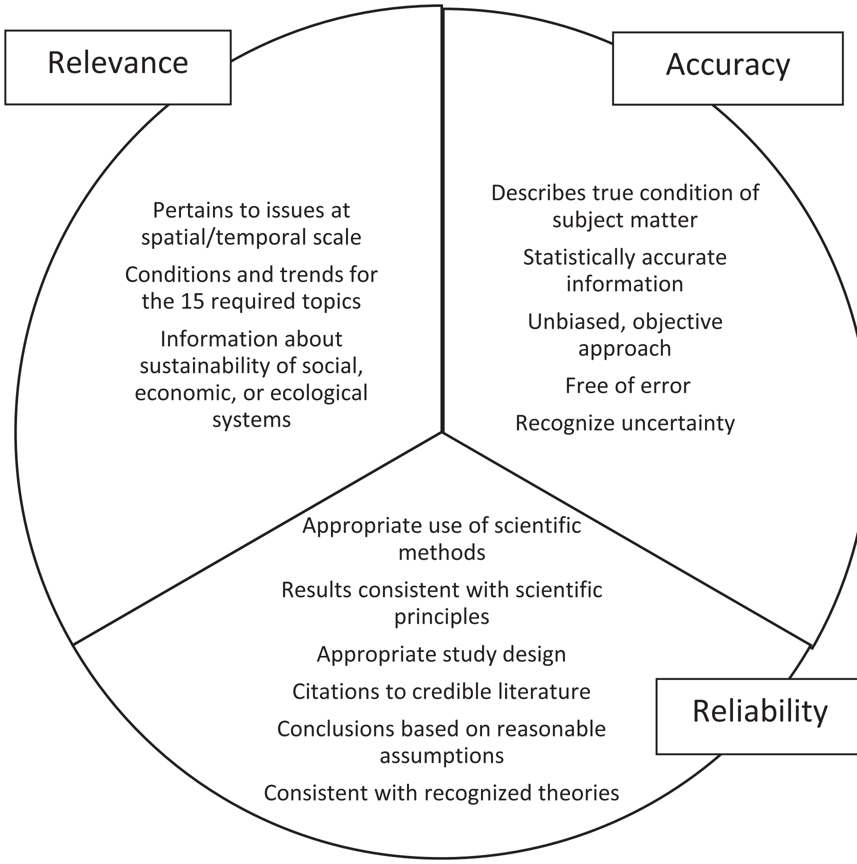

Figure 2. Criteria for determining best available scientific information (BASI). Source: Forest Service Handbook 1909.12.07.12

or other federal agencies, information prepared by universities, national research networks, and other reputable scientific organizations, and data or information from public and governmental participation (FSH 1909.12.07.13).

At the US Forest Service, two regional science synthesis efforts were initiated to assist forest planners in identifying BASI for their assessments. The first synthesis included the Sierra Nevada, southern Cascades, and Modoc plateau areas of California, and informed plan revisions on three national forests (Long et al. 2014). The second synthesis is currently underway as part of the Northwest Forest Plan area planning process, which covers 17 national forests and five Bureau of Land Management units across parts of the Cascade and coastal ranges of Washington, Oregon, and northern California. Once drafted, the synthesis report underwent independent third-party peer review, in addition to public review, and is currently under revision (Spies et al. 2017). Science synthesis efforts represent a noteworthy approach to developing BASI for use in forest assessments, creating a role for public engagement, and for employing a bioregional approach to assembling the latest science for use by multiple forests.

Methods

We used an exploratory case study approach to examine four national forest planning units that were revising their forest plans under the 2012 rule. Information on the USFS website helped us determine the planning status of each national forest as of spring 2015. The primary selection criterion was completion of the assessment process by spring 2015. We also strove to select national forests from different regions. Based on these criteria, we selected the Chugach National Forest (Alaska), Cibola National Forest (New Mexico), Inyo National Forest (California), and the Nantahala and Pisgah National Forests (North Carolina). Table 1 displays characteristics of each national forest planning unit in our sample.

Our research approach relied on content analysis of documents and interview data. We began by conducting a chapter-by-chapter analysis of each forest's assessment report to identify and characterize the information presented. We recorded page counts for each of the 15 assessment topics specified in the 2012 rule. In some cases, the chapters directly aligned with the required topics (Figure 1). In other cases, we had to make a more subjective characterization of the chapter contents. We also noted and analyzed any references to the use of best available science.

Second, as part of the document review, we analyzed data sources used in the assessment. For each assessment report, we identified all of the items cited in the reference section. We then coded each cited item according to the type of publishing entity and the type of document. Every cited item was placed in one category for each coding exercise. For each cited item, we determined the appropriate categories by examining the information in the citation entry and (when necessary) directly reviewing the item or gathering information on the publishing entity. We grouped publishing entities into five types: government; non-government; scientific, scholarly, or peer-reviewed; universities; and unknown or other (Table 2). This categorization approximates the rigor of scientific review, but there is overlap in categories. Most scholarly journals require a double-blind peer-review process, where reviewers and authors are

Table 1. Characteristics of national forests in the study.

Management unit(s)	Geography	Total acreage* (millions of acres)	Notes on use and resources	Designated early adopter?	Most recent previous plan revision	Notes on current plan revision
Chugach National Forest Alaska Region (R10)	Southcentral Alaska: major geographic areas are Cooper River, Prince William Sound, and eastern Kenai Peninsula	6.26	Subsistence, timber, recreation, mining. Human use concentrated in Kenai area. Very limited road coverage and use in other areas. Habitat for all 5 Pacific salmon species	Yes	2002	Managed by a planning team housed within unit
Cibola National Forest Southwest Region (R3)	West-Central New Mexico: Eight noncontiguous parcels organized around distinct mountainous areas known as "sky islands"	2.11	Recreation, timber, cultural heritage, range. Surrounding region experiencing population growth and demographic changes. Pinyon-juniper & ponderosa pine are predominate vegetation types	Yes	1985	Managed by a planning team housed within unit. Does not include 4 associated national grasslands
Inyo National Forest Pacific Southwest Region (R5)	Eastern California & West Nevada: Two noncontiguous parcels at intersection of Sierra Nevada, Great Basin, and Mojave Desert areas	2.07	Water supply, hydroelectricity, recreation, timber, range. Nearly 47% of total area is wilderness. Focus on wildland fire management. Substantial variation in vegetation type, habitat, and elevation	Yes	1988	One of three early adopters in R5. Coordination through a regional planning team, with separate planning teams for each unit. Each unit releases its own assessment & forest plan. Joint EIS for 3 units
Nantahala & Pisgah National Forests Southern Region (R8)	Western North Carolina: Blue Ridge region of Appalachian Mountains	2.48	Timber, recreation, cultural/historical heritage, water development. Located in Blue Ridge National Heritage Area. Hardwood forest with high species diversity	No	1987	Both units will use same revised plan. Managed by planning team housed at NF in NC headquarters

*Total acreage includes NFS-owned land and acreage under other ownership within each unit. Source: USDA Forest Service 2015b.

Table 2. Categories for coding type of publishing entity.

Publishing entity	Description of coding criteria
Government	Federal, tribal, state, or local governments in the United States; foreign governments; international intergovernmental groups such as the United Nations and affiliates. Includes peer-reviewed and non-peer-reviewed materials
Non-government	Materials not published by a government agency, university, or peer-reviewed entity. Includes businesses, consulting firms, and advocacy groups
Scientific scholarly or peer reviewed	Associations, societies, journal publishers, university presses, or other entities that produce peer-reviewed scientific or scholarly material
Universities	Materials from universities that may or may not be subject to rigorous academic peer review. Includes university or college departments, programs, laboratories, and centers, and theses and dissertations from universities
Unknown or other	News organizations or other undefined groups; disposition of publisher could not be determined

unknown to each other. University and government agency scientific documents often require peer review, but the level of rigor of the review may be variable. It was not possible to discern the level or type of peer review or scientific rigor for each category.

For the type of document, we sorted the references into 12 categories: academic book; non-academic book; conference proceeding; correspondence; database; scientific journal; news; technical report; statute or regulation; thesis or dissertation; website; and unknown (Table 3).

Our final data collection activity was qualitative interviewing with members of the planning teams at three of the forests in our study. We conducted nine semi-structured interviews (nine people in total; three interviews each from three forests). Unfortunately, we were not able to recruit interview participants from the Cibola planning effort. Potential interview participants were identified through the list of preparers included in each assessment document. Interviewees were subject matter experts who had contributed material to the assessment reports, along with planning staff officers or coordinators. Interview questions explored the overall structure of the assessment process, the role of the planning directives, the overall organization of the forests' plan revision efforts, and approaches to identification and use of best available science. Interviews were audio-recorded, transcribed, and analyzed using content analysis with a coding framework developed by the study team. Content analysis is a method that uses codes, or labels that assign meaning to descriptive or inferential data collected during a study (Miles et al. 2014). The codes are used to retrieve and organize similar data and aid the researcher in relating data to research questions, theoretical concepts, and themes (Araujo 1995; Miles et al. 2014).

Results

We present results of our analysis in three sections: 1) required topics; 2) sources and types of information; and 3) identifying and using BASI.

Required topics in the forest assessment

The number and percent of pages devoted to each required topic is presented in Table 4. We did not include introductory front matter in the page counts. A 0* entry means that the assessment report did not

Table 3. Categories for coding type of document.

Document type	Description of coding criteria
Academic book	An item printed, bound, distributed as a book, or released as an e-book by a peer-reviewed/scholarly entity
Non-academic book	An item printed, bound, distributed as a book, or released as an e-book by an entity whose primary orientation is not peer reviewed/scholarly
Conference proceeding	Papers, abstracts, and talks presented at a conference and published in a conference proceeding collection
Correspondence	Letters or emails written by individuals of any affiliation
Database	Raw data or data analysis tools/software; online databases
Scientific journal	A peer-reviewed article in a scholarly journal
News	Articles in newspapers (print or online) and news magazines
Technical report	Technical and research reports, white papers, policy papers, fact sheets, briefings
Statute, regulation, and planning documents	Federal, state, or local laws and rules; EISs; management plans; strategic plans
Thesis or dissertation	Advanced degree projects and papers
Website	One or more webpages on a non-database website, including encyclopedias with narrative entries
Unknown	The type of document could not be discerned

Table 4. Page counts and percentages of total pages for 15 required assessment topics.

		Number of pages (pct. of total pages in report)				Pct.
Topic #	Assessment topics (per 36 CFR 219.6)	Chugach	Cibola	Inyo	N&P	Avg.
1	Terrestrial ecosystems, aquatic ecosystems, and watersheds	66 (22.9%)	51.5 (11.2%)	38.5 (21.0%)	29 (15.7%)	**17.7**
2	Air, soil and water resources and quality	17 (5.9%)	88 (19.2%)	9 (4.9%)	19 (10.3%)	**10.1**
3	System drivers (processes, disturbance regimes, and stressors)	40 (13.9%)	21 (4.6%)	15 (8.2%)	7 (3.8%)	**7.6**
4	Baseline carbon stocks	7 (2.4%)	6 (1.3%)	4 (2.2%)	7 (3.8%)	**2.4**
5	Threatened, endangered, candidate species; potential species of conservation concern	12 (4.2%)	36 (7.9%)	24 (13.1%)	4 (2.2%)	**6.8**
6	Social, cultural, and economic conditions	21 (7.3%)	71 (15.5%)	14 (7.7%)	8 (4.3%)	**8.7**
7	Benefits obtained by people (ecosystem services)	49 (17.0%)	0* (0.0%)	2.5 (1.4%)	4 (2.2%)	**5.1**
8	Multiple uses and their contributions to economies	0* (0.0%)	26 (5.7%)	15 (8.2%)	17 (9.2%)	**5.8**
9	Recreation settings, opportunities, and access, and scenic character	29 (10.0%)	39 (8.5%)	15.5 (8.5%)	21 (11.4%)	**9.6**
10	Renewable and nonrenewable energy and mineral resources	17 (5.9%)	18 (3.9%)	3.5 (1.9%)	8 (4.3%)	**4.0**
11	Infrastructure	2 (0.7%)	12 (2.6%)	9.5 (5.2%)	10 (5.4%)	**3.5**
12	Areas of tribal importance	2 (0.7%)	13 (2.8%)	4.5 (2.5%)	3 (1.6%)	**1.9**
13	Cultural and historical resources and uses	3.5 (1.2%)	40 (8.7%)	7 (3.8%)	23 (12.4%)	**6.6**
14	Land status and ownership, use, and access patterns	8 (2.8%)	17 (3.7%)	7 (3.8%)	9 (4.9%)	**3.8**
15	Designated areas, potential/need for new designations	15 (5.2%)	20 (4.4%)	14 (7.7%)	16 (8.7%)	**6.5**
	TOTAL	**288.5**	**458.5**	**183**	**185**	**100**

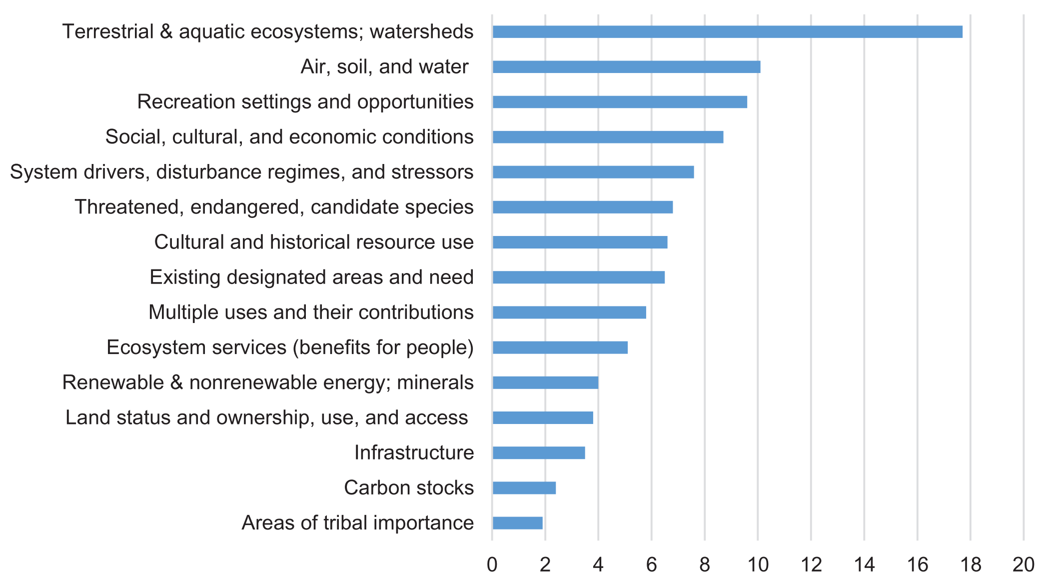

Figure 3. Average percentage of pages devoted to each topic in each forest assessment for all forests combined

have any pages that were specifically devoted to the topic, but references to the topic were instead interspersed throughout the report and it was too difficult to separate them from other topic page counts.

Two of the national forests (Inyo and Nantahala-Pisgah) published assessment reports that consisted of 15 chapters that directly reflected each of the required topics. Meanwhile, the Chugach and Cibola took a different approach; some of the chapter topics aligned with the topic requirements in the 2012 rule, but other required topics were broken up and distributed among multiple chapters. For example, the Chugach had one chapter for areas of tribal importance and one chapter for land status and ownership, but divided the terrestrial and aquatic ecosystems and watersheds

Table 5. Percent allocation of predominant topics among four forest assessments.

Rank	Chugach topics	Pct.	Cibola topics	Pct.	Inyo topics	Pct.	N&P topics	Pct.
1	Terrestrial and aquatic ecosystems	23%	Air, soil, and water	19%	Terrestrial and aquatic ecosystems	21%	Terrestrial and aquatic ecosystems	16%
2	Benefits obtained by people (ecosystem services)	17%	Social, cultural, and economic conditions	16%	Threatened and endangered species	13%	Cultural and historic resources	12%
3	System drivers, disturbance regimes, and stressors	14%	Terrestrial and aquatic ecosystems	11%	Recreation settings and opportunities	9%	Recreation settings and opportunities	11%
4	Recreation settings and opportunities	10%	Cultural and historic resources	9%	System drivers, disturbance regimes, and stressors	8%	Air, soil and water	10%
5	Social, cultural, and economic conditions	7%	Recreation settings and opportunities	9%	Multiple uses	8%	Multiple uses	9%
Total		71%		63%		59%		59%

into five chapters, one each for watersheds, fish, wetlands, vegetation, and wildlife, and these chapters were integrated with material discussing soils and carbon stocks. Two forests did not have any pages specifically devoted to one required topic each (benefits obtained by people for the Cibola, and multiple uses for the Chugach), but these subjects were still referenced in the context of the other topics.

For all four assessments combined, the required topic with the largest average percentage of pages was terrestrial and aquatic ecosystems and watersheds (17.7%), followed by air, soil, and water resources (10.1%) and recreation opportunities (9.6%) (Figure 3).

Terrestrial and aquatic ecosystems and watersheds comprised the largest section of the assessment for three of the four forests. Air, soil, and water was especially prominent for the Cibola National Forest, and all of the forest assessments covered recreation evenly. In contrast, the three required topics with the smallest page counts, on average, were areas of tribal importance (1.9%), carbon stocks (2.4%), and infrastructure (3.4%). Benefits obtained by people (ecosystem services) had the most variable coverage, with one of the shortest sections for three of the four forest assessments, but the second longest topic for the Chugach National Forest. In all four assessment documents, benefits obtained by people were mentioned throughout the document in sentences or paragraphs at too fine a scale for this analysis to count.

We found some variation among the forest assessments in terms of the extent to which a forest focused on a particular topic (Table 5).

For the Chugach National Forest, the top five topics comprised more than 70% of the assessment, with the bulk emphasizing terrestrial and aquatic ecosystems, which reflects the importance of salmon habitat. The Chugach was the only forest to emphasize ecosystem services as a predominant framework to capture benefits obtained by people. However, other forests may have captured this topic under the category of multiple uses. Disturbance regimes (fire and invasive species) were also important for the Chugach. The Cibola National Forest was unique in their emphasis on air, soil, and water as well as social, cultural, and economic conditions and cultural and historic sites. Because water access is very important in the southwest, the predominance of this topic is not surprising. For the Inyo National Forest, the topic of threatened and endangered species was prominent, while topics related to recreation and disturbance regimes (fire, invasive species, and other ecosystem stressors) were also important. Meanwhile, cultural and historical resources were prominent in the Nantahala and Pisgah National Forests, along with recreation.

Although the 2012 rule provides a list of 15 distinct required topics, these topics overlap and are not discussed in complete isolation from one another. As we found in our analysis, it is difficult to discuss multiple uses without also discussing benefits obtained by people; air, soil, and water resources; recreation; and terrestrial and aquatic ecosystems and watersheds. In our analysis, we often found that an assessment chapter devoted to a required topic also contained information that closely resembled material discussed elsewhere. In particular, we found the chapters on multiple uses and benefits obtained by people to be largely redundant, given the other topics that were also included in the report.

Sources and types of information in the forest assessment

To understand the sources and types of information used in the assessments, we conducted a systematic examination and tally of citations by publication source and type. Overall, government sources were the most commonly cited information source (51.8%), followed by scientific scholarly publications (30.7%) (Table 6).

Table 6. Citations based on information source for forest assessments.

	Count (Percent)				
Publishing entity	Chugach	Cibola	Inyo	Nantahala & Pisgah	TOTAL (Mean)
Government	239 (53.6%)	159 (49.8%)	131 (49.8%)	109 (54.0%)	**638 (51.8%)**
Scientific scholarly or peer reviewed	155 (34.8%)	82 (25.7%)	82 (31.2%)	63 (31.2%)	**382 (30.7%)**
Non-government	21 (4.7%)	39 (12.2%)	24 (9.1%)	18 (8.9%)	**102 (8.7%)**
Universities	30 (6.7%)	39 (12.2%)	19 (7.2%)	11 (5.5%)	**99 (7.9%)**
Unknown or other	1 (0.2%)	0 (0%)	7 (2.7%)	1 (0.5%)	**9 (0.9%)**
TOTAL	446	319	263	202	**1230**

Table 7. Citations based on document type for forest assessments.

	Count (Percent)				
Document type	Chugach	Cibola	Inyo	Nantahala & Pisgah	**TOTAL**
Technical report	174 (39.0%)	121 (37.9%)	108 (41.1%)	73 (36.1%)	**476 (38.5%)**
Scientific journal article	129 (28.9%)	47 (14.7%)	63 (24.0%)	48 (23.8%)	**287 (22.8%)**
Academic book	28 (6.3%)	36 (11.3%)	20 (7.6%)	15 (7.4%)	**99 (8.2%)**
Statute, regulation, or planning document	43 (9.6%)	26 (8.2%)	23 (8.8%)	12 (5.9%)	**104 (8.1%)**
Website	33 (7.4%)	42 (13.2%)	3 (1.1%)	13 (6.4%)	**91 (7.0%)**
Database	17 (3.8%)	25 (7.8%)	17 (6.5%)	18 (8.9%)	**77 (6.8%)**
Conference proceeding	10 (2.2%)	3 (0.9%)	6 (2.3%)	18 (8.9%)	**37 (3.6%)**
Non-academic book	4 (0.9%)	9 (2.8%)	10 (3.8%)	0 (0.0%)	**23 (1.9%)**
Correspondence	0 (0.0%)	7 (2.2%)	5 (1.9%)	4 (2.0%)	**16 (1.5%)**
Thesis or dissertation	8 (1.8%)	2 (0.6%)	2 (0.8%)	1 (0.5%)	**13 (0.9%)**
News	0 (0.0%)	1 (0.3%)	3 (1.1%)	0 (0.0%)	**4 (0.4%)**
Unknown	0 (0.0%)	0 (0.0%)	3 (1.1%)	0 (0.0%)	**3 (0.3%)**
TOTAL	446 (100.0%)	319 (100.0%)	263 (100.0%)	202 (100.0%)	**1230 (100.0%)**

A large portion of the government sources included US Forest Service publications (average of 28%), which were more commonly cited than other federal government sources (average of 12%) or state and local governments (average of 11%). Some variation exists among the forests in our sample, but the trends were consistent in terms of reliance on government sources and scholarly peer-reviewed publishers for the majority of citations (82.5% combined average for both categories). The Chugach relied to a greater degree on scholarly publications than other forests. The Cibola had the highest proportion from non-governmental organizations and trade groups (12.2%). The Inyo and the Nantahala and Pisgah mirrored the group average.

Next, we explored citations by the type of document referenced. We found that technical reports were the most common type of document cited in the assessments, with an average of 38.5% (Table 7).

The technical report classification is broad and includes technical and scientific reports, policy briefings, white papers, and other types of information (sometimes referred to as gray literature). All four forests were consistent in the ratio of technical reports cited. The second most common document type was the scientific journal article, with an average of 23%, although the Cibola assessment featured far fewer than the other forests. All of the forests cited a wide variety of regulations, statutes, and planning documents, (e.g., water quality regulations, county comprehensive plans, environmental impact statements, state resource management plans, and forest plans). The Cibola assessment featured the greatest variety of document types, relying on websites and academic books more than the other forests. The Nantahala and Pisgah assessment relied more heavily on conference proceedings. The least commonly cited document types, on average, were news articles (0.4%), theses or dissertations (0.9%), and correspondence (1.5%). Although there is a separate category for websites, documents in many of the other categories were readily available online.

Table 8. Approaches to identifying and using BASI from interview data.

BASI approach	Chugach	Nantahala/ Pisgah	Inyo
Literature review (e.g. Google Scholar for scholarly literature)	x	x	x
Forest Service reports, monitoring data	x	x	x
Personal expertise/training/judgement	x	x	x
Existing dataset/database	x		x
Nearby Forest Service research station		x	x
Nearby university		x	
Host data sharing meeting (partners and stakeholders)		x	
Meet with scientists		x	
Post draft documents on wiki site for public review/editing			x
Other public review opportunity		x	
Gray ("non-peer-reviewed") literature, traditional knowledge			x

Identifying and using best available scientific information in the forest assessment

In interviews, respondents were asked how they identified and obtained BASI for their assessment. Table 8 displays the different approaches used by three of the four forests.

Literature reviews and searches, Forest Service reports and datasets, and personal scientific expertise were mentioned by all nine respondents as primary ways that they identified and obtained BASI. Literature reviews focused on identifying peer-reviewed journals, conference proceedings, or agency reports. Existing datasets and nearby Forest Service research stations and universities were also relied upon. The Sierra Nevada science synthesis effort, which informed the Inyo National Forest assessment, took nearly 18 months to complete (Long et al. 2014). The Inyo also posted draft documents on a wiki site for public review and editing. All nine interviewees stated that their assessment team used the Draft Planning Directives, but also mentioned that the directives were not clear, save for the focus on organizing around the 15 topics. No respondent mentioned specific guidance beyond the draft directives on how to identify BASI. The final directives do specifically address the definition of BASI, as discussed above (Figure 2). Gray literature and traditional knowledge presented challenges, as it at times conflicted with peer-reviewed information. Two respondents mentioned that they wanted to incorporate this type of information, but were unsure how to do so.

Assessments must document what information was determined to be BASI, explain the basis for that determination, and explain how the information was applied to the issues considered (36 CFR

219.3). Our analysis of the assessment documents reveals that all documents discuss the use of high-quality and valid scientific information, citing criteria such as clearly defined and well-developed methodology; standardized methodology; logical conclusions; and reasonable inferences (Chugach National Forest 2014; Inyo National Forest 2014; Nantahala and Pisgah National Forests 2014; Cibola National Forest and National Grasslands 2015). The assessments for all forests mention their reliance on information relevant to their specific forests and issues. Only the Nantahala-Pisgah assessment presented a hierarchy of information sources, with peer-reviewed journal articles the highest, followed by government documents and reports, monitoring datasets, theses and dissertations from universities, and expert opinion where facts were not known through the other sources.

Discussion

The 2012 forest planning rule requires that each national forest or grassland conduct a scientific assessment to guide plan development. We found that assessment reports were disproportionately heavy in science related to terrestrial and aquatic ecosystems, and more limited in treatment of infrastructure, land ownership and access patterns, cultural heritage, and areas of tribal importance. Recreation was the only topic to receive consistent attention across all four forests, although the topic was overshadowed by terrestrial and aquatic ecosystems. We may only speculate about why terrestrial and aquatic ecosystem information was the most prevalent in all four forests, but it is consistent with agency administrative hiring practices since the 1980s that have emphasized recruitment of ecologists, biologists, and other biophysical scientists, compared to social scientists, for example (Thomas and Mohai 1995). The abundance of agency specialists in these topic areas may reinforce the relative importance of terrestrial and aquatic ecosystems compared to other topic areas, such as recreation, social science, or cultural resource management. This has been confirmed by a national assessment of interdisciplinary planning team composition (Cerveny et al. 2011). Ensuring that assessment teams include broad and diverse disciplinary experts will help address this challenge, recognizing that some forests may not have access to necessary disciplinary specialists. It is also possible that some of the topics (e.g., ecosystem services, tribal and cultural resources, land status and use patterns) simply do not have as much relevant and available information as other topics.

The benefits obtained by people (ecosystem services) topic received little or no explicit coverage in all but one assessment. The limited coverage of ecosystem services may make sense because it was not even considered an area of research until the late 1990s, so there would be less existing information on certain important ecosystem service topics (e.g., pollination, stormwater attenuation, medicinal resources, and spiritual and historical significance) compared to recreation, threatened and endangered species, and other traditional assessment topics (Blahna et al. 2017). Previously, "forest benefits to people" were considered elements of "multiple use" and planners might have addressed these benefits under the "multiple use" topic. Ecosystem services (ES) are often categorized into four classes: provisioning, regulating, cultural, and supporting. Timber, recreation, wildlife, and other traditional forest planning topics all fall into one of these four classes. Another reason for lack of coverage of ecosystem services may be that planners could not differentiate the normal assessment topics from the ecosystem service classes. Efforts to help planning team members understand ecosystem services approaches and how they can be used to inform the planning process may be warranted, and the rule's current requirement for only using existing data in assessments may need to be revisited (Blahna et al. 2017). For example, implementation teams working on ecosystem services may consider the benefits of providing specific tools, frameworks, and guidelines for integrating ecosystem services models into the forest planning process. In addition, critical issues and topics (e.g., newly listed threatened or endangered species, or changing recreation behaviors) that forest plans need to address may change from one planning cycle to the next.

The specific required topics may not be universally appropriate for every planning unit. Planners felt obligated to address all 15 topics, but the lack of coverage for some topics suggests that the topic was not deemed relevant or meaningful for their plan, there was no available data on the topic, or it was unclear how the topics could be covered. Variability in application of the directives, and acknowledgment of local context and conditions, is consistent with the overall Forest Service approach toward decentralized decision-making (Kaufman 1960; Tipple and Wellman 1991; Koontz 2007) and localized interpretation by planning teams, similar to "street-level" bureaucrats who create de facto policy through everyday practice (Sabatier et al. 1995; Lipsky 2010; Trusty and Cerveny 2012). Kaufman (1960) observes the traditional Forest Service practice of maintaining control of heterogeneous and geographically dispersed management units by issuing centralized directives that provide parameters (or "side boards") within which line officers have some leeway to make decisions. This tendency toward uniformity and "pre-formed" decisions may result in some inefficiencies and omissions. The implied obligation to cover all 15 topics may have resulted in some assessments that distract from the most important management issues for the unit. This will be especially important during the next stage of planning—revision or amendment—where the assessment data will be used to analyze different management scenarios. Approaches for identifying and analyzing the most relevant assessment data that address the key environmental problems or social conflicts that confront each planning unit will be needed (Blahna et al. 2017). This is especially important for topics like human benefits (ecosystem services) and multiple uses, which cut across all of the other topical areas and are not as easily categorized in assessments. Recent efforts to engage the public in science synthesis efforts in support of forest planning suggest that there may be an important role for the public to help prioritize forest assessment topics.

The most common sources of information were government sources, followed by scholarly academic sources. Many of the agency sources were peer-reviewed scientific studies, which appear to be especially useful because of the topical specificity or geographic focus (relevance). Although not all technical reports are peer reviewed, they may be more accessible and usable compared to scholarly journal articles, which may require planning team members to interpret the findings and make inferences for relevance to local conditions. This finding is consistent with previous research examining the information needs and sources of Forest Service fire managers (Ryan and Cerveny 2011) and recreation managers (Ryan and Cerveny 2010). Fire managers relied heavily on agency information sources. Although managers in the study noted the availability of high-quality, relevant information, they faced significant barriers in terms of time, funding,

and personnel to access and use that information. Similarly, recreation managers also relied on agency information sources, but indicated strong preferences for enhanced interactions with agency scientists, including collaborative research, conferences, and a desire for agency researchers to reach out more directly to managers to ensure their research was relevant and useful. With regard to forest assessments, engagement with scientists is particularly important for topics where little research is available. Assessment teams may want to consider additional ways to interact with scientists and others to create functioning communities of practice related to science exchange for forest planning. In the same way, agency scientists may consider forging new and enduring relationships with planners and managers that could generate new science that is of immediate relevance.

The 2012 planning rule and its directives provide criteria for BASI, and we found similarities across all forests in the most common approaches to identifying BASI, in addition to other approaches, such as data sharing meetings, a wiki review site, and requests for a science synthesis. Information from non-peer-reviewed sources was more difficult for planners to assess and evaluate, and it is not clear how this information was incorporated into each assessment. Teams may not have the capacity to separately evaluate and assess the many different types and sources of information, and so they rely on hierarchical ranking approaches (peer-reviewed sources being highest rank) to streamline the evaluation. Planning teams clearly value peer-reviewed and agency-generated information, and it may be that they are simply identifying information that is "available" and using the "best" of that based on their judgments. This may result in situations where the science expertise on each team could influence BASI decisions. As discussed above, consideration of the makeup and membership of the assessment team is important here, as well as increased transparency regarding the process for determining science relevance and quality.

Conclusion

Implementation of the US Forest Service 2012 planning rule is still in its early stages. Our study illustrates that forest planners use a variety of approaches to address required topics, and do rely on BASI as they develop their forest assessments. While each national forest assessment included the 15 required topics, we found considerable variation in coverage, which suggests that planners may emphasize topics most relevant to their forest, or that variation exists in terms of what science or planning team expertise is available or deemed desirable. The predominance of science related to terrestrial and aquatic ecosystems in the assessments compared to other topics warrants further inquiry in order to learn whether this asymmetry is based on policy, availability of information, existing expertise, or other factors. Efforts to include the public in the process of prioritizing topics for the assessments could also be evaluated. The reliance on government sources for scientific information suggests that agency-supported science is either more accessible or more relevant to the planning team. It also suggests that there may be benefits to bolstering "communities of practice" for key topical areas covered by forest assessments that bring together university and agency scientists with managers.

The appearance of science in an assessment report is important, but the actual *use* of science in planning may be more important. Although our findings are not generalizable to all national forests, they do provide an understanding of plan assessment activities for those in the early phases of forest planning, whose efforts are likely to inform and influence other national forests. Our goal was to provide an early glimpse of plan revision efforts in order to highlight important lessons learned and create a foundation for future research. For example, do planners find that the required topics provide useful guidance for developing their assessments? How can planners become more confident in knowing what BASI is, and how to identify and use it? Is additional guidance needed for incorporation of traditional knowledge and other information? Of particular interest is whether the "science synthesis" information is useful to forest planners in addressing their forest assessment needs, given the significant agency resources devoted to developing science syntheses. Finally, how is information from the assessment used in forest plan revision (development and selection of management options) and monitoring efforts? While draft environmental impact assessment (EIS) reports are available in various stages, as of this writing only one final Record of Decision (ROD) has been issued for a forest plan undergoing revision under the 2012 rule. Thus, it remains to be seen how scientific information will be incorporated in development of alternatives, impact statements, and final management decisions.

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The Interaction of Fire, Fuels, and Climate across Rocky Mountain Forests

TANIA SCHOENNAGEL, THOMAS T. VEBLEN, AND WILLIAM H. ROMME

Understanding the relative influence of fuels and climate on wildfires across the Rocky Mountains is necessary to predict how fires may respond to a changing climate and to define effective fuel management approaches to controlling wildfire in this increasingly populated region. The idea that decades of fire suppression have promoted unnatural fuel accumulation and subsequent unprecedentedly large, severe wildfires across western forests has been developed primarily from studies of dry ponderosa pine forests. However, this model is being applied uncritically across Rocky Mountain forests (e.g., in the Healthy Forests Restoration Act). We synthesize current research and summarize lessons learned from recent large wildfires (the Yellowstone, Rodeo-Chediski, and Hayman fires), which represent case studies of the potential effectiveness of fuel reduction across a range of major forest types. A "one size fits all" approach to reducing wildfire hazards in the Rocky Mountain region is unlikely to be effective and may produce collateral damage in some places.

Keywords: fire ecology, forest management, forest health, Rocky Mountain forests, climate

The interaction between climate, fuels, and the frequency and severity of wildfires across Rocky Mountain forests is complex. A comprehensive understanding of the relative influence of fuels and climate on wildfires across this heterogeneous region is necessary to predict how fires may respond to a changing climate (Dale et al. 2001) and to define effective fuel management for controlling wildfires in this increasingly populated region (USDA 2002). The annual area burned by wildfires has apparently increased during the last few decades across North America, and in the southern Rocky Mountain region in particular, possibly in response to recent climate change and the gradual accumulation of fuels following decades of effective fire suppression (figure 1; Grissino-Mayer and Swetnam 2000). However, more complete modern records, and an increase in land under federal protection since the 1960s, may also have contributed to this apparent trend over the last half-century. Nonetheless, the United States recently experienced a series of big fire years: According to the National Interagency Fire Center (*www.nifc.gov*), wildfires in 1988, 2000, and 2002 burned 3.0 million, 3.4 million, and 2.8 million hectares (ha), respectively, Most of these fires took place in the western United States, which is characterized by fire-prone ecosystems.

In an effort to mitigate the risk to life and property from wildfires and the high cost of fighting fire throughout the western United States, fuel reduction has become an important forest and fire management tool. In 2002, thinning and prescribed-fire projects were carried out across 1 million ha of federal land as part of the US National Fire Plan (*www.fireplan.gov*) to reduce the fire hazard and to restore historical species composition and stand structures. The goals of fire-hazard reduction and ecological restoration may converge in some ecosystems, yet they may be incompatible in others (Veblen 2003).

The idea that decades of fire suppression have promoted unnatural fuel accumulation and subsequent unprecedentedly large, severe wildfires across western forests was developed primarily from experience in dry ponderosa pine (*Pinus ponderosa*) forests in the US Southwest, the interior West, and the Sierra Nevada (Covington and Moore 1994, Caprio and Swetnam 1995, Moore et al. 1999). Historically, short-interval, low-severity surface fires maintained sparse, open stands in most dry ponderosa pine forests (Swetnam and

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Baisan 1996). With fire suppression, young fire-intolerant trees can establish during lengthened fire intervals. Denser stands provide "ladder" fuels at intermediate heights that carry fire up into continuous canopy fuels, promoting unprecedentedly large, catastrophic fires. This system has presented a strong case for thinning to reduce the fire hazard and to restore historical stand structure.

Ecological restoration and fire mitigation are urgently needed in dry ponderosa pine forests, where previous research supports this management action. However, we are concerned that the model of historical fire effects and 20th-century fire suppression in dry ponderosa pine forests is being applied uncritically across all Rocky Mountain forests, including places where it is inappropriate (e.g., USDA 2002, White House 2002). Of particular concern is President Bush's Healthy Forests Initiative, which identifies unnatural fuel buildup as a widespread risk across the West: "Today, the forests and rangelands of the West have become unnaturally dense, and ecosystem health has suffered significantly. When coupled with seasonal droughts, these unhealthy forests, overloaded with fuels, are vulnerable to unnaturally severe wildfires. Currently, 190 million acres [77 million ha] of public land are at increased risk of catastrophic wildfires" (White House 2002, executive summary). This initiative was recently enacted as HR 1904, the Healthy Forests Restoration Act of 2003.

The relative contribution of fuels and climate to recent fire activity across forest types throughout the western United States is hotly debated (e.g., see *Conservation Biology,* vol. 15 [2001]). It is easy to identify either local situations in which fire suppression has allowed unusual fuel accumulations or, by contrast, those in which fuel conditions remain within the historical range and the effects and frequency of fire are controlled primarily by weather conditions, not by fuels. What is lacking is a broad synthesis of the geographical patterns in historical fire regimes, and of 20th-century changes in these regimes, addressing these key questions:

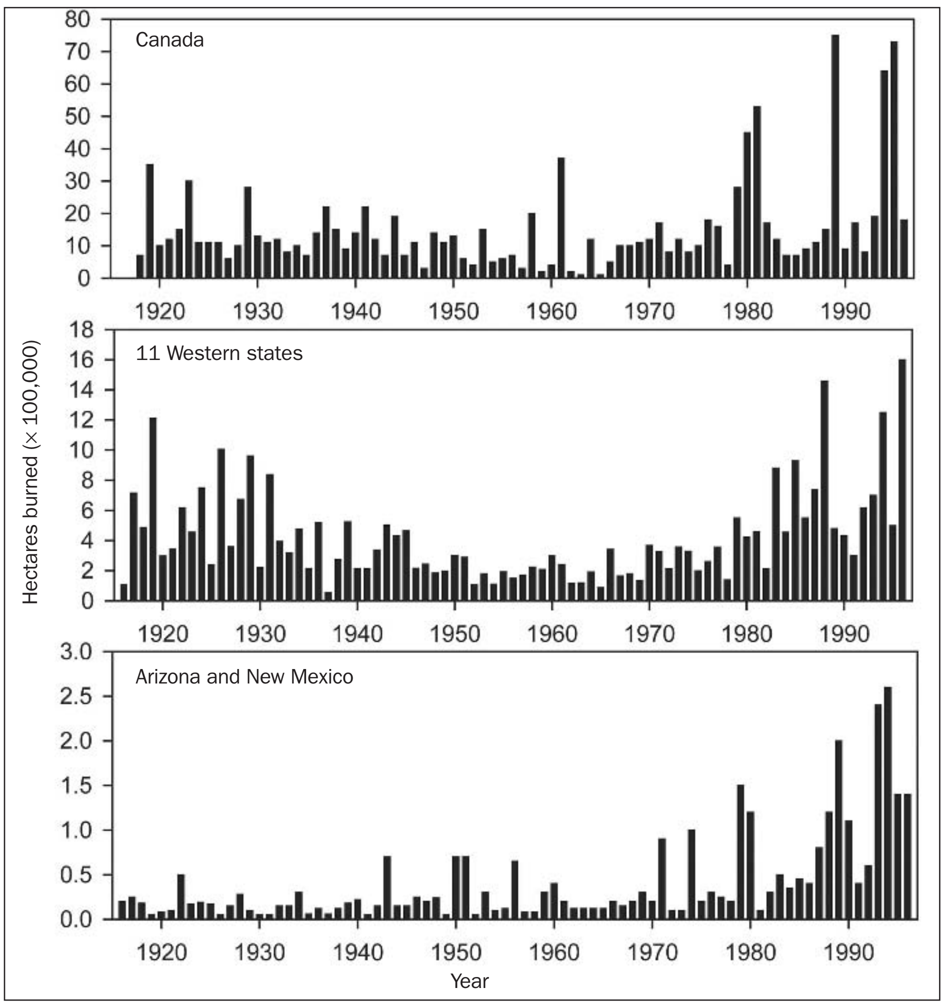

***Figure 1. Area burned by wildfires in different regions under federal protection across North America. The apparent increase in the extent of fires over the last century is most pronounced in the southwestern United States (Arizona and New Mexico), although we urge caution in interpreting these trends. Source: Grissino-Mayer and Swetnam (2000); reprinted with permission from* The Holocene.**

- Where, in what ecosystem types, and to what degree have fuels increased with fire suppression across the Rocky Mountain region (Arizona, New Mexico, Colorado, Utah, Wyoming, Montana, and Idaho)?
- Where are forest restoration treatments appropriate, and how will fire respond to fuel-reduction treatments in different forest types?
- Where and when is the influence of short-term (i.e., seasonal and annual) climatic variation expected to override the effectiveness of fuel treatments?

To address these questions, we synthesize current understanding of the different types of fire regimes (defined by the historical range of variability in fire size, severity, and frequency) that occur across the Rocky Mountain region. The fire regime is a central concept in fire ecology and is essential for understanding the character, effect, and variability of disturbance patterns across regions. Our analysis of different fire regimes is based on the classic fire triangle of weather, fuels, and ignition, which identifies the factors controlling combustion. All three factors must be present in a form conducive to combustion, or fire will not occur. However, the inherent variability, and therefore the limiting role, of these three ingredients is dramatically different among forest types and geographic regions. For example, we argue below that fuel types and amounts are less limiting to fire spread in subalpine forests than in low-elevation forests, but suitably dry weather conditions for fire spread in subalpine forests occur infrequently. Hence, variability in seasonal and annual climate is more limiting and has a greater influence on fire extent and severity in these generally cool, moist ecosystems.

In contrast, periods of several months of warm, dry weather occur almost annually in most southwestern ponderosa pine forests, leaving fuels sufficiently desiccated for extensive fires to occur annually. Given the higher frequency of weather conditions that desiccate fuels in this ecosystem, factors that affect fuel type, quantity, and configuration are more limiting than climate in controlling this fire regime. Variations in local site productivity, and in the time elapsed since the last fire event, affect fuel accumulation in the dry, low-elevation ponderosa pine forests. Annual climatic variation affects fuels indirectly in these forests both through short periods of above-average moisture availability, which enhance the production of fine fuels (e.g., leaves, grasses, forest litter), and through fuel-desiccating drought. But overall, climate is more limiting in subalpine forests, where short-term (i.e., months to a few years) variability in climate primarily affects fire severity and spread through fuel desiccation and wind, not fuel abundance. In contrast, the fire regime in dry ponderosa pine woodlands is more limited by annual variability in fine fuel amounts and by ladder-fuels related to the time elapsed since the last fire. Ignition sources also may be important, at least locally, but in this study we do not identify spatial patterns in this component of the fire regime. Assuming instead that ignition sources are always available, we evaluate the relative importance of variability in short-term climatic variation and in fuel quantity and configuration.

We identify three major types of historical fire regimes (Agee 1998): (1) high severity, (2) low severity, and (3) mixed severity. In addition to developing a general theoretical framework for assessing controls on local fire regimes, we summarize the lessons learned from three recent large wildfires (the 1988 Yellowstone fires and the 2002 Rodeo-Chediski and Hayman fires). These case studies reveal the potential effectiveness of fuel reduction under varying climate conditions across a range of major forest types and historical fire regimes. Finally, we develop coarse estimates of the spatial extent of the three major historical fire regimes to broadly quantify heterogeneity in fire regimes and responses to fire suppression across the Rocky Mountain region.

To develop coarse estimates of the proportion and extent of historical fire regimes across the Rockies, we rely on research reported in the peer-reviewed literature to group major forest types that historically experienced each of the three major fire regimes we discuss. Because it is relatively difficult to define the spatial extents of different fire regimes at this scale, we rely on two independent maps of forest cover to highlight general trends and degrees of uncertainty in the relative proportion of major fire types across the Rocky Mountain region. In the first analysis, forest types are based on a map of Küchler's potential natural vegetation (PNV) groups (climax vegetation types that are expected, given the occurrence of natural disturbances such as fire, based on site characteristics such as soils, climate, and topography), modified by Schmidt and colleagues (2002). In our reclassification of these data, we combine eight PNV groups into three main forest types: (1) ponderosa pine (pine forest and Great Basin pine), (2) mixed ponderosa pine (pine–Douglas fir, Douglas fir, grand fir–Douglas fir, and Southwest mixed conifer [Arizona, New Mexico]), and (3) spruce–fir (spruce–fir and spruce–fir–Douglas fir). In the second analysis, forest types are based on a map of current cover types, which Schmidt and colleagues (2002) developed by combining the Forest and Range Resource Planning Act map of US forest type groups with AVHRR (Advanced Very High Resolution Radiometer) satellite imagery. In our reclassification of these data, we combine the current cover types into three main forest types, similar to those obtained by combining the PNV groups: (1) ponderosa pine, (2) Douglas fir, and (3) spruce–fir–lodgepole pine.

In this summary, we assume a one-to-one correspondence between forest types and fire regimes; however, as we emphasize throughout the text, this is a considerable oversimplification. Nonetheless, this summary reveals coarse levels of heterogeneity in fire regimes across the Rocky Mountain region, unaccounted for in current forest policy debates. Other endeavors to define fire regimes at this scale include the work of Schmidt and colleagues (2002), who developed a map of historical fire regimes and departures from historical conditions throughout the continental United States for strategic fire-planning purposes, but who relied primarily on managers' expert knowledge rather than on peer-reviewed empirical studies in defining fire regimes. In addition, McKenzie and colleagues (2000) developed a regional model of fire frequency within the interior Columbia River basin, based on a large fire-history database from the western United States.

Overall, our analysis highlights the heterogeneity of forest types and fire regimes across the Rocky Mountain region. Further, it provides insight into pressing management questions of when and where various fuel treatments are consistent with the goal of ecological restoration, and where such treatments are likely to be successful in reducing the size and severity of wildfires. We focus on the Rocky Mountain region; however, the spatial and geographic heterogeneity in fire regimes across this region is also evident throughout the West (e.g., Agee 1998).

High-severity fire regimes

High-severity or stand-replacing fires are defined by the death of canopy trees, in contrast to low-severity fires, which do not kill overstory trees. High-severity fires typically burn the treetops (crown fires) but may also kill trees through very hot surface fires, which primarily burn the forest floor.

High-elevation subalpine forests in the Rocky Mountains typify ecosystems that experience infrequent, high-severity crown fires (Peet 2000, Veblen 2000). The forest types that occur in the subalpine zone range from mesic spruce–fir forests to drier, dense lodgepole pine stands; and xeric, open woodlands of limber and bristlecone pine. The most extensive subalpine forest types are composed of Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), and lodgepole pine (*Pinus contorta*), all thin-barked trees easily killed by fire.

Extensive stand-replacing fires occurred historically at long intervals (i.e., one to many centuries) in subalpine forests (Romme 1982, Kipfmueller and Baker 2000, Veblen 2000, Schoennagel et al. 2003), typically in association with infrequent high-pressure blocking systems that promote extremely dry regional climate patterns (Romme and Despain 1989, Renkin and Despain 1992, Bessie and Johnson 1995, Nash and Johnson 1996). Persistent high-pressure blocking systems affect regional temperature and precipitation patterns throughout the Rockies and may respond to global climate anomalies (Baker 2003). Regional synchrony of large, high-severity fires across subalpine forests corroborates the idea that high-elevation forest fires respond to broad scale synoptic climate (Nash and Johnson 1996, Kipfmueller and Baker 2000, Veblen 2000, Baker 2003). In moist high-elevation forests, successive seasons of drought can initiate large, stand-replacing fires (Balling et al. 1992, Kipfmueller and Swetnam 2000). In these generally cool subalpine environments, significant drought events are infrequent, which prevents the frequent occurrence of large, high-severity fires. Although they occur infrequently, drought-induced large fire events account for the greatest percentage of the area burned in subalpine forests (figure 2; Bessie and Johnson 1995).

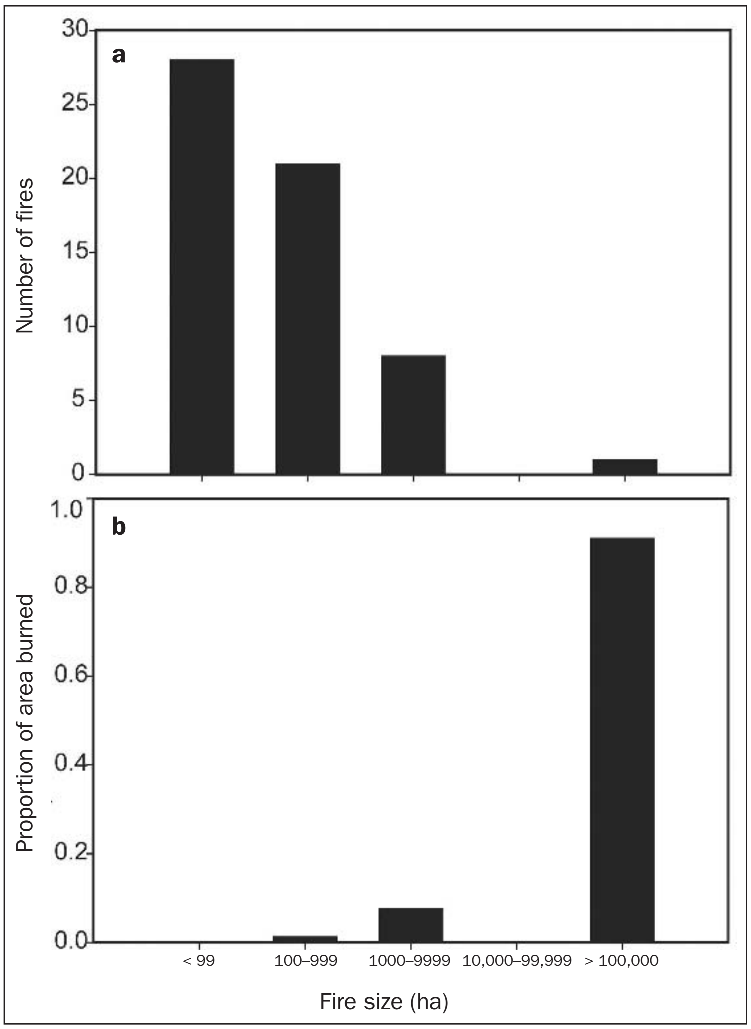

Figure 2. (a) Histogram of the occurrence of different size classes of stand-replacing fires in Yellowstone National Park (1895–1991). (b) Proportion of the total area burned in each size class for the same period (1.0 = 100% of total area). Although large stand-replacing fires (i.e., fires that burn more than 1000 hectares) are infrequent, they are the dominant influence on subalpine forests. Data are from Balling and colleagues (1992).

Subalpine forests typically experience stand-replacing crown fires, rather than low-severity surface fires, because they lack fine fuels on the forest floor but have abundant ladder fuels that carry fire into the treetops. These dense, closed-canopy forests typically support sparse understory vegetation, and the short, stout needles of subalpine trees compact tightly on the forest floor, creating a poor substrate for fire spread (Swetnam and Baisan 1996). This is in stark contrast to the warmer, open-canopied, productive forests at lower elevations, which support abundant, well-aerated fine fuels on the forest floor (Swetnam and Baisan 1996). Although fine surface fuels are sparse in subalpine forests, ladder fuels are abundant. Shade-tolerant fir and spruce trees have abundant lateral branches, which easily carry fire up into the canopy. By contrast, shade-intolerant lodgepole pines have few lateral branches, but these trees tend to grow in very dense stands that thin over time, contributing to abundant dead ladder fuels (figure 3). The abundance of ladder fuels, the proximity of crowns, and the lack of abundant, spatially continuous fine surface fuels all promote high-severity crown fires that dominate subalpine forests.

The low abundance of small fuels, and the relatively high abundance of large dead and live fuels, explains why fires are infrequent but typically large in subalpine forests. Fuel moisture levels respond to ambient environmental conditions and are critical in determining fire potential. Small-diameter dead fuels dry quickly; for example, 1-hour fuels (particles less than 0.6 centimeters [cm] in diameter) approach equilibrium with ambient relative humidity within an hour. By contrast, dead branches, logs, or other large, slow-drying materials (7.6 to 20.3 cm in diameter) are known as 1000-hour fuels because they require 1000 hours to equilibrate (figure 4). Live fuels dry even more slowly than dead fuels and are influenced most strongly by sustained periods of drought. Because of the paucity of small dead fuels such as needles and grasses in subalpine forests, short-duration drying episodes generally do not create sufficiently dry conditions to sustain a fire. However, prolonged dry weather conditions (about 40 days without precipitation) can sufficiently dry live fuels and larger dead fuels to carry large, intense fires once they are ignited (figure 5). Conditions necessary for large fires are infrequent and often coupled with the occurrence of lightning. This suggests that Native Americans probably did not have a major influence on fires in the subalpine forest types, except in some localized areas.

The recent period of consistent, effective fire suppression in remote high-elevation sites, which has lasted 50 years at most, represents only a small portion of typical fire-free intervals in subalpine forests. Studies of fire history show that long fire-free periods (as long as, or longer than, the fire exclusion period during the 20th century) characterized the fire regimes of these forests before Euro-American settlement (Romme 1982, Romme and Despain 1989, Kipfmueller and Baker 2000, Veblen 2000, Schoennagel et al. 2003). Therefore, it is unlikely that the short period of fire exclusion has significantly altered the long fire intervals in subalpine forests (Romme and Despain 1989, Johnson et al. 2001, Veblen 2003). Furthermore, large, intense fires burning under dry conditions are very difficult, if not impossible, to suppress (Wakimoto 1989), and such fires account for the majority of area burned in subalpine forests (figure 2; Romme and Despain 1989, Bessie and Johnson 1995). At lower elevations within its range, lodgepole pine may also experience occasional small surface fires (Kipfmueller and Baker 2000), but their spatial extent and frequency are not well quantified. Suppression of smaller, less intense fires under moderate climate conditions probably has had little influence on the dominant fire regime in subalpine forests (Johnson et al. 2001, Veblen 2003). Our understanding of the dominant fire regime in these high-elevation, cool forests leads us to conclude that any recent increases in area burned in subalpine forests are probably not attributable to fire suppression. Evidence from the subalpine forests of Yellowstone indicates that fires of comparable size to the 1988 fires occurred in the early 1700s (Romme and Despain 1989).

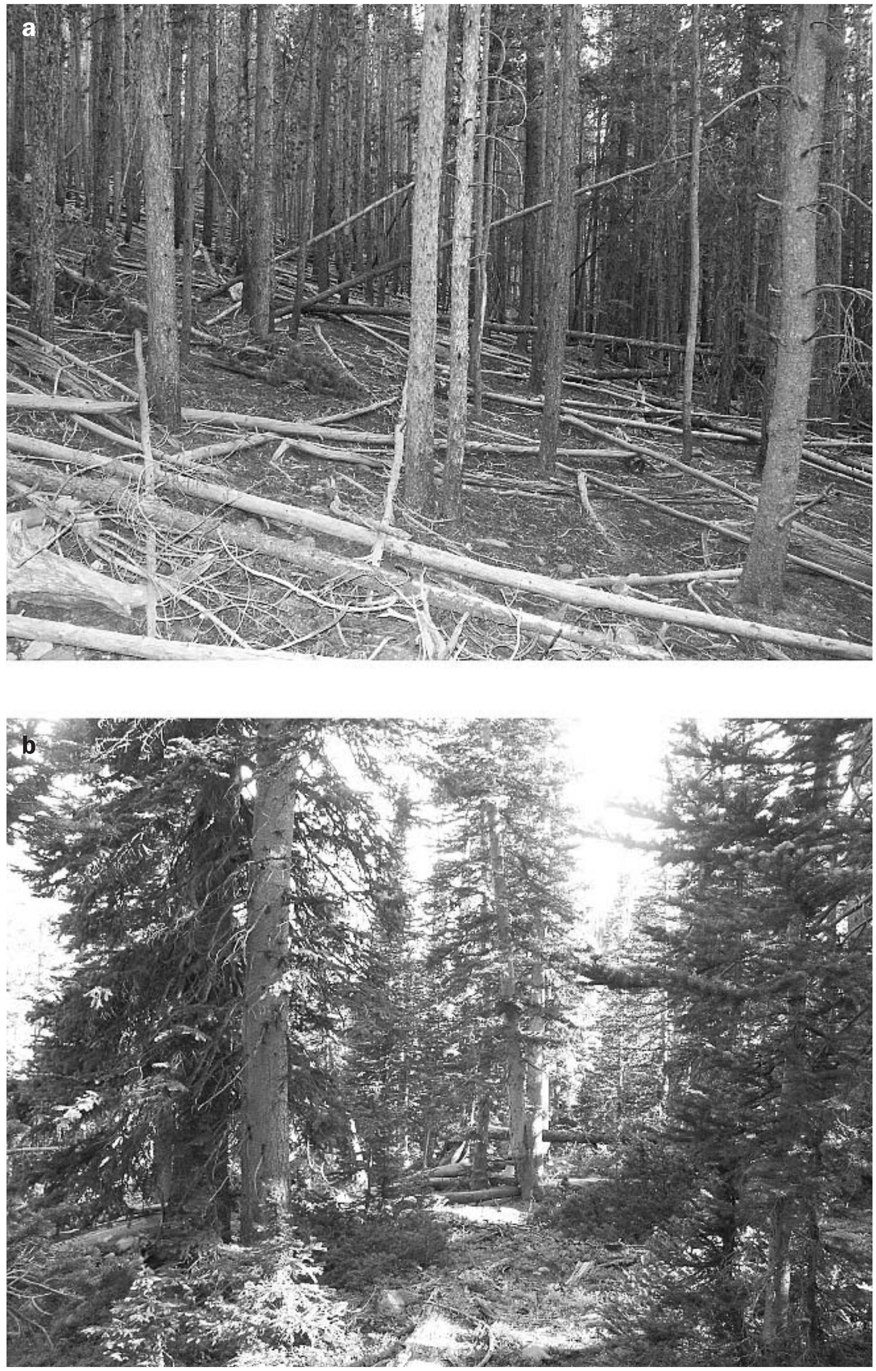

Figure 3. Typical subalpine forest stand structure, which easily carries fire into the canopy, promoting high-severity crown fires. (a) Lodgepole pine stand with sparse understory fuels and high tree densities. (b) Spruce–fir stand with abundant live ladder fuels throughout the vertical profile. Photographs: Tania Schoennagel.

Moreover, there is no consistent relationship between time elapsed since the last fire and fuel abundance in subalpine forests (Brown and Bevins 1986), further undermining the idea that years of fire suppression have caused unnatural fuel buildup in this forest zone. For example, lodgepole pine stands experience high rates of self-thinning that contribute large dead fuels as stands mature (Kashian 2003). However,

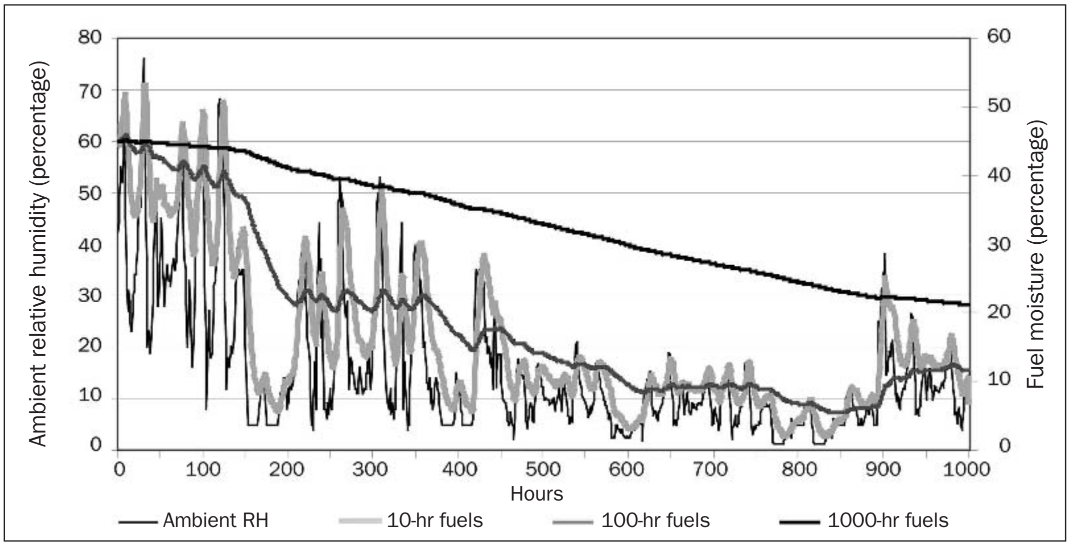

Figure 4. A theoretical example illustrating differences in fuel-moisture time lags for small (10-hour), intermediate (100-hour), and large (1000-hour) fuels. Small fuels dry out rapidly and respond more quickly to short-term variability in ambient relative humidity, while large fuels exhibit a more lagged response, requiring much longer dry periods to reach similar dryness.

the legacy of wood from the prefire stand contributes abundant loads of large fuel to young postfire stands (Romme 1982). Bessie and Johnson (1995) report little variation in total fuel loads, relative to variation in weather, in subalpine forests of different ages. No evidence suggests that spruce–fir or lodgepole pine forests have experienced substantial shifts in stand structure over recent decades as a result of fire suppression. Overall, variation in climate rather than in fuels appears to exert the largest influence on the size, timing, and severity of fires in subalpine forests (Romme and Despain 1989, Bessie and Johnson 1995, Nash and Johnson 1996, Rollins et al. 2002). We conclude that large, infrequent stand-replacing fires are "business as usual" in this forest type, not an artifact of fire suppression.

Case study: The 1988 Yellowstone fires. In 1988, according to the National Interagency Fire Center, more than 700,000 ha burned in mostly high-elevation subalpine forests throughout Wyoming, Montana, and Idaho. Yellowstone National Park was the focus of public attention during these fires. Some 40% of the park burned, much of it at high severity (Turner et al. 1994). Drought, which had started years earlier, extended beyond its immediate region during the summer of 1988. From 1977 to 1989, a strong Pacific North America pattern developed, creating a blocking ridge over the northwestern United States that reduced winter snowpack across Montana and Wyoming (Baker 2003). Low winter snowpack in 1988, followed by an unusually dry, hot, and windy summer, contributed to extreme burning conditions in the park (Balling et al. 1992). Precipitation in July and August was only 20% of normal levels; relative humidity fell to 6%; and strong, dry, gusty winds (60 to100 kilometers [km] per hour) spread multiple fires ignited by humans and lightning.

Variation in daily area burned was highly correlated with the moisture content of 100-hour (2.5- to 7.6-cm diameter) and 1000-hour dead fuels (Turner et al. 1994). Once fuels reached critical moisture levels later in the season, the spatial pattern of the large, severe stand-replacing fires was controlled by weather (wind direction and velocity), not by fuels, stand age, or firefighting activities (Minshall et al. 1989, Wakimoto 1989, Turner et al. 1994). Variation in fuel abundance and topography (including formidable barriers such as the Grand Canyon) had little influence on the severity or direction of the fire when fuel moistures were critically low (Turner et al. 1994). Stand-replacing fire affected stands of all ages, including some as young as 7 years old (Schoennagel et al. 2003).

Contrary to popular opinion, previous fire suppression, which was consistently effective from about 1950 through 1972, had only a minimal effect on the large fire event in 1988 (Turner et al. 1994). Reconstruction of historical fires indicates that similar large, high-severity fires also occurred in the early 1700s (Romme and Despain 1989). Given the historical range of variability of fire regimes in high-elevation subalpine forests, fire behavior in Yellowstone during 1988, although severe, was neither unusual nor surprising.

Summary: High-severity fire regimes in subalpine forests. Subalpine forests that experience infrequent, high-severity fires cover approximately 32% to 46% of the forested area in the Rocky Mountain region, which encompasses the three major forest types discussed in this article (table 1). The following insights are drawn from analyses of historical fire regimes and contemporary fire behavior in subalpine forests.

- Infrequent, high-severity, stand-replacing fires dominate the historical and contemporary fire regime in these forests.
- Climatic variation, through its effects on the moisture content of live fuels and larger dead fuels, is the predominant influence on fire frequency and severity.
- Dense trees and abundant ladder fuels are natural in subalpine forests and do not represent abnormal fuel accumulations.

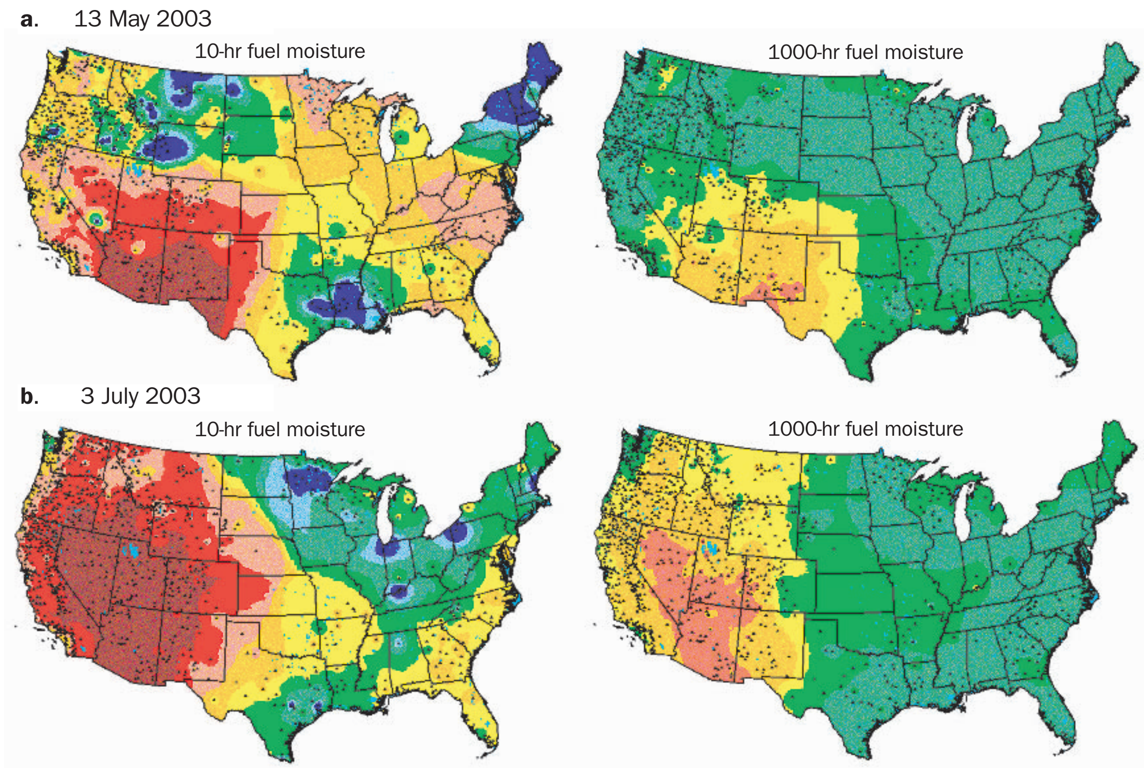

Figure 5. Maps of fuel moisture for small (10-hour) and large (1000-hour) fuels, showing responses to (a) short-term (1- to 2-day) and (b) longer-term (1- to 2-month) drying conditions in the southwestern United States. Large fuels dry sufficiently to carry fire only under longer drying conditions, while smaller fuels may dry sufficiently to carry fire under short-term or moderate drying conditions. The maps were developed by the National Interagency Fire Center (17 June 2004; **www.fs.fed.us/land/wfas/wfas10.html**).

- Fire suppression has had minimal influence on the size, severity, and frequency of high-elevation fires.
- Mechanical fuel reduction in subalpine forests would not represent a restoration treatment but rather a departure from the natural range of variability in stand structure.
- Given the behavior of fire in Yellowstone in 1988, fuel reduction projects probably will not substantially reduce the frequency, size, or severity of wildfires under extreme weather conditions.

Low-severity fire regimes

In marked contrast to the infrequent, high-severity fire regimes characteristic of subalpine forests, many low-elevation ponderosa pine forests historically experienced frequent, low-severity fires. A meta-analysis of 63 fire histories from similar-size southwestern ponderosa pine sites (10 to 100 ha) indicates that surface fires returned at mean intervals of 4 to 36 years (based on fire dates recorded for more than 10% of the sampled trees; Swetnam and Baisan 1996), an order of magnitude shorter than the intervals for subalpine forest stands. Some low-elevation ponderosa pine stands in Colorado, near the Plains grasslands, show evidence of 8- to 10-year intervals for fire returning to the same small stand or tree before the 1900s (Veblen et al. 2000). In the Black Hills of South Dakota, the mean fire interval was 20 to 23 years at each of four low-elevation ponderosa pine sites (about 100 ha each) for the period from 1388 to 1900 (Brown and Sieg 1996). Although detailed comparison of fire-interval statistics across study sites is problematic because of differences in the extent of the study area and the intensity of sampling, these studies clearly indicate a significant difference in fire interval and severity between low-elevation, dry ponderosa pine forests and high-elevation, moist subalpine forests.

Frequent, low-severity fire regimes occurred predominantly in dry, low-elevation ponderosa pine forests that were formerly open woodlands with abundant, contiguous fine fuels in the understory. This surface fuel layer, dominated by grasses and long cast needles, dries easily and thus promotes the spread of frequent surface fires. Historically, climate, fine-fuel abundance, and fire were highly interrelated in dry, low-elevation ponderosa pine forests. El Niño–Southern Oscillation (ENSO) patterns correlate tightly with the incidence of synchronous, low-severity fires in dry, low-elevation forests of the Southwest (Swetnam and Baisan 1996, Grissino-Mayer and Swetnam 2000, Kitzberger et al. 2001). The ENSO cycle alternates between El Niño and La Niña conditions at

Table 1. Two coarse estimates of the extent and proportion of three major forest types across the Rocky Mountain region (Arizona, New Mexico, Colorado, Utah, Wyoming, Montana, and Idaho). The first estimate is based on a map of Küchler's potential natural vegetation groups, modified by Schmidt and colleagues (2002). The second estimate is based on a map of current cover type developed by Schmidt and colleagues (2002). A different historical fire regime is associated with each of the three forest types, although the correspondence is not exact.

Forest type	Area (hectares)	Percentage of total	Associated severity of historical fire regime
Based on PNV groups			
Ponderosa pine (pine forest, Great Basin pine)	8,201,600	17.7	Low
Mixed ponderosa pine (pine–Douglas fir, Douglas fir, grand fir–Douglas fir, Southwest mixed conifer)	23,176,200	49.9	Mixed
Spruce–fir (spruce–fir, spruce–fir–Douglas fir)	15,056,000	32.4	High
Total	46,433,800	100.0	
Based on current cover types			
Ponderosa pine	13,009,100	36.7	Low
Douglas fir	6,176,000	17.4	Mixed
Spruce–fir–lodgepole pine (lodgepole pine, fir–spruce)	16,287,200	45.9	High
Total	35,472,300	100.0	

PNV, potential natural vegetation.

Note: Total is the forested area in the Rocky Mountain region defined by the three major forest types listed. Some other forest types, such as piñon-juniper woodlands, are not included.

2- to 6-year frequencies. In the southern Rockies, El Niño years are characterized by wetter-than-average winter and spring conditions, which enhance the growth of fine fuels (especially grasses). Drier-than-average La Niña years typically follow, desiccating abundant fine surface fuels. Time-lag analysis shows that dry, low-elevation ponderosa pine forests commonly experience more extensive fires when wetter conditions 1 to 3 years before a fire are followed by dry conditions during the year of the fire. Infrequent or anomalous prolonged drought conditions are not the primary factor promoting fires in dry, low-elevation pine forests, as they are in subalpine forests. Summers in the low-elevation forests are typically dry enough to promote low fuel moisture levels that would permit ignition, although the abundance and continuity of fine surface fuel historically were the primary limiting factors (Swetnam and Baisan 1996, Rollins et al. 2002).

Unlike the historical fire regime in subalpine forests, the fire regime in dry, low-elevation ponderosa pine forests has been significantly altered as a result of fire suppression and its effects on historical fuel structure (Arno and Gruell 1983, Swetnam and Baisan 1996, Veblen et al. 2000). Before fire suppression, the frequent, low-severity surface fires in these forests kept dry ponderosa pine stands sparse and open by killing young, newly established trees. With fire suppression and livestock grazing (which reduces the amount of grass fuel), fire intervals have lengthened, and dense stands have developed in which fine grass fuels are less abundant and dense ladder fuels are capable of carrying fire up into the canopy (figure 6). Consequently, high-severity fires potentially can occur in dry ponderosa pine forests, where historically they were rare because of the sparse ladder fuels and the lack of contiguous tree crowns. This pattern has been well documented on the basis of fire scars, repeat photography, and stand age structures, especially for forests in Arizona and New Mexico (Covington and Moore 1994, Allen et al. 1998, Mast et al. 1999, Moore et al. 1999), for some sites in the Colorado Front Range (Veblen and Lorenz 1991, Brown et al. 1999, Kaufmann et al. 2000), and for portions of the Bitterroot Range in Montana (Gruell 1983, Arno et al. 1995). As a consequence of fire suppression, the size and occurrence of high-severity fires has increased in this forest type. Reduction of ladder fuels through mechanical thinning and prescribed fire can effectively reduce the unprecedented occurrence of extensive crown fires and restore the historical surface fire regime in dry, low-elevation ponderosa pine forests (Covington et al. 1997, Allen et al. 2002, Fule et al. 2002).

Case study: The 2002 Rodeo-Chediski fire complex. The Rodeo-Chediski fire, which burned 189,095 ha in northern Arizona from 18 June through 7 July 2002, was the largest Arizona fire in recorded history. The area burned was dominated by ponderosa pine, with isolated pockets of mixed conifers at higher elevations along the Mogollon Rim, where the northern half of the fire burned. Fire-history studies conducted before the fire, in nearby ponderosa pine stands, record frequent surface fires with mean fire intervals of 7 to 10 years (based on fires recorded by more than 10% of sampled trees in 10- to 100-ha study areas; Swetnam and Baisan 1996). In 2002, high-severity crown fire affected 48% of the Rodeo-Chediski fire area, an extent of severe burning that is unprecedented in the low-elevation, dry ponderosa pine forests of this area.

The summer of 2002 marked the fourth year of drought in the Southwest. That May had been the second driest on record across Arizona and New Mexico in 108 years. Levels of fuel moisture before the fire were unusually low: 7% in 1000-hour fuels, as low as 2% in 10-hour (0.6- to 2.4-cm diameter) and 100-hour fuels, and below critical thresholds in live pine and brush fuels (Wilmes et al. 2002). The Haines index is a measure of lower-atmosphere stability and dryness correlated with wildfire growth. Low values (2 or 3) indicate moist, stable conditions; the highest values (5 or 6) represent dry, unstable conditions that favor moderate to high fire activity. The Haines index was 6 on many days during the Rodeo-Chediski fire.

Prescribed fire, salvage logging in previously burned stands, and fuel-reduction treatments (including the removal of slash, or woody debris, from branches and treetops) were effective in reducing fire severity and spread in the Rodeo-Chediski fire, even under extreme weather conditions (figure 7; Wilmes et al. 2002), as predicted by restoration research in Arizona (Fule et al. 2002). High-severity crown fires affected 35% of the stands that had been treated within the last 15 years, compared with 55% of the untreated stands. The average stand density of treated and untreated stands was 387 and 1108 trees per hectare, respectively. All prefire fuel treatments appeared to lower burn severity except for precommercial treatments, which increased it. In precommercial treatments, slash (branches and tree tops) was lopped and scattered throughout the stand, which contributed to higher fuel loads than those in untreated stands. Areas that had high forage production and low tree density experienced less severe burning during the Rodeo-Chediski fire, suggesting that open stands with abundant fine surface fuels were more resistant to high-severity canopy fire (figure 8). Overall, burn severity was positively correlated with overstory tree density (Wilmes et al. 2002). This outcome, in clear contrast with the findings from Yellowstone (where weather rather than fuel type and arrangement influenced fire behavior), highlights the heterogeneity of forest types and fire effects across the Rocky Mountain region.

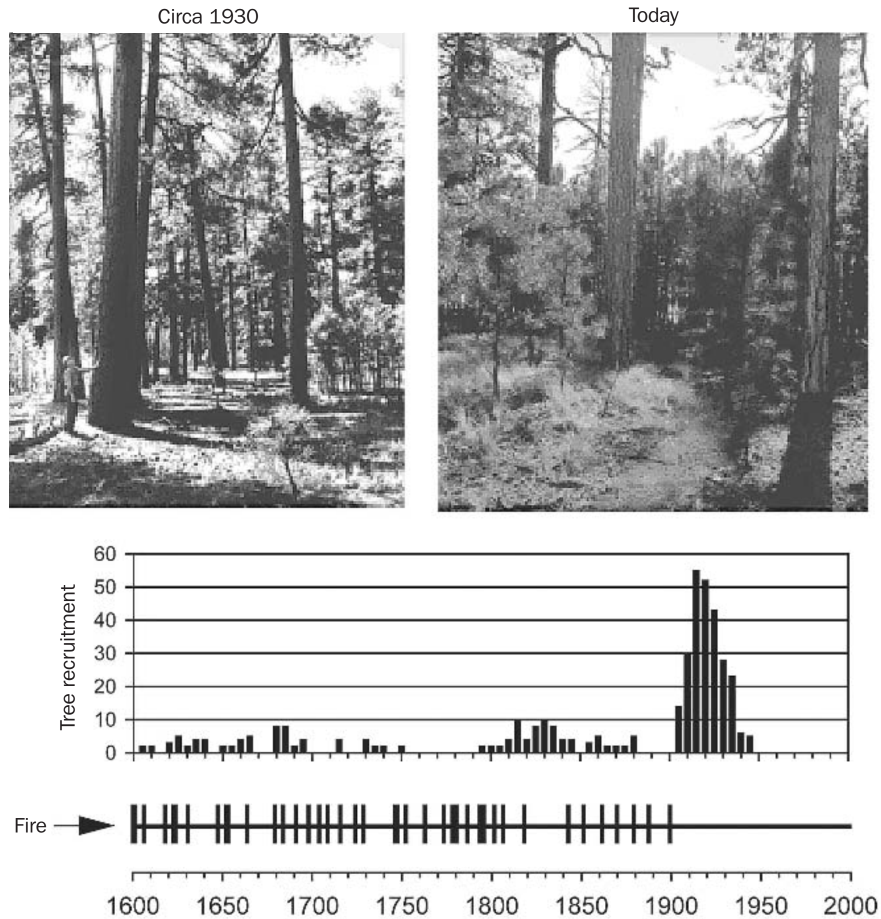

Figure 6. A comparison of historical and contemporary stand structure of dry ponderosa pine stands from the Jemez Mountains of New Mexico, and the relationship of this change to the frequency of low-severity surface fires. Source: Modified from Allen et al. 1998.

Summary: Low-severity fire regimes in low-elevation ponderosa pine forests. Dry, low-elevation ponderosa pine forests in the Rocky Mountain region, which were historically characterized by frequent low-severity fire regimes, make up an estimated 19% to 37% of the forested area that encompasses the three forest types discussed in this article (table 1). Such historically sparse forests, subject to high-frequency fires, comprise much of the ponderosa pine forest in Arizona and New Mexico but only a small fraction of the ponderosa pine forest in the central and northern Rockies. Regional modeling of fire regimes, based on a large fire-history database from the western United States, similarly predicts decreasing fire frequency from southern to northern latitudes (McKenzie et al. 2000). Important lessons about fire regimes in dry, low-elevation ponderosa pine forests are listed below.

- The historical fire regime in these forests was characterized by frequent, low-severity surface fires.
- Historically, the frequency, size, and severity of fires were largely controlled by spatial and temporal variation in fine fuels.
- Fire suppression has significantly increased tree densities and ladder fuels in low-elevation ponderosa pine forests.
- As a consequence of this change in stand structure, unprecedented high-severity fires now occur.

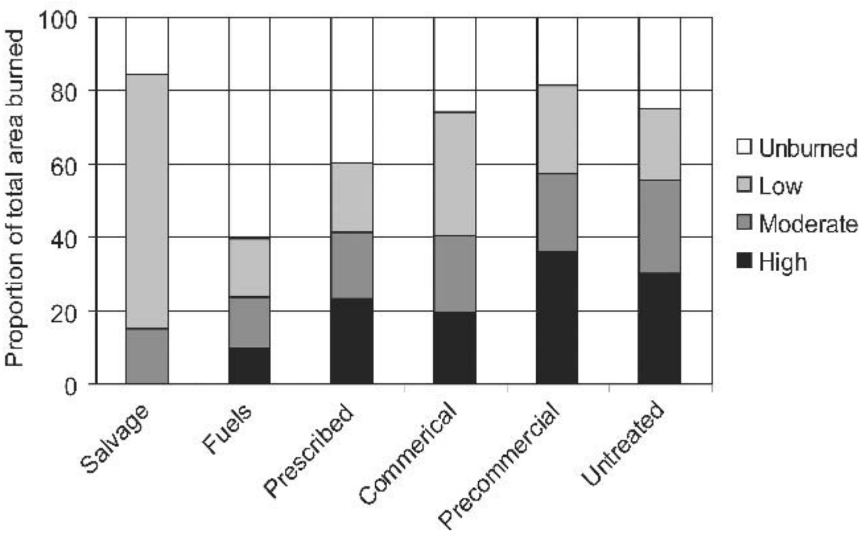

Figure 7. Proportion of different prefire fuel treatments burned at different severities during the Rodeo-Chediski fire in the Apache-Sitgreaves National Forests, Arizona, 2002. Burn severity, defined by the Burned Area Emergency Rehabilitation team **(www.fs.fed.us/r3/asnf/salvage/publications/proj_record/001_rodeo_baer_report_7-29-02.pdf),** ***ranges from unburned (surface fire with little or no canopy damage, tree foliage unscorched) through low severity (some tree crowns scorched but most trees not killed) and moderate severity (variable tree mortality, foliage scorched but not consumed) to high severity (complete tree mortality, foliage completely consumed). Fuel treatments are defined as salvage (removal of trees after a fire), fuels (thinning, chipping, and pile burning), prescribed fire (broadcast burning), commercial (removal, seed cut, regeneration, harvest, partial removal, final cut, or thinning), or precommercial (thinning with chipping, lopping, or both; no slash removal). Data are from Wilmes and colleagues (2002).***

- Fuel-reduction treatments involving mechanical thinning and prescribed fire are likely to be effective in mitigating extreme fire behavior and restoring this forest type to the historical fire regime.

Mixed-severity fire regimes

Mixed-severity fire regimes are intermediate between the infrequent, high-severity fire regimes of high-elevation subalpine forests and the frequent, low-severity fire regimes of dry, low-elevation ponderosa pine forests. Both high- and low-severity fires can occur at varying frequencies in mixed-severity fire regimes. This fire regime occurs predominantly at mid elevations, where topographic variation creates a complex moisture gradient resulting in a mosaic of tree species and densities that is sometimes referred to as mixed conifer forest. There is also evidence of mixed-severity fire regimes that predate fire suppression in some forests dominated by ponderosa pine, and even in pure or nearly pure ponderosa pine stands at low to mid elevation (Veblen and Lorenz 1986, Mast et al. 1998, Kaufmann et al. 2000, Ehle and Baker 2003).

Historically, forests that experienced mixed-severity fire regimes had variable densities of ponderosa pine, Douglas fir (*Pseudotsuga menziesii*), grand fir (*Abies grandis*), and western larch (*Larix occidentalis*), depending on their location. These forests constituted a mosaic of even-aged stands resulting from stand-replacing fire, interspersed with uneven-aged stands that experienced low-severity surface fires and episodic tree regeneration (Arno 1980, Brown et al. 1999, Kaufmann et al. 2000). Pre-1900 stand-replacing fires in these forest types have been documented by historic photographs and by the occurrence of even-age stand structures whose age corresponds to that of fire scars on adjacent trees (Gruell 1983, Veblen and Lorenz 1986, 1991, Arno et al. 1995, Swetnam and Baisan 1996, Shinneman and Baker 1997, Mast et al. 1998, Brown et al. 1999, Kaufmann et al. 2000, Ehle and Baker 2003). Low-severity fires are also well documented by historic photographs, fire scars, and all-age stands that include centuries-old trees, although these surface fires usually occurred less frequently than in the lower-elevation dry ponderosa pine forests described above (Arno 1980, Veblen and Lorenz 1991, Swetnam and Baisan 1996, Brown et al. 1999, Moore et al. 1999, Kaufmann et al. 2000, Veblen et al. 2000). The relative importance of surface versus crown fires and the size of these post-disturbance patches in shaping forests of mixed-severity fire regimes remain uncertain and have probably varied spatially and temporally.

Since the late 19th century, the densities of relatively fire-intolerant and shade-tolerant species, such as Douglas fir and grand fir, have increased in response to the suppression of low-severity fires in areas that historically experienced mixed-severity fire regimes (Arno et al. 1995, Kaufmann et al. 2000). Increases in density probably have occurred more commonly at lower elevations, on drier aspects, and adjacent to grasslands where frequent, low-severity fires were more dominant historically. Sites that previously supported denser stands because of favorable topographic and edaphic conditions have probably changed less as a result of fire suppression; those sites historically experienced stand-replacing fires, and high stand densities are a normal part of the postfire recovery process (Veblen and Lorenz 1986, Arno et al. 1995, Mast et al. 1998, Kaufmann et al. 2000, Ehle and Baker 2003). With fire suppression, forests that historically experienced mixed-severity fire regimes have developed a more homogenous forest structure across the landscape, resulting in larger areas of continuously dense forest and perhaps in larger patches of crown fire than were witnessed historically. In some areas, tree regeneration following logging of these forests in the late

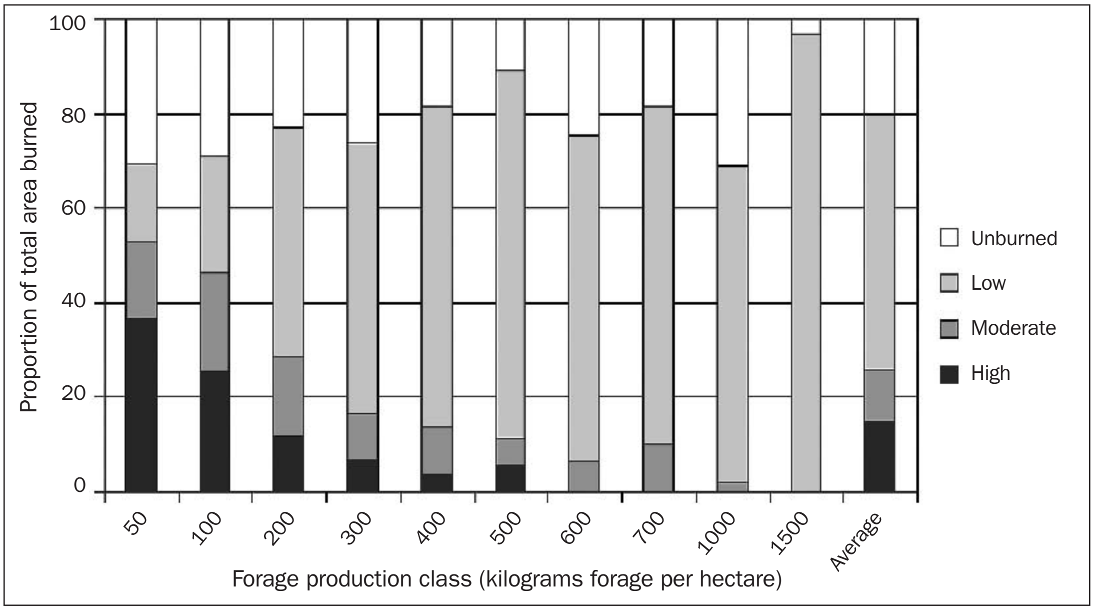

Figure 8. Proportion of different forage production classes burned at different severities during the Rodeo-Chedeski fire in relation to forage production classes for Carlisle and Town Tank allotments on the Lakeside Ranger District, Apache-Sitgreaves National Forest. Data are from Wilmes and colleagues (2002).

19th and early 20th centuries has contributed to high stand densities (Veblen and Lorenz 1986, Kaufmann et al. 2000). Overall, fire suppression has probably significantly affected only sites within the mixed conifer zone at lower elevations, on drier aspects, and adjacent to grasslands where fires historically were more frequent. Therefore, current fire regimes and stand densities in mixed conifer forests are likely to be within the historical range of variability, or at least are not likely to be as far outside this range as those in the dry ponderosa pine forests discussed above (Veblen 2003). However, additional research is needed on the causes of variability in mixed-severity fire regimes and the attendant effects of fire suppression.

In mixed-severity fire regimes, climate and fuels interact in a complex manner to control the frequency and severity of fires. Arno (1980) describes this interaction in mixed-severity fire regimes: “Under severe burning conditions, especially with strong winds, fires sometimes crowned and covered sizeable areas. When conditions moderated, fire would creep along the ground, with occasional flare-ups. Often the major fires burned at several intensities in reaction to changes in stand structure, fuel loadings, topography, and weather. The result was a mosaic of fire effects on the landscape” (p. 463). In mixed-severity regimes, in contrast to the previous two types of fire regime discussed, both climate and fuels (surface and ladder fuels) vary considerably and are important drivers of fire frequency and severity. We look to the example of the Hayman fire to tease apart these interactions in more detail.

Case study: The 2002 Hayman fire. The Hayman fire burned a 55,915-ha area southwest of Denver, Colorado, where previous fire history and forest structure studies (Brown et al. 1999, Kaufmann et al. 2000), mechanical fuel treatments, and burns (wild and prescribed) had occurred. Making use of this unplanned experiment, researchers assessed the relative effect of fuels and climate on fire behavior in the area, which had a historical mixed-severity fire regime (Finney et al. 2003).

Short-term drought during the 5 years before the fire created important antecedent conditions. In particular, below-normal precipitation and unseasonably dry air masses had persisted since 1998, when drier-than-average La Niña conditions began to develop. These conditions persisted intermittently through the spring of 2002. As a consequence, the Colorado Front Range received low snow during the winters of 2001 and 2002, with snowpack recorded in May 2002 at less than 50% of normal levels. By spring 2002, measurements of large-fuel moisture (moisture in 100-hour and 1000-hour fuels) in mid- to low-elevation forests of the southern Rockies were among the driest in the previous few decades, dipping as low as 3% when typically they exceed 12% (Graham 2003).

The size and severity of the Hayman fire can largely be explained by the extreme fire activity during two separate periods associated with sustained, exceptionally dry, forceful winds. First, on 9 June, the fire grew from 485 to 24,700 ha (43% of the total fire size); later, on 18 June, it traveled 5 miles along its southeastern flank (figure 9). During these two periods, mean relative humidity dipped below 8%, maxi-

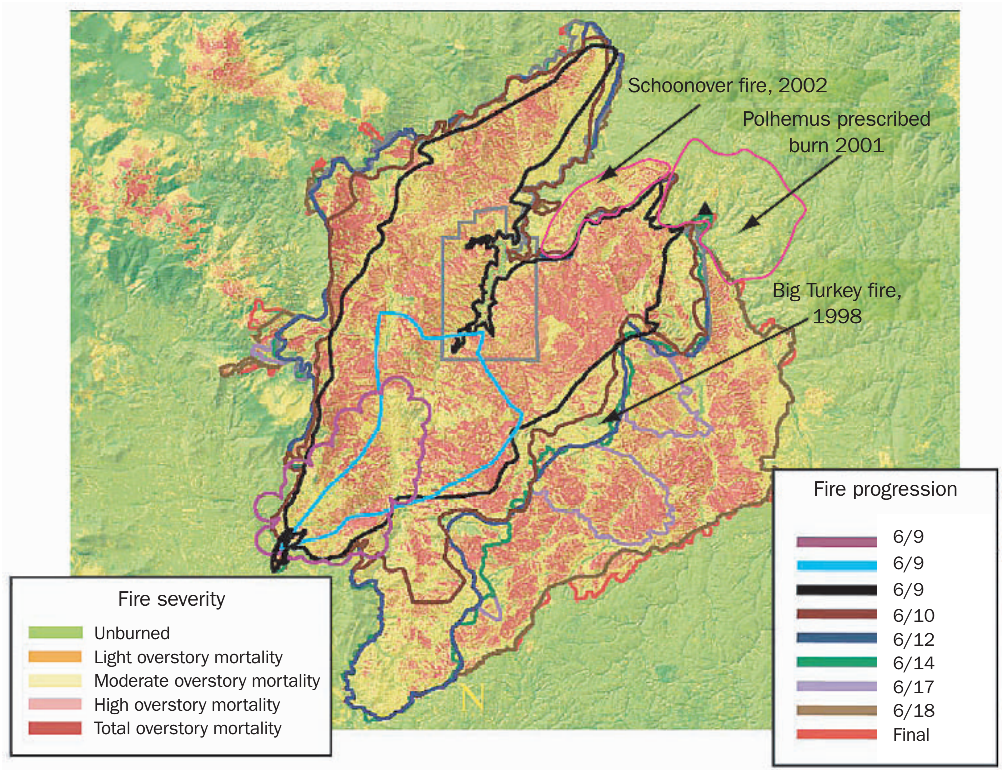

Figure 9. Map of the Hayman fire progression during the period 9–18 June 2002. Note the significant progress of the fire on 9 June (black line) and 18 June (brown line). Not all days are shown, because fire perimeters on slow-growth days overlapped previous days. Burn severity classes are based on the difference-normalized burn ratio from the US Geological Survey's National Burn Severity Mapping Project. Gray line represents the Cheesman Reservoir boundary, pink lines represent the perimeter of recent burns. Source: Modified from Finney and colleagues (2003).

mum wind gusts reached 84 miles (135 km) per hour, and the Haines index was 6, marking very dry, unstable conditions conducive to high fire spread. Both periods produced extensive torching, crown fire, and spotting (firebrands thrown in advance of the fire). These high-activity periods terminated with the passage of fronts followed by upslope winds that substantially increased ambient relative humidity (Finney et al. 2003).

During the substantial fire-progression days of 9 and 18 June, most fuel treatments had very little impact on the severity or direction of the fire (Finney et al. 2003). On 9 June, for example, the burned area included more than 2400 ha that had experienced previous prescribed fires or other fuel-reduction treatments. These treatments, which included previous wildfires (in 1963 and 1998), prescribed fires (in 1990, 1992, 1995, and 1998), and numerous stand modifications with and without subsequent slash removal (table 2), had virtually no effect on the Hayman fire. This is in marked contrast to the behavior of the Rodeo-Chediski fire, whose severity was affected by previous fuel-reduction treatments even under extreme climate and weather conditions. In the Hayman fire, extreme weather conditions overwhelmed the effectiveness of most fuel treatments. However, the fire stopped abruptly at the edge of the area that had been burned by two fires months to weeks before, in fall 2001 (Schoonover fire) and May 2002 (Polhemus prescribed burn), where very little fuel had accumulated during a spring of extreme drought (figure 9; Finney et al. 2003). Overall, the direction, severity, and size of the fire on extreme days were mostly explained by high wind and low relative humidity (table 3), with little effect of past fire or thinning activity. The Hayman review team concluded that "fuel modifications generally had little influence on the severity of the Hayman Fire during its most significant run on June 9th" (Finney et al. 2003) but acknowledged that the small size of these treatments contributed to their lack of effectiveness. On days of moderate fire growth, however, fuel modifications did influence fire spread and severity; of these modifications, recent wild or prescribed fires and thinning with slash removal were most effective. In an example of the interactions between fuels and climate, on 17 June the Hayman fire split into two runs on either side of the area burned by the Big Turkey fire in 1998 (figure 9); however, when the weather became more extreme the following day, this effect on fire shape and extent was obliterated (figure 9; compare 17 June and 18 June perimeters).

Table 2. Distribution of fire severity classes among fuel-modified areas on moderate slopes (defined as slopes of less than 30%) that burned in the Hayman fire on 9 June 2002.

Level of prefire fuel modification	Area (ha)	Fire severity class (percentage): Unburned	Low	Moderate	High
Unmodified	9128	4	18	8	70
		Recent modifications (after 1990)			
Wildfires	5	0	0	25	75
Prescribed fires	291	6	20	11	63
Fuel treatment	0	NA	NA	NA	NA
Improvements and treatment	160	0	19	7	74
Improvements, no treatment	253	3	12	9	76
Harvest and treatment	657	5	14	10	71
Harvest, no treatment	236	0	1	33	66
Plantation	55	0	8	5	87
		Older modifications (before 1990)			
Wildfires	Unknown	NA	NA	NA	NA
Prescribed fires	34	17	50	8	25
Fuel treatment	2	0	86	14	0
Improvements and treatment	0	NA	NA	NA	NA
Improvements, no treatment	592	1	14	8	77
Harvest and treatment	1	0	16	9	75
Harvest, no treatment	384	3	27	2	68
Plantation	127	0	27	10	63

Source: Finney et al. 2003.

Summary: Mixed-severity fire regimes in the Rocky Mountain region. Mixed-severity fire regimes account for an estimated 17% to 50% of the forested area in the Rocky Mountain region that encompasses the three major forest types discussed in this article (table 1). These forests experience the most complex type of fire regime and the least understood. Nonetheless, we have learned several important lessons about mixed-severity fire regimes in Rocky Mountain forests.

- The historical fire regime in these forests is complex, including both low-severity surface fires and infrequent high-severity crown fires.
- Both fuels and climate have major influences on the frequency, severity, and size of fires.
- Fire suppression has had variable effects on fuel densities in mixed-severity fire regimes, with the greatest impacts on sites that formerly supported open woodlands.
- The occurrence of high-severity crown fires is not outside the historical range of variability, although their size and frequency may be increasing.
- Extreme climate and weather conditions can override the influence of stand structure and fuels on fire behavior.
- Fuel-reduction treatments (mechanical thinning and prescribed burning) may effectively reduce fire severity under moderate weather conditions, but these treatments may not effectively mitigate fire behavior under extreme weather conditions and may not restore the natural complexity of historical stand and landscape structure.

Implications for fire mitigation and restoration

What does an understanding of the spatial variation in dominant controls on wildfire frequency and severity mean for ecological restoration and for effective fuel treatments to reduce the threat of large, severe wildfires? The Yellowstone fires in 1988 revealed that variation in fuel conditions, as measured by stand age and density, had only minimal influence on fire behavior. Therefore, we expect fuel-reduction treatments in high-elevation forests to be generally unsuccessful in reducing fire frequency, severity, and size, given the overriding importance of extreme climate in controlling fire regimes in this zone. Thinning also will not restore subalpine forests, because they were dense historically and have not changed significantly in response to fire suppression. Thus, fuel-reduction efforts in most Rocky Mountain subalpine forests probably would not effectively mitigate the fire hazard, and these efforts may create new ecological problems by moving the forest structure outside the historic range of variability (Veblen 2003, Romme et al. 2004).

In contrast, for many low-elevation, dry ponderosa pine forests, it is both ecologically appropriate and operationally possible to restore a low-severity fire regime through thinning and prescribed burning (Covington et al. 1997, Allen et al. 1998, 2002). Fuels rather than climate appear to be the most significant factor affecting fire spread and severity in these forests. Fire suppression in dry ponderosa pine forests appears

Table 3. Comparison of the mean and range of weather indices associated with the type (high, moderate, or low) of fire-growth days during the Hayman fire, 9 June to 18 June 2002.

		Mean (range)			
Fire-growth days	*n*	Relative humidity (percentage)	10-minute average wind (kph)	Maximum wind gust (kph)	Haines index
Low	4	36.6 (8–68)	11.2 (0–30.4)	22.4 (3.2–57.6)	3.7 (2–6)
Moderate	4	27.6 (5–76)	11.2 (0–28.8)	24 (1.6–54.4)	4.2 (2–6)
High	2	7.8 (5–15)	16 (1.6–48)	38.4 (3.2–134.4)	5.7 (5–6)

kph, kilometers per hour.

Note: The Haines index, ranging from 2 to 6, measures the moisture and stability of the lower atmosphere; low values indicate moist, stable conditions, and high values indicate dry, unstable conditions conducive to fire. The two high fire-growth days occurred on 9 and 18 June. High- and moderate fire-growth days are identified on the Hayman fire progression map (figure 9); low fire-growth days are those omitted from the map because fire perimeters were not significantly different from previous days. Data are summarized from Finney and colleagues (2003).

to have contributed to an unprecedented buildup of fuels and to the occurrence of high-severity fires. Indeed, the objectives of fire mitigation and forest restoration generally converge in forests of this type.

Perhaps the most difficult forests to assess are the mid-elevation forests that historically were characterized by mixed-severity fire regimes. Because mixed-severity fire regimes are most complex and least well understood, we must exert caution in developing simple prescriptions for wildfire mitigation that may not bring predictable results under extreme climate conditions. Our analysis reveals that fire regimes, climate, fuel type and abundance, and stand structure vary significantly across the Rocky Mountain region. As a consequence, the heterogeneous forests in this region require very different approaches to restoration and wildfire management (Gutsell et al. 2001). Clearly, policymakers need to incorporate ecological heterogeneity into their decisions in order to implement sound forest management policy.

In addition to the fuel-management operations described above, we need new research to clarify the geographic variation in fire regimes across different forest types in this large, heterogeneous region. There is great geographical variation in the distribution of the three broad fire regimes defined here. In Montana, for example, subalpine forests cover roughly 40% of the forested area, while in Arizona the extent of these forests is significantly smaller and they are more isolated on scattered mountaintops. At a regionwide scale, it is difficult to define the precise extent of these different fire regimes and their spatial location (and especially to distinguish between the low-severity and mixed-severity fire regimes), as illustrated by the variation between the estimates based on PNV groups and those based on current cover type (table 1). There is also significant variation in fire regimes within each of the three broad fire-regime classes in response to local topography and landscape position, and there are other important vegetation types not covered in this brief article (e.g., piñon-juniper woodlands; Romme et al. 2003).

A "one size fits all" approach to reducing wildfire hazards in the Rocky Mountain region is unlikely to be effective and may even produce collateral damage in some places. We do not advocate delaying action until all of the ecological questions have been answered; in many places, there is an urgent need and a solid ecological basis for restoration and fire-mitigation efforts. In other areas, however, where the ecological basis for aggressive fuel reduction is inadequate or lacking, uncritical extrapolation of models from other systems may cause more harm than good.

Acknowledgments

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Adapt to more wildfire in western North American forests as climate changes

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Wildfires across western North America have increased in number and size over the past three decades, and this trend will continue in response to further warming. As a consequence, the wildland–urban interface is projected to experience substantially higher risk of climate-driven fires in the coming decades. Although many plants, animals, and ecosystem services benefit from fire, it is unknown how ecosystems will respond to increased burning and warming. Policy and management have focused primarily on specified resilience approaches aimed at resistance to wildfire and restoration of areas burned by wildfire through fire suppression and fuels management. These strategies are inadequate to address a new era of western wildfires. In contrast, policies that promote adaptive resilience to wildfire, by which people and ecosystems adjust and reorganize in response to changing fire regimes to reduce future vulnerability, are needed. Key aspects of an adaptive resilience approach are (*i*) recognizing that fuels reduction cannot alter regional wildfire trends; (*ii*) targeting fuels reduction to increase adaptation by some ecosystems and residential communities to more frequent fire; (*iii*) actively managing more wild and prescribed fires with a range of severities; and (*iv*) incentivizing and planning residential development to withstand inevitable wildfire. These strategies represent a shift in policy and management from restoring ecosystems based on historical baselines to adapting to changing fire regimes and from unsustainable defense of the wildland–urban interface to developing fire-adapted communities. We propose an approach that accepts wildfire as an inevitable catalyst of change and that promotes adaptive responses by ecosystems and residential communities to more warming and wildfire.

wildfire | resilience | forests | wildland–urban interface | policy

Wildfire is a key driver of ecosystem change that increasingly poses a significant threat and cost to society. In western North America (hereafter, the West), warming, frequent droughts, and legacies of past management combined with expansion of residential development have made social–ecological systems (SESs) more vulnerable to wildfire. As the annual area burned has increased over the past three decades, we are confronting longer fire seasons (1, 2), more large fires (3, 4), a tripling of homes burned (5), and more frequent large evacuations. In 2016, the Fort McMurray Fire in Alberta, Canada and the Blue Cut Fire in southern California prompted evacuation orders for a combined total of more than 160,000 people. The costs of wildfire have also risen substantially since the 1990s. The US Congress appropriated \$13 billion for fire suppression and \$5 billion for fuels management in fiscal years 2006–2015 (6). Other societal costs, including real estate devaluation, emergency services, and postfire rehabilitation, total up to 30 times the direct cost of firefighting (7).

Notwithstanding these costs, many plants, animals, and ecosystem services benefit from fire, and those dependent on frequent fire have been negatively affected by the significantly reduced burning resulting from fire suppression, as compared with the period before European settlement

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(8). However the response of ecosystems to increases in wildfire activity and warming in the coming decades is not well understood. Broad heterogeneity among western forest landscapes in terms of biophysical environment, past management, human footprint, and the role of fire and future warming creates a complicated playing field. Managing ecosystems, people, and wildfire in a changing climate is a complex but critical challenge that requires effective and innovative policy strategies (9, 10).

Our key message is that wildfire policy and management require a new paradigm that hinges on the critical need to adapt to inevitably more fire in the West in the coming decades. Policy and management approaches to wildfire have focused primarily on resisting wildfire through fire suppression and on protecting forests through fuels reduction on federal lands. However, these approaches alone are inadequate to rectify past management practices or to address a new era of heightened wildfire activity in the West (11–14).

In delivering this message, we focus specifically on the distinction between specified, adaptive, and transformative resilience (15, 16). Rigorous definition and critical assessment of resilience to wildfire are needed to develop effective policy and management approaches in the context of climate change. We suggest an approach based on the concept of adaptive resilience, or adjusting to changing fire regimes (e.g., shifts in prevailing fire frequency, severity, and size) to reduce vulnerability and build resilience into SESs. Adaptive resilience to wildfire means recognizing the limited impact of past fuels management, acknowledging the important role of wildfire in maintaining many ecosystems and ecosystem services, and embracing new strategies to help human communities live with fire. Our discussion focuses on western North American forests but is relevant to fire-influenced ecosystems across the globe. We emphasize that long-term solutions must integrate relevant natural and social science into policies that successfully foster adaptation to future wildfire.

Why Has Coping with Wildfire Become Such a Challenge?

Three primary factors have produced gradual but significant change across western North American landscapes in recent decades: the warming and drying climate, the build-up of fuels, and the expansion of the wildland–urban interface (WUI; the zone where houses meet or intermingle with undeveloped wildland vegetation).

In terms of climate, wildfire activity is closely tied to temperature and drought over time scales of years to millennia (2, 17–19). Globally, the length of the fire season increased by 19% from 1979 to 2013, with significantly longer seasons in the western United States (1). Since 1985, more than 50% of the increase in the area burned by wildfire in the forests of the western United States has been attributed to anthropogenic climate change (20). Increases in the number of wildfires and area burned in most forested ecoregions of the West are a result of rising temperatures, increased drought, longer fire seasons, and earlier snowmelt (1–4, 21). Specifically, since the 1970s the frequency of large fires has increased most dramatically in the forests of the Northwest (1,000%) and Northern Rocky Mountains (889%), followed by forests in the Southwest (462%), Southern Rockies (274%), and Sierra Nevada (256%), in response to earlier snowmelt and a longer fire season (21). Based on spatial overlays in western United States forests of large wildfires since 1984 (Monitoring Trends in Burn Severity, available at www.mtbs.gov/dataaccess.html and Existing Vegetation Types, available at https://www.landfire.gov/vegetation.php), we found that in northern regions with dramatic increases in fire activity (the Canadian Rockies, Middle Rockies, and Idaho Batholith ecoregions) cold/wet subalpine forests predominantly burned. These forests characteristically burn at high severity and have not experienced a significant build-up of fuels. Overall, cold/wet forests account for about a quarter of total forest burning in the US West since 1984.

Fire suppression, in addition to past logging and grazing and invasive species, has led to a build-up of fuels in some ecosystems, increasing their vulnerability to wildfire. For example, drier, historically open coniferous forests in the West ("dry forests") have experienced gradual fuels build-up in response to decades of fire suppression and other land-use practices (8, 22, 23). Historically, predominantly frequent, low-severity fires killed smaller, less fire-resistant trees and maintained low-density dry forests of larger, fire-resistant trees. Large, high-severity fires now threaten to convert denser, more structurally homogeneous dry forests to nonforest ecosystems, with attendant loss of ecosystem services (24). However, only forests in the Southwest show a clear trend of increasing fire severity over the last three decades, and only a quarter to a third of the area burned in the western United States experienced high severity during that time (25, 26). Although fuels build-up in dry forests can increase the area burned because of higher contagion, the 462% increase in the frequency of large fires in southwestern forests since the 1970s is also a result of an extension of the fire season by 3.6 mo [the average for the western United States is 2.8 mo (21)]. Overall, dry forests account for about half of the total forest burning in the western United States since 1984.

Alongside these increases in warming and fuels, the WUI has expanded tremendously in the past few decades, augmenting wildfire threats to people, homes, and infrastructure. Between 1990 and 2010, almost 2 million homes were added in the 11 states of the western United States, increasing the WUI area by 24% (27). Currently, most homes in the WUI are in California (4.5 million), Arizona (1.4 million), and Washington (1 million) (27). Since 1990, the average annual number of structures lost to wildfire has increased by 300%, with a significant step-up since 2000 (28). About 15% of the area burned in the western United States since 2000 was within the WUI, including a 2.4-km community protection zone, with the largest proportion of wildfires burning in the WUI zone in California (35%), Colorado (30%), and Washington (24%) (Fig. 1) (27). Additionally, almost 900,000 residential properties in the western United States, representing a total property value more than $237 billion, are currently at high risk of wildfire damage (29). Because of the people and property values at risk, WUI fires fundamentally change the tactics and cost of fire suppression as compared with fighting remote fires and account for as much as 95% of suppression costs (28). Together, these gradually changing variables—climate change, fuels build-up, and residential development—interact with rapid combustion to increase wildfire risks and costs to society and some ecosystems substantially.

Potential Consequences of Future Wildfire

Wildfire activity is predicted to increase in the West over the next century (20, 30, 31). This anticipated ramp-up in burning and possible directional changes in fire regimes (e.g., increases in fire frequency, severity, and/or size) could transform the composition, structure, and function of many forest (8, 32, 33), shrubland, and grassland ecosystems (34). Changes in temperature and precipitation in semiarid shrublands and grasslands may reduce fuel availability subsequently, to the extent that fire occurrence, size, and severity in such areas will eventually decline (35). Thus, although fire activity is projected to increase in the West in the near term (i.e., in the next few decades), longer regional trends will depend on feedbacks between vegetation and fire as well as on anthropogenic alterations in vegetation and land use (36, 37).

Increased exposure of communities to wildfire is also expected with additional warming. More than 3.6 million ha, or almost 40% of the current WUI in the western United States, is predicted to experience moderate to large increases in the probability of wildfire in the next 20 y (Fig. 2). This increase is in addition to the growing wildfire risk to developed nonurban areas (e.g., energy production) and infrastructure (e.g., power lines, pipelines) that define a broader wildland–development

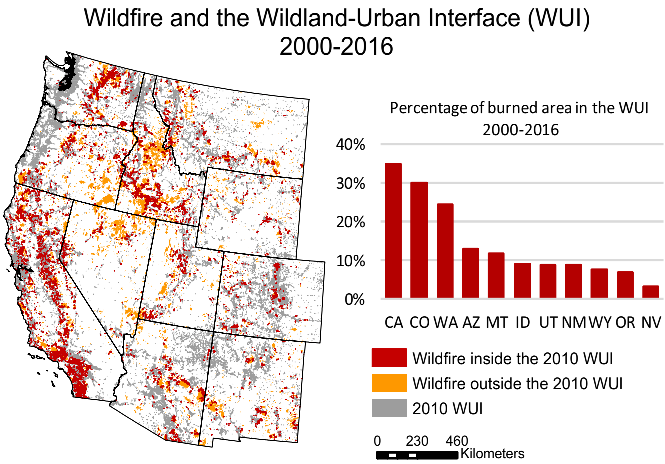

Fig. 1. **(*Left*) Area burned by wildfires between 2000 and 2016 across the western United States inside and outside the 2010 WUI including a 2.5-km community protection zone (27). (*Right*) About 15% of the WUI burned during this period, with largest proportions of the WUI burning in California, Colorado, and Washington.**

interface. Continued WUI growth will further increase human exposure to wildfires (38) and anthropogenic ignitions (37, 39). By midcentury, 82 million people in the western United States are likely to experience more and longer "smoke waves," defined as consecutive days of high, unhealthy particulate levels from wildfires (40). Climate change and increasing exposure of existing and future development to wildfire and smoke present a dangerous and vexing problem for residents, local officials, fire fighters, and managers.

Gradual but significant changes in climate, fuels, and the WUI affect wildfire impacts on ecosystems and society but are difficult to recognize and are challenging to alter meaningfully. There often is a lack of political will to implement policies that incur short-term costs despite their long-term value or to change long-standing policies that are ineffective. For example, few jurisdictions have the will or means to restrict further residential development in the WUI, although modifying and curtailing residential growth in fire-prone lands now would reduce the costs and risks from wildfire in the long term. Furthermore, although the impacts of fire suppression on fuels build-up are now well understood, fire-suppression policies still dominate current fire management (13). Projected global warming of at least 1.1–3.1 °C in the coming century offers a unique opportunity to change policy and the course of our response to wildfires (41). A paradigm shift now in approaches to WUI development and management of fire and fuels can yield tremendous benefits to society later.

Specified, Adaptive, and Transformative Resilience to Wildfire

Resilience is increasingly invoked as a guiding principle in strategies that address the social and ecological dimensions of wildfire. The US Forest Service's National Cohesive Wildland Fire Management Strategy (42) specifically addresses the need to bolster social and ecological resilience to increasing wildfires. Although often invoked in wildfire management and policy, resilience is defined inconsistently or neglects social or ecological contexts, despite the need for uniformity and specification in setting goals and evaluating progress (43, 44).

Defining resilience to wildfire in an SES is especially challenging in the WUI, where people, ecosystems, and wildfire interact over multiple spatial and temporal scales (12). An SES is the intersection and interdependence of biophysical units and associated people and institutions. Resilience in an SES generally has been defined as the capacity to absorb disturbance so as to retain essential structures, processes, and feedbacks and to adapt to and reorganize following disturbance (45). These perspectives of resilience, absorbing versus adapting to disturbance, offer different guiding principles for policy and management in responding to wildfire and measuring success over different planning timelines (44). Here we outline a consistent framework that defines resilience to wildfire in coupled SESs based on the concepts of specified resilience and general resilience, the latter of which includes adaptive and transformative approaches (Table S1) (15, 16, 44).

When climate trends or disturbance regimes are relatively stable and well-characterized and planning horizons are short (years), specified resilience or restoration is an appropriate guiding principle. "Specified resilience" refers to the buffer capacity of a system to retain its identity after a well-specified disturbance (16). Specified resilience reflects the concept of ecological resilience, which refers to the capacity of a system to absorb or tolerate disturbance without shifting to a qualitatively different state controlled by a different set of processes (46). In terms of wildfire, specified resilience applies when fire characteristics are within the bounds of historical range of variability (HRV) of disturbance regimes and a burned forest recovers without converting to another state, e.g., to a nonforest state such as a persistent grassland. In a social context, specified resilience is evident when a community recovers economically and rebuilds similar structures in similar locations following a wildfire (44, 47). Management guided by specified resilience often values recent ecological and social dynamics, particularly when the goal is the conservation of particular species or landscapes. Such management is often informed by short temporal windows of HRV, or "recent HRV" (rHRV) (Fig. 3). This approach can be useful for responding to fires in the short term. However, when social and environmental conditions change rapidly, this approach may foster management goals that are unrealistic or unsustainable in the long run (48, 49).

When climate and wildfire trends are changing and planning horizons are intermediate (decades), general resilience is a more appropriate and desirable guiding principle. "General resilience" refers to the capacity of an SES to adapt or transform in response to unknown shocks or disturbances outside the rHRV (16). Adaptive resilience incorporates aspects of change, reorganization, learning, and adaptability in response to changing climate and disturbance regimes and is an on-going process achieved by harnessing adaptive capacity. In an ecological context, adaptive resilience refers to actively or passively supporting species compositions and fuel structures that are better adapted to a warming, drying climate with more wildfire. Management of specified resilience maintains ecosystems within the rHRV,

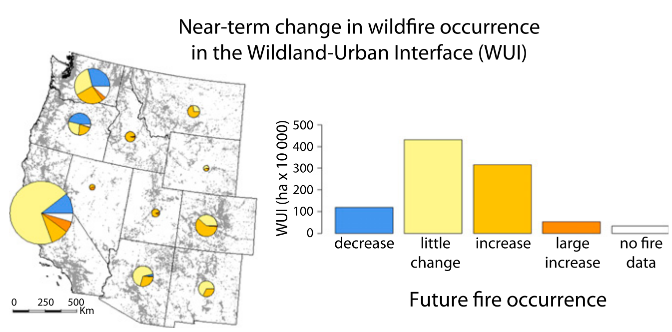

Fig. 2. **(*Left*) Area of the WUI in the conterminous western United States, classified according to projected near-term changes in fire occurrence. The size of each pie is scaled relative to the area of the WUI (both intermix and interface) in each state, based on data from Martinuzzi, et al. (27). Within each pie, slices represent the proportion of WUI area overlapping the five categories of projected fire occurrence for the period 2010–2039, based on data from Moritz, et al. (30). (*Right*) The bar chart summarizes the area of the WUI projected to experience each level of change in fire occurrence in the western United States.**

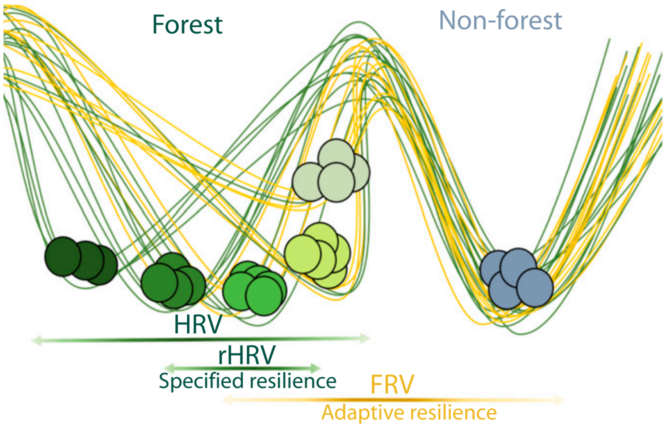

Fig. 3. Conceptual ball-and-basin representation of specified and adaptive resilience across a forested landscape. Lines defining basins depict the ranges of variation in fire regimes across forest types. Sets of green balls reflect the variation in abundance and composition within different forest types, and the set of blue balls represents nonforest ecosystems. Specified resilience of forests to wildfire is maintained within basins that fall within an rHRV of fire regimes over recent decades to centuries, typically derived from historical documents, remotely sensed data, and tree-ring data. Longer definitions of HRV reflect variation in fire regimes over the last 4,000–5,000 y, when present-day forest types were established in most regions; these data are derived from paleoecological reconstructions. Adaptive resilience to changing fire regimes is reflected within basins that fall within the FRV (yellow). Under the FRV, shifts to nonforest ecosystems remain unlikely in some cases (lower green balls) and more likely in other cases with easier transition to nonforest basin (higher green balls). Changes in the severity, frequency, and size of fire regimes and long-term regeneration following fire events reflect adaptive responses to changing fire regimes and climate conditions across broad scales.

whereas managing for adaptive resilience considers how changing disturbance regimes may favor suites of traits that are better adapted to a future range of variability (FRV) (Fig. 3) (22). Alignment of fire regimes with adaptive regeneration traits of native vegetation defines a safe operating space (50). The HRV can still play a role by providing insight into how adaptive traits align with changing disturbance regimes to confer adaptive resilience, but under the FRV the safe operating space is shifting (Fig. 3) (50, 51, 52). In a social context, communities exhibiting adaptive resilience engage in ecological, psychological, social, and policy processes that set the community on a trajectory of change to reduce future vulnerability (Fig. 4) (53). Strategies may include changing building codes to make structures more fire-resistant, planning communities to avoid or withstand future wildfire, or providing incentives, education, and resources to reduce vulnerability to future wildfire (47). Adaptive resilience also involves institutional learning, where past management approaches to wildfire evolve.

When climate and wildfire trends are significantly altered from historical trends and/or variability, and planning horizons are long (century), transformative resilience may be necessary. "Transformative resilience" refers to planned fundamental change in response to drastically altered disturbances that have the potential to create broad-scale, systemic shifts in ecological states or radical shifts in values, beliefs, social behavior, and multilevel governance. Examples might include significant regional changes in ecosystem states and associated loss of ecosystem services and/or the relocation of communities of people away from wildfire-prone areas (44, 54). Rapid, planned social–ecological transformation is rare and difficult to implement because of uncertainties about future risk, inflexible institutions and behaviors, and the high cost of transformative action (55).

Although distinct, these approaches to resilience may be nested. Promoting specified resilience may make some forests better poised for adaptive resilience as climate changes, but in some forests or conditions specified resilience may not be effective as climate changes (e.g., refs. 56, 57). Allowing postfire shifts from forest to grassland or shrubland may increase adaptive resilience to changing wildfire and climate conditions. Approaches to adaptive resilience could reduce the need for transformation if efforts keep pace with climate and wildfire trends or may help pave the way toward inevitable social–ecological change. Embracing specified resilience may be the easiest, most familiar path with the least uncertainty, but this approach is short-sighted and could come at the cost of adaptation to future wildfire as climate change continues.

Taking an adaptive resilience approach now is critical, because specified resilience, although useful in some contexts, will become a less useful guiding principle as we exceed HRVs. Adaptive resilience means adjusting to changing fire regimes and climate—in both social and ecological systems—by taking advantage of opportunities to moderate potential impacts and cope better with the consequences. Adapting to wildfire sooner rather than later provides the widest benefits to society at the least cost. If we do not adapt to wildfire now, disruptive and unintended transformations of SESs in the West may ensue.

How Policy and Management Can Promote Adaptive Resilience to Wildfire

Current approaches to managing wildfire focus primarily on controlling fire through suppression and secondarily focusing on managing fuels build-up in forests. Within the context of current and future trends in wildfire, we evaluate the following three approaches in terms of their promise for fostering adaptive resilience in ecosystems and residential communities living with more wildfire: (*i*) managing fire, (*ii*), managing fuels, and (*iii*) promoting adaptive capacity (Fig. 5).

Fig. 4. Wildfires are catalysts of change that promote adaptive resilience by communities and ecosystems to future wildfires. (*A* and *B*) Example of adaptation in communities. (*A*) A home burned in the 2010 Fourmile fire, Boulder County, CO, which at the time was the most destructive fire in Colorado history in terms of home loss. (*B*) A home that survived the 2016 Cold Springs fire, where many residents managed structural and vegetative fuels around their home to reduce fire hazard after the Fourmile fire through Boulder County's Wildfire Partners program. (*C* and *D*) Heterogeneity in wildfire severity promotes diversity in postfire regeneration and fuels in the 2002 Rodeo-Chediski fire, Coconino and Navajo counties, AZ (*C*) and the 2016 Canyon Creek fire, Grant County, OR (*D*). Photographs courtesy of REUTERS/Alamy Stock Photo (*A*), Wildfire Partners (*B*), Tom Bean/Alamy Stock Photo (*C*), and M.A.K. (*D*).

Managing Wildfire

Suppressing Fewer Fires and Prescribing More Burning. Increasing the use of prescribed fires and managing rather than aggressively suppressing wildland fires can promote adaptive resilience as the climate continues to warm. Many dry forests currently experience significantly less burning than in the period just before European settlement (8, 35, 58). In recognition of the fire-dependence of many ecosystems, the 1995 Federal Wildland Fire Management policy ushered in the first federal policy aimed at reintroducing more wildfire on public lands; that policy remains in effect today. US federal agencies actively managed an average of 75,000 ha of lightning-caused fires per year under the Wildland Fire Use policy from 1998–2008 and currently burn about 1 million hectares per year with prescribed fires (58). However, prescribed fires still constitute only about 10% of the treatments implemented by the US Forest Service in the West and burn about one-third of the area burned by wildfires (National Interagency Fire Center, https://www.nifc.gov/). In the United States and Canada, suppression remains the primary approach to wildfire, with more than 95% of all wildfires suppressed (28). Continued aggressive fire suppression is counterproductive to building adaptive resilience to increasing wildfire in the long term (13, 14).

Using Fire to Foster Adaptive Resilience to Climate Change. In some systems, fire today attenuates future fire effects, because flames that burn dead and live fuel limit where and how severely subsequent fires burn, at least for a time (59–61). Fires often create complex patterns of burn severity that create variation in postfire regeneration and fuels (62–67). As fire regimes shift over time, individual fire events filter for species adapted to changing fire and climate conditions (68). Strategic planning for more managed and uncontrolled wild fires on the landscape today (69) may help decrease the proportion of large and severe wildfires in the coming decades and may enhance adaptive resilience to changing climate. Prescribed fires, ignited under cooler and moister conditions than are typical of most wildfires, can reduce fuels and minimize the risk of uncontrolled forest wildfire near communities. In contrast to wildfires, prescribed fire risks are relatively low, and more than 99% of prescribed fires are held within planned perimeters successfully (58).

Challenges to increasing use of managed and prescribed fires vary from the public's limited experience with smoke and wildfire to significant direct health impacts of smoke on vulnerable populations, including children, the elderly, and low-income communities (40, 70, 71). Some smoke hazards can be reduced through careful planning and management of fire, public health monitoring, and provisioning of health services for vulnerable populations. Public perceptions of fire are also an important hurdle, given the success of Smokey Bear's fire-prevention campaign and because most urban and suburban residents have very limited experience with wildfire compared with rural residents of the early 20th century. Therefore, public education programs that demonstrate the inevitability of wildfire will be a key aspect of living with increasing fire in the West. We need to develop a new fire culture. Despite these and various legal and operational challenges (58), the benefits of prescribed fire and managed wildfires to ecosystems and communities are high (72). Promoting more wildfire away from people and prescribed fires near people and the WUI are important steps toward augmenting the adaptive resilience of ecosystems and society to increasing wildfire.

Managing Fuels

Limiting Reliance on Fuels Treatments to Alter Regional Fire Trends. Managing forest fuels is often invoked in policy discussions as a means of minimizing the growing threat of wildfire to ecosystems and WUI communities across the West. However, the effectiveness of this approach at broad scales is limited. Mechanical fuels treatments on US federal lands over the last 15 y (2001–2015) totaled almost 7 million ha (Forests and Rangelands, https://www.forestsandrangelands.gov/), but the annual area burned has continued to set records. Regionally, the area treated has little relationship to trends in the area burned, which is influenced primarily by patterns of drought and warming (2, 3, 20). Forested areas considerably exceed the area treated, so it is relatively rare that treatments encounter wildfire (73). For example, in agreement with other analyses (74), 10% of the total number of US Forest Service forest fuels treatments completed 2004–2013 in the western United States subsequently burned in the 2005–2014 period (Fig. 6). Therefore, roughly 1% of US Forest Service forest treatments experience wildfire each year, on average. The effectiveness of forest treatments lasts about 10–20 y (75), suggesting that most treatments have little influence on wildfire. Implementing fuels treatments is challenging and costly (7, 13, 76, 77); funding for US Forest Service hazardous fuels treatments totaled \$3.2 billion over the 2006–2015 period (6). Furthermore, forests account for only 40% of the area burned since 1984, with the majority of burning in grasslands and shrublands. As a consequence of these factors, the prospects for forest fuels treatments to promote adaptive resilience to wildfire at broad scales, by regionally reducing trends in area burned or burn severity, are fairly limited.

Targeting Fuels Treatments in Ecosystems with Fuel Build-Up and on Private Lands. Strategically targeting treatments in areas where fuels build-up has increased the expected burn severity may augment the adaptive resilience of those ecosystems to increasing wildfire. For example, treating drier forests, where the likelihood of fire is

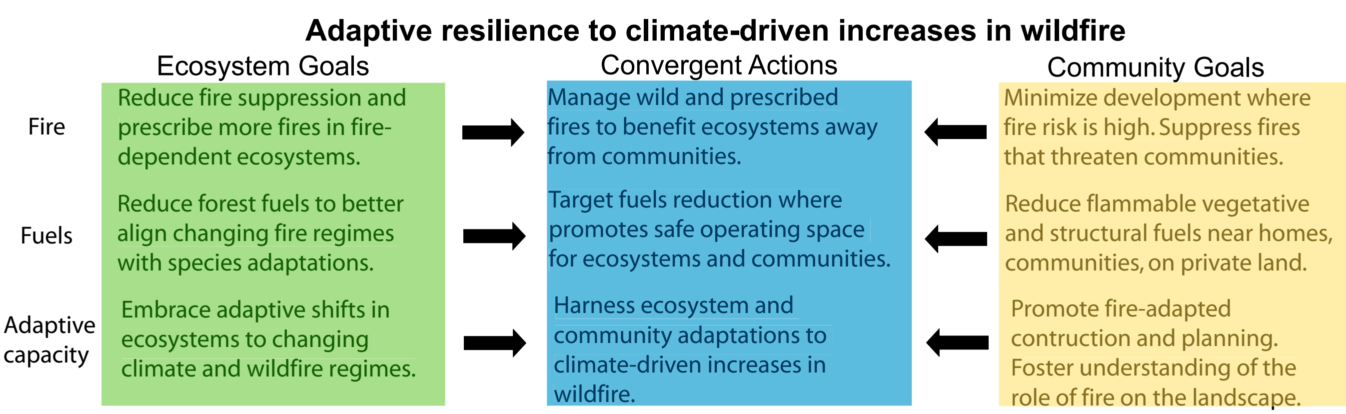

Fig. 5. Convergent actions that promote adaptive resilience to climate-driven increases in wildfire in the West by ecosystems and communities, based on goals related to management of fire, fuels, and adaptive capacity.

high, may also increase opportunities to modify wildfire behavior and postfire recovery. Burn severity has increased because of past fire suppression and fuels build-up in low-elevation dry forests adapted to predominantly frequent, low-severity surface fires (8, 11, 22, 25, 78, 79). In these forests, fuels treatments that remove midstory and understory fuels through thinning and prescribed fire can reduce fire intensity, severity, and rate of spread and may promote adaptive resilience to more frequent fire. Such forests were preferentially treated under the National Fire Plan in 2004–2008 (80). Thinning may effectively restore more frequent, low-severity fire in some dry forests, but when thinning is combined with the expected warming, unintended consequences may ensue, whereby regeneration is compromised and forested areas convert to nonforest (56, 57, 81). Strategic placement of treatments to promote low-severity fire at ecotones between dry and mesic forest distributions may help facilitate postfire migration of species better adapted to warmer, drier conditions.

Midelevation mixed conifer forests, or mesic forests, which typically experienced broad variance in fire frequency and severity, may also benefit from fuels treatments that reduce the likelihood of large patches of high-severity fire and facilitate the migration of species adapted to drier, warmer conditions (77). In contrast, cold/wet forests, such as high-elevation subalpine forests, are adapted to high-severity fire that historically recurred at relatively long (~100–300 y) intervals (19, 82, 83) and have not experienced unprecedented fuels build-up in recent decades. Severe wildfires have occurred for millennia across a broad range of forests and shrublands, and in many ecosystems species are adapted to severe fire (17, 19, 84, 85), although postfire regeneration may be comprised by drier, warmer conditions (86).

Fuel-reduction treatments also hold promise for locally reducing wildfire hazard around WUI communities if treatments are strategically located to protect homes and the surrounding vegetation. Fuel reduction on federal lands and in municipal watersheds is a primary management tool that has limited application in the WUI, where the majority of land is privately owned (87). Home loss to wildfire is a local event, dependent on structural fuels (e.g., building material) and nearby vegetative fuels (88, 89). Therefore, fuels management for home and community protection will be most effective closest to homes, which usually are on private land in the WUI where ignition probabilities are likely to be high (37). Programs that facilitate the targeted removal of fuels from private land, such as community chipping programs, have been highly successful in some areas, at relatively low cost. The Wyden and Good Neighbor authorities and federal programs, such as the US Forest Service Collaborative Forest Landscape Restoration Program, take an "all-lands" approach to forest management through collaboration with landowners and communities. These policies and programs are roadmaps for augmenting fuel-management efforts across land ownerships. These and other more ambitious policies that facilitate significant fuels management on private land, on a par with fuel-reduction efforts on federal lands, are needed. New policies that facilitate private-land fuels management are critical to augment significantly the adaptive resilience of communities to increasing wildfire.

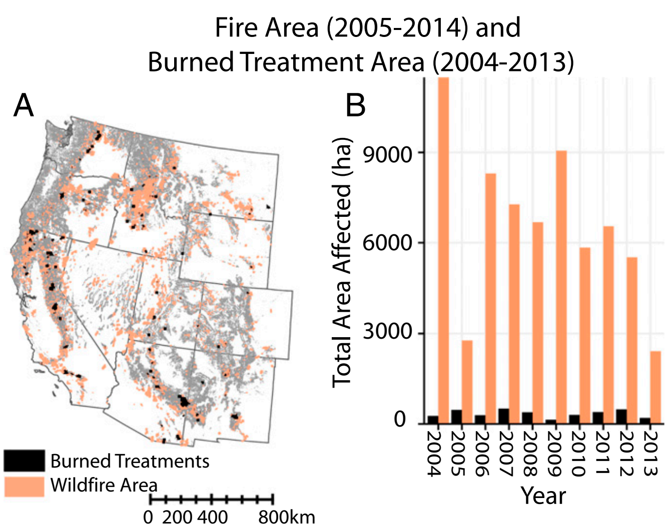

Fig. 6. (*A*) Spatial distribution and area of US Forest Service fuels treatments from 2004–2013 and wildfire from 2005–2014 across forests and woodlands in the western United States. About 3% of the total treated area and 10% of the total number of treatments burned in the period 2005–2014. (*B*) Annual total wildfire area and total burned treatment area. Data are from the following: (1) US Forest Service fuels treatments: Hazardous Fuel Treatment Reduction Polygon (https://data.fs.usda.gov/geodata/edw/datasets.php), (2) Wildfires >1000 ac: Monitoring Trends in Burn Severity Burned Areas Boundaries (www.mtbs.gov/dataaccess.html), (3) Wildfires ≤1000 ac: GeoMAC Historic Fire Perimeters (https://rmgsc.cr.usgs.gov/outgoing/GeoMAC/historic_fire_data/).

Promoting Adaptive Capacity

Fostering and Embracing Adaptive Shifts in Ecosystems. Management of fire and fuels will help some ecosystems withstand more frequent fires and possibly may reduce the risk of larger, more severe fires that may compromise forest recovery. Such efforts will be significant in high-value ecosystems or locations, in helping slow the pace of change and providing a chance for ecosystems and species to adapt to changing fire regimes. The HRV concept can guide management in identifying ecological vulnerabilities and adaptation strategies to changing disturbance regimes (Fig. 3) (50, 51, 52). However, quantifying ecological objectives outside the HRV will be increasingly important in guiding management as fire regimes and climate continue to change (90, 91). Given such uncertainties, management must be adaptive and iterative, and monitoring will be critical to assessing progress. Given the vast area of fire-prone forests in the West, management can directly affect only a small portion of forests. In the majority of forested ecosystems beyond our effective reach, we will have to accept and even embrace changing ecological conditions. While some forests may be entering decades of significant change with high tree mortality in response to drought, wildfire, insect outbreaks, and legacies of past management (86, 92), they also are in the process of adjusting to new conditions to which they will be better adapted and that may challenge our existing philosophies of and approaches to conservation.

Creating Fire-Adapted Communities. The majority of home building on fire-prone lands occurs in large part because incentives are misaligned, where risks are taken by homeowners and communities but others bear much of the cost if things go wrong. Therefore, getting incentives right is essential, with negative financial consequences for land-management decisions that increase risk and positive financial rewards for decisions that reduce risk. For example, shifting more of the wildfire protection cost and responsibility from federal to state, local, and private jurisdictions would better align wildfire risk with responsibility and provide meaningful incentives to reduce fire hazards and vulnerability before wildfires occur. Currently, much of the responsibility and financial burden for community protection from wildfire falls on public land-management agencies. This arrangement developed at a time when few residential communities were embedded in fire-prone areas. Land-management agencies cannot continue to protect vulnerable residential communities in a densifying and expanding WUI that faces more wildfire (12). The US Government Accountability Office questioned the US Forest Service's prioritizing protection of WUI communities that lie under private and state jurisdictions and has argued for increased financial responsibility for WUI wildfire risk by state and local governments (93). This shift in obligation would enhance adaptive governance and could increase the motivation to pursue adaptive resilience of WUI communities to increasing wildfire (94).

Another promising approach for increasing adaptive resilience of WUI residents to wildfire is the promotion of fire-adapted planning in communities. Providing incentives for counties, communities, and homeowners to plan fire-safe residential development for both existing and new homes and discouraging new development on fire-prone lands will make communities safer (89, 94–96). Communities can use land-use and development codes that encourage developers to set aside open space and recreational trails as fuel breaks and require ignition-resistant construction materials in fire-prone settings. For example, San Diego, California enforces strict brush management regulations; the Flagstaff, Arizona fire department uses a WUI development code to protect properties; and Santa Fe, New Mexico applies stringent fire-safe regulations on new developments to protect its watershed (97). Programs such as the Community Planning Assistance for Wildfire (CPAW; planningforwildfire.org), funded by the US Forest Service and private foundations, offer assistance to communities in the form of advice on land-use planning and detailed mapping of wildfire risk. Another example is California, which employs a statewide Fire Hazard Severity Zone map to guide development plans and building codes that reduce wildfire risk. With 84% of potential WUI lands in the West still undeveloped (98), land-use planning now has high potential to reduce the vulnerability of communities to future wildfire. Furthermore, fire-adapted planning may increase management options in terms of how, where, and when fire can be used as a tool for reducing the spread of wildfires into communities and rejuvenating fire-dependent ecosystems, thus increasing the adaptive resilience of communities and ecosystems to more wildfire.

Strengthening and expanding programs such as Fire Adapted Communities, Fire Adapted Communities Learning Network, Firewise Communities USA, and FireSmart Canada will also help communities become more fire-adapted. Capacities to assume these responsibilities will vary significantly among homeowners, communities, and local jurisdictions with markedly different risks and resources (99–101). For example, home hazard mitigation programs and community planning tools are more successful in communities at the fringe of urban areas that have more financial resources and often have a greater trust in government than in more isolated, resource-dependent WUI communities, immigrant non–English-speaking communities, or tribal and First Nations communities (101). Although some tax incentives and rebates are available for wildfire risk mitigation on and around homes, more comprehensive programs that include broader incentives and support are needed for meaningful and widespread impacts. Efforts that combine wildfire-specific efforts with other community capacity-building efforts may leverage the networks that enable communities to act on shared notions of risk (102).

Overall, a shift in resources from the defense of the WUI from wildfire to the mitigation of wildfire hazards and risks in advance of events will build a safe operating space for fire-prone communities that increases adaptive resilience to wildfire. Encouraging development away from fire-prone areas, reducing fuels on private lands in and near communities, and retrofitting and building homes to withstand ignition will increase the adaptive capacity for managing more wildfire (89), similar to adaptive approaches for other natural hazards such as flooding and earthquakes (12). Communities and institutions are long-lived, and disruptive events such as wildfires create windows of opportunity that can shift rules, norms, and expectations to increase adaptive resilience to future wildfires.

Conclusions

Policies that foster adaptive resilience enable WUI communities and fire-prone ecosystems to adjust to increased wildfire risk and reduce future vulnerability. Adaptive resilience provides a realistic framework as the climate warms and wildfires increase, but how will we know if we are achieving adaptive resilience to future fires? On the societal front, minimizing the costs of suppression in the WUI, the number of homes lost to wildfire, the area burned in the WUI, and the number of smoke-related health problems are some metrics. Developing state- or county-wide maps of fire hazard, home survivability rating, and the adaptive capacity of communities would be useful tools in developing this framework.

Some ecosystems will survive and thrive as they adapt to novel future conditions, but not all will. Embracing rather than resisting ecological change will require a significant paradigm shift by individuals, communities, and institutions and will challenge our conservation philosophies. Wildfire is an important catalyst of responses to climate change by communities and ecosystems. Patterns of wildfire are changing with rising global temperatures, and will accelerate in the future. What we can do now is focus management efforts on the places where intervention is needed to slow the pace of change and thereby give particular species and ecosystems a chance to adapt. We also can change how we build, live, and work in fire-prone landscapes to keep our communities safe, healthy, and vibrant.

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Invited Paper

Wildlife Conservation Planning Under the United States Forest Service's 2012 Planning Rule

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ABSTRACT In 2012, the United States Forest Service (USFS) promulgated new planning regulations in accordance with the National Forest Management Act (NFMA). These regulations represent the most significant change in federal forest policy in decades and have sweeping implications for wildlife populations. We provide a brief overview of the history of the NFMA planning regulations and their wildlife provisions and review the current science on planning for effective wildlife conservation at the landscape scale. We then discuss the approach to wildlife conservation planning in the 2012 rule and compare it to alternatives that were not selected and previous iterations of the planning rule. The new planning rule is of concern because of its highly discretionary nature and the inconsistency between its intent on the one hand and operational requirements on the other. Therefore, we recommend that the USFS include in the Directives for implementing the rule commitments to directly monitor populations of selected species of conservation concern and focal species and to maintain the viability of both categories of species. Additional guidance must be included to ensure the effective selection of species of conservation concern and focal species, and these categories should overlap when possible. If the USFS determines that the planning unit is not inherently capable of maintaining viable populations of a species, this finding should be made available for scientific review and public comment, and in such cases the USFS should commit to doing nothing that would further impair the viability of such species. In cases where extrinsic factors decrease the viability of species, the USFS has an increased, not lessened, responsibility to protect those species. Monitoring plans must include trigger points that will initiate a review of management actions, and plans must include provisions to ensure monitoring takes place as planned. If wildlife provisions in forest plans are implemented so that they are enforceable and ensure consistency between intent and operational requirements, this will help to prevent the need for additional listings under the Endangered Species Act and facilitate delisting. Although the discretionary nature of the wildlife provisions in the planning rule gives cause for concern, forward-thinking USFS officials have the opportunity under the 2012 rule to create a robust and effective framework for wildlife conservation planning.

KEY WORDS at-risk species, coarse-filter, fine-filter, focal species, forest planning, monitoring, viability.

In April 2012, the United States Forest Service (USFS) issued its final planning rule in accordance with requirements of the National Forest Management Act of 1976 (NFMA; 77 FR 21162). The 2012 rule took over 2 years to complete and included extensive public involvement, consultation through forums with scientists and policy experts, and environmental analysis conducted in accordance with the National Environmental Policy Act of 1969 (NEPA; USFS 2012). The new rule represents the most substantive change in federal forest policy in 30 years, with sweeping implications for wildlife. We review the administrative history of the planning rule, explore the provisions that affect the conservation of wildlife and biodiversity, and discuss how careful implementation could lead to more efficient and effective wildlife management. To provide a context for interpreting the changes that will come with implementation of the new rule, we begin with a short administrative history, and then provide a conceptual framework for interpreting the management implications of the rule. We also consider the intersection of the NFMA and the Endangered Species Act (ESA) and look at the implications of this rule change for ESA decision making. We conclude with a series of observations and recommendations for how the wildlife profession might help ensure that sound science and practical policy are effectively wed as the planning rule is implemented across the nation's public forest lands over the years to come.

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A BRIEF HISTORY OF THE 2012 PLANNING RULE

The NFMA created a 3-tiered, regulatory approach to planning. At the highest tier, national-level regulations govern the development and revision of second-tier forest plans. Site-specific plans for projects and other activities make up the third tier, and they must be consistent with both sets of higher-level regulations. Forest plans typically make zoning and suitability decisions and regulate various activities within a forest area, therefore acting as a gateway through which subsequent project-level proposals must pass. They do not, however, authorize or mandate site-specific projects. Instead, plans address issues such as the prioritization of various multiple-use goals, requirements for managing resources such as wildlife, watersheds, or soils, and the determination of which land is suitable for timber cutting, along with allowable volume and the choice of harvesting and regeneration methods.

Efforts to revise the rules governing Forest Service planning have been many, and debate has been intense, resulting in considerable confusion regarding the requirements, process, and legal provisions underlying recent forest planning and management. During development of the 2012 rule, the USFS operated under the 1982 planning rule (47 FR 43026), despite the issuance of more recent rules in 2000 (65 FR 67514), 2005 (70 FR 1023), and 2008 (73 FR 21468). The 2000 rule, developed by the Clinton administration with guidance from a Committee of Scientists (Committee of Scientists 1999), was deemed by the subsequent administration too "costly, complex, and procedurally burdensome" (77 FR 21162: 21164) to implement, and the USFS reverted to planning under the terms of the 1982 rule. Both the 2005 and the 2008 rules were enjoined by the courts because of a failure to meet legal requirements. The agency had argued that planning regulations did not have environmental impacts and thus did not require analysis under the NEPA and the ESA, but this argument failed to survive judicial review (*Citizens for Better Forestry v. USDA* 2007, 2009). A desire to address these persistent weaknesses and to avoid a similar judicial outcome is evident in the language of and justification for the 2012 rule.

One of the most controversial and highly litigated aspects of previous USFS planning rules has been the regulations written in accordance with the NFMA's requirement to "provide for a diversity of plant and animal communities based on the suitability and capability of the specific land area in order to meet overall multiple-use objectives" (16 USC 1604[g][3][B]). To interpret the diversity provision and other requirements of the NFMA, a Committee of Scientists was convened in 1977, in accordance with requirements of the NFMA, to assist with the development of the first planning rule (issued in 1979 and revised in 1982). The diversity regulations in the 1982 rule required that "fish and wildlife habitat shall be managed to maintain viable populations of existing native and desired non-native vertebrate species in the planning area" (36 CFR 219.19). The reference to "viable populations," drawn directly from fundamental principles of population biology, embedded specific, scientific intent into the Forest Service's planning and management responsibilities.

Subsequently, this provision caused significant controversy and drove change in forest management (Corbin 1999, Duncan and Thompson 2006). For example, compliance with the viability provision initiated litigation over the northern spotted owl (*Strix occidentalis caurina*), and led the courts to reject forest plans in the Pacific Northwest for failure to protect the viability, not only of the owl, but also of other species associated with late-successional forests (Duncan and Thompson 2006). Implementation of the 1982 rule relied primarily on the selection of management indicator species, like the northern spotted owl, meant to serve as surrogates to indicate management impacts on a broader suite of unmeasured species. Most forests indirectly assessed the status and trends of management indicator species by measuring habitat amount, a controversial practice that has been the subject of numerous court cases (Corbin 1999). Nonetheless, the use of habitat as a proxy for population status was established in court as not necessarily prohibited by the 1982 regulations (*Inland Empire Public Lands Council v. USFS* 1996).

In the 1990s, the USFS made several attempts to revise the planning rule, and in 1997 a second Committee of Scientists was convened. Its recommendations served as the basis for the 2000 rule, which maintained the viability requirement and extended it to all plant and animal species. The Committee of Scientists suggested a combination of coarse-filter approaches, which focus on the maintenance of ecosystems defined in terms of dominant vegetation types and their successional stages (see Hunter 1990), and fine-filter approaches, which involve direct species-specific measurements of population status and trends (Haufler et al. 1996, Committee of Scientists 1999). Specifically, the 2000 rule required that focal (see below) and at-risk species be monitored using fine-filter approaches. Diversity provisions of the 2000 rule were never implemented, because in 2001 the USFS reverted to the 1982 rule, using a transitional provision in the 2000 rule, while the Bush administration initiated revisions to the planning rule. Both the 2005 and 2008 rules relied entirely upon a coarse-filter approach to wildlife conservation. Contrary to assertions from the scientific community (Noon et al. 2003, 2005), the USFS argued that maintenance of broad ecosystem diversity (as represented by coarse-filter approaches) would adequately protect species and address their diversity obligations under the NFMA. These rules did not require any fine-filter, species-specific planning or monitoring. When the 2005 and 2008 rules were enjoined, the court gave the USFS the option of using the 2000 or the 1982 rule. The USFS chose to use the provisions of the 1982 rule, including the viability provision, through the transitional language in the 2000 rule. In its justification of the most recent planning effort, the USFS claims that the 1982 rule is out-of-date in its scientific foundations, planning procedures, and social values, and is too complex, expensive, and procedurally burdensome to implement (77 FR 21162).

CONCEPTUAL BASIS FOR WILDLIFE CONSERVATION PLANNING

In addressing asserted shortcomings of the 1982 rule, the Forest Service adopts an approach to wildlife conservation that hinges primarily on the assessment, analysis, management, and monitoring of habitat. The 2012 Programmatic Environmental Impact Statement for the planning rule states, "The best opportunity for maintaining species and ecological integrity is to maintain or restore the composition, structure, ecological functions, and habitat connectivity characteristics of the ecosystem. These ecosystem components, in essence, define the coarse-filter approach to conserving biological diversity" (USFS 2012:126). This contrasts with the 1982 and 2000 rules that emphasized population viability.

A Combined Coarse-Filter/Fine-Filter Approach

Most wildlife ecologists believe that effective biodiversity conservation planning requires an appropriate balance between habitat-based, coarse-filter approaches and insights from fine-filter, species-level assessment and monitoring (Noon et al. 2009). The 2012 Programmatic Environmental Impact Statement for the planning rule recognizes the limits of the coarse-filter approach stating, "initially at least, some amount of direct species measurement may be needed to assess the effectiveness of the ecological conditions provided under the coarse-filter approach in achieving the goal of conserving the biological diversity of the area" (USFS 2012:124). The impact statement goes on to propose that fine-filter strategies "can be focused on the few species of special concern whose habitat requirements are not fully captured by coarse-filter attributes." However, this language understates the importance of a complementary fine-filter approach. Research indicates that the coarse-filter approach is unlikely to provide a reliable basis for multi-species conservation planning (Cushman et al. 2008), only limited testing of the approach's validity has occurred (Noon et al. 2009), and the monitoring of a select group of species using a fine-filter approach is necessary to determine the efficacy of coarse-filter approaches (Committee of Scientists 1999, Flather et al. 2009). A recent review of the degree to which coarse-filter models can be used to infer animal occurrence concluded that ". . . the observed error rates were high enough to call into question any management decisions based on these models" (Schlossberg and King 2009:609). These authors went on to state, ". . . [coarse-filter] models oversimplify how animals use habitats, and the dynamic nature of animal populations" (Schlossberg and King 2009:609).

Defaulting to vegetation type as a descriptor of a species' habitat has a long history in ecology. It has been driven largely by pragmatism—vegetation is much easier to measure and characterize than prey resources or nest sites, for example. The practice continues because detailed vegetation maps exist for most of the United States based on either extensive ground-surveys or remotely sensed imagery (e.g., USFS LandFire Program). However, vegetation is often a poor proxy for more influential, but difficult to measure resources, and the frequent failure of vegetation-based habitat models to predict a species' distribution and abundance may be because of limitations of this assumed relationship (Van Horne 2002, Cushman et al. 2008). Even with more detailed data on habitat characteristics, unmeasured and unknown factors will still affect populations. For these reasons, population status of focal and at-risk species must be directly assessed. Therefore, a coarse-filter approach based primarily on dominant vegetation communities will have limited ability to predict the distribution and abundance of many wildlife species and effectively address the diversity provisions of the NFMA; this requires both coarse- and fine-filter approaches.

Selecting Species for Fine-Filter Assessment

Striking a balance between coarse-and fine-filter assessments of biological diversity has challenged forest managers for decades. Even if the fine-filter approach was restricted to vertebrates, monitoring the status of all species is not feasible, thus previous planning rules have restricted USFS requirements to an assessment of a small subset of species occurring across the planning area. This pragmatic constraint was recognized in the 1982 planning rule with the designation of management indicator species, species assumed to reflect the effects of management on their populations as well as the populations of many unmeasured species. However, the notion that a single species can serve as an indicator for a suite of species is an untested premise and generally not supported by research studies or ecological theory (Noon et al. 2009, Cushman et al. 2010). The concept that some species act as direct surrogates of others is untenable unless those species share similar population drivers (Cushman et al. 2010).

Instead of management indicator species, the second Committee of Scientists recommended the use of "focal species" (Committee of Scientists 1999) to evaluate status and trends of plant and animal diversity, generally. The Committee of Scientists proposed that focal species would commonly be selected on the basis of their functional role in ecosystems (e.g., they serve keystone functions [Mills et al. 1993], they are indicators of exposure to key stressors [Caro and O'Doherty 1999], they have a role as engineers of ecological processes [Jones et al. 1994], or play an important role in food web dynamics [Soule et al. 2005]). For federal public lands, Noon et al. (2009) suggest a combined coarse-filter and fine-filter approach, with the latter focusing on monitoring threatened, at-risk, and rare species, along with a modest number of focal species selected with complementary and comprehensive functional roles as described above. Systematic approaches exist for identifying and prioritizing an informative subset of species for fine-filter assessment and monitoring. For example, Regan et al. (2008) suggest selecting species based on existing schemes, such as The World Conservation Union (IUCN) Red List, Nature Serve, Partners in Flight databases, and federal or state listings, combined with an assessment of the degree and spatial and temporal characteristics of known threats. Nevertheless, uncertainties regarding the ability to generalize inferences drawn from any subset of species make the selection process of fundamental importance to the successful implementation of the fine-filter approach.

Improved Techniques for Fine-Filter Monitoring

One argument against direct assessment of wildlife populations is that it is not financially feasible. Traditional monitoring programs and viability analyses have been based on estimates of demographic parameters such as abundance, density, survival, and reproductive rates (Beissinger and McCullough 2002). Estimates of these parameters are expensive, require extensive field surveys, often involve capture and marking of individual animals, and are available for only a small number of species. However, indirect estimates of a species' status and trend based on their spatial distribution can provide defensible surrogate measures (MacKenzie and Nichols 2004, Manley et al. 2004). Focusing on distribution, rather than traditional measures of population size and growth rate, greatly increases the efficiency of broad-scale monitoring programs (Noon et al. 2012). Advancements in wildlife monitoring, based on detection/non-detection data, including the use of sign surveys, genetic evaluation, and historical presence–absence survey data decrease the cost of monitoring changes in distribution, which can be inferred from the proportion of sample units at which the species is detected (MacKenzie et al. 2006). One of the most significant advances in detection/non-detection monitoring is the ability to confirm the presence of a species at a survey site based on its genetic signature (e.g., in hair or scat; Waits 2004, Schwartz et al. 2006). The July 2005 issue of the *Journal of Wildlife Management* devoted a special section to the application of presence–absence sampling in wildlife monitoring (Vojta 2005), including an application to National Forest System lands (Manley et al. 2005). One variable estimated by these models is the area occupied by a species, a measure of a species' spatial distribution. Temporal and spatial patterns in detection/non-detection monitoring data allow inference to changes in animal abundance (MacKenzie and Nichols 2004), the single most influential parameter that provides insights into likelihood of species persistence (Lande 1993). Thus, previous arguments citing the practical limitations of the fine-filter approach are blunted by recent technical and statistical research, much of it inspired by the difficulty and expense of implementing earlier approaches to fine-filter assessments under the 1982 planning rule.

Political and Administrative Barriers to Effective Biodiversity Conservation Planning

In the past, very few if any management indicator species have been monitored in a manner that would allow a reliable assessment of their response to management (Noon et al. 2009). Managers cite the lack of monitoring data as a critical limitation in understanding cumulative impacts to species (Schultz 2012). Aside from cost and the technical challenges discussed above, funding and implementation of reliable, species-specific monitoring has been a significant challenge on National Forest System lands because of political reasons. Maintaining the political and fiscal will to support long-term monitoring programs is difficult (Doremus 2008, Biber 2011). In addition to the challenges of chronic under-funding, management agencies face disincentives to implementing robust species-level monitoring plans because monitoring data may reveal the negative impacts of management. For example, documenting the impacts of timber harvest or fuels reduction activities on sensitive wildlife species often highlights conflicts between different agency mandates, each of which enjoys strong political and social support. In addition, funds allocated to monitoring may draw funds away from projects that result in immediate job creation, the provision of marketable goods such as timber, the attainment of fuels reduction and restoration goals, or other accomplishments that can be reported to Congress in a timely manner. Furthermore, an agency could face legal challenges if it makes enforceable monitoring commitments that it does not have the funding to implement. However, at least as they are typically drafted, monitoring plans are difficult to enforce in court, obviating the need to fully implement intended programs. The judiciary usually finds commitments to monitor land-use plans not subject to review under the parameters of administrative law, and even when reviewed in court, determinations regarding the adequacy of monitoring data are traditionally left to the expertise of administrative agencies (Biber 2011).

Several other issues make understanding management effects on wildlife populations problematic. For example, the USFS has often monitored impacts to species at the project level (Schultz 2010), a spatial scale with generally small population-level effects. Small effect sizes require high statistical power for their detection. The disparity between the scale at which population responses can be detected and the scale of individual management actions leads to persistent problems in assessing impacts to species viability (Ruggiero et al. 1994). Monitoring impacts to habitat must be done cumulatively and at multiple spatial scales to assess whether small-scale habitat changes affect individual organisms, interrupt landscape connectivity affecting multiple populations, or synergistically interact with other small-scale disturbances, resulting in broad-scale effects.

Finally, the integrity of any monitoring plan, coarse- or fine-filter, depends on the articulation of clearly stated objectives and triggers to management actions. A trigger point is a threshold value for a monitoring state variable (e.g., percent area occupied by a given focal species within a national forest planning area) that, when exceeded, triggers a particular management response. A monitoring program without triggers selected a priori to call attention to trends provides little more than a retrospective time series of data with no feedback—and therefore little value—to the management decision-making process (Noon 2003). Furthermore, the efficacy of a monitoring program cannot be assessed at adoption without pre-defined trigger points. Trigger points can be most objectively set up-front, before the difficult management changes that might result from crossing such points are proximate. This is especially true if effects are analyzed exclusively at project scales, masking broader trends. In such cases, declines in population size or habitat quality, for example, may occur incrementally with no recognition of impact until a decline in species status is clearly established via listing under the ESA (Schultz 2010). To provide value to the forest planning process, a monitoring program must establish, a priori, the magnitude of change in the monitoring state variable that would trigger a review of management practices.

In summary, a comprehensive wildlife assessment framework would include a combination of both coarse- and fine-filter approaches. It would commit to monitoring at-risk and focal species using recent advances in monitoring approaches that make species-specific monitoring more financially feasible and efficient than it has been in the past (Noon et al. 2012). As required for effective and meaningful adaptive management, monitoring would occur at multiple spatial scales and use pre-defined triggers to meaningfully evaluate the consequences of management actions and to inform future management decisions.

AN OVERVIEW OF THE 2012 PLANNING RULE'S DIVERSITY PROVISIONS

The planning framework for the 2012 final rule involves a 3-step process: assessment; plan development, amendment, and revision; and monitoring (36 CFR §219.5 [2012]). It requires the use of the "best available scientific information to inform the planning process" (36 CFR §219.3 [2012]) and identifies restoration and watershed protection as agency priorities, while emphasizing the contributions of sound forest management to ecological, social, and economic sustainability (36 CFR §219.8 [2012]). Because restoration requires: 1) an assessment of the current system state relative to desired future conditions; 2) measurement of the system state subsequent to management activities; and 3) a comparison of the observed to desired state, restoration is critically dependent on monitoring. In this section, we discuss the approach in the 2012 rule and the alternatives that were considered but not selected in the agency's decision process.

Assessment and Planning

Section 219.9 outlines the approach for providing for diversity of plant and animal communities. It explains that the USFS is adopting "a complimentary ecosystem and species-specific approach," or a combined coarse- and fine-filter approach. Paragraph (a) outlines the coarse-filter requirements to maintain ecosystem integrity and diversity: plans "must include plan components . . . to maintain or restore the ecological integrity of terrestrial and aquatic ecosystems and watersheds in the plan area" and "maintain or restore the diversity of ecosystems or habitat types throughout the plan area" (ecological integrity and diversity are defined in §219.19 of the 2012 rule). Plan components must function to maintain or restore ecosystem structure, function, composition, connectivity, key ecosystem characteristics, rare species communities, and native tree diversity. A commitment to restore or maintain landscape connectivity to facilitate movement, migration, and dispersal is a significant addition to the planning rule. Paragraph (b) outlines the fine-filter approach. It begins by explaining that the responsible official must determine whether the plan components under part (a), the coarse-filter requirements, will provide the necessary conditions to contribute to the recovery of species listed as threatened or endangered under the ESA, or species that are proposed or candidate species for listing. Additionally, the responsible official must determine whether the coarse-filter approach is sufficient for maintaining viable populations of "species of conservation concern." These are species known to occur in the plan area, other than those listed, proposed, or identified as candidate species under the ESA, that are selected by the Regional Forester based on "substantial concern about the species' capability to persist over the long-term in the plan area" (36 CFR §219.9[c] [2012]). If the coarse-filter is deemed to be insufficient, the responsible official must include species-specific plan components (e.g., buffer areas around nest sites), that will contribute to the recovery of populations of species of conservation concern, as well as federally listed, proposed, and candidate species. If the coarse-filter is assumed adequate, no further species-level consideration is employed in planning. Yet how responsible officials will be held accountable for such decisions is unclear. The burden of proof for determining the effectiveness of the coarse-filter approach is not addressed. These species-specific requirements represent the USFS commitment to the fine-filter approach in section 219.9.

Notably, the new rule eliminates the requirement for maintaining viable wildlife populations, in contrast to the 1982 rule's viability provision for vertebrates and the provisions of the 2000 rule that would have extended the requirement to other species. Since the agency only commits to maintaining the viability of species of conservation concern, under the 2012 rule the USFS has no obligation to address the decline of any species not listed, proposed, or a candidate under the ESA, unless the responsible official, in this case the Regional Forester, expresses substantial concern about its persistence. Thus, any number of species could pass from secure to endangered status before any federal intervention would be required. However, in contrast to the 1982 rule, the agency can commit to maintaining viable populations of non-vertebrates by identifying them as species of conservation concern.

Historically, the diversity provisions of the NFMA have been one of the most controversial aspects of the planning rule, and the issue of how the USFS should address the clearly established public values associated with wildlife conservation often has been overshadowed by legal and technical arguments about the practicality of specific approaches to viability assessment. For example, over the course of the drafting and judicial review of multiple rules, considerable disagreement existed as to whether a requirement to maintain viable populations of all species, or just vertebrate species, or just at-risk species was an attainable goal. Understandably, the USFS has been reluctant to commit the agency to a species viability standard with which demonstrating compliance is difficult. At any point in time, all species have some non-zero probability of extinction; thus, viability can never be guaranteed. Viability is a probabilistic concept that invokes a specific level of risk over a stated time horizon, and proponents of the viability standard have had difficulty explaining to the public—and sometimes to their colleagues in wildlife management—how probabilistic events can be addressed in legally enforceable standards.

Nonetheless, in its 2012 record of decision, the agency commits to maintaining the viability of species of conservation concern, arguing that the combination of coarse- and fine-filter approaches it proposes are scientifically defensible, will adequately protect biodiversity on its lands, and will not be too costly to implement (77 FR 21162). However, the planning rule does not specify how viability will be assessed or what information will be used to assess a species' viability. Additionally, species identified as being of conservation concern could experience sharp range restrictions, since the regulations no longer require viable populations to be well-distributed, as was the case under the 1982 rule. Instead, the new rule defines of a viable population as one that "continues to persist over the long term with sufficient distribution to be resilient and adaptable to stressors and likely future events" (36 CFR §219.19 [2012]).

Finally, the USFS may absolve itself of responsibility for species-level conservation if the agency determines that maintaining a viable population of a species of conservation concern is beyond the capability of the plan area. In this case, which might result from stressors extrinsic to the planning area, such as climate change or the loss of habitat in other regions, the responsible official is required to document the basis for that decision and include plan components that contribute to the maintenance of a viable population across multiple land ownerships, in coordination with other managers and private parties working across jurisdictional boundaries, to the extent practicable.

Monitoring

Monitoring requirements are outlined in section 219.12. The planning rule requires a monitoring program for each National Forest, which can be developed jointly across forests and must be developed in coordination with the Regional Forester and the Research and State & Private branches of the agency. Plan monitoring programs must include questions and indicators; for diversity, these include indicators addressing the status of ecological conditions and the status of focal species, defined in the rule as "a small subset of species whose status permits inference to the integrity of the larger ecological system to which it belongs and provides meaningful information regarding the effectiveness of the plan in maintaining or restoring the ecological conditions to maintain the diversity of plant and animal communities in the plan area. Focal species would be commonly selected on the basis of their functional role in ecosystems" (36 CFR § 219.19 [2012]). Regional Foresters are to develop "broader-scale monitoring" for questions that are relevant at scales larger than the planning area. In all cases, monitoring information is to be compiled, evaluated, made available to the public, and used to inform adaptive management of the plan area. Thus, the new rule adopts, for the first time, a multi-scaled approach for monitoring and codifies the intent, although not the process, for implementing a transparent and data-driven approach to adaptive management. Although the adoption of a focal species approach based on functional roles in sustaining ecosystem processes reflects the logic of the 2000 rule, the 2012 rule draws no connection between the monitoring of focal species and the conservation of their roles in the ecosystem. The new rule does not include a requirement to maintain the viability of focal species, despite the fact that it is the status of these species that is meant to indicate whether the USFS is successfully maintaining and restoring ecosystem diversity and integrity. Additionally, the 2012 rule does not provide a requirement to monitor species of conservation concern, despite their established vulnerability to local extirpation. Consequently, the fine-filter approach to monitoring is explicitly separated from the fine-filter approach for biodiversity conservation.

Alternatives Not Selected

Although a review of the key provisions of the planning rule provides direct insight into the place of wildlife conservation in the future of forest planning and management, examination of the alternatives not selected reveals the underlying logic, pivotal choices, and philosophical foundations of the Forest Service's interpretation of the NFMA and reconceptualization of its institutional role and responsibilities to the public. The USFS considered several other alternatives in its Programmatic Environmental Impact Statement, in addition to the selected alternative (i.e., the final rule), which was a modified version of Alternative A. Alternative B closely followed the 1982 rule, notably in regards to the viability provision (". . . fish and wildlife habitat shall be managed to maintain viable populations of existing native and desired non-native vertebrate species in the planning area . . ." [36 CFR 219.19]). The agency provides a lengthy rationale for not selecting Alternative B, focusing on the defects of the 1982 viability provision (see 77 FR 21162:21168). This rationale also pertains to the selection of the final rule (modified Alternative A), which dropped the 1982 viability provision with the exception of "species of conservation concern" (see below). The agency states the 1982 rule "included planning procedures that do not reflect current science or result in unrealistic or unattainable expectations because of circumstances outside of the Agency's control, particularly for maintaining the diversity of plant and animal species" (77 FR 21162:21169). The USFS further justifies dropping the requirement to maintain species viability by stating, "[T]here are limitations on the Agency's authority and the inherent capability of the land" (77 FR 21162:21169). It notes that forest clearing in South America and habitat fragmentation in the Rocky Mountains on private land affect the agency's ability to maintain viable populations on National Forest System lands. For reasons such as these, the agency notes, the USFS cannot ensure a species' existence in the planning area when circumstances outside of its control may be contributing to population declines. It also notes that managing for the habitat of a single species sometimes impinges on management requirements for a species listed under the

ESA, or on other necessary activities the agency must undertake to comply with statutory requirements. Furthermore, the agency writes, some forests simply cannot support viable populations of species that are rare and far-ranging, like wolverines (*Gulo gulo*), and require more habitat than is available on a single National Forest unit.

Alternative C included no specific provisions for biodiversity conservation beyond the minimum requirements of the NFMA. This alternative was highly discretionary, leaving decisions about the requirements for assessment, planning, and monitoring to the USFS Directives' System (i.e., the agency's handbook and manual), whose provisions are not legally binding. The high degree of discretion in this alternative, according to the agency, would have resulted in too much variation in implementation: "There would be no certainty with regard to the inclusion of any plan components beyond the minimum required by this Alternative, and a potential lack of consistency across the National Forest System" (77 FR 21162:21170).

Alternative D "was designed to evaluate additional protections for watersheds and an alternative approach to addressing the diversity of plant and animal communities" (77 FR 21162:21170). This alternative required watershed-scale assessments of climate change vulnerability and designation of key watersheds to anchor the assessment and maintenance of the ecological status of aquatic, riparian, and terrestrial components of watersheds (USFS 2012). Establishing connectivity between habitats and discrete populations of species would also have been required. The alternative maintained and extended the 1982 viability requirement, stating the National Forests would provide for viable populations of native and desired non-native species in each planning area. The USFS was required to use the best available science to determine ecological conditions necessary to support viable populations, as informed by the "current and likely future viability of focal species within the planning area" (USFS 2012:F-9). To address the agency's concern that it cannot ensure the viability of populations on its lands, Alternative D included language that required the Secretary of Agriculture to provide notice to the public and allow for public comment if the agency determined it could not provide for viable populations of native or desired non-native species in a plan area. Furthermore, the agency was required to provide for viability of such a population to the maximum extent practicable and to take no actions that would increase the likelihood of extirpation of a population in the planning area. As with the selected alternative, Alternative D required monitoring of the status and trends of focal species, but with the additional requirement that triggers be identified for focal species' monitoring that would initiate a review of planning and management decisions to achieve compliance with the viability standard. This alternative explicitly stated that population surveys of focal species would be conducted using presence–absence data, occupancy modeling, genetic monitoring, or count-based methods. Alternative D was not selected because of the high anticipated planning and monitoring costs (77 FR 21162). The record of decision states that many plans already incorporate elements of this alternative, but that it is too prescriptive to allow for efficient, effective, and flexible management of all National Forests (77 FR 21162).

Finally, Alternative E was highly prescriptive in terms of requirements for public notification, assessment, and monitoring. It would have required specific monitoring questions, indicators, and triggers for changes in management action. The diversity requirements would have been similar to those in the selected alternative, but with more emphasis on monitoring of species' status and trends. The alternative was rejected for the same reasons as Alternative D.

MANAGEMENT IMPLICATIONS AND RECOMMENDATIONS FOR IMPLEMENTING THE 2012 PLANNING RULE

In theory, the new planning rule could be implemented in a robust way, drawing on the best available science to protect plant and animal diversity on National Forest System lands. However, the primary change introduced by the 2012 rule is the considerable discretion afforded centralized authorities, particularly at the regional level, in how general provisions will be implemented. Based on the management history of the USFS, numerous aspects of the 2012 planning rule are of concern, primarily because they defer many fundamental details to the interpretation of officials who may lack scientific background and disciplinary depth in wildlife biology and may have disincentives to prioritize wildlife. A number of scientists and scientific societies (including The Wildlife Society) commented on the draft rule and noted that it leaves more decisions about diversity conservation to agency discretion than did the 1982 rule. Forest Service officials must strike a fine balance between prescriptive standards and discretion or flexibility in a rule that is meant to guide planning years into the future on the entire National Forest System. Although some discretion is necessary, a rule must be sufficiently prescriptive to ensure that the National Forests do not implement a loosely written and unenforceable standard with so much variability across management units as to compromise the conservation of biological diversity.

Discretion, Authority, and Responsibility in Wildlife Conservation

Highly discretionary mandates are especially problematic for protecting resources such as wildlife that, without clear substantive requirements, have historically received less attention in land management. The 1897 Organic Act gives the USFS wide discretion by providing an open-ended mandate to secure water flows and provide timber. The Multiple Use Sustained Yield Act (MUSYA), passed in 1960, expanded the factors that the USFS must consider in planning, including wildlife conservation. However, the language in the MUSYA does not require the USFS to conserve wildlife in any specific fashion, only to consider the wildlife resource when planning for multiple-use. The concept of multiple-use, according to the courts, "breathes discretion at every pore" (*Perkins v. Bergland* 1979). Wildlife never gained serious consideration in forest management under the MUSYA, in part because of the agency's deference to state wildlife agencies, which have generally focused on game species and sport fisheries.

We have consistently heard many USFS personnel argue that their primary responsibility is to manage the habitat on USFS lands, whereas actual populations are the domain of the states. However, the USFS clearly has the power to manage wildlife on its lands. The United States Constitution's Property Clause (Art IV, section 3) gives Congress proprietary and sovereign powers over its property, and it may delegate decisions regarding federal lands to executive agencies. The Supreme Court has repeatedly observed that this power over federal land is "without limitations" (*United States v. San Francisco* 1940). The Court's expansive reading of the Property Clause also extends to managing wildlife on federal lands. The dispositive case is *Kleppe v. New Mexico* (1976), where the Court states, "the 'complete power' that Congress has over public lands necessarily includes the power to regulate and protect the wildlife living there" (426 U.S. 529: 541). Of course, the states also manage wildlife on federal lands, but as made clear in *Kleppe*, "those powers exist only in so far as [their] exercise may be not incompatible with, or restrained by, the rights conveyed to the Federal government by the Constitution." (426 U.S. 529: 545). Though the USFS seldom chooses to assert its full wildlife management powers, the Courts continue to emphasize the Property Clause's application to wildlife (see, e.g., *Wyoming v. United States* 2002).

Concerns about wildlife were one of the central factors precipitating the passage of the NFMA in 1976, and the USFS has a clear responsibility under the Act to manage for biodiversity. The Act's legislative history shows that its diversity provision was meant to require "Forest Service planners to treat the wildlife resource as a controlling, coequal factor in forest management and, in particular, as a substantive limitation on timber production" (Wilkinson and Anderson 1987:296). When the NFMA was passed, it included language stating that the USFS has a responsibility to be "a leader in assuring that the Nation maintains a natural resource conservation posture that will meet the requirements of our people in perpetuity" (16 U.S.C. §1600[6]) and an explicit requirement to protect plant and animal diversity. To ensure that the agency's new requirements were effectively translated into administrative regulations, Congress required the agency to convene a Committee of Scientists to inform the writing of these regulations, which were finalized in 1982 (16 U.S.C. §1604[h][1]).

Timber harvest on the National Forests, nonetheless, continued to increase steadily, until the late 1980s. At that time, citizen enforcement, frequently manifest through appeals and litigation based on substantive provisions like the 1982 rule's viability standard and the ESA, was a major factor that led to significant declines in timber production (from >13 million board feet/year in the late 1980s to <2 million in the early 2000s). Legal exposure created by the suite of substantive requirements to protect biological diversity under the NFMA and ESA forced the agency to address wildlife conservation, something that had not come to pass under the MUSYA. However, even in the 1990s, pressure to prioritize timber production over the protection of wildlife remained strong because of internal biases, financial incentives, and Congressional intervention (Wilkinson 1992, Government Accountability Office 1997, Corbin 1999).

Although agency culture and priorities have shifted over time, biodiversity conservation still may conflict with activities like timber harvest, fuels reduction, recreation, or energy development, all of which the USFS has strong economic and political incentives to promote. Literature in political science and economics predicts that when given conflicting tasks by Congress, such as the multiple use mandate, agencies systematically prioritize high incentive and measurable goals over those that are lower incentive and more difficult to measure (Biber 2009). A highly discretionary NFMA diversity regulation could lead the USFS to prioritize higher incentive and measurable goals that are supported by political interests.

Given this reality, even when regulations for protecting plant and animal diversity are well meaning and scientifically sound, if they are not specific, measurable, binding, and enforceable, history suggests that effective wildlife conservation planning will end up as a secondary objective (Houck 1997). Specific, mandatory language is needed to protect wildlife on the National Forests, a point not lost on the first Committee of Scientists, who wrote the following in 1979, "It is simply not possible to carry out the planning requirements of NFMA in accordance with a set of regulations that contain nothing but generalities" (44 FR 53967: 53968). Such specificity, said the Committee, is what the NFMA requires. Historically, the NFMA's diversity provision and its associated regulations have provided an effective counterbalance to competing agency demands and political pressures. However, without more specific requirements, the administrative discretion in the 2012 rule's diversity provisions will lead to varied implementation across management units, give managers who are not committed to wildlife conservation the leeway to pursue other management goals without concern for biodiversity, and leave managers who are committed to protecting biodiversity without a solid, legal framework to help them withstand internal and external pressures to prioritize other factors.

Although the diversity provisions in the 2012 planning rule itself are highly discretionary, the agency, through the Directives system, could adopt standards and practices for wildlife conservation that are more prescriptive and would help to ensure that the rule is implemented in a more robust fashion and informed by the best available science. We urge the agency to implement the rule in a manner that closes the gap between the stated purpose of maintaining ecological integrity and diversity, and the highly general and discretionary operational provisions in the rule that are meant to achieve these purposes. The Wildlife Society and other professional organizations can play an important role in guiding this process, and for this purpose, we offer a series of recommendations that would strengthen the key wildlife provisions in the 2012 rule.

Coarse-Filter Contributions

Coarse-filter approaches, typically focused at broader spatial scales than fine-filter strategies, are aimed at communities, ecosystems, or landscapes (Schwartz 1999). Their central role in the 2012 rule complements fine-filter provisions and commits the USFS to multi-scaled assessment and monitoring efforts. Coarse-filter conservation strategies often rely on habitat predictors (e.g., dominant vegetation and landform) derived from satellite imagery (e.g., the California Wildlife Habitat Relationships System, http://www.dfa.ca.gov/biogeodata/cwhr). Under this approach, all appropriate habitats within a planning unit that intersect the species' geographic range are typically assumed to support the species. This assumption is often based on anecdotal occurrence data because the spatial extent of coarse-filter strategies often constrains the agency's ability to implement probability-based survey designs. The consequence is that commission errors are likely, which can lead to the erroneous conclusion that animal diversity is being maintained when it is not. Although these concerns limit the ability the coarse-filter approach to serve as a substitute for fine-filter assessments, a management objective to sustain dominant vegetation communities and their successional stages at broad spatial scales is an essential aspect of a comprehensive approach for sustaining biological diversity. In the context of the diversity requirements of the 2012 rule, measures of the effectiveness of the coarse-filter are presented in terms of species' metrics (e.g., number of rare and imperiled species conserved, presence of apex consumers, species richness, etc.). Therefore, verifying the efficacy of the coarse-filter approach requires some level of direct species-level assessment, and a comprehensive diversity policy requires a carefully balanced coarse-filter/fine-filter strategy.

Implementing the Fine-Filter Approach

We are concerned with the limited commitment to conduct fine-filter (species-level) assessments in the new rule. We found little scientific evidence to suggest that maintaining the diversity and integrity of a combination of habitat types "will provide the ecological conditions for the long-term persistence of most species within the plan area" (36 CFR §219.9). The Committee of Scientists stated, "Habitat alone cannot be used to predict wildlife populations" and "diversity is sustained only when individual species persist; the goals of ensuring viability and providing for diversity are inseparable" (Committee of Scientists 1999, Chapter 3:19,38). For this reason, the fine-filter species assessment is critical.

The rule is inaccurate in the way it portrays its coarse- and fine-filter approaches. It claims that the coarse-filter approach, along with the inclusion of fine-scale habitat management requirements for species that are not adequately protected, constitutes a combined coarse-filter/fine-filter approach. This discussion misconstrues fine-filter species conservation approaches, which entail direct assessment at the species level, including monitoring state variables such as a species' abundance, density, survival, birth rate, or occupancy. Managing fine-scale habitat components for a given species is not the same as fine-filter assessment.

The USFS defines focal species, in part, based on their functional significance to ecosystem processes (36 CFR §219.19[2012]). The planning rule requires the selection and monitoring of focal species "to assess the ecological conditions required under §219.9 . . ." (§219.12[a][5][iii]), and it is this aspect of the rule that holds the most promise as a genuine, complimentary fine-filter approach to wildlife conservation planning. The USFS defines ecological conditions as "the biological and physical environment that can affect the diversity of plant and animal communities, the persistence of native species, and the productive capacity of ecological systems" (36 CFR §219.19[2012]). An emphasis on monitoring species with known or suspected functional significance to ecosystems process and sustainability is appropriate. Ecosystem resilience is strongly related to native species diversity and functional redundancy (the degree to which multiple species perform similar ecosystem functions [Naeem et al. 2009]). In general, ecosystems with greater native species diversity are more resistant to disturbance, recover more quickly following disturbance, and are less likely to experience irreversible changes than species-poor communities (Cottingham et al. 2001, Hooper et al., 2005, Naeem et al. 2009). Furthermore, species loss ranks among the most severe global change stressors, with effects comparable to those of climate warming, acidification, and elevated carbon dioxide (Hooper et al. 2012). Therefore, it is inconsistent with the stated intent of §219.9 to maintain or restore ecological conditions not to include a commensurate requirement to maintain viable populations of focal species.

Another central requirement of the 2012 rule is the mandate to contribute to the recovery of proposed, candidate, and listed ESA species and to protect viable populations of species of conservation concern. Section 219.9 requires that species-specific habitat management components be built into plans if the responsible official determines that coarse-filter approaches are insufficient for maintaining viable populations of species of conservation concern, and ESA species, within the plan area. We are concerned that, as presently construed, the rule does not require the monitoring of these species. Thus, it is unclear what information will be used to determine if a species maintains a viable population within the plan area, or if it requires additional species-specific conservation actions. Because the coarse-filter approach may be insufficient to provide insights into the status and trend of species (Cushman et al. 2008), some direct species-level monitoring is necessary. Without such monitoring, the USFS's approach is problematic; by the time evidence of further decline for these already at-risk species is found, threats may have significantly increased.

Ideally, the rule would have committed to population-level monitoring and viability for both focal species and species of conservation concern. Extending the viability requirement to focal species, selected in part because of their known or suspected functional significance, is a logical way to address the ecosystem integrity goals of the rule. Further, monitoring species of conservation concern will provide essential information to assess their viability. These changes, incorporated into the Directives, would connect the commitment to species-level conservation with the mandate for adaptive management and bring greater cohesion to the disjointed diversity provisions in the 2012 rule. In addition, all species-level monitoring should include trigger points so that significant declines in either focal species or species of conservation concern would initiate reviews of management policies.

Selecting Species of Conservation Concern and Focal Species

The process for selecting focal species and identifying species of conservation concern, separately or in concert, is not detailed in the rule. The rule simply states that the selection of species of conservation concern will be based on the best available science and evidence of substantial concern about their long-term persistence in the plan area. The Record of Decision indicates that further guidance will be provided in the Directives, but that the Department of Agriculture expects species to be identified based on existing classifications of risk, such as NatureServe conservation status or those listed as threatened or endangered under state law (77 FR 21162:21218). In addition to referencing NatureServe and state law, we recommend the agency also consider IUCN red-list species that are not already listed under the ESA, and high priority species identified in State Wildlife Action Plans; if such species are not selected, a rationale for failing to designate them as species of conservation concern should be required.

Criteria for focal species selection include the species' functional roles in the ecosystem and sensitivity to changing conditions, management activities, particular threats, or desired ecological conditions (77 FR 21162). This is consistent with recommendations of the most recent Committee of Scientists' Report (Committee of Scientists 1999). Additional guidance in the Directives will be necessary to establish and maintain consistency and efficacy across management units in the selection of focal species. Noon et al. (2009) provide useful guidance on focal species selection for fine-filter assessments on federal public lands. Furthermore, we see no reason that species identified as species of conservation concern cannot also be identified as focal species, providing a ready avenue for conceptual integration of the fine-filter approaches under the new planning rule.

Establishing a step-down process to identify and prioritize species for fine-filter monitoring that reflects the reality of Forest Service monitoring budgets remains a major challenge. This topic goes beyond the scope of our paper, but to initiate discussion, we suggest that identifying the core species (Magurran and Henderson 2003) that are 1) persistent members of a given management unit; 2) functionally significant; and 3) at risk in that unit may be a first step in developing a manageable species set.

Developing Informative Monitoring Programs

The planning rule requires forests to develop monitoring programs that will include a set of questions and indicators to track change, measure management effectiveness, and assess progress towards desired future conditions. The rule only commits to monitoring focal species, which as mentioned above, may include species of conservation concern (the fine-filter approach). It also requires monitoring a select set of ecological conditions in accordance with the objectives of §219.9 (the coarse-filter approach). The Regional Forester is required to develop a broad-scale monitoring plan to address issues relevant at a scale larger than a single National Forest. The content of the broad-scale monitoring plan is at the discretion of the Regional Forester, and s/he is required to coordinate with other jurisdictions, other branches of the USFS, and the public. Additionally, monitoring plans may be coordinated across units. The responsible officials are to conduct biennial evaluations of monitoring information and adjust management activities as necessary.

At the outset, the discussion of species monitoring in the Record of Decision (77 FR 21162:21232–21233) is confusing and suggests a critical misunderstanding by the USFS of environmental monitoring. The Record of Decision (77 FR 21162:21233) states, "The final rule does not require monitoring species population trends. Species population trend monitoring is costly, time intensive, and may not provide conclusive or relevant information." This perspective is at odds with the general understanding in the scientific literature of environmental monitoring. For example, Suter (1993:505) states that monitoring is the "measurement of environmental characteristics over an extended period of time to determine status or trends in some aspect of environmental quality." Monitoring of an appropriate state variable (e.g., occupancy) is conducted at regular intervals to assess both the current state and time trend in some ecological resource (e.g., a species' population [Noon 2003, Nichols and Williams 2006])—that is, the stated purpose of monitoring is to estimate temporal trends.

Provisions in the rule encourage the development of robust monitoring strategies. However, our primary concern is whether these strategies will be developed, funded, implemented, and designed in such a way that they inform adaptive planning. As noted previously, monitoring has been chronically underfunded by federal agencies. The rule requires development of a monitoring plan but does not specify a particular standard of quality or utility of monitoring data. Since Congress annually sets the agency's budget, the USFS cannot commit to funding monitoring at a particular dollar amount. However, committing a certain percentage of planning dollars to monitoring may be possible so that the USFS can address its commitment to adaptive management.

Following the United States Supreme Court's decision in *Norton v. SUWA* (2004), enforcing monitoring requirements of federal land use plans is difficult. In language easily extendible to NFMA plans, that case held that commitments to monitor in Bureau of Land Management land use plans are not generally binding or reviewable under the parameters of administrative law. The Court noted that monitoring requirements could perhaps be written in such a way as to make them enforceable, if they were written as clear and binding commitments. In some cases, when monitoring activities are clearly required before undertaking certain activities, monitoring can be enforceable in court (Blumm and Bosse 2007). However, because requiring or enforcing funding levels or data quality standards for monitoring programs is generally difficult, oversight will be necessary to ensure that monitoring occurs in a way that it clearly assesses management and restoration actions.

We recommend that multi-party oversight boards be established to aid in the design of monitoring programs, contribute to the selection and prioritization of monitoring state variables, provide science consistency checks, provide interpretations of the monitoring data, suggest when changes to management practices are needed, and advocate for consistent funding. Because monitoring data will unlikely be subject to judicial review, oversight from a multi-party stakeholder monitoring board could increase the likelihood that monitoring will provide reliable information and useful insights into future decision making (Nie and Schultz 2012). Such boards must consider how monitoring data will inform decision making and the level of statistical certainty required to trigger a change in management actions.

All species-level monitoring should include trigger points so that significant declines in either focal species or species of conservation concern will initiate reviews of management policies. If trigger points are not identified, monitoring data may not feed back into adaptive planning and decision making (Noon 2003). Triggers will be critical for species-level monitoring and for any evaluation of species viability. Monitoring enforceability also would be substantially increased if forest plans included requirements that before approving any major projects, such as those requiring an Environmental Impact Statement, the responsible official find that monitoring programs are being implemented and that no trigger points have been exceeded without corrective action.

Maintaining Current Populations and Adequate Distribution of Species

Whether the planning rule intentionally allows for local extirpation of species or range reductions in cases where this might be avoided is unclear, but the decline and loss of species from the planning area is an allowable outcome of USFS management under the new rule. Aside from the loss of a broader viability requirement, this is the most significant change from the 1982 rule: the replacement of language requiring that viable populations be well-distributed, with the definition of a viable population as one that "continues to persist over the long term with sufficient distribution to be resilient and adaptable to stressors and likely future events" (36 CFR §219.19 [2012]). The impact of the change stems from the fact that what constitutes a "sufficient distribution" is not defined in the rule, providing broad discretion to the responsible official and obfuscating the well-established relationship between geographic distribution and persistence likelihood (e.g., Harris and Pimm 2008).

Furthermore, the rule establishes that the USFS does not need to protect viable populations, as required in the 1982 rule, if this is not within the "inherent capability of the plan area," a vague concept that is never defined in measurable terms. In this case, the USFS is held to a much lower conservation standard: documenting the rationale for such a determination and working across land ownerships to create management standards and guidelines to maintain or restore conditions that will contribute to maintaining a viable population of the species within its range (36 C.F.R. §219.9(b)(2)(i) [2012]). The USFS also states, "the individuals of a species of conservation concern that exist in the plan area will be considered to be members of one population of that species" (77 FR 21162:21217). In light of this, whether the agency is committing to maintaining a viable population of a species of conservation concern when it is not within the inherent capability of a single plan area to protect a viable population is not entirely clear. Depending on how the agency interprets these standards, it might never have to commit to maintaining a viable population of a low-density, wide-ranging species, but it might have to commit to maintaining multiple viable populations of species with more constricted ranges.

To address ambiguities in the 2012 viability requirements, we recommend that the USFS explicitly recognize the importance of maintaining a wide geographic distribution for species viability. Species that are widely distributed across the landscape are much less likely to experience spatially correlated disturbance events (den Boer 1981). Maintaining the distribution and viability of rare or widely distributed species and populations will require close coordination among administrative units. Guidance should be included in the Directives indicating that the agency should assess viability (perhaps employing more efficient distributional analyses based on occupancy [Noon et al. 2012]) across ownerships and plan units, when this will enhance the likelihood of persistence for individual species. When the USFS determines that maintaining a viable population of a species is not within the inherent capability of the plan area, the agency should solicit scientific comment and review. This review will help ensure that the agency is aware of all relevant scientific information that may conflict with their determination and will better prepare the agency to defend its decisions against possible legal challenge. In cases where the USFS determines that providing for a viable population of a species that relies upon National Forest System lands for its habitat is not within the capability of the plan area, we recommend that the agency task itself with restoring populations, to the maximum extent practicable. At the least, a standard should be included in the Directives that directs the agency not to authorize or permit activities that reduce the viability of any species of conservation concern.

Development on private land and other activities external to National Forest System lands may affect species such that the USFS cannot alone ensure their viability. A critical question is to what extent should this compel the USFS to compensate for declines in species status due to factors outside of their control. Recall that the NFMA emphasizes the National Forests' role in conserving resources for the American people, in perpetuity. It does not imply that this objective is restricted to National Forest System lands. There is ample historical precedent for the USFS to consider what is happening outside of its jurisdiction and proactively respond on the National Forests (Nie and Miller 2010). In the view of the first chief of the USFS, Gifford Pinchot, 1 rationale for establishment of the National Forests was to compensate for unsustainable management of resources on private lands (Wilkinson 1992). Pinchot was focused on unsustainable timber harvest at the time, but the reasoning applies widely to other natural resources on USFS lands based on changing public values and priorities. The USFS, in its 2012 rule, emphasizes its responsibility to maintain and restore ecosystem diversity and integrity, and diverse plant and animal communities are fundamental to ecosystem integrity (Naeem et al. 2009). If development on private land is adversely affecting biodiversity, the USFS has a greater, not lesser, responsibility to protect species on its lands. This compensation principle will become even more significant given predictions of private land development in the future, with much of this development projected in the wildland urban interface (Nie and Miller 2010). The National Forests, and federal lands in general, will become more important to wildlife in increasingly developed landscapes. Therefore, the "inherent capacity" clause of the 2012 rule should be used rarely, if at all, and if used, be subject to scientific and public review. The USFS must recognize its increasingly important mission to conserve the nation's forest and grassland ecosystems during the current period of rapid global change and species loss, when unpredictable transformations of ecosystems may be the "new normal" (Barnosky et al. 2012).

Considerations Regarding the Relationship Between the NFMA and the ESA

Important intersections exist between biodiversity conservation requirements under the NFMA and the ESA, which work together as part of this nation's biodiversity conservation policy. Wildlife provisions in forest plans are a significant factor in ESA decision making (see below), and ESA decisions have profound and far-reaching implications for forest management. Ideally, viability protection on National Forests would serve as an early warning signal that a species may be heading towards local extirpation or extinction. A proactive approach to address risks to a species' viability could avoid costly and polarizing ESA decisions that might limit management flexibility for the USFS.

On the National Forests, currently 430 species are listed under the ESA as threatened or endangered, and an additional 60 species are candidates for listing (USFS 2011:14). More than 647,000 ha of terrestrial habitat and 35,000 km of stream habitat on USFS lands are designated as critical habitat under the ESA (USFS 2011:14). For these and other reasons, the 2012 planning rule emphasizes the connections between forest planning and the ESA more than previous regulations:

> The [Department of Agriculture] anticipates that plan components, including standards or guidelines, for the plan area would address conservation measures and actions identified in recovery plans relevant to T&E [threatened and endangered] species. When implemented over time, these requirements would be expected to result in plans that will be proactive in the recovery and conservation of the threatened, endangered, proposed, and candidate species in the plan areas. These requirements will further the purposes of section 7(a)(1) of the ESA, by actively contributing to threatened and endangered species recovery and maintaining or restoring the ecosystems upon which they depend (77 FR 21162:21215).

One way in which the USFS can actively contribute to species conservation and recovery is by providing wildlife and habitat-based standards in individual National Forest plans. The NFMA requires the incorporation of standards and guidelines in land and resource management plans (16 U.S.C. 1694). Standards are mandatory constraints on USFS projects and activities and are used to achieve or maintain desired conditions and planning objectives, to avoid or mitigate undesirable environmental impacts, and to meet applicable legal requirements (76 FR 8480). Guidelines, as commonly applied, also constrain decision making but allow for some deviation from rules as long as the intent of the guideline is achieved (76 FR 8480).

The types of wildlife and habitat-based standards used in forest planning differ in scale, specificity, and complexity. Some standards cover multiple National Forests, such as the Northwest Forest Plan's Aquatic Conservation Strategy (discussed below) and the Inland Native Fish Strategy. The latter, covering at one point 22 National Forests, is used to protect native fish and their habitats in eastern Oregon and Washington, Idaho, western Montana, and portions of Nevada. It does so by using several riparian management objectives, standards, guidelines, and monitoring requirements (USFS 1995). The Inland Native Fish Strategy's standards and guidelines replaced conflicting direction in multiple National Forest plans, except when those forests provided for more protection for inland native fish habitat. Standards can also be applied forest-wide, such as requiring that all snags over a certain size be retained or that a specified percentage of old growth be maintained on a National Forest. Other standards apply to particular management areas or zones as delineated in a land use plan; they often permit or prohibit various uses, such as grazing or the application of herbicides in a municipal watershed zone.

An enduring debate continues over the appropriate role of standards in forest planning. The 2012 rule requires every plan to include standards as 1 of 5 plan components (36 C.F.R. §219.7), but it leaves their application to the discretion of the responsible official, with the expectation that further direction will be provided in the Directives system (77 FR 21162:21206). Regarding the diversity of plant and animal communities, the rule requires standards or guidelines be used "to maintain or restore ecological conditions within the plan area to contribute to maintaining a viable population of the species within its range" (36 C.F.R. §219.9). Standards for wildlife protections should play a significant role in the new forest plans that will be written under the 2012 regulations. Legally binding and enforceable standards promote accountability and provide increased certainty about future management actions. Without them, there is an increased risk that wildlife protections will give way to other agency pressures and priorities.

Forest plan standards can play significant roles in decisions to list or delist a species under the ESA. One of the 5 factors to be considered by the wildlife regulatory agencies that enforce the ESA (the National Oceanic and Atmospheric Agency [NOAA] Fisheries and the U.S. Fish and Wildlife Service [USFWS]) in making ESA listing decisions is "the inadequacy of existing regulatory mechanism[s]" (16 U.S.C. §1533). Vague, voluntary, speculative, and unenforceable measures found in plans are generally not considered a sufficient regulatory mechanism (*Oregon Natural Resources Council v. Daley* 1998). Instead, federal wildlife agencies and the courts typically assess whether a plan contains specific and legally enforceable standards having regulatory force. Forest plan standards also can be relevant for determinations made by the wildlife regulatory agencies under section 7 of the ESA, which requires federal agencies to undergo consultation with the wildlife agencies to ensure their projects will not cause jeopardy to a listed species.

Several cases have been decided in which NOAA Fisheries and the USFWS made a no-jeopardy determination under section 7 of the ESA or decided not to list a particular species because a forest plan contained binding standards and other regulatory mechanisms to protect the petitioned species. One example is the decision not to list the Queen Charlotte goshawk (*Accipiter gentilis laingi*) in southeast Alaska. Roughly 80% of this region is managed by the Tongass National Forest, and petitioners argued that old-growth logging in the region posed a threat to goshawks. Standards and other regulatory mechanisms specified in the 2007 Tongass Land Management Plan were significant factors in the decision by the USFWS to not list the goshawk (72 FR 63133). The USFWS also emphasized the legally binding and enforceable nature of Tongass forest planning standards in its 1997 status review of the species (USFWS 2007), and the Department of the Interior asked the USFS to retain the Conservation Strategy in the 2008 Tongass Forest Plan Amendment. The USFS also recognizes the significance of these wildlife standards. Possible changes to the Strategy, according to Undersecretary of Agriculture Harris Sherman, "could hamper the plan's ability to maintain viable populations of plant and wildlife species [and] this could lead to the need for USFWS to reconsider its previous determinations regarding the goshawk . . ." (Sherman 2011:8).

The Aquatic Conservation Strategy, part of the Northwest Forest Plan, provides another example of the interactions between binding standards and the ESA (USFS and Bureau of Land Management 1994). The purpose of the Aquatic Conservation Strategy is to maintain and restore the ecological health of watersheds in the northwestern National Forests. The Strategy includes several binding standards and guidelines that apply to key watersheds, riparian reserves, required watershed analyses, and watershed restoration. In biological opinions written in accordance with section 7 of the ESA, NOAA Fisheries equates Aquatic Conservation Strategy consistency with no-jeopardy findings, a practice that has satisfied the courts (*Pacific Coast Federation of Fishermen's Associations v. National Marine Fisheries Service* 2001). Standards such as these can be used to protect wildlife while also achieving the restoration and watershed protection purposes of the 2012 rule.

The lack of enforceable standards and clear conservation commitments made in forest plans also has been a factor influencing decisions to list a species. In these cases, NOAA Fisheries and the USFWS determine that a forest plan fails to provide sufficiently certain, binding, and detailed protection to a species to count as an adequate regulatory mechanism. One of the most significant decisions in this regard is provided by the listing of Canada lynx (*Lynx canadensis*) as threatened in 2000 (65 FR 16052). The species was classified as a sensitive species by the USFS before listing, but most National Forests with lynx did not have population viability objectives or management standards and guidelines in place at the time (63 FR 37005). The fact that forest plans in effect at the time did not provide enough protection and guidance for the conservation of the lynx is a primary reason why the species was listed. The USFWS determined that these forest plans permitted several actions that cumulatively could cause a significant threat to lynx persistence across its range (63 FR 37005). The USFS responded to the listing by amending multiple national forest plans to incorporate various lynx standards and guidelines (USFS 2007). Currently, the USFS does not have to engage in ESA consultation with the USFWS on a project-by-project basis if these projects comply with these binding and enforceable lynx standards. Another prominent example is the 2010 decision to list the greater sage-grouse (*Centrocercus urophasianus*) as warranted-but-precluded, meaning the species is warranted for listing but precluded from actually being listed because of funding limitations (75 FR 13910). The USFS manages roughly 8% of the sagebrush habitat significant to the species. Greater sage-grouse were designated by the USFS as a sensitive species on USFS lands across the range of the species, and 14 of these forests designated the bird as a management indicator species (75 FR 13910:13979). But of the 33 National Forests managing greater sage-grouse habitat, "16 do not specifically address sage-grouse management or conservation in their Forest Plans, and only 6 provide a high level of detail specific to sage-grouse management" (75 FR 13910:13980). The lack of detailed protections and the variation among National Forest plans in the greater sage-grouse area was an important factor in making the warranted-but-precluded determination (75 FR 13910).

Enforceable wildlife standards and protections on the National Forests also play a role in delisting species from the ESA. One of the few species to be delisted under the ESA is the Robbin's cinquefoil (*Potentilla robbinsiana*), an endemic plant found in the White Mountains of New Hampshire, in areas managed exclusively by the White Mountain National Forest (67 FR 54968). The USFS was able to assist in the recovery of this species by restricting entry to particular areas of the National Forest, relocating trails, and entering into a Memorandum of Understanding with the USFWS. This Memorandum of Understanding included provisions related to habitat protection and monitoring, and it served as a long-term commitment by the USFS to conserve this plant, irrespective of its status and potential delisting under the ESA (USFS and USFWS 1994). The USFS regulations also prohibited removing, destroying or damaging plants that are classified as threatened, endangered, rare, or unique (36 C.F.R 261.9). All of these specific actions and commitments—the protective actions taken by the White Mountain National Forest, the plant regulations, and the Memorandum of Understanding—served as an adequate regulatory mechanism for delisting the species by the USFWS.

A more controversial example is the proposed delisting of the Yellowstone distinct population segment of grizzly bears (*Ursus arctos horribilis*). The lack of regulatory mechanisms to protect grizzly bear habitat on National Forest System lands was 1 reason why the species was listed in 1975 (40 FR 31734). A conservation strategy for the bear was written pursuant to its recovery plan to provide adequate regulatory mechanisms after the bear's delisting. The USFS amended 6 forest plans to incorporate the habitat standards and other provisions in the conservation strategy. The USFWS considers these standards to be adequate regulatory mechanisms for the purpose of delisting grizzly bears, but much of the debate and litigation over the delisting decision centers on the sufficiency of these standards. A district court found the delisting impermissible, partly because the amended forest plans contained discretionary and legally unenforceable guidelines, rather than binding standards, in the bear's primary conservation area (*Greater Yellowstone Coalition v. Servheen* 2009). The Ninth Circuit disagreed with the lower court on this matter and found the standards, as applied by the USFS within the primary conservation area, to be sufficient under the ESA because they are a legally enforceable part of National Forest plans, and management of these forests must be consistent with their governing forest plans (*Greater Yellowstone Coalition v. Servheen* 2011).

The 2012 rule also requires that forest plans provide the ecological conditions to "contribute to the recovery" of listed threatened and endangered (T&E) species (77 FR 21162:21215, 36 C.F.R. §219.9). The USFS has an expectation that forest plans would use standards or guidelines "to address conservation measures and actions identified in recovery plans relevant to T&E species" (77 FR 21162:21215). Better use of ESA recovery objectives could lead to more proactive, integrated, and strategically coordinated forest plans.

We recommend that more guidance be provided as to how synergies might be developed between forest and ESA recovery planning. Scott et al. (2005:386) show that "most listed species will require continuous management action in order to maintain their recovered status." These "conservation-reliant species" can only be maintained as a self-sustaining population in the wild "if ongoing management actions of proven effectiveness are implemented" (Scott et al. 2005:386). The Memorandum of Understanding and revised forest plan for Robbin's cinquefoil provide this sort of ongoing protection to a conservation-reliant species, and similar standards in forest plans could do the same for other T&E species on the National Forests.

The number of ESA listing decisions will only increase in the future, given the September 2011 settlement between the USFWS and environmental groups requiring the agency to make listing decisions on over 800 species, including 262 candidate species, for which such decisions have been delayed (Center for Biological Diversity 2012). Altogether, another 1,000 listing decisions will possibly have to be made by 2020 (Rylander 2012:10018). Furthermore, conservation scientists, the IUCN, and the Intergovernmental Panel of Climate Change all predict increases in the number of species threatened with extinction (Scott et al. 2010). For these reasons, the impact of ESA listing decisions on National Forest management is likely to increase over time. The use of binding standards in forest plans would likely serve to decrease the number of species listed as threatened and endangered and promote delisting decisions in the future.

If implemented in a robust fashion, the NFMA's diversity mandate will serve as a precautionary and proactive approach to wildlife conservation. In contrast, the ESA provides a more reactive and crisis-based approach to decision making, since the law's protective measures are usually not initiated until a jeopardized species is listed, and by that time, it is already in the proverbial emergency room. Federal wildlife agencies take an average of 11 years to list species (Greenwald et al. 2006), frequently after their long-term viability is in doubt (Wilcove et al., 1993, Neel et al. 2012, Rylander 2012). Waiting until a species is on the brink of extinction before taking protective measures creates unnecessary risks to a species and increases the controversies, costs, and restrictions associated with their recovery. Furthermore, funding is inadequate to meet the needs of species that are already listed, are candidates for listing, or have been petitioned for listing (Scott et al. 2010). Strong wildlife provisions under the NFMA could provide an earlier, proactive response to species declines, lessening the trend for more listings under the ESA. Allowing populations to decline towards listing is not good policy ecologically, politically, or economically. It will only reduce management flexibility for states, private citizens, and federal agencies and will further burden managers implementing the already underfunded ESA.

CONCLUSIONS

Given clear guidance in the Directives and sufficient funding, the 2012 planning rule has the potential to be a highly effective framework for wildlife conservation on National Forest System lands. It commits the Forest Service to a formal adaptive management process, adopts a landscape perspective as the primary context for forest planning, strives to find an appropriate balance between coarse- and fine-filter approaches to the assessment of biological diversity, and codifies the need to monitor focal species at multiple spatial scales. These are all significant advances that signal the Forest Service's commitment to a new planning rule that is responsive to the status and trends of ecological systems, as well as the expectations of the nation for the wise stewardship and conservation of public lands and resources.

Although we are confident that the rule can be implemented so as to effectively conserve wildlife populations, we are concerned about the ambiguity of the plan's diversity provisions and the level of discretion permitted when interpreting and implementing the plan's most fundamental actions: identifying focal species, monitoring status and trends, establishing triggers for adaptive management, and taking action to sustain viable populations. Effective implementation of the rule will require a commitment to direct monitoring of focal species, species of conservation concern, and ESA species, as well as a commitment to maintaining their viability. Without this commitment, the provision to sustain biological diversity is incoherent and unlikely to be effective. Triggers will have to be established for monitoring of species to signal when a review of management approaches is necessary. Without an assessment of the effects of management actions via monitoring, the agency cannot fulfill its obligation to manage adaptively. When private land development or other more distant factors affect the viability of species, the USFS should place more, not less, emphasis on providing for viable populations to the extent practicable. The design of monitoring programs, determinations about the inherent capability of the land, and selection of focal and species of conservation concern should be based on the best available scientific information.

The language of the new rule is more discretionary than the 1982 rule, and it removes the requirement to maintain viable populations of all vertebrate species. Although this is unquestionably a significant change in regulatory language, some might argue the 2012 rule merely codifies the way the USFS has managed for diversity since 1982. In practice, management indicator species seldom have been monitored directly in a way that allowed for a clear understanding of their response to management actions, and the USFS has been managing for Regional Forester Sensitive Species by relying primarily on habitat measurements as proxies for the species' current status. In effect, the 2012 rule simply makes it more explicit that this relaxation of the standards established in the 1982 rule will be the USFS's accepted standard for managing for diversity—to focus primarily on coarse-filter approaches, with the expectation that currently abundant species will remain abundant, and that sensitive but stable wildlife populations will, by and large, persist. The problem with this approach is that the NFMA includes clear requirements to provide for a diversity of plant and animal species, not just a range of ecological conditions that may or may not support diversity. In the end, habitat is a meaningless concept if it is never occupied by actual individuals of the species in question.

With the new rule, the USFS faces a new set of decisions that it can address from a position of power, with greater discretion over its approach to wildlife, and forest management in general. It has the opportunity to improve upon past efforts to conserve wildlife and biological diversity, or it could retreat from the responsibilities established in the NFMA and the 1982 rule. At this juncture, the USFS and the broader community of foresters and wildlife managers should pause to consider whether a relaxation of standards—most notably with respect to population viability—and the consequent lessening of agency responsibility and authority is in the best interest of the nation or the agency itself. We respectfully argue that conservation of the nation's biological wealth, including the persistence of viable populations of wildlife species, is an important service that a strong and professional USFS can and should provide to the American public. To the extent that the agency uses its new discretion to lessen its responsibility to wildlife and its exposure to controversy and criticism, the 2012 rule is likely to represent a retreat from an essential public responsibility and a blow to the wildlife profession. But to the extent that the agency signals its leadership on these issues by voluntary committing itself to a nationwide, science-based, and outcome-oriented program of adaptive management of both forest ecosystems and their full complement of species, the 2012 rule will signal a new era of leadership, where increased discretion is used to elevate intent and expectations, accept greater responsibility, and provide energetic leadership in the conservation and management of the nation's public lands and wildlife.

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Review and assessment of LANDFIRE canopy fuel mapping procedures

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Executive Summary

The LANDFIRE procedures for quantifying and mapping canopy fuel characteristics follow generally accepted scientific procedures in the fields of fuel science and remote sensing. Accuracy of LANDFIRE canopy fuel products is low, but consistent with constraints imposed by the very large (national) extent of the effort and the high inherent variability of the characteristics being mapped. Other canopy fuel mapping efforts have achieved greater accuracy than LANDFIRE's products, but at greater cost per acre mapped, and by employing methods that can't be applied at LANDFIRE's extent. The problem of low map accuracy of LANDFIRE canopy fuel products is a greater problem for project-level geospatial fire analyses than for the national-level analyses which LANDFIRE was designed to support. Insufficient accuracy can be resolved by end users through a routine process of critique and calibration (refinement using local information) and refreshing (to account for changes in the landscape since the effective date of LANDFIRE products). Work is now underway to develop a standard procedure for critiquing and calibrating LANDFIRE data layers and to refresh the LANDFIRE data to the present time. These efforts will improve accuracy for both project- and national-level analyses.

Artificial seams in LANDFIRE data products may exist both within and between map zones. The problem of data seams is very difficult to resolve once the data have been published by LANDFIRE, but are unavoidable given the scale and constraints of the project. The utility of LANDFIRE data for national-level analyses is not significantly compromised by these seamlines, but regional- and project-level analyses may suffer from the difficult-to-remove seams.

This report is organized around seven potential shortcomings or problems with canopy fuel related LANDFIRE data products:

- canopy cover values are too high,
- data discontinuities exist within and between map zones,
- canopy bulk density values are too low for use in FARSITE,
- canopy base height is too high to generate crown fire,
- treelist data sources may not be best for canopy fuel calculations
- alternative canopy fuel calculation programs may produce different results
- Refreshing and calibrating LANDFIRE data is needed

Canopy Cover values too high

The canopy cover values used in the LANDFIRE process were obtained from the National Land Cover Dataset (NLCD). The NLCD dataset was produced using a Classification and Regression Tree (CART) analysis relying on a method combining satellite remote sensing and field data. Unfortunately, there are several cover-related quantities measured by ecologists and used by fire modelers; the different quantities are frequently interchanged, erroneously.

As used in fire modeling software and envisioned by fire behavior specialists, canopy cover is the proportion of the forest floor covered by the vertical projection of tree crowns. Some field methods estimate this quantity without bias, but the most common field measurement technique uses a spherical densitometer that actually measures a quantity sometimes called canopy *closure*—the proportion of the sky hemisphere obscured by vegetation when viewed from a single point. Canopy closure is usually a higher value than canopy cover; canopy cover rarely exceeds about 70 percent, whereas canopy closure often approaches 100 percent. Refer to the FireWords glossary of fire science terminology (Scott and Reinhardt 2007) for more details (available at www.fs.fed.us/fmi/downloads/firewords.html). It is not clear if this is the reason for the discrepancy between the NLCD canopy cover values and on-the-ground experience. Nonetheless the canopy cover values produced by NLCD are acknowledged by the LANDFIRE developers to be too high relative to the quantity used by existing fire models.

Canopy Cover is a key LANDFIRE variable because it is used as an independent variable for estimating a wide range of dependent variables like fuel model and canopy bulk density. As directly used in fire modeling programs, canopy cover is used to estimate wind adjustment factor and fine dead fuel moisture. The wind adjustment factor sub-model in fire modeling systems is relatively insensitive to the magnitude of apparent errors in the canopy cover maps. The dead fuel moisture model, however, is more sensitive to errors in canopy cover. In an unpublished analysis, LANDFIRE's Matt Reeves[1] found that correcting the apparent canopy cover error using an alternative approach resulted in a dead fuel moisture decline of roughly 2 percentage points across example landscapes. This change in fuel moisture led to modest changes in potential fire behavior as simulated with FlamMap[2], but a factor-of-two increase in fire growth using FARSITE[3], a significant increase.

[1] Matt Reeves is a GIS Specialist and leads the LANDFIRE Fuels Team, stationed at the Missoula Fire Sciences Laboratory.

[2] FlamMap is software that maps potential fire behavior across a landscape for a single specified weather condition, and has features that allow simple fire growth simulation, identification of fire travel paths, and locating fuel treatments. Available at www.firemodels.org

[3] FARSITE is software that simulates the growth of one fire for one projected weather scenario. Available at www.firemodels.org.

Moreover, canopy cover mapping errors may lead to significant indirect fire modeling effects. Because canopy cover is a keystone variable, these indirect effects are difficult to quantify. If canopy cover is overestimated, LANDFIRE may subsequently map the incorrect fuel model, incorrect CBD, incorrect CBH, etc., all of which can strongly affect fire modeling outputs in a geospatial fire analysis. Using the current LANDFIRE fuel mapping procedure, Tobin Smail[4] believes these indirect effects may be small, because they are so heavily calibrated by end users before publication of the data.

Unfortunately, most of the direct and indirect effects of overestimating canopy cover tend to under-predict fire behavior; the effects are not necessarily compensating. For example, overestimating canopy cover in forested areas can lead to slight underestimation of midflame wind speed, slight-to-moderate overestimation of dead fuel moisture content, choosing a too-benign fuel model (one with little or no live fuel, for example). Together, these factors conspire to underestimate surface fire behavior. Overestimating canopy cover can potentially lead to overestimating canopy bulk density in the LANDFIRE process, which in some cases can partially balance the underestimation.

Because it is used as an independent variable, the importance of an accurate canopy cover layer in the LANDFIRE process should not be underestimated. Matt Reeves reports that a newer type of FIA plot allows independent calculation of canopy cover for FIA plots installed since 2005. This new method appears to agree well with the unbiased (but infrequently used) line-intercept field method of estimating canopy cover, whose values correlate very well with what is expected in the fire behavior models, without manipulation. If enough of such plot data is available, it may be possible for LANDFIRE to generate canopy cover maps using this new approach, with significant improvement in fire modeling. Such improved canopy cover maps may also affect dependent LANDFIRE maps such as CBD.

[4] Tobin Smail is a LANDFIRE fuel specialist based at the Missoula Fire Sciences Lab.

Seamlines within and between map zones

LANDFIRE data is “gapless” because it maps fuel and vegetation characteristics across all ownerships across the U.S. That is a critical feature because important aspects of geospatial fire analysis (fire growth modeling and mapping potential fire behavior and effects) require gapless coverage of not only the analysis area but of a large buffer around the area as well. However, despite using a consistent methodology across the U.S., LANDFIRE data is not “seamless” in the sense that obvious artifacts of the mapping process are evident in surface and canopy fuel layers. Seams in LANDFIRE maps can arise from two sources. First, a seam can exist along map zone boundaries, even if the satellite imagery were the same in both map zones, because different protocols and different fuel and fire experts can be used in each map zone. Second, a seam may exist *within* a map zone due to the developers’ need to stitch satellite scenes into a composite image for a whole map zone. This procedure is similar in nature to stitching together digital photos to make a panorama—if the exposure is not the same for each photo, then the boundary between photos becomes obvious in the final panorama. In the LANDFIRE process, if those separate satellite images are of similar quality (captured during times of similar atmospheric conditions, for example) then the compositing process works well and a seam may not be created. However, the separate images may differ in many respects (primarily atmospheric conditions) such that the information contained in one image may differ from another image for the same pixel. The CART analysis assumes that all variation in the images is due to on-the-ground differences, not atmospheric differences unrelated to actual differences on the ground. When used in subsequent CART analyses, the boundaries where the two images were merged can become a noticeable data seam where the map indicates a strong change in value that is not actually present on the landscape. This is a difficult problem to reconcile; there is no easy way to remove such a seam—it’s in the base imagery that the data layers are built upon, and it runs along an artificial (satellite image) boundary. Such data seams can also exist in the inherited NLCD canopy cover data used in the LANDFIRE fuel mapping process, but those seams are generally “hidden” along natural terrain features such as rivers and major ridgelines where changes in vegetation structure are not uncommon. Despite being hidden, such seams can produce disconcerting data discontinuities in the final map.

For example, Charley Martin[5] provided this LANDFIRE CBD data for southern Oregon, which shows the distribution of CBD values in a small watershed that crosses a map zone boundary (figure 1).

[5] Charley Martin is a Fire Ecologist with the Bureau of Land Management’s Medford District, Oregon. Charley has been closely involved in LANDFIRE’s calibration workshops and participated in a separate project to assess accuracy of LANDFIRE fuel maps.

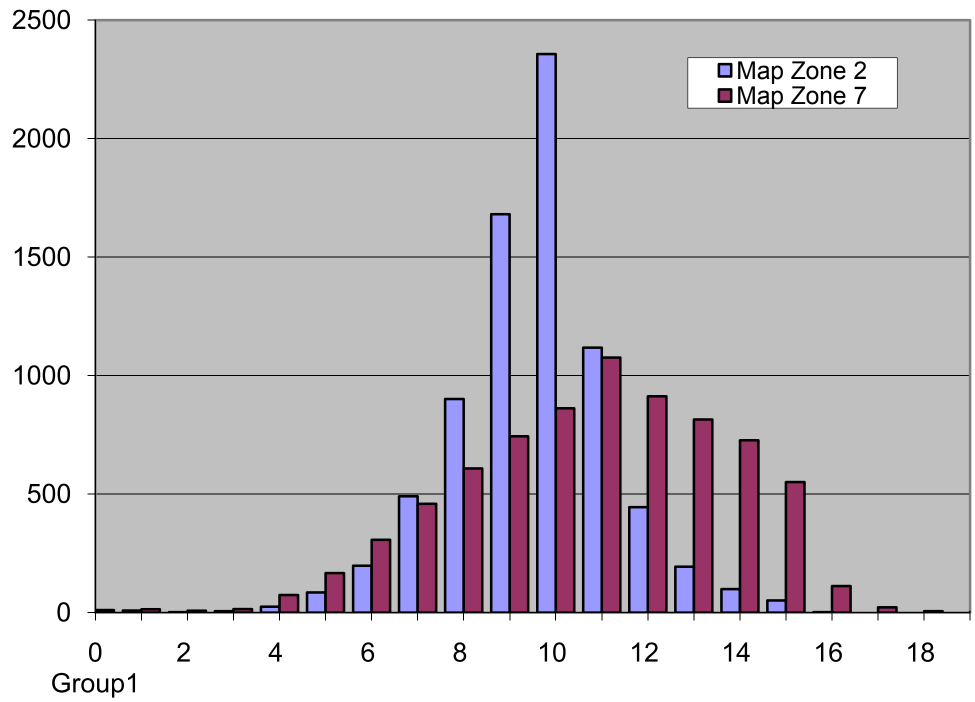

Figure 1 -- Distribution of CBD values (kg/m3 * 100) for a southern Oregon watershed that crosses two map zones .

The expectation, based on field experience in the watershed, is that the distributions should have the same shape. Information such as this can help in a calibration exercise designed to force the map zones into similar distibutions, but there is no way to know which distibution is “correct”. The following map (figure 2) shows the nature of the data discontinuity on the CBD map. Similar data seams are evident in nearly all LANDFIRE maps for this watershed.

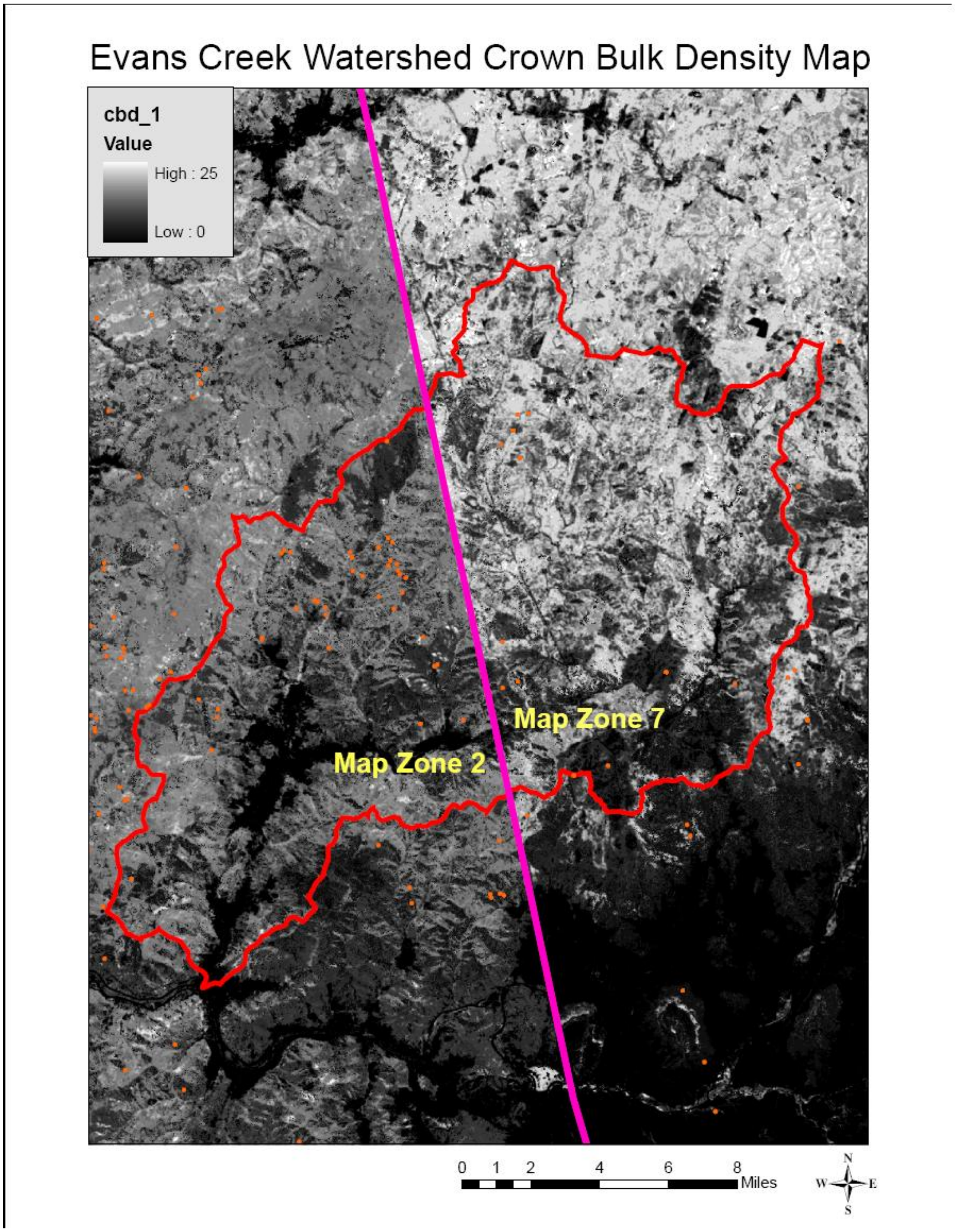

Figure 2 -- LANDFIRE map of CBD in a watershed crossing the map zone 2/7 boundary. CBD is shown as higher in the Map Zone 7 portion of the watershed; on-the-ground experience does not support that result.

Rick Stratton[6] of Systems for Environmental Management[7] is currently working on a procedure for calibrating and updating LANDFIRE data for use in some fire modeling

[6] Rick Stratton is a Fire Modeling Specialist with Systems for Environmental Management, based at the Missoula Fire Sciences Lab.

systems. Fire Program Analysis (FPA) and the USDI National Park Service are co-funding that work. The current in-preparation version of Stratton's work does not yet suggest a method for mitigating seams. Charley Martin, with the BLM in Oregon, is trying an approach that smoothes the data on both sides of the seam to reduce its effect. Such an approach is visually appealing on a map, but does not effectively deal with the problem.

Alternative approaches to using remote sensing imagery in creating the LANDFIRE data layers may reduce the intensity or extent of seams. An alternative approach, which avoids seamlines by conducting the mapping and analysis one strip (satellite image) at a time, rather than one map zone at a time, is being used to generate LANDFIRE maps in Alaska. This approach requires that field data be well-distributed across the area, because sufficient field data must exist within each image, not just the map zone. Assuming such data exist, this approach may work well to avoid seamlines and improve accuracy.

Seamlines in LANDFIRE data primarily affect project-level analyses, but regional- and national-level analyses may also be affected. Even a national analysis like FPA is broken into smaller units (FPUs and FMUs) for analysis and comparison. If the analysis unit is small compared to the map zone or satellite imagery, then the potential exists for the data discontinuities to affect results. The larger the analysis area, the smaller the effect seamlines will have on the results.

[7] Systems for Environmental Management (SEM) is a private, nonprofit research and education foundation based in Missoula Montana. In conjunction with federal partners, SEM has developed a host of fuel, weather and fire behavior modeling software and procedures, which are available at www.fire.org.

CBD too low for crown fire in the FARSITE family

Users of the Mark Finney's[8] family of geospatial fire analysis programs (FARSITE, FlamMap, FSPro[9], FSIM[10]) have long noted that values of canopy bulk density (CBD) produced by treelist methods are too low to generate the expected amount of crown fire in their simulations. LANDFIRE has used a prototype of FuelCalc, which applies a treelist method for estimating CBD, to generate its CBD map, so the complaint has been extended to LANDFIRE CBD maps. A general rule-of-thumb was developed to cope with this apparent disconnect: double the LANDFIRE treelist-generated values for use in Finney's geospatial programs to achieve the expected results.

Early CBD mapping procedures (Selway-Bitterroot, Gila wilderness areas) were developed before any plot-level methods of estimating CBD had been developed. Instead of relying on observed CBD at plots, the early efforts instead populated CBD by working backward from expected fire behavior to determine the CBD values that produce that behavior using a given fire model. Given the lack of plot-level CBD observations available at the time, the approach was reasonable. Nonetheless, that procedure produces a value that is good only for the particular fire model used. In this case, the fire model used, FARISTE, produces fire behavior quite different than all others developed since then (except FARSITE's geospatial relatives).

Since those initial mapping efforts, our ability to estimate CBD has improved considerably, and is codified in FuelCalc[11], Fuels Management Analyst Plus (FMAplus[12]), and the Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS), all of which use a treelist approach. Such methods are based on decades of biomass research. The CBD algorithm in FuelCalc was conservatively designed to over-estimate rather than under-estimate CBD (by using the highest CBD found in any 11-ft layer of a canopy as the value for the whole plot, which is commonly more than twice the average bulk density). Comparison of predicted CBD with meticulously observed CBD (Scott and Reinhardt 2005) has generally verified the utility of the approach for estimating CBD in various stand structures. The values the treelist method produces fall squarely in the

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[8] Mark Finney is a research forester at the Missoula Fire Sciences Lab. Mark is the developer of a suite of geospatial fire modeling software tools, including FARSITE, FlamMap, FSPro, and FSim.
[9] FSPro is online software that simulates the likelihood of fire spread across a landscape by simulating fire growth under a large sample of possible future weather scenarios. FSPro is an integral component of the Wildland Fire Decision Support System (WFDSS).
[10] FSIM is prototype software that simulates the likelihood of fire growth and behavior across a landscape for a sample of possible weather conditions and for a sample of possible escaped-fire frequencies and locations. FSIM simulations are being considered for use in FPA, and are also used in prototype quantitative wildland fire hazard and risk assessments.
[11] A prototype version of FuelCalc designed by Elizabeth Reinhardt at the Missoula Fire Sciences Lab was used by the LANDFIRE fuel staff. A more complete version of FuelCalc is currently under development.
[12] FMAplus is commercially available software produced by Don Carlton of Fire Program Solutions LLC, available at www.fireps.com.

range of values that Agee's (1996) analysis found would lead to crown fire. The treelist methods generate CBD values work well in all fire modeling software programs except Finney's geospatial family, including BehavePlus[13], FMAplus and NEXUS[14]. FlamMap and FARSITE offer users a choice of crown fire modeling methods to use: the original "Finney (1998)" method, and a method similar to that used in NEXUS, which is labeled "Scott and Reinhardt (2001)" in those programs (figure 3).

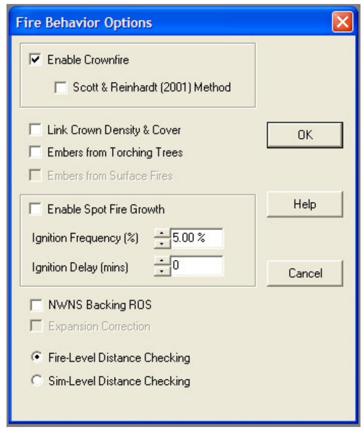

Figure 3--The *Model | Fire Behavior Options* dialog box in FARSITE, showing the checkbox that allows calculation of crown fire similar to the method described in Scott and Reinhardt (2001).

Users have generally found that using LANDFIRE or other treelist-generated CBD data with the crown fire option set to "Scott and Reinhardt" produces very reasonable results for crown fire occurrence, but not when using the "Finney 1998" default setting.

Scott (2006) suggests that the significant difference in fire model outputs (fire type, crowning index, etc.) between Finney's geospatial fire models and the others can be attributed to an error in modeling logic made initially in the Canadian Forest Fire Behavior System and subsequently used in Finney's programs. The error in modeling logic had little practical effect as implemented in the Canadian prediction system, so it went unnoticed; the same logic error when implemented in the U. S. system, however, has led to great differences in predicted fire behavior. See Scott (2006) for a detailed discussion of this topic.

The problem that LANDFIRE-generated CBD may be too low for use in Finney's geospatial fire models is best addressed by the fuel and fire modeling community, not by LANDFIRE. For users who wish to use those programs in their default setting (or those using FSPRO and FSIM, which do not yet have an option to use the Scott and Reinhardt 2001 method), the current rule of thumb may be appropriate. Otherwise,

[13] BehavePlus is software that allows simulation of fire behavior and effects for a specific point in space and time. Available at www.firemodels.org

[14] NEXUS is software that allows simulation of crown fire potential for a specific point in space and time. Available at www.fire.org

many users report that using the “Scott and Reinhardt” switch with LANDFIRE CBD maps produces acceptable results.

CBH too high for crown fire

At first glance this issue appears similar to the above issue with CBD, but in reality it is much more difficult to address. Unlike CBD, CBH is difficult to define in such a way that it can be measured in the field or estimated from a treelist. Moreover, CBH is not strongly correlated with other stand characteristics, making it difficult to produce reliable maps using the LANDFIRE approach (or any mapping approach, for that matter). For example, within any given forest type, CBH can be low in areas with low canopy cover, because there may have been little self-pruning in such a low-density stand, or CBH can be high if the stand has low cover because it was thinned. The LANDFIRE procedure can only broadly distinguish those cases.

Such difficulties led to the development of an alternative method of estimating CBH based on expert opinion. (Note that this is a similar approach taken for the Selway-Bitterroot and Gila mapping projects when faced with a lack of available CBD data.) The fuel and fire behavior experts did not offer their opinion of CBH directly, but instead were asked to identify the weather conditions that typically lead to torching (because CBH is used to predict when torching will occur). From that information, along with the fuel model and canopy cover already assigned, the CBH that leads to torching is then identified by working backward through the crown fire initiation model. This expert opinion CBH therefore depends on the fuel model and canopy cover for the area, as well as the weather conditions identified by the experts. Any errors in mapping of those layers, and any changes or adjustments made by users to those layers invalidate this CBH estimate—transition to torching would no longer take place at the identified threshold.

Fortunately, unlike with CBD, all point-based and geospatial fire models, regardless of developer, use CBH in the same way, so estimates of CBH made this way are valid in all U.S. fire modeling programs.

The difficulties with estimating CBH to simulate transition to crown fire cannot be resolved by LANDFIRE. The fire modeling community may need to find a different approach that is more amenable to mapping and less dependent on surface fire behavior (see Cruz and others 2004).

In most fire modeling systems, especially in Finney's geospatial models, the downside of conservatively estimating a low CBH is small compared to the downside of estimating a CBH that is too high. Until fire modeling uses a different approach, a stop-gap measure that LANDFIRE could employ is to modify the FuelCalc procedure for estimating CBH to identify the height of the lowest biomass of any density. Responsibility for this task lies not with LANDFIRE but with the fuel and fire behavior modeling community.

Treelist data sources

LANDFIRE has gathered treelist data from a variety of sources that use a variety of inventory methods. Two tree inventory methods are generally used: fixed-area plots and variable-radius plots (some tree inventories combine both methods). The treelist-based calculation methods used by LANDFIRE in FuelCalc are designed to be used with fixed-area plots of approximately 0.1 ac in size. The developers of that method felt that variable-radius plot may not adequately represent stand structure of a plot because it emphasizes sampling of large trees at the expense of small trees. For canopy fuel estimation, the contribution of a large number of small trees can be much more important than a small number of large trees, so it is important to have as much information as possible for those trees. Moreover, the trees sampled at a variable radius plot can be very far apart from each other, so their individual crown characteristics may not necessarily reflect growing conditions near the plot center.

Nonetheless, a large amount of treelist data available to LANDFIRE is of the variable-radius or hybrid plot type. The magnitude of potential problems with using variable-radius plots is unknown. In theory, the CBD predicted for a variable radius plot is probably slightly lower than if a fixed-area plot had been established at the same location, but this is impossible to know without research comparing the two approaches at the same plot.

For this report, a comparison of fixed-radius and variable-radius plot types was conducted using a dataset for a single even-aged ponderosa pine/Douglas-fir stand in western Montana. The dataset consisted of a complete list of tree attributes, including (X,Y) coordinates, of every tree on a square, 100 x 100 m (1-ha) plot. (The plot was established in 2006 by Elizabeth Reinhardt[15] to eventually test the use of upward-looking LIDAR for estimating canopy fuel characteristics.) From this complete dataset we established four virtual sample points within the megaplot, each located 25 meters from the edge. At each of these sample points we identified which trees would be counted in fixed- and variable-radius plots of different sizes. We then computed the average canopy fuel characteristic across the four sample points for each plot size. The results are summarized below. The results for a one-tenth-acre fixed-radius plot are shown in bold for emphasis. Plot sizes are listed in descending order of "size"; plots at the top sample a larger number of trees than plots at the bottom.

[15] Elizabeth Reinhardt is a research Forester at the Missoula Fire Sciences Lab. Elizabeth has led or participated in the development of several fuel, fire behavior and fire effects modeling systems, including FOFEM, FFE-FVS, NEXUS, and FuelCalc.

Table 1 -- Mean canopy characteristics (n = 4) for various plot types and plot sizes. The highlighted row indicates the plot type and size recommended by the developers of FuelCalc.

Fixed-radius Plots							
Plot Id	CBD (kg/m3)	CFL (t/ac)	CBH (ft)	SH (ft)	CC (percent)	Basal Area (ft2/ac)	Trees Per Acre
0.50 ac	0.054	3.0	22	86	38	102	278
0.25 ac	0.058	2.9	23	85	38	110	239
0.20 ac	0.058	2.8	23	85	38	105	253
0.10 ac	**0.065**	**3.1**	**22**	**83**	**39**	**109**	**298**
0.05 ac	0.086	4.0	29	85	46	143	310
0.02 ac	0.099	4.2	42	85	53	173	183
0.01 ac	0.148	6.2	38	72	58	239	233
Variable-radius Plots							
	CBD (kg/m3)	CFL (t/ac)	CBH (ft)	SH (ft)	CC (percent)	Basal Area (ft2/ac)	Trees Per Acre
BAF10	0.065	2.9	39	87	37	115	110
BAF20	0.088	3.7	38	84	45	135	151
BAF30	0.104	4.3	37	85	49	157	150
BAF40	0.110	5.0	37	86	56	190	207
BAF50	0.109	4.8	38	85	51	175	170
BAF60	0.130	5.7	38	86	57	210	204

Plot size appears to matter significantly for the fixed-radius plots—CBD ranged from 0.054 kg/m3 for the half-acre plots to 0.148 kg/m3 for the hundredth-acre plots, a factor of three difference (for the very same plot centers). These averages mask the increasing variability as plot size decreased—CBD at the four half-acre plots ranged from 0.046 to 0.065 kg/m3, whereas the hundredth-acre plots ranged from 0.000 to 0.366 kg/m3. This situation resulted in increasing CBD values with decreasing plot size, but that is unlikely to be a universal truth. In fact, one of the four plots was located such that many trees were found on the hundredth-acre plot, whereas the others had few or none.

The variable radius plots did not tend to underestimate CBD compared to the fixed-radius plot, an unexpected result. In fact, the BAF10 (variable-radius plot with 10-factor prism), a common BAF used in vegetation sampling, produced an estimate of CBD similar to the tenth-acre plot. In fact, larger BAFs, which sample fewer trees but puts more weight on each, tended to increase the estimated CBD. While very encouraging, this result applies to this one even-aged stand; a similar result may not be found for more complex fuel structures.

Canopy fuel load, canopy cover, and stand height estimated from variable-radius plots was also similar to the fixed-radius plots.

Canopy base height estimates differed significantly between the fixed-radius plots and the variable-radius plots, which predicted much greater CBH. This is likely due to the fact that the variable-radius plots do not adequately sample small trees, so tend to under-predict biomass in the lower part of the canopy. This effect would be even greater in more complex stand structures than present in this analysis. A hybrid plot with both variable- and fixed-radius plot elements could mitigate this effect.

Finally, the variable-radius plots underestimated tree stem density relative to the fixed-radius plots, again due to the under-sampling of small trees.

In summary, this analysis supports the conclusion that variable-radius plots under-sample small trees. That is, in fact, the purpose of that plot design. For this even-aged stand, the under-sampling of small trees led to underestimation of CBH, but not CBD, CFL, or SH. Only a more exhaustive analysis with other stand structures will confirm or refute this result.

Canopy fuel calculation programs

LANDFIRE used a customized-prototype version of FuelCalc coded by Larry Gangi[16] to estimate CBD and CBH. Other fuel analysts have used the Fire and Fuels Extension to FVS or Fuels Management Analyst Plus for making the same estimates. The canopy fuel calculations in FuelCalc and FFE-FVS[17] were designed by Elizabeth Reinhardt, and FMAplus was also patterned after those programs. All three programs use the same general approach to estimating CBH and CBD, but there are slight differences in the equations used in each tool, and slight differences in certain parameters and internal models. For example, the user has control over whether any of the biomass of broadleaf tree species is factored into the CBD and CBH estimates. In theory, differences in output generated from these three programs should be small.

As a quick test of this assumption, Charley Martin's dataset of 700 FIREMON[18] plots was run through both FuelCalc and FMAplus. The resulting differences between the programs were larger than expected—FMAplus consistently over-predicted relative to FuelCalc (Figure 4).

[16] Larry Gangi is a computer programmer with Systems for Environmental Management. Larry has also served as software developer for the FOFEM and FireMon software tools.
[17] FFE-FVS is software to simulate vegetation growth and quantify fuel characteristics over time. Available at www.fs.fed.us/fmsc/fvs/description/ffe-fvs.shtml.
[18] FIREMON is software to catalog and monitor fuel and vegetation characteristics. Available at www.fire.org

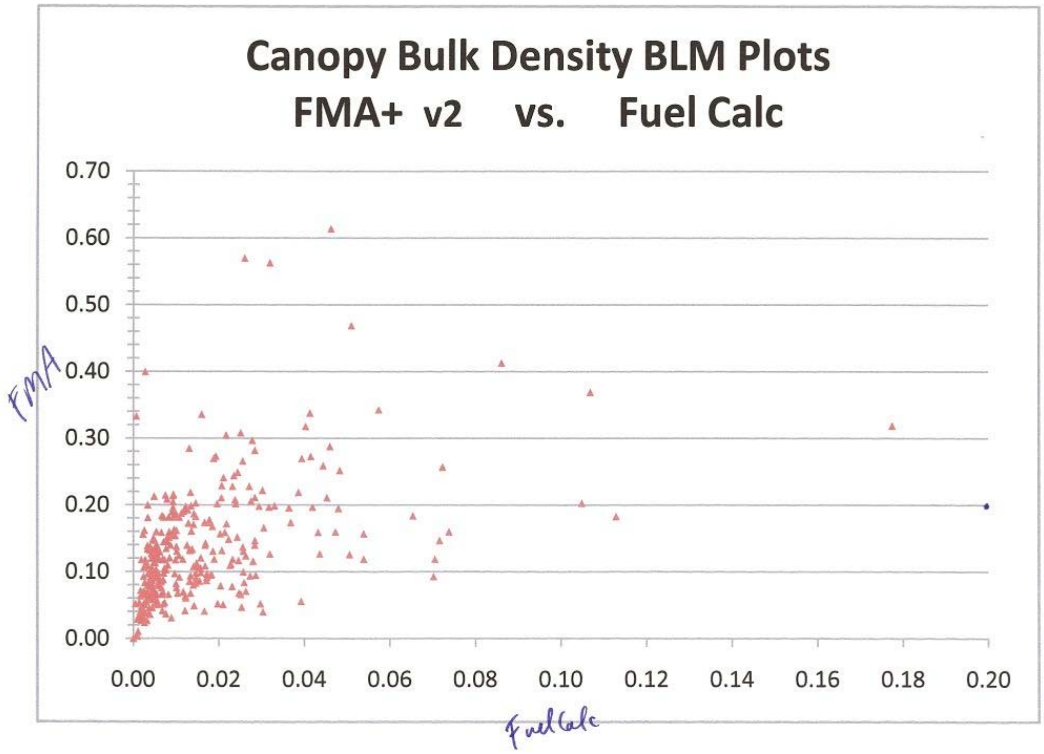

Figure 4 -- Predicted CBD (kg/m3) for FMAplus (Y-axis) and FuelCalc (X-axis). FMAplus overpredicts relative to FuelCalc, but it is not possible to know which is closest to "observed".

It is impossible at this point to know which is more accurate or reliable, but the FMAplus CBD values did not seem unreasonably high. (I don't have enough experience with the vegetation and fire behavior in the study area to confirm that conclusion, though.) It is possible that FMAplus is over-emphasizing the contribution of broadleaf species to canopy bulk density, or that FuelCalc is under-emphasizing those species.

I have forwarded this finding to Elizabeth Reinhardt, lead developer of FuelCalc, for further investigation. At this point I surmise that the FuelCalc-FMAplus comparisons were not apples-to-apples; user settings controlling different aspects of the calculation may not have been equal. FuelCalc remains the standard government application for quantifying canopy bulk density; LANDFIRE can rely on its output in mapping efforts.

The fuel and fire behavior modeling community should investigate this issue by thoroughly analyzing the outputs from a common set of treelist inputs for a variety of calculation tools. Any differences in output should be explained, and recommendations for resolving differences among the various programs should be provided.

Refreshing and calibrating LANDFIRE data

Two important limitations result from LANDFIRE's national extent: early date of validity (*ca*. 1999), and poor project-level accuracy for some fire planning applications. Refreshing data to the current year is a critical task before applying LANDFIRE's spatial data for any analysis, whether national-, regional- or project-level in extent. Improving local-level accuracy is important for project-level planning, but is not required (and may in fact hinder) regional- and national-level analyses by mixing adjusted and unadjusted data. LANDFIRE and others are addressing these issues by publishing procedures for calibrating and adjusting LANDFIRE data using a variety of *ad hoc* software tools.

To address the first limitation, LANDFIRE has developed a data-refresh plan to reflect landscape changes due to fire biennially. To jump-start the process, NIFTT[19] has conducted and nearly completed a Rapid Refresh of LANDFIRE data—a first-cut refreshing of LANDFIRE data to reflect landscape changes between 1999 and 2007. The products of this effort are expected to be replaced by a more thorough refreshing on a two-year cycle. In addition, the entire LANDFIRE mapping process will be repeated on a 10-year cycle. This procedure should ensure that high quality, up-to-date landscape data is always available. See the LANDFIRE Operations and Maintenance Business Case and Plan at http://www.landfire.gov/documents_updatedprod.php for more information.

Two separate efforts are underway to address the adjustment of LANDFIRE data to meet the needs of project-level analysis. One effort, co-funded by FPA and the National Park Service, is being carried out by Rick Stratton of Systems for Environmental Management. The product of that effort will be a document describing a process for critiquing and adjusting LANDFIRE data. A draft of this document will be available soon.

Second, NIFTT is continuing development and training of software tools and developing a training package designed to help users to download and prepare LANDFIRE. Two tutorials are available, and a course is being developed.

The DataPrep tutorial shows users how to prepare LANDFIRE data for use in NIFTT tools. This tutorial does not address adjustment or calibration of spatial data; it simply instructs users on how to download, clip, and re-project LANDFIRE data for use in a project-level analysis.

The LANDFIRE Data Access Tool tutorial describes the use of this tool for obtaining LANDFIRE data. This tutorial also does not address calibration and adjustment of the data itself.

[19] NIFTT is the National Interagency Fuel Technology Transfer team, co-funded by LANDFIRE and the National Interagency Fuel Coordination Group.

Finally, a course titled “GIS Tools for Wildland Fire and Fuels Planning” is under development. The course will teach students to download and edit LANDFIRE data for use in NIFTT’s GIS tools.

The combination of the NIFTT courses and tutorials and Stratton’s NPS/FPA-funded process for critiquing and editing LANDFIRE data should be enough guidance for most users.

Discussion

At times during their development process, LANDFIRE faced the choice of producing data that was consistent with biological science (for example, producing CBD values based on methods derived from the biomass literature) or producing data specifically adjusted so that could be consumed by a fire behavior modeling tool (CBD values manipulated so they work better in FARSITE). The LANDFIRE philosophy for the current effort was to base all data maps on the best available biological science, knowing that adjustment would be required for certain models. This is the only scientifically supportable approach. Should LANDFIRE's best biological estimate of a certain quantity end up not working well in a fire model, a quick investigation would indicate whether the problem was with the data, with the model, or with the fire modeling science. LANDFIRE should take steps to adjust any data layers it produces that are not consistent with scientifically valid field data, as they did for canopy cover values. In other cases, the fuel and fire modeling community may need to make accommodations in their fire models for the biologically estimated data.

Although users may need to critique and calibrate LANDFIRE data for use in project-level analysis, the goal of producing a nationally-consistent dataset is met without such effort. The scope of a critique and editing effort should be tied to the extent of analysis to be conducted. A national-level analysis would require a nationally consistent critique and calibration effort – LANDFIRE has already accomplished this task. A mid-scale analysis (state or region, for example) should have a critique and calibrate effort at the same scale, or none at all – mixing base LANDFIRE data for some areas with critiqued and calibrated data for others may lead to spurious results.

LANDFIRE's success at producing biologically based fuel and vegetation maps has created a situation where fire modeling difficulties can be addressed by the fire modeling community. Before LANDFIRE, without consistently created maps, fire modeling errors were always attributed to problems with the data, with no consideration for problems with the model. Geospatial fire modeling systems have been developed with a very rigid fire behavior model—no way to accommodate model error. (FARSITE has model-side spread rate adjustment factors, but other geospatial fire modeling tools do not). As a result, calibration of the fire model has always focused on changing the underlying data. When based on reliable fuel maps and weather data, many FARSITE simulations under-predict fire growth and behavior. The approach to improve simulation accuracy has been to adjust the data: reduce canopy base height, increase canopy bulk density, increase wind speed, etc. Unless there is specific evidence of a data accuracy problem, adjusting the data to suit the model is not the best approach to calibration. Instead, the fire modeling community should focus on adjusting parameters in the fire model itself. Few such adjustment factors currently exist in geospatial fire models, especially the emerging FSPro and FSIM.

Conclusion

LANDFIRE has done an admirable job integrating emerging fuel and fire modeling technologies into their mapping efforts. Given the large extent of the project, high inherent spatial variability of the characteristics being mapped, emerging (and sometimes contradictory) nature of the fuel and fire modeling technologies involved, and time constraints, no better map products could have been produced. More accurate, seamless maps can be produced at greater cost and smaller scale than required by LANDFIRE's mission. Most remaining problems with LANDFIRE data for local-level projects can be addressed through a process of calibration and adjustment. Both FPA and LANDFIRE are funding the development of procedures for accomplishing that task.

Two significant problems can potentially be addressed by LANDFIRE. First, LANDFIRE can explore whether the new canopy cover estimation techniques developed for recently placed FIA plots can be used to generate a LANDFIRE-produced forest canopy cover map to replace the inherited NLCD maps. The adjustment of this map will significantly improve fire modeling by facilitating better estimates of wind adjustment and dead fuel moisture, both of which depend on forest canopy cover. Second, in an effort to reduce data discontinuities caused by seamlines, LANDFIRE can consider strip-based mapping for any future efforts (as opposed to the present zone-base). LANDFIRE mapping for Alaska is already planning to use strip-based approach, and will serve as a good test of that approach.

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Review

Management for Mountain Pine Beetle Outbreak Suppression: Does Relevant Science Support Current Policy?

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Abstract: While the use of timber harvests is generally accepted as an effective approach to controlling bark beetles during outbreaks, in reality there has been a dearth of monitoring to assess outcomes, and failures are often not reported. Additionally, few studies have focused on how these treatments affect forest structure and function over the long term, or our forests' ability to adapt to climate change. Despite this, there is a widespread belief in the policy arena that timber harvesting is an effective and necessary tool to address beetle infestations. That belief has led to numerous proposals for, and enactment of, significant changes in federal environmental laws to encourage more timber harvests for beetle control. In this review, we use mountain pine beetle as an exemplar to critically evaluate the state of science behind the use of timber harvest treatments for bark beetle suppression during outbreaks. It is our hope that this review will stimulate research to fill important gaps and to help guide the development of policy and management firmly based in science, and thus, more likely to aid in forest conservation, reduce financial waste, and bolster public trust in public agency decision-making and practice.

Keywords: bark beetle; clearcut; climate change; climate change adaptation; daylighting; *Dendroctonus ponderosae*; forest pest management; monitoring; sanitation; thinning

1. Introduction

Insect outbreaks are increasing in size and severity on a global scale [1]. In North America alone, three massive insect outbreaks occurred within the last two decades, all involving native bark beetles in conifers [2]. Of these, the mountain pine beetle (*Dendroctonus ponderosae*) outbreak is an order of magnitude larger than any previously recorded. A variety of factors, natural and anthropogenic, converged to result in these dramatic events [2]. Each outbreak has not only had severe ecological effects, but each has also triggered human responses that, for better or for worse, have resulted in additional impacts along with massive expense [3]. Predictions are that outbreaks of bark beetles will become more frequent and severe in the future [4,5] indicating an imperative need to critically assess the efficacy and impacts of our approaches to their management.

Outbreaks of bark beetles are not new. They have been occurring for millennia and have played a major role in shaping coniferous forest ecosystems of the world. While considerable research has been conducted on controlling bark beetles, massive gaps in knowledge remain. In particular, there is a disturbing dearth of rigorous replicated empirical studies assessing the effects of various management strategies, particularly timber harvest treatments, for bark beetle outbreak suppression. Even fewer studies have focused on how such treatments meet explicit goals or affect forest structure, function and future outbreak dynamics [6]. Particularly pertinent at this time, there is a lack of information to address forest adaptation to climate change in light of increasingly “out of historic norm” behavior of bark beetles. Despite this, there is a widespread belief in the policy arena that timber harvesting is an effective and necessary tool to address beetle infestations. That belief has led to proposals for, and enactment of, significant changes in federal environmental laws to encourage more timber harvests. Our question is, does that belief have a sound grounding in current science?

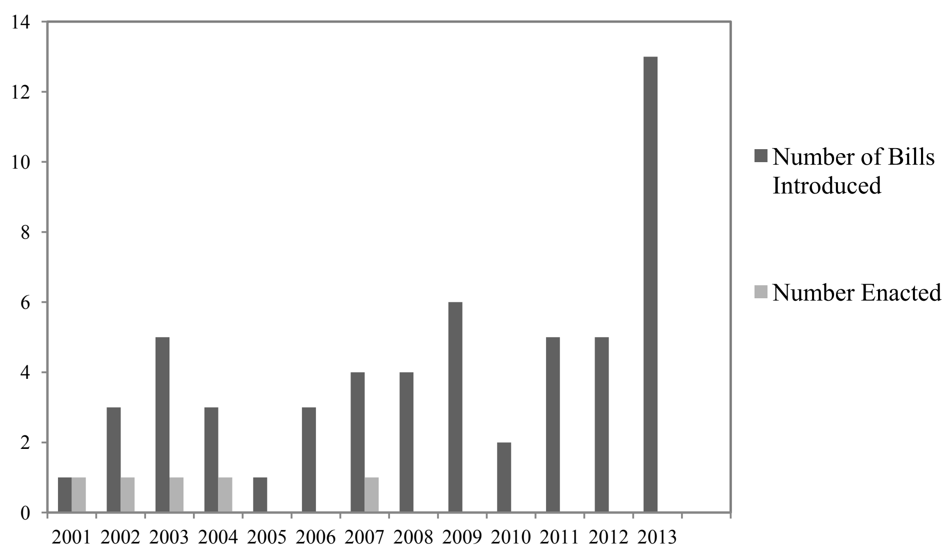
In this review, we focus on mountain pine beetle as an exemplar to critically evaluate the state of science behind the use of timber harvest treatments for bark beetle suppression during outbreaks. The mountain pine beetle was chosen because it is the most studied, most intensively managed, and most aggressive of the irruptive bark beetles. It has also responded strongly to climate change, resulting in a recent massive outbreak of unprecedented size that, in turn, has initiated numerous human responses, mostly involving implementation of timber harvests. It has also initiated many policy changes with many more currently in the pipeline.

We begin with an overview of the current policy situation. We then briefly review the biology of mountain pine beetle to form a foundation for understanding the factors that initiate and maintain outbreaks and how anthropogenic factors are contributing to current problems. We then describe the primary timber harvest treatments used to suppress bark beetle outbreaks and examine how well relevant science and ecological principles support their use. We conclude with a discussion on how well policy reflects the actual state of current science and identify where significant gaps between science and practice occur particularly in light of climate change. We also discuss the need to use advanced tools, including genetics and remote sensing, to adapt old practices to new situations-particularly in the realm of climate change adaptation. It is our hope that this review will stimulate research to fill important gaps and to help guide the development of policy and management firmly based in science, and thus, more likely to aid in forest conservation, reduce financial waste, and bolster public trust in public agency decision-making and practice.

2. The Current Policy Situation

There have been many recent proposals to streamline, reduce, or eliminate perceived legal obstacles to implementing timber harvests to address beetle epidemics on federal public lands (Figure 1). Between the 107th Congress (January 2001) and the 113th Congress (present), we found 55 bills that were introduced where at least one goal of the legislation was to increase timber harvests in order to respond to beetle infestations (Figure 1). Most of these proposals focused on the US Forest Service, which manages the majority of forests on federal public lands.

Figure 1. Number of bills involving timber sales that included bark beetle control that were introduced and/or enacted from 2001 to 10 July 2013.



Some of these proposals have been enacted. By far, the most important legal change has been the Healthy Forest Restoration Act of 2003 (HFRA). HFRA reduced the level of environmental analysis required for certain timber projects under the National Environmental Policy Act (NEPA), specifically by limiting the number of alternatives that the Forest Service was required to analyze. It also significantly restricted the ability of members of the public to challenge certain timber projects in court (by making participation in the agency's administrative process a precondition for filing suit). Further, it sought to streamline the Forest Service's internal administrative process for considering citizen challenges to certain timber projects. HFRA applies nationally to all National Forest System and Bureau of Land Management lands, and has resulted in forest treatment projects on an average of 220,000 acres of federal land per year since its enactment [7]

HFRA authorizes this streamlined process for timber projects on "Federal land on which…the existence of an epidemic of disease or insects, or the presence of such an epidemic on immediately adjacent land and the imminent risk it will spread, poses a significant threat to an ecosystem component, or forest or rangeland resource, on the Federal land or adjacent non-Federal land" [8,9].Moreover, while other types of HFRA projects in old growth forests are subject to limitations intended to protect old growth structure and large trees, timber projects to address insect epidemics can occur in old growth forests without those limitations [10,11].

HFRA also sets up a special experimental management process to develop better management methods for beetle infestations. After a long list of findings by Congress about the risks of beetle infestations in US forests, Congress authorized up to 250,000 acres of "applied silvicultural assessment and research treatments" on National Forests that would be categorically excluded from NEPA; these treatments could include timber harvesting [12,13]. HFRA section 401(b)(3) [14] requires that these applied silvicultural assessments and treatments must be peer reviewed by non-agency scientists.

HFRA is not alone. Another enacted bill created exemptions from environmental laws to allow timber harvest projects in a geographically limited area. As part of a massive supplemental appropriations act to address recovery from the September 11, 2001 terrorist attacks, Congress exempted a series of timber harvest projects in the Black Hills of South Dakota from any and all environmental laws; the law specifically stated that the projects were intended to reduce both fire risk and beetle infestations [15].

Other recent enactments create additional incentives for timber harvests intended to address beetle infestations. Congress permitted state forestry agencies to perform beetle control timber harvest projects on federal lands in Colorado and Utah under what is called "Good Neighbor Authority" [16]. These state forestry agencies must also implement "similar and complementary" services on state land adjacent to federal land in order to use the authority. Additionally, in the 2008 Farm Bill, Congress expanded subsidies for the production of "renewable biomass" energy to include timber produced from projects intended to reduce or contain disease or insect infestation [17].

There have been many more recent proposals for additional changes. Congress has considered multiple bills to expand the scope of HFRA. One proposal would require the Forest Service to implement at least one insect and disease control pilot project in at least one subwatershed in every national forest in a state that is "subject" to an insect or disease epidemic [18–24].Congress has also considered many other changes to encourage timber harvesting to control beetle infestations besides expanding HFRA. Some proposals would expand the exemptions to the Forest Service's Roadless Rule (which prohibits commercial timber projects and road construction in unroaded areas of National Forests) in order to allow more timber projects that are intended to address beetle infestations; some of these projects would be exempt from judicial review [25–27].

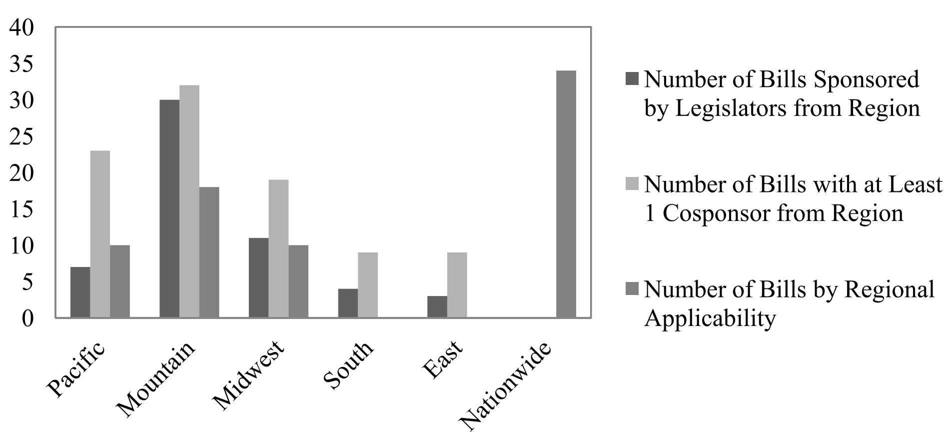
Congress has considered giving additional benefits under the Clean Air Act for "renewable biomass" produced from timber projects on federal lands, including projects intended to control beetle infestations [28,29], giving grants and other subsidies for beetle control timber projects [30], extending the Good Neighbor Authority to more states [31–33], and reducing or eliminating the fee that private timber contractors pay for timber contracts in exchange for agreements to implement restoration work, such as culvert removals, road improvements, or invasive weed removal, if the project provides insect control and other forest management benefits [26]. Finally, two bills have proposed that designation of additional federal lands as protected wilderness be paired with exemptions of beetle-related timber projects from environmental laws [34,35].

Throughout this policy debate, members of Congress and major stakeholders have regularly stated that timber harvest on federal lands is a necessary component of efforts to fight beetle infestations and control outbreaks and that additional flexibility under environmental laws is necessary for agencies to pursue these timber harvest projects [36–41].

Likewise, the U.S. Forest Service and other U.S. federal land management agencies have prescribed timber harvests as a necessary component of beetle control. For example, the Forest Service's Western Bark Beetle Strategy calls for the agency to "reduce the number of trees per acre and create more diverse stand structures to minimize extensive epidemic bark beetle areas" by using thinning and other harvest treatments [42]. While the Forest Service has applauded HFRA as "very helpful" in addressing beetle outbreaks (U.S. Forest Service, Review of the Forest Service Response: The Bark Beetle Outbreak in Northern Colorado and Southern Wyoming, September 2011), available at [43], agency leaders do not look favorably upon all legislative proposals to weaken environmental laws to facilitate timber harvest for beetle control. For example, Tom Tidwell, Chief of the Forest Service, criticized recent bipartisan legislation [25] because it would "shortchange the environmental review process, cut out public engagement and collaboration…and override roadless protections." (Testimony from House Subcommittee on Public Lands and Environmental Regulation Legislative Hearing on H.R. __, H.R. 1294, H.R. 818, H.R. 1345, H.R. __, and H.R. 1442 available at [44].

Given the geographic concentration of federal public lands in the West, most of the bills have a specific focus on western states, and were introduced or supported by westerners (Figure 2). But that is not universally the case. Two of the proposals to expand the scope of HFRA were sponsored by Representative Markey, a Democrat from Massachusetts [19,23]. Moreover, support for these bills is bipartisan, showing that the belief that timber harvest can address beetle infestations crosses the political spectrum. Of the 55 total bills, 17 were sponsored by Democrats alone, 21 sponsored by Republicans alone, and 17 had bipartisan sponsors. Markey himself has received very high ratings from the League of Conservation Voters, with a 94% lifetime score from the group.

Figure 2. Bill sponsorship, co-sponsorship, and applicability by region. (Pacific = CA, OR, W, AK, HI; mountain states = MT, ID, NV, WY, UT, CO, AZ, NM; Midwest = ND, SD, NE, KS, MN, IA, MO, WI, IL, IN, MI, OH; SOUTH = TX, OK, AR, LA, KY, TN, MS, AL, GA, FL, SC, NC, VA, WV; east = ME, NH, VT, MA, NY, RI, CT, NJ, DE, MD, PA).



The 55 bills introduced since 2001 show that many legislators, particularly those from western states, believe that timber harvests are a necessary tool to address beetle infestations. This belief has led to the enactment of laws that reduce compliance burdens under NEPA and other federal environmental laws. There are many more proposals for additional significant changes to federal environmental laws to encourage more timber harvests for beetle control. While "there is certainly a tremendous amount of social and political pressure to 'do something' about beetles," there is also growing concern by many that timber harvests for beetle control are expensive and ineffective and that long-term impacts on forests are unknown [42 citing Ann Merwin, director of policy and government affairs for the Wilderness Society]. The policy debate demonstrates the need to critically examine how well these treatments work and place policy in the context of the best available science.

3. A Mountain Pine Beetle Primer

The mountain pine beetle is native to pine forests in western North America [45]. During outbreaks, it can kill millions of trees across extensive areas. The ability to cause such widespread mortality has led it to be described as the most destructive forest pest on the continent [46]. Indeed, economic and aesthetic impacts of outbreaks can be severe. From a manager's perspective, outbreaks are often perceived as a symptom of poor "forest health", while ecologists more often view outbreaks as natural ecological processes integral to the maintenance and resilience of the forest. These differing human perceptions have led to conflicting and ambiguous management goals as well as scientific, social, and political conflict.

The mountain pine beetle is polyphagous on pines (*Pinus*) [45]. It attacks not only native pines but also exotic pines used in ornamental landscaping. Within the natural range of the beetle, only *P. jeffreyi* appears to be avoided, likely due to its unusual chemistry [45]. Pines are well defended and are not easy targets for the beetle. They produce constitutive defenses consisting of resin that can flush the tiny beetles from trees, often drowning them [47–49]. Pines also produce induced defenses in the phloem comprised of resin containing elevated concentrations of toxic monoterpenes [49,50]. Induced defenses develop in response to attack, and thus, involve a lag time of one or more days to develop and can last for a month or more even when trees are killed [51].

To contend with a defensive host, the mountain pine beetle has evolved a complex chemical communication system it uses to coordinate a mass attack on a tree [52]. A female beetle will land, begin to tunnel, and release an aggregation pheromone that attracts conspecifics of both sexes to the tree. Subsequent arrivals release additional pheromone increasing attraction to the tree [53]. If enough beetles respond, the tree can be overwhelmed in just a few days. As defenses are depleted, the beetles release an anti-aggregation pheromone which repels late arriving beetles and acts to reduce intra-specific competition among brood [53]. At this point, the tree has reached "a point of no return" [54]. It will not recover and will slowly die, although it may remain green for nine months or more due to translocation of water to needles by capillary action in the xylem.

The number of beetles needed to kill a tree varies and depends, in part, on the strength of its defenses [55]. In general, as the strength of defenses increase so does the number of beetles needed. Several factors influence the strength of tree defenses. Trees weakened by drought, disease or damage can be overwhelmed by only a few hundred beetles while very vigorous trees may require many hundreds or even thousands [56]. Genetics of the host tree also play an important role. Within a tree species, different genotypes result in differing levels of resistance and susceptibility [57,58]. Genetic differences are even more pronounced when considering differences in defenses among *Pinus* species [59,60].

The ability of tree defenses to affect mountain pine beetle success varies by whether the beetle is in endemic (non-outbreak), incipient (building) and eruptive (outbreak) phases. During the endemic phase, when beetle populations are low, host tree defenses are the major constraint in the ability of beetles to kill trees. However, tree defenses become inconsequential once the threshold to the incipient stage has been surpassed [61]. When numbers are low, beetles attack smaller diameter trees with low defenses. However, once populations rise to the incipient stage, beetles choose larger, healthier, resource-rich trees, despite their superior defenses [61]. Because larger trees have thicker phloem resources to support larval development, they support greater beetle productivity which results in positive feedback that helps fuel the expansion of the outbreak. Thus, host tree traits (primarily host defenses and diameter class) that determine which trees are killed when populations are low, may be unimportant or even have an opposing effect on beetle success when populations are high [61].

It is often reported in the press that mountain pine beetle populations are cyclical. This is not the case. The population dynamics of insects that develop cyclical outbreaks are typically dominated by *delayed* negative density dependent feedback involving regulation by natural enemies and induced resistance mechanisms [62]. This type of feedback results in predictable intervals (cycles) between outbreaks although the amplitude of population peaks can vary due to spatiotemporal variation in abiotic conditions. Bark beetle dynamics, instead, are driven by alternations of negative density dependent and positive density dependent feedbacks resulting in sporadic unpredictable population eruptions primarily driven by threshold effects and typically triggered by abiotic factors, particularly climate [61–63]. It is critical to distinguish between cyclical and eruptive population dynamics as insects exhibiting these two types of dynamics demand different management and monitoring approaches. In particular, eruptive dynamics are triggered by abiotic factors typically outside the realm of human manipulation.

Mountain pine beetle can remain in non-outbreak phase for very long periods of time, even when forests are composed of suitable age classes of host trees and in a condition often considered to be highly susceptible and "unhealthy". Outbreaks occur *only* when multiple thresholds involving temperature, tree defenses, and brood productivity are surpassed that allow positive feedbacks to amplify across several scales [2,64]. While outbreak development is complex, the primary elements that must exist are an abundance of suitable hosts *and* a trigger [63]. Triggers for mountain pine beetle that allow population amplification and subsequent widespread outbreak initiation are warm temperatures and drought, conditions that often co-occur [65]. There can also be a substantial lag period, even several years, from the initiation of the abiotic factors that trigger an outbreak to when populations actually amplify [65,66]. However, once a threshold number of beetles is surpassed, the outbreak becomes self-perpetuating.

While forest conditions alone do not cause outbreaks, certain forest conditions can support larger and more severe outbreaks once they are initiated. Mountain pine beetle attacks only pines (except in rare instances where it "bleeds over" into spruce) [67], and typically only those larger than ca. 15 cm in diameter [68]. Therefore, forests comprised mainly of large diameter pine can be at higher risk of widespread mortality when a trigger occurs than are forests comprised of young, small diameter pine or composed of a mix of tree species including non-pines [68]. Processes that homogenize forest structure and composition such as abnormally widespread stand replacement events (e.g., fires of 1910, Yellowstone 1988) or particular types of forest management (e.g., some timber harvest practices, fire suppression) that alter forest composition and structure over large areas, can contribute substantially to the extent and severity of an outbreak once it is initiated. Processes that result in heterogeneity, such as "normative" wildfires and bark beetle outbreaks, and some land management practices (e.g., restoration treatments focused on restoring a mosaic structure of forest stands of different age classes) tend to reduce outbreak severity and extent by reducing the amount of contiguous susceptible hosts [68].

Climate acts as a trigger for mountain pine beetle outbreaks for a very good reason. Like all insects, mountain pine beetle is poikilothermic-it cannot regulate its body temperature, and thus, all its metabolic rates and vital functions are dependent upon the temperature of its environment [69]. As temperatures rise, feeding, activity, development and reproductive rates increase. Importantly, this also means that the length of the mountain pine beetle life cycle is determined by temperature [69]. Under optimal thermal conditions, development is univoltine (one year). A univoltine cycle allows synchronized emergence of brood adults in mid-late summer, supporting not only mass attacks, but also attacks at a time that allows subsequent offspring to enter winter as cold-hardened larvae [70,71]. Cold hardening is a gradual process that occurs as temperatures fall in autumn. Once larvae are cold hardy it can take temperatures as low as −40 °C to kill significant numbers [72]. However, cold air incursions in fall when beetles are not yet cold hardened or in spring when larvae have lost cold hardening in preparation for transitioning to the adult stage can result in widespread mortality. This can halt an outbreak if subsequent conditions are no longer favorable for the beetle. However, if favorable conditions return, beetle populations rebuild. Importantly, outbreaks require a univoltine life cycle combined with moderate winter temperatures [73].

In areas where temperatures are too cool to support a univoltine life cycle, a semivoltine (longer than one year) life cycle occurs [73]. A semivoltine life cycle is maladaptive for the beetle in several ways. First, adaptive seasonality is disrupted, increasing the percentage of brood that enter winter in stages vulnerable to freezing (eggs, pupae and adults). Additionally, mortality increases when beetles must pass through two winters and feed on a food source increasingly depleted in moisture, nutrients, and symbiotic fungi [74]. Warm periods support not only greater brood production and survival in areas typically suitable for the beetle, but also allow a transition from a semivoltine to a univoltine life cycle in areas otherwise too cool. This increases the spatial extent of suitable habitat and tree mortality. Thus, abnormally warm periods can vastly increase the total area suitable for the beetle and play a major contribution to the synchronicity and coalescence of outbreaks across regions [2,65].

Drought can also play an important role in outbreak initiation. Host tree defense mechanisms are compromised during drought allowing beetles to more easily attack trees [2,75]. Tree defenses are major constraints when beetles are in non-outbreak phase. However, drought-weakened trees can support population amplification until a point where stand level densities surpass a critical threshold. Once this threshold is passed, tree defenses lose their importance in regulating beetle populations [61]. Very importantly, drought stresses large numbers of trees at a regional scale. This results in large numbers of trees that are easier for the beetles to kill, further supporting outbreak intensification [65,76].

Recent studies have found that drought occurring years or even decades before the outbreak can influence outbreak initiation. Furthermore, prolonged drought stress appears to pre-condition trees to be more susceptible, an effect that can continue for years after normal precipitation has returned [58,65,77]. There also appears to be a genetic component to tree sensitivity to drought, and subsequently, susceptibility to beetles. In two studies, one conducted in whitebark pine and the other in ponderosa pine, differences in growth of surviving trees and trees killed by beetles over the last century suggest that adaptive differences to changes in climate exist. In the whitebark pine study, the trees studied were co-dominants and not significantly different in diameter age or mean growth over their lifetimes [58]. However, trees that were killed exhibited faster rates of growth in the first half of the century suggesting they were better adapted to the cooler wetter conditions of that period. The surviving trees had greater growth in the latter half of the century when conditions were warmer and drier. Millar *et al.* [58]) suggested that the beetle-caused tree mortality in the stands they studied resulted in a strong natural selection event that removed trees less fit under our current climate while leaving those more well-suited.

Likewise, Knapp *et al.* [77] found genotypes of ponderosa pine that were slow-growing in the two to three decades prior to the outbreak were much more vulnerable to beetle infestation than those that were fast-growing, again suggesting the beetle may act as a selective agent shifting genetic structures in stands over time to those most suited to prevailing climatic conditions. In lodgepole pine, trees of similar age and diameter growing intermixed in the same stand and under the same conditions exhibited different levels of sapwood moisture that were highly correlated with susceptibility to beetle attack [74] hinting at genetic differences in water efficiency. Those with lower sapwood moisture were attacked and killed by the beetle while those with higher sapwood moisture were not [74].

While mountain pine beetle has developed outbreaks for millennia, the current outbreak is far outside the historic norm [2,78]. The unprecedented size and severity of this outbreak is due to a combination of increasingly favorable climate for the beetle and forest conditions. Warming trends have supported the development of a univoltine cycle in many areas that previously were too cool and have resulted in greater beetle productivity and survival [79]. This has led to massive tree mortality, not only in areas previously favorable for the beetle, but also in areas previously suboptimal or unusable. Warmer temperatures and high population levels have also supported expansions of the beetle's range hundreds of kilometers further north in British Columbia and eastward across Alberta [80–82]. In these new locations, the beetle is infesting naïve hosts including (in the eastern expansion) a novel species, jack pine [80,82]. These naïve hosts exhibit lower defenses to beetle attack [83] as well as similar chemical compositions to natural hosts [84] promoting establishment. Predictions are that the beetle will continue to move across the continent through the boreal forest and finally into eastern pine forests [78].

Warming has also allowed the beetle to move higher in elevation where it is devastating whitebark pine, a tree that is foundational to the western North American subalpine ecosystem and that was previously protected from the beetle by cold [73,85]. Movement into the subalpine has been supported by overall warmer temperatures and milder winters allowing the beetle to switch from a semivoltine to a univoltine life cycle while simultaneously reducing winter mortality [85–87]. The resulting mortality to whitebark pine in many areas, particularly the greater Yellowstone Ecosystem, has been so severe the tree is now proposed for listing as an endangered species [88]. The tree is already listed as an endangered species in Canada due to the combined effects of mountain pine beetle and white pine blister rust [89].

4. Mountain Pine Beetle Outbreak Suppression

Treatments used to mitigate the effects of mountain pine beetle are grouped into three broad categories. Treatments that strive to reduce or eliminate beetle populations are termed direct controls [90]. Treatments aimed at increasing tree vigor and altering stand conditions to be less favorable for beetles are called indirect controls [90,91]. Prophylactic treatments aim to protect high value individual trees or stands of trees from infestation. Salvage, while often included in beetle management programs does not actually reduce or impact beetle populations-it is the removal of dead trees for economic or other reasons and often involves removal of trees that are already 'empty' of beetles and thus has no impact on beetle population size. Because our focus is on how well science supports the use of timber harvests (including tree felling and destruction of trees in place) to reduce or suppress bark beetle outbreaks, we will focus primarily on direct and indirect controls concentrating on these treatments.

Direct control includes sanitation treatments such as removing single trees or small patches of trees that are infested with the insect, clearcutting (also called block harvesting) and prescribed burning of infested trees, as well as fell and burn, trap trees, debarking, and application of insecticides or toxins such as MSMA (monosodium methanearsonate). Sanitation cuts attempt to remove most or all beetles in an area by removing infested trees before the beetles developing within them can emerge and disperse [90,92]. Prescribed burns, fell and burn, debarking, and toxin applications attempt to destroy beetles in infested trees on-site. Trap trees are trees that are baited with attractant pheromone baits in an attempt to draw beetles into specific areas where they are concentrated into the baited trees which are subsequently taken to the mill or destroyed. Each of these methods relies on killing as many beetles as possible in order to lower beetle population thresholds below which they can maintain outbreak dynamics.

Indirect controls are primarily silvicultural in nature. The main treatment used for mountain pine beetle is thinning. Thinning is thought to act by reducing inter-tree competition for water, nutrients, and light, enhancing greater tree vigor, and thus defenses against the beetle [93]. Thinning treatments are also thought to reduce successful beetle attacks by altering microsite conditions by increasing temperatures on bark surfaces on bark in summer and decreasing them in winter, as well as disrupting beetle communication by increasing wind flow [94,95]. A new treatment recommended for reducing bark beetle infestation is "daylighting" which involves removing trees and shrubs from around trees that are to be protected to increase light on the tree's stems to disrupt beetle colonization. Other silvicultural treatments include removal of beetle-suitable hosts (mature trees and old growth) and conversion of stands from species preferred by beetles (pines) to species that are not hosts or converting stands that are primarily pine to a mixed species composition [91,92]. Most of these approaches involve, completely or partially, the use of timber harvests.

4.1. Efficacy of Direct Controls

Direct control treatments are extremely expensive in time, effort and resources. They address only one aspect of an outbreak which is the amount of beetles present in a stand or area. Because they do not address the underlying conditions that support an outbreak (climate, tree condition/stress) their effects are considered a holding action until conditions shift to being less favorable for the beetle [92].

Direct control efforts must be maintained at a high level on an annual basis until the outbreak ceases [3,90,96]. It is highly controversial whether direct controls are effective in reducing tree mortality in the short-term, and if they can be effective in halting or suppressing outbreaks in the long-term.

One of the biggest problems in assessing the utility of direct controls is a general lack of monitoring or *post hoc* assessments of the outcomes of implementing these practices. Despite decades of direct control and large-scale implementation of these practices, few rigorous studies on its efficacy have been done and there remains no agreement among scientists or foresters regarding its ability to reduce beetle populations or losses of trees. Studies conducted prior to the current outbreak have variously concluded that direct treatments may merely act to delay infestation of susceptible stands [97], or that if used correctly, can be effective [98,99]. Many studies found that while some treatments slowed the rate of infestation, overall, they had little to no impact on mountain pine beetle populations [97,100–104].

The US and Canadian governments have spent hundreds of millions of dollars in direct control efforts to address the current outbreak. However, assessments of the efficacy of these efforts are nearly non-existent and only a few studies on assessments have been published. The few that have been published are reviewed here. Although much of our review addresses how well science supports US policy, we use primarily studies conducted in Canada as few studies have been published on direct control measures during the current outbreak in the US.

Nelson *et al.* [3] evaluated the efficacy of five direct control treatments in British Columbia roughly midpoint in the portion of the current outbreak as it progressed in that province. The assessment was extremely short-term and looked only at the response of beetles in the year immediately post-treatment. However, it provides one of the very few broadscale assessments ever conducted of the efficacy of direct controls during an outbreak. The treatments assessed were applications of MSMA, trap trees, fell and burn, and clearcutting. The study was split into three geographic regions to account for potential sources of variability due to location and different background levels of beetles. The northern-most region was at the margin of the beetles range (expansion zone) and possessed relatively low beetle populations, while the central and southern regions had higher beetle populations and were known to have supported high beetle populations historically. The study found that, overall, sites receiving MSMA treatments exhibited higher infestation intensities (a metric based on kernel density estimators) than randomly selected untreated sites with similar characteristics. This was particularly pronounced in the southern region. Results for trap tree treatments showed substantial variability within and among regions. A reduced infestation rate in response to treatment was observed more often than not in the northern area where beetle pressure was low. However, in the central and southern regions where beetle pressure was higher, the range of infestation intensities was similar for treated and untreated sites although a larger number of comparisons found higher infestation intensities in the treated sites. The overall conclusion was that MSMA and trap tree treatments may be effective, but not reliably, and only when beetle pressure is low and environmental conditions are not highly favorable for the beetle.

Results for fell and burn were also variable. In the northern region, intensities were lower overall in treated *vs.* untreated sites. However, in the central area, treated areas tended to have greater infestation intensities. In the southern area, no discernible effect of treatment was seen. Therefore, like with trap trees, fell and burn appeared to sometimes be effective, but only when populations of beetles were low, and became increasingly unreliable as beetle pressure increased and the infestation moved into outbreak phase.

Removal of trees in patches was studied only in the central region. No significant effect of treatment was detected. Clearcuts were assessed in the central and southern areas and were found to lead to a significant reduction in infestation intensity. In almost all cases, infestation intensities were lower in treated *vs.* untreated areas. However, this was likely due to the removal of all living trees (potential subsequent hosts) that survived the beetle as well as the infested trees. The overall conclusion of the study was that mitigation treatments are effective when populations are low to moderate and if infested trees can be kept to 2.5 or fewer per hectare. Efficacy was also recognized to be contingent upon a high level of accuracy in detecting infested trees and wide-scale and continuous implementation of treatments. However, with only one year of data, the authors could not predict how long treatments would need to be sustained to remain effective, nor what effect beetle pressure from surrounding areas might have on the subsequent fate of treated stands. No follow up study has been published to report how these treatments fared as the outbreak progressed.

Fell and burn has been a stalwart component of the direct control efforts against mountain pine beetle in Canada during the current outbreak, particularly on the advancing front as the beetle expands its range eastward. Coggins *et al.* [105] examined the efficacy of fell and burn treatments to "stabilize" such infestations (*i.e.*, prevent expansion) using field plot data from sites at the expanding edge of the mountain pine beetle infestation in 2008 in eastern British Columbia and western Alberta. The authors used multiple modeling scenarios along with ground data to demonstrate how infestations may develop with and without mitigation, and to predict how long mitigation may need to be maintained to be effective given different levels of infestation and detection accuracy. They found non-mitigated plots experienced more tree mortality due to the beetle and that infestations in these plots expanded more rapidly. The higher the expansion factor (means rate of increase, e.g., 2 would indicate a doubling of the population each year) the greater the detection accuracy that was required to maintain a static population. When a beetle population had an expansion factor of 5.1 (high), an 80% detection rate was required, whereas with a population with an expansion factor of 1.1 (very low), the minimum detection rate could be as low as 10% and still be effective. The authors also modeled how long it would take to achieve population stability given different levels of infestation. On average, across their stands, with a 70% detection accuracy rate, mitigation would take 11 years, at 80% 6 years, and at 90% 3 years. The actual mean mitigation efficiency at their sites was found to be 43%, a level at which no control could occur. They concluded that the stabilization of mountain pine beetle populations is possible, but only with a much higher detection accuracy than commonly occurs coupled with an intense level of mitigation maintained potentially over a very long timeframe.

Wulder *et al.* [96] looked at the effectiveness of sustained mitigation on slowing the beetle's expansion in western Canada. The results were difficult to assess because of the unevenness of application of mitigation treatments (for example, in one year only 68% of sites slated for mitigation were treated) and differences in background beetle populations. However, such a situation is typical and thus may represent the reality of many on-the-ground direct control efforts. One site where little mitigation was conducted early on, did exhibit a strong increase in tree mortality due to the beetle that declined once extensive mitigation efforts were implemented. However, overall, the conclusion was that mitigation must be extensive and continuous to work and may only be effective when populations are low to moderate.

Trzcinski and Reid [104] studied the trajectory of beetle populations in treated and untreated zones in Banff National Park from 1997–2004. The Park used a combination of pheromone-baited trees and fell and burn to remove as many beetles as possible from treatment zones—they also conducted prescribed burns to reduce beetle numbers and lodgepole pine hosts. The area colonized by the beetle increased rapidly over this time period in both the untreated and treated zones. After four years of treatment, control measures did not reduce the area affected by beetles and infestations continued to expand at a similar rate in both zones. The authors estimated that between 45% and 79% of infested trees had failed to be detected in the treated areas. This equated to *only* 0.7–3.7 infested trees remaining per thousand ha yet still was sufficient to support subsequent rapid beetle population growth.

A general consensus of these studies is that suppression of a beetle outbreak would require massive sustained efforts with extremely high detection rates to succeed. It has been estimated that 97.5% of beetles in an area must be killed to merely stabilize a mountain pine beetle population [90]. Even a small increase in survival above this value can allow a substantial increase in population size. For example, if mortality drops to 95%, this would allow a population to *double* in size annually. If the goal is not just to stabilize a population, but to reduce it, mortality of beetles would need to be higher than 97.5%, a goal that is highly unlikely given the vast areas that would need to be treated on a continual basis when conditions are favorable for outbreak development. Even if 100% removal of infested trees from an area was feasible, the migration of beetles into treated stands from surrounding areas allows reestablishment and subsequent tree mortality further decreasing the potential for effective direct control.

The on-the-ground reality is that direct control efforts typically fall far below the levels needed to stabilize, let alone control, mountain pine beetle populations. In the above cited studies, rates of detection in mitigated stands ranged from 45%–79%. These situations are not unusual. Direct control treatments are laborious, extremely costly and time consuming, and require high levels of training. Logistical difficulties, including proper seasonal timing, access, inclement weather, and lack of trained personnel, increase the odds that they will not be effective. The high financial cost of such efforts coupled with a volatile market for sawtimber, pulp and pellets further complicates the use of direct controls. Importantly, outbreak development is extremely swift and the amount of mitigation required can rapidly outstrip the ability of managers to respond.

During an outbreak the number of trees killed annually is often in the millions and infestations may cover hundreds of thousands of hectares [90]. Carroll *et al.* [90] presents an example of the degree of mitigation that would be required for an outbreak that covers 300,000 hectares with a rate of increase of 2 (the population doubles in one year-a conservative rate for an outbreak). In this case, 150,000 ha of infested trees would need to be removed each year just to maintain a *static* beetle population–this would still allow tree mortality to occur for many years, potentially until most or all mature trees were killed. In reality, such a high level of detection and mitigation is impossible. Given that the goal of direct management is to reduce populations and protect trees, the effort that would be needed to actually reduce such a high beetle population would require an even more unlikely effort.

Studies in other bark beetle systems also have found that a high degree of detection accuracy and intensity of mitigation is required to reduce beetle numbers. Fahse and Heurich [106] found that control of *Ips typographus*, a less aggressive European bark beetle, requires a detection and removal level of around 80% to be effective. They concluded that direct control efforts are useless and should be dropped if survival probabilities of the beetle after treatment are above 20%–30%. This estimate is in line with those developed in studies on mountain pine beetle in North America and highlights the challenge the high reproductive capacity of bark beetles poses when conditions are favorable for outbreak development.

It is not just the difficulty of dealing with the extreme spatial extent of outbreaks and the challenge of detection and treatment that makes the efficacy of direct control measures unlikely, but also the time frame over which direct controls must be maintained. Carroll *et al.* [90] estimated that to control a population involving 10,000 infested trees with expansion factor of 2 (conservative) and with a detection and removal rate of 80% (difficult), it would take at least 10 years of annual treatment to reduce the population to a single tree. If the population was tripling or quadrupling, a more likely scenario during an outbreak, it would take 18 or 41 years, respectively. A costly, intensive detection and treatment program lasting that long, assuming sufficient trees even remained to be infested, would be unlikely [90].

Carroll *et al.* [90] emphasized three requirements for direct controls to be effective in treating *individual* infestations: infestations must be detected early, efforts must be applied quickly and intensively, and control programs must be maintained continuously until the desired population level is achieved. Because of the cost and intensity of treating individual infestations, the US Forest Service recommends that direct control measures only be applied to higher value stands [92]. However, treating individual infestations or stands during outbreaks can fail because of the regional nature of outbreaks. Outbreaks are driven by abiotic factors that affect entire regions (warm temperatures and drought). Thus, they consist of many infestations that occur synchronously across a very large area. These infestations often coalesce to form vast expanses where beetle populations are extremely high. These characteristics mean that many stand level efforts are prone to failure due to high beetle pressure and migration into treated areas by beetles from surrounding areas. Given that treating entire regions is impossible, and that many treatments are not in line with other land use objectives, direct control efforts may in some cases, not be worth their costs. The consensus of studies and retrospectives over the course of several outbreaks is that even after millions of dollars and massive efforts, suppression using direct controls has never been effectively achieved, and at best, the rate of mortality to trees was reduced only marginally [90,101,102,105]

4.2. Efficacy of Indirect Controls

Thinning is the primary indirect control measure used to manage the mountain pine beetle. It is generally considered a preemptive measure to be implemented prior to the initiation of a mountain pine beetle outbreak, although it is increasingly employed to reduce damage by the insect during outbreaks. It is often touted as a global panacea for problems with pest bark beetles. One type of thinning is even termed “beetle-proofing” [107], further reinforcing the view among managers, the public, and policy makers, that this approach is failsafe. While overall, evidence suggests that thinning can reduce mortality of trees due to mountain pine beetle, the outcome is frequently more variable than is often recognized or reported. This is particularly true when outbreak populations are involved.

So how exactly does thinning work, and how well does thinning hold up under outbreak conditions? Surprisingly, the mechanism(s) by which thinning affects beetle activity in forest stands is still not well understood. Two, non-mutually exclusive, lines of thought exist. One hypothesis is that thinning increases tree vigor, and thus tree defenses, by reducing competition among trees for light, nutrients and water [93,108]. Intuitively, this makes sense, and indeed, immediate impacts of thinning on reducing water stress have been seen [109]. Likewise, increases in growth and photosynthetic rates also have been observed post-thinning, albeit after a lag period of one or more years [107,109,110]. Increases in growth and vigor are predicted to increase the amount of energy that trees allocate to defense, leading to greater resistance to beetle attack through increased resin and monoterpene production. In fact, the initial impetus for the use of thinning to manage mountain pine beetle came from an early study that found that ponderosa pines in thinned stands produced more defensive resin [93]. However, subsequent studies have reported a variety of responses in resin production as well as growth in response to thinning. For example, Zausen *et al.* [111] found that ponderosa pines in the thinned stands exhibited lower water stress but also produced less resin. This, along with the thicker phloem (greater food resources) found in trees in thinned stands, indicates they might be not only more susceptible to attack but also a more productive resource for beetles. In contrast, McDowell *et al.* [112] found greater resin flow in thinned stands. Both studies were conducted in southwestern US ponderosa pine forests indicating that the variable responses observed were not due to major regional differences in hosts. Six and Skov [113], in a study conducted in ponderosa pine in the northern Rocky Mountains looking at effects of thinning and burning treatments, found that resin flow was highest in trees in burn treatments, intermediate in controls, and lowest in thinned treatments. Raffa and Berryman [114] tracked the fate of trees over time during an outbreak and found no significant difference between resin flow for lodgepole pines that survived attack vs those killed by the beetle.

A number of studies have noted a reduction in beetle caused-mortality of trees immediately after thinning treatments were applied and before trees had time to respond physiologically to lower stocking densities. This timing suggests that the effects of thinning may have more to do with microsite conditions than to changes in tree vigor or defense. These observations led to the second line of reasoning that thinning affects beetle activity through changes in microsite conditions.

Thinning alters temperature, light intensity and wind speed within a forest stand; factors that can have major effects on insect behavior and success. A number of studies have tried to describe how shifts in microsite conditions due to thinning may influence mountain pine beetle activity. Bartos and Amman [94] investigated how incident solar radiation, wind speed, wind direction and temperature were altered by thinning and whether changes affected beetle responses to stands. They did not conduct statistical analyses on their data; however, there was a trend for south sides of trees in thinned stands to be warmer, and ambient temperatures in thinned stands to be overall warmer during parts of the day. Incident solar radiation was higher in the thinned stand. It is not known if bark temperature affects beetle attack behavior, although higher temperatures on south sides of trees in thinned stands have been suggested to be deleterious to beetle development [94]. However, this speculation does not account for differences in local environmental conditions. For example, at cool sites, increased temperatures and insolation could ostensibly support better beetle development by increasing thermal units sufficiently to support a univoltine life cycle.

Light intensity affects the flight behavior of mountain pine beetles [115]. However, if and how different levels of light in treated and untreated stands affect beetle attack behavior is unclear. It has been hypothesized that a reduced propensity for flight in darker stands might concentrate beetles for mass attack, while beetles may be more likely to disperse in open stands [116].

The hypothesis that light has a strong effect on mountain pine beetle behavior, particularly in reducing attacks, has led to a new treatment called daylighting. This approach is currently being implemented on a broad scale by federal and western state agencies. Daylighting involves removing trees and vegetation from around trees that are targeted for retention and is believed to work by repelling beetles from the boles of trees by increasing light and solar radiation [117]. While widely recommended, the efficacy of this treatment is unknown; there are no published studies on its effects on bark beetles.

Changes in wind speed and direction due to thinning have also been suggested to alter beetle behavior by disrupting beetle communication via disruption of pheromone communication. Schmid *et al.* [118] found no statistically significant differences in horizontal and vertical wind patterns in thinned and unthinned stands. However, disruption of pheromone plumes by greater wind speeds may affect communication and thus the potential for successful attacks [95]. Ultimately, we need to look at actual population dynamics of beetles in treated and untreated stands to understand if microsite effects hold under epidemic conditions. MacQuarrie and Cooke [119] found that, under outbreak conditions, mountain pine beetle populations exhibited density-dependent dynamics and that thinning did not change the epidemic equilibrium. In this study, population growth curves did not exhibit responses that would be expected if microsite conditions played a role in beetle behavior. It is evident that more research is needed to understand how these effects ultimately influence tree mortality due to beetle attack.

While we may not have a complete understanding of how thinning works, it is clear that this practice can have a significant effect on mountain pine beetle infestations. Several studies have reported striking differences in mortality to trees caused by beetles in thinned *vs.* un-thinned forests (reviewed in [120,121]). In contrast, only a small number of studies have reported failures. However, the disparity in numbers of successes and failures must be placed within a broader context. Many studies assessing the efficacy of thinning have been conducted under non-outbreak conditions. Their results do not reflect how stands perform during an outbreak. Additionally, failures are often not reported, dismissed as a result of poor management 'next door' or targeted for management without evaluation. This is unfortunate because thinned stands that fail may have particular characteristics that could inform a better understanding and application of this approach.

Studies conducted during outbreaks indicate that thinning can fail to protect stands. In Colorado, thinning treatments in lodgepole pine implemented in response to the outbreak that began in the 90s often only slowed the spread. Klenner and Arsenault [122] reported high levels of mortality due to the mountain pine beetle across a wide range of stands densities in lodgepole pine in British Columbia during the same outbreak. They noted that silvicultural treatments were largely ineffective in reducing damage to the beetle. Preisler and Mitchell [123] found that once beetles invaded a thinned stand the probability of trees being killed there can be greater than in unthinned stands and that larger spacings between trees in thinned stands did not reduce the likelihood of more trees being attacked. Whitehead and Russo [107] reported on the performance of 'beetle-proofed' (stands thinned to an even spacing of about 4–5 m between mature trees) and un-thinned stands in five areas in western Canada during approximately the same time period. These treatments were successful in protecting stands when they were combined with intensive direct control measures (removal of infested trees) in the areas surrounding the thinned units, but failed if units were exposed to beetle pressure from the neighboring area—a situation most thinned stands experience during an outbreak.

Unfortunately, long-term replicated studies monitoring beetle responses to thinned forests from non-outbreak to outbreak to post-outbreak phase are virtually non-existent. One large fully-replicated long-term study was initiated in 1999 under non-outbreak conditions and continues to track beetle activity [113]. In this study, mountain pine beetle was low in all treatments in the period leading up to the outbreak, but increased in some controls and burn treatment replicates as the outbreak developed. Although more trees were killed overall in control units during the outbreak, all controls still retained a greater number of residual mature trees than did thinned stands as they entered the post-outbreak phase [124].

Two factors contribute substantially to our inability to assess how well thinning performs under outbreak conditions. One, very few thinning treatments are monitored after implementation over either the short- or the long-term. Thus, for the vast majority of stands that have been treated, we have no data on how well they perform once an outbreak of the insect initiates (or for that matter, even under non-outbreak conditions). Second, stands that become infested, thinned or otherwise, are often targeted for intensive suppressive management and are cut without assessment or data collection. This even includes studies and sites that are intended to inform management. For example, at the sites studied by Whitehead and Russo [107], infested trees were being removed from the study sites even before data collection for their study could be completed. The long-term study discussed previously [113,124] is under continual pressure to be logged to remove beetle kill even though the site lies within an experimental forest designated specifically for studies assessing the outcomes of forest management.

5. What are the Goals?

When we manage forests, we do so in an attempt to achieve one or more outcomes, preferably with minimal negative effects on non-target resources. To be effective, management must have explicit and appropriate goals as well as clear metrics for success. Ideally, management is monitored to assess how well it meets its goals, where it falls short, and whether and how it can be improved. This approach is called adaptive management and implies an iterative process through time whereby we learn from the outcomes of our actions and base future actions on improving performance [125].

Not only outcomes, but the costs of management must be factored into decision making. These include direct financial costs as well as the less tangible (at least in dollar values) effects on ecosystem services and functions. By considering the full cost of management along with benefits as verified through monitoring and evaluation, we lessen the risk of failure, financial waste, and unnecessary negative environmental impacts.

In assessing how well we meet goals when managing for mountain pine beetle, we must ask several questions. Do our management practices actually control the beetle during outbreaks? Do the outcomes justify the financial and ecological costs? And, what long-term impacts do these treatments have on forests and their ability to adapt to climate change? These questions are difficult to answer. Only limited data are available on the short-term efficacy of direct and indirect controls, and information on long-term effects is virtually nonexistent. The results of short-term assessments can be difficult to interpret. For example, often only the proportion or numbers of trees killed by beetles post-treatment are reported. This does not allow a complete evaluation of outcomes. A study may report that 75% of trees in controls are killed by the beetle, whereas only 10% are killed in thinned stands. At first glance, this appears to be a resounding success in saving trees. However, if we approach this situation from a pretreatment perspective, our interpretation of success may change. In this example, 400 mature trees existed in each plot prior to treatment. After treatment, 100 mature trees remain in the thinned plots (300 trees have been removed by thinning). Doing the math, we find that once the beetles have run their course, more residual living trees (100) actually remain in the control plot than in the thinned plot (90) and, in fact, humans have contributed more to tree mortality than have the beetles. In the case of silvicultural intervention, humans typically must expend considerable effort and expense. They also choose the trees that remain, and thus the structure and composition of the remaining forest. This may result in very different trajectories for residual forests as discussed below.

When we include pre-treatment conditions as well as post-treatment responses we can assess the management efficacy from a more informed position. For instance, in a retrospective study investigating the effects of management on spruce beetle, researchers found that post-infestation, untreated stands had more live spruce trees and greater basal areas. When comparing only residual large spruce, final densities in both stand types were similar [126]. Six [124] found higher numbers of mature living trees remained in control stands of ponderosa pine than in thinned stands post-mountain pine beetle outbreak. In a study in Canada focusing on stocking density of living lodgepole pine post-outbreak, the authors found that, even in hard hit stands, stocking density in post-outbreak unmanaged stands was sufficient to maintain desired levels of productivity [127]. Klutsch *et al.* [128] in a study conducted in lodgepole pine forests in Colorado, found greater mortality of trees due to the beetle in more densely stocked stands. However, while the density and basal area of lodgepole pine in infested plots declined 62% and 71%, respectively, the number of trees that remained and their size distribution post-outbreak indicated that lodgepole pine would remain the dominant overstory tree. In another study in Colorado, the beetle killed 60%–92% of overstory lodgepole pine. However, these stands retained residual overstory trees as well as advance regeneration. Furthermore, untreated stands were predicted to return to pre-outbreak stocking levels approximately 25 years sooner than treated stands [129]. Other studies have found similar results for both lodgepole and ponderosa pine [130–134]. These studies highlight a seldom considered impact of mountain pine beetle- that it can act as a natural thinning agent and seldom removes all mature trees during outbreaks. These effects are an important part of the ecological role that the beetle plays in western pine forests [135].

It is also important to recognize there can be significant differences in long-term forest trajectories for stands thinned by beetles *vs.* those thinned by humans. When humans thin, they select for particular size classes, often favoring the retention of larger, older trees, selecting toward one desired tree species, and often ‘thinning from below’ which removes advanced regeneration (small trees) [123,136]. Thinning prescriptions also typically call for relatively even spacing between residual trees [92,107,121]. Mountain pine beetle, on the other hand, often selects the largest trees during outbreaks (with exceptions; [121,123,131]) which can lower the mean diameter of the stand [128]. However, beetles often leave sufficient numbers of large diameter trees to maintain a dominant overstory of pine. Beetles also leave substantial amounts of advanced regeneration to replace the mature trees that arekilled [121,129]. Spacing among trees after an outbreak is uneven, resulting in a clumpy network of living trees [129]. Patches where all trees are killed are seldom extensive and add to a mosaic structure as forests recover post-outbreak. Heterogeneous stand and mosaic forest structures are more typical of natural conditions and can support greater biodiversity and resilience against fire and subsequent beetle outbreaks [137–139]. In contrast, intensive thinning treatments by humans typically favors the retention of mature pines. Over time, these pine-dominated stands grow, they are predicted to have increased susceptibility and potential for tree mortality from future mountain pine beetle outbreaks [123,136].

Very importantly, the beetle exercises selectivity in the trees it kills. While extremely high numbers may override this selectivity, evidence is accumulating that, even under outbreak conditions, beetles choose trees that have particular qualities. Beetles commonly select trees for attack that exhibit lower growth rates, defenses, and higher water stress [58,74,77]. While these factors can be influenced both locally and regionally by site conditions and climate, much of the variation in these properties within individual stands that affect bark beetle choice likely has a genetic basis. Outbreaks can result in strong natural selection against trees with phenotypes (and likely genotypes) favorable for the beetle and for those that possess unfavorable qualities [58,77]. However, when humans thin forests, trees are removed according to size, species, and density, without consideration of genetics. Thus, trees best adapted to surviving beetle outbreaks are as likely to be removed as those that are not.

When humans thin forests, they typically manage for resistance and resilience, rather than adaptation which involves genetic change. It is very important to distinguish between resistance, resilience, and adaptation, as each have different goals and operate on different temporal scales [140]. Resistance is a short-term holding action where we try to maintain an existing state. Approaches focusing on resistance often require massive interventions and increasing physical and financial investments over time. Such approaches may set forests up for future outbreaks [136] and even catastrophic failure as they surpass thresholds in a warming climate [140]. In contrast, practices that promote resilience attempt to allow forests the ability to adjust to gradual changes related to climate change and to recover after disturbance. However, like resistance, resilience is not a long-term solution. In the long term, forests must be able to adapt to change. Adaptation involves genetic change driven by natural selection. Currently, much of forest management, including bark beetle management, focuses on resistance and resilience, mainly through direct and indirect management, respectively. However, neither approach allows for true adaptation. For long term continuity of our forests, it will be imperative to begin to incorporate this aspect of management into our approaches.

We also need to reassess the ecological role of bark beetles, including the mountain pine beetle, in our forest ecosystems. As has been well demonstrated by a century of fire suppression, the dampening or suppression of natural disturbance can alter forest trajectories in undesirable ways, many of which can be irreversible. Although beetle outbreaks, like fire, can have negative impacts on timber values and aesthetics, their natural role in many forest ecosystems is seldom considered and beetle suppression is often perceived as something that must be conducted at all costs. However, as with fire, suppression of beetles over the long term may alter forests in ways that are not desirable or sustainable. While intensive management for bark beetle suppression is called for in some situations such as in the wildland urban interface, it may not be appropriate in many other areas where natural processes including natural selection are needed to maintain a dynamic and functional forest.

6. What are the Needs in Research and Monitoring?

There is clearly a need to better understand how well management programs aimed at reducing mountain pine beetle work, particularly under outbreak conditions, and what impacts these treatments have on forests in both the short and long term.

Perhaps the biggest area of need is in monitoring. Monitoring is essential to understanding whether mountain pine beetle treatments work, and in which contexts, but as noted above there has been all too little long-term monitoring of the effectiveness of various treatment efforts. This is a failing among both agencies and researchers. Agencies often do not have strong incentives to conduct long-term monitoring: Monitoring is costly; external and internal political pressures focus on short time frames; and monitoring may produce information that conflicts with agency goals or missions. It is also difficult to get strong public pressure to force agencies to conduct the necessary monitoring, particularly when the public has been led to believe that outbreaks are strictly the result of a lack of management. Even for scientists, long-term monitoring projects are not encouraged by short-term funding time frames and professional incentives or norms; monitoring is often not viewed as "real" science, and the long-time frames required for monitoring to result in significant gains in information are often longer than the time frames used for professional advancement (e.g., completion of a dissertation, tenure review) [141].

Addressing the shortage of monitoring for beetle treatments may, therefore, require far more than simply trying to provide additional funds (even assuming additional funding is politically feasible). Scientists can help by encouraging and rewarding projects that involve long-term monitoring. Agencies might try to establish units that are focused specifically on monitoring forest health, insulating monitoring projects from adverse political or bureaucratic pressure [141]. Finally, tools that might reduce the cost of monitoring significantly, such as retrospective studies and remote sensing, should be used to complement traditional monitoring and decrease its costs.

Monitoring is all the more essential if forest health management in general, and beetle treatments in particular, are truly to be guided by adaptive management. The high levels of uncertainty and dynamism associated with beetle infestations and the effectiveness of beetle treatments make adaptive management a very appealing tool to reduce uncertainty and allow us to respond to changes in global climate and forest ecosystems. But adaptive management requires monitoring to be successful [141], monitoring that is currently not occurring even as agencies conduct massive beetle treatments and propose to pursue even more.

There is also a real need to increase research on management efficacy and, in particular, how our approaches affect forest adaptation including genetic responses of trees to climate and the role in bark beetle selectivity and fitness. With a changing climate we will need to develop new approaches rather than trying to force old methods of questionable efficacy onto new conditions.

Unfortunately, most funding for research on bark beetles is very short-term, sometimes even as short as on an annual cycle, and thus cannot hope to address the complexities of beetle responses to treatments. Funding cuts to research personnel, particularly in agencies like the US Forest Service, have exacerbated this problem exactly at the time when the need for rigorous research is increasing at a rapid pace. The US Forest Service has recognized that long-term planning must include explicit goals to increase forest resilience and adaptation to disturbance, including outbreaks of the mountain pine beetle. However, with extreme cuts to budgets and personnel, they are highly constrained to meet these needs at this time. Likewise, cuts in federal funding to agencies such as United States Department of Agriculture and the National Science Foundation concurrently reduce the ability of academic researchers to address these problems.

7. Aligning Policy to Science

Our survey of the relevant literature finds that there is significant uncertainty about whether the most commonly used beetle timber harvest treatments are, indeed, effective. Yet there has been little discussion of this uncertainty in the relevant policy debates. Politicians have instead latched on to beetle timber treatments as a cure-all for beetle infestations and have pushed to weaken or eliminate environmental laws that are perceived to be obstructing these treatments. Agencies such as the US Forest Service, to their credit, have been more nuanced in their support for bills that package beetle timber harvest treatments with weakened environmental laws; they have opposed several proposals to alter environmental laws to allow more treatments, but on the other hand, the agencies have at times also aggressively pushed for the implementation of treatments.

It seems clear that the policy debates–both in the agencies and in Congress–need to be better informed by science. Researchers should be more proactive in communicating their understandings of the current science to policymakers. This does not mean that researchers need to take a position pro or con vis-à-vis beetle treatments, or even vis-à-vis specific legal proposals. In the face of uncertainty, aggressive beetle timber harvest treatments may be warranted in some instances. However, policymakers should be aware of uncertainty when they are making the relevant decisions and should also be more willing to include the voices of scientists in the development of policy.

Given the uncertainty about the effectiveness of many beetle timber harvest treatments, the high financial costs of those treatments, the impacts on other environmental resources and values, and the possibility that in the long-run those treatments may interfere with the ability of North American forests to adapt to climate change, our position is that weakening or eliminating environmental laws to allow more beetle timber harvest treatments is the wrong choice for advancing forest health in the United States. Indeed, given the uncertainty, the costs, and the possibilities of both short-term harm to other resources and long-term ineffectiveness, we believe that the current structure of thoughtful, detailed environmental review for these projects is, in general, appropriate. If agencies believe that they need to be able to react quickly to specific infestations with treatments, and that this quick reaction is incompatible with existing legal procedures, we encourage the agencies to adopt overall programmatic environmental reviews based on the principles of adaptive management. Agencies should be able to build (or tier) on these programmatic reviews to respond quickly to individual events as needed. However, the programmatic reviews should allow the agency to build in the monitoring, replication, and variance of treatments that are essential for successful adaptive management [142].

8. Conclusions

The manner in which policy makers have accepted beetle timber harvest treatments as a panacea for responding to bark beetle outbreaks in North American forests raises a number of red flags. As ecosystems and places that have economic, social, and cultural value to human communities are altered by climate change, there is a risk that people will overreact because of a need to "do something" to respond to change, and to give themselves some sense of control over broader forces that appear to be out of control. That pressure, to "do something", might also interact with the uncertainty about which choices are effective and appropriate (as with beetle timber harvest treatments) to create an opportunity for political pressures to force the adoption of particular choices that benefit specific interest groups [143]. It is perhaps no accident that the beetle treatments that have been most aggressively pushed for in the political landscape allow for logging activities that might provide revenue and jobs for the commercial timber industry. The result is that the push to "do something," uncertainty, and political pressures might lead us to act to respond to climate change before we understand the consequences of what we are doing, in the end producing more harm than good.

Our argument here is not to forgo management, but rather that management should be led by science and informed by monitoring. Both direct and indirect management for bark beetles have their place. However, to manage our forests in a way that best ensures their long-term function while wisely using limited financial resources, policy makers and the public need a clearer understanding of current science and gaps.

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Conflicts of Interest

The authors declare no conflict of interest.

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Chapter 10

c0050
Carbon Dynamics of Mixed- and High-Severity Wildfires: Pyrogenic CO_2 Emissions, Postfire Carbon Balance, and Succession

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s0010
10.1 MIXED-SEVERITY FIRES: A DIVERSITY OF FUELS, ENVIRONMENTS, AND FIRE BEHAVIORS

p0010 Recent increases in global temperatures are projected by some research to increase the frequency and severity of wildfires in certain regions, particularly those experiencing warmer, drier summers (McKenzie et al., 2004; Flannigan et al., 2006). While the annual area burned in most forests of western North America remains well below historical levels (see Chapter 9), many areas have experienced significant increases in annual burning, particularly from 1970 to 1986 (Westerling et al., 2006), prompting concerns about the additional release of carbon, primarily in the form of carbon dioxide. However, concerns over a positive feedback between wildfire-caused carbon emissions and temperature increase must be considered in the context of the physical magnitudes of pyrogenic carbon emissions and the respective constituents of forest carbon storage from which they are derived. Here I discuss the factors influencing the combustion of different constituents of forest carbon storage and how rates of fuel combustion vary among fires of low, medium, and high severity. This chapter also addresses the relationship of fuel reduction treatments with regard to reducing fire severity and carbon emissions at the potential expense of forest carbon storage. Finally, I discuss postfire carbon emissions from the decomposition of fire-killed biomass, postfire forest succession, and the eventual recovery of forest carbon storage.

The Ecological Importance of Mixed-Severity Fire: Nature's Phoenix.

p0015 Rates of pyrogenic carbon emission from wildfires can be highly variable among mixed-severity wildfires. The consumption of each respective component of forest fuel is strongly determined by individual particle geometry, often expressed as the surface area-to-volume ratio for the purposes of quantifying the amount of fuel that is likely to be consumed. Combustion generally occurs at the surface of the fuel particle, and the size of each particle and its surface area-to-volume ratio control the amount of heat required for ignition and consumption. Fuels with large surface area-to-volume ratios, such as grasses and pine needles, require less heat for ignition and combustion. Conversely, large fuels with low surface area-to-volume ratios, such as standing trees, as well as snags, downed logs, and other forms of coarse woody debris, require considerably more energy for ignition and combustion. Fuel particle size also influences the rates of moisture absorption and release, as smaller fuel particles release moisture more rapidly than larger particles in response to increasing atmospheric vapor pressure deficits, as well as in response to the thermal energy brought about by an approaching flaming front. Consequently, large fuels are much more likely to burn during the smoldering stage, in which the emissions of combustible gases and vapors are too low to support flaming combustion (Lobert and Warnatz, 1993).

p0020 Fuel consumption also is influenced by the compactness of the fuel bed, in part because of the two-stage process of consumption through pyrolysis and combustion. While these processes are nearly simultaneous, pyrolysis occurs first and is the heat-absorbing reaction that converts fuel elements such as cellulose into char, carbon dioxide, carbon monoxide, water vapor, highly combustible vapors and gases, and particulate matter (DeBano et al., 1998; Ward, 2001; Ottmar, 2014). Pyrolysis is followed by combustion, in which escaping hydrocarbon vapors are released from the surface of the fuels and are oxidized. Thus fuel compaction presents a tradeoff between heat transfer and oxygen diffusion. Highly compacted fuels facilitate a more efficient transfer of heat between fuel particles while limiting the diffusion of oxygen and, by extension, limiting consumption. Conversely, low fuel compaction allows for high diffusion of oxygen, albeit with a low diffusion of heat between fuel particles (Hardy et al., 2001). Fuel consumption also is influenced by the spacing, or continuity, of fuels across the forest floor (Finney et al., 2010) (Figure 10.1).

p0025 While the amount of consumption that is to be expected can be strongly determined by the fuel's physical and chemical characteristics, it is also a function of climate and topography. Regional climate exerts a top-down influence on fire frequency through seasonal patterns of temperature and precipitation (Littell et al., 2010), whereas local factors such as topography, vegetative composition, and fuel loads exert a bottom-up influence on fire behavior (Perry et al., 2011; Miller et al., 2012). Topography can influence the species composition of a forest, the composition and accumulation of fuels from a forest, and the topographically mediated content of fuel moisture. Among landscapes at elevations dominated by ponderosa pine (*Pinus ponderosa*) in eastern Oregon and Washington, white fir (*Abies concolor*) and grand fir (*Abies grandis*) are more common on north-facing slopes

f0010 **FIGURE 10.1** Aerial view of a smoke plume. *(Photo courtesy of M. Welling, Max Planck Institute for Chemistry.)*

because of the cooler and moist conditions that result from less incoming solar radiation (Cowlin et al., 1942). Stand composition and structure interact with the edaphic (pertaining to soils) moisture gradients to determine patterns of fire severity (Hessburg et al., 2000; Miller, 2003; Hessburg et al., 2004). In areas north of the Klamath Mountains in northwestern California, north-facing slopes may burn with mixed severity, whereas south-facing slopes can burn with mixed or low fire severity. However, the opposite occurs in the more xeric (dry) forests of the Klamath Mountains, wherein mixed-severity fires have historically dominated on south- and west-facing aspects, whereas low-severity fires were dominant on north- and east-facing aspects (Taylor and Skinner, 1998). Extreme weather conditions can override these effects, however, as was the case in the Biscuit Fire of 2002 in southwest Oregon; hot, dry winds from the northeast drove the fire, thereby eclipsing much of the influence of topographic positions (Thompson and Spies, 2010). Other fires with severe conditions have shown a stronger response to topographic controls, such as the Megram Fire in northern California (Jimerson and Jones, 2000).

p0030 The expected fuel consumption for a given level of fire severity is often expressed as a combustion factor (CF). A CF is the proportion of a biomass constituent that is expected to be consumed in a wildfire. CFs vary with respect to different biomass components such as live foliage, litter, stem, branches, shrubs, and soil. CFs can also vary as a function of fire severity: Lower levels of fire severity typically result in lower levels of combustion for each respective constituent of forest carbon storage. Note, however, that the use here of the term "fire severity," expressed as the proportion of mortality observed in overstory trees, can be misleading when used as a determinant of fuel combustion. Fuel combustion often ~~is~~ determined by fire intensity, a measure of energy output

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from a fire (Keeley, 2009). A fire of relatively low intensity could conceivably result in a fire of medium or even high severity if it occurred among trees with relatively low tolerance to fire. Because this is a book concerned about forest ecosystems with mixed- and high-severity fire regimes, however, we are largely dealing with ecosystems that have evolved at least some adaptations to moderate- or high-severity fire.

p0035 An improper use of a CF in estimating the carbon emissions of a given fire can produce vastly different estimates of pyrogenic carbon emissions. Worldwide, forests store about 45% of terrestrial carbon (861 ± 66 pg carbon) in soils, ~42% in above- and belowground live biomass, ~8% in dead wood, and ~5% in litter (Bonan, 2008). Given the magnitude of carbon stored in, say, dead wood, a poorly derived CF for dead wood can have a considerable impact on the resulting estimates of carbon dioxide emissions. Estimates of average pyrogenic carbon emissions for a given time period can produce a considerable range of values, some of which can be over four times higher than those of others (Wiedinmyer and Neff, 2007; Ghimire et al., 2012), in part because of methodological differences in the approaches used to estimate biomass accumulation and area burned, as well as different approaches used by different studies to obtain CFs.

p0040 Here I discuss factors controlling the combustion of different constituents of carbon storage in forest ecosystems and how these constituents can influence, and can be influenced by, different levels of fire severity in forested landscapes with mixed- and high-severity fire regimes. I also discuss the indirect impacts of wildfire through the long-term carbon emissions of fire-killed biomass and how emissions after wildfire can influence the source-sink dynamics throughout a postfire landscape.

s0015 10.2 DUFF, LITTER, AND WOODY DEBRIS COMBUSTION

p0045 Duff carbon comprises the dead organic matter found in the O_a (almost complete decomposition) through the O_e (moderate composition) horizons, whereas litter comprises the dead materials found in the O_i horizon (undecomposed plant parts) and includes small, woody fragments <0.51 cm in diameter, also known as 1 h fuels. Small, woody debris consists of particles 0.51-2.54 cm in diameter, also known as 10-h fuels. While only a small fraction of total forest carbon storage, these components of carbon storage on the forest floor often constitute the majority of combusted fuel for fires of all severities. Campbell et al. (2007) estimated that duff, litter, and small, downed, woody debris consumption constituted about 60% of direct carbon emissions in the Biscuit Fire of 2002. High rates of combustion among these components are consistent with the principle that fuels with large surface area-to-volume ratios have higher CFs than fuels with lower surface area-to-volume ratios, much of which can be attributed to the short time periods required for woody materials (1- to 10 h fuels) to dry out. Seasonal variation in fuel moisture can thus have a considerable impact on carbon emissions. Knapp et al. (2005) found that early season burns, in which fuel

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moisture was higher, left approximately five times more litter and duff unconsumed in areas where fire passed over the forest floor than late season burns.

p0050 Noting that this pool of carbon storage is destined for biogenic emission to the atmosphere in the absence of wildfires is important. Pools of litter, foliage, and small, downed wood are thought to have a mean residence time of 10-20 years (Law et al., 2001), and while a portion of this eventually transitions into more stable forms of soil carbon storage, much of it is lost through decay. Furthermore, much of the carbon stored in a pool with such high turnover should equate to a subsequent reduction in heterotrophic (requiring organic matter for food) respiration until these pools become recharged by the addition of leaf litter and small, woody debris (Campbell et al., 2007).

p0055 Because additional energy is necessary to remove water before combustion is possible, more energy is required to propagate flaming combustion in moist fuels than dry fuels (Nelson, 2001). In theory (Finney et al., 2013), as well as in some modeling studies (Hargrove et al., 2000; Miller and Urban, 2000), the probability that fire will propagate to neighboring fuels is reduced at higher fuel moisture levels. Knapp et al. (2005) found that the amount of area within the fire perimeter burned, and greater patchiness of early season burns conducted under higher fuel moisture conditions, are consistent with these model predictions. Thus the combustion of large, woody debris (1000-h fuels) can be particularly sensitive to fuel moisture. Estimates of combustion of downed, coarse, woody debris suggest that the majority of carbon contained therein will remain after the fire, with CFs of 0.04 for low- and very-low-severity fires and up to 0.08 and 0.24 for medium- and high-severity fires (Table 10.1). CFs are even lower for standing coarse, woody debris, ranging from 0.02 for low- and very-low-severity fires to 0.04 and 0.12 for medium- and high-severity fires (Table 10.1).

p0060 Interestingly, levels of fuel consumption for woody debris, duff, and litter exhibit a surprisingly high level of similarity at different levels of fire severity, even among different forest types (Table 10.2). CFs for woody debris (including all diameter classes) averaged 0.56, 0.63, and 0.79 for low-, medium-, and high-severity fires, respectively (Table 10.2). Average duff combustion (0.46) was lower than average woody debris combustion among stands burned by low-severity fires, but it was higher in stands burned by medium- and high-severity fires, with average CFs of 0.70 and 0.90, respectively (Table 10.2). The highest rates of combustion were observed in litter biomass, which had CFs of 0.68, 0.7[illegible] 0.95 for low-, medium-, and high-severity fires, respectively (Table 10.2).

s0020 10.3 LIVE FOLIAGE COMBUSTION

p0065 Estimates of live, crown foliage combustion are difficult because few studies have attempted to distinguish between crown consumption and noncombustive mortality (Wyant et al., 1986; McHugh et al., 2003; Hull Sieg et al., 2006; Campbell et al., 2007; Keyser et al., 2008). While live foliage can be consumed

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t0010 **TABLE 10.1 Constituents of Biomass Storage and Combustion Factors[a] for the 2002 Biscuit Fire in the Rogue River-Siskiyou National Forest in Southwestern Oregon**

C Storage Constituent	C Storage (kg C ha^{-1})	High CF (%)	Medium CF (%)	Low CF (%)	Very Low CF (%)
Foliage					
Large conifers	3242	0.69	0.27	0.08	0.02
Large hardwoods	1698	0.58	0.29	0.12	0.03
Small conifers	1863	0.89	0.76	0.44	0.01
Small hardwoods	417	1.00	0.80	0.50	0.00
Grass and forbs	2	1.00	0.76	0.75	0.70
Branch					
Large conifers	9858	0.05	0.02	0.00	0.00
Large hardwoods	4350	0.05	0.02	0.01	0.00
Small conifers	609	0.64	0.69	0.41	0.00
Small hardwoods	579	0.79	0.63	0.40	0.00
Bark					
Large conifers	11,199	0.20	0.06	0.03	0.01
Large hardwoods	4523	0.22	0.11	0.03	0.01
Small conifers	597	0.70	0.70	0.42	0.01
Small hardwoods	69	0.79	0.63	0.40	0.00
Bole					
Large conifers	57,419	0.00	0.00	0.00	0.00
Large hardwoods	30,748	0.00	0.00	0.00	0.00
Small conifers	288	0.61	0.68	0.40	0.00
Small hardwoods	700	0.79	0.63	0.40	0.00
Dead wood					
Large standing	5927	0.12	0.04	0.02	0.02
Small standing	1642	0.61	0.68	0.40	0.00
Large downed	9324	0.24	0.08	0.04	0.04

Continued

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TABLE 10.1 Constituents of Biomass Storage and Combustion Factors for the 2002 Biscuit Fire in the Rogue River-Siskiyou National Forest in Southwestern Oregon—Cont'd

C Storage Constituent	C Storage (kg C ha^{-1})	High CF (%)	Medium CF (%)	Low CF (%)	Very Low CF (%)
Medium downed	1798	0.79	0.73	0.67	0.62
Small downed	1543	0.78	0.58	0.61	0.62
Forest floor and soil					
Litter	9499	1.00	0.76	0.75	0.70
Duff	6335	0.99	0.51	0.54	0.44
Soil to 10 cm	45,500	0.08	0.04	0.04	0.02

Litter consists of materials in the O_i horizon, and duff is in the O_e and O_a horizon. Soil is all mineral soil to a depth of 10 cm, including fine roots. For live trees, small is a <7.62 cm diameter at breast height (DBH); large is a >7.62 cm DBH. For dead wood, small is 0.51-2.54 cm, medium is 2.54-7.62 cm, and large is a >7.62 cm diameter.

np0010 [a]*Data from Campbell et al. (2007).*

by wildfires, foliage can also be scorched and damaged by direct contact with or indirect convective heating from flames, leading a yellowing or browning of foliage. Once scorched, the foliage is usually killed and subsequently falls to the ground.

p0070 Understory and shrub-layer vegetation can have a significant impact on foliage consumption, but these effects depend on species composition. In the 2002 Biscuit Fire, open conifer forests with a predominantly sclerophyllous (trees and shrubs with hard, thick leaves) shrub understory experienced the most crown mortality (Thompson and Spies, 2009). Conversely, an assessment of the foliar moisture content of several grass and nonsclerophyllous shrub species suggested the possibility that the presence of a grass and/or shrub in the understory could reduce flame height throughout most of the fire season (Agee et al., 2002). If true under field conditions of fire ignition and development, such a finding would suggest a possible caveat to the common assumption that fuels with high surface area-to-volume ratios are among the most combustible and efficiently burning fuel types. The abundance of foliage fuel found throughout densely stocked, uniform forests, however, clearly has a high probability of combustion capable of propagating fires with high subsequent mortality. In a mixed conifer system in the Sierra Nevada range, North and Hurteau (2011) examined the effects of "thin from below" treatments, in which trees of a given diameter are removed to minimize the presence of ladder fuels that could t0015 **TABLE 10.2** ~~Mortality Factors (MFs) and~~ Combustion Factors (CFs) for Woody Debris (WD), Litter, and Duff Fuels for Different Forest Species Groups and Levels of Fire Severity[a]

	Low Severity			Medium Severity			High Severity		
Dominant Vegetation	*WD CF*	*Litter CF*	*Duff CF*	*WD CF*	*Litter CF*	*Duff CF*	*WD CF*	*Litter CF*	*Duff CF*
Pinyon/juniper	0.56	0.63	0.48	0.62	0.77	0.77	0.81	0.97	0.97
Douglas-fir	0.53	0.70	0.47	0.60	0.73	0.81	0.81	0.97	0.97
Ponderosa pine	0.52	0.65	0.54	0.65	0.75	0.84	0.82	0.96	0.97
Fir/spruce/mountain hemlock	0.53	0.60	0.44	0.63	0.76	0.69	0.77	0.92	0.83
Lodgepole pine	0.68	0.50	0.21	0.77	0.56	0.33	0.96	0.72	0.42
Hemlock/sitka spruce	0.59	0.75	0.54	0.58	0.76	0.51	0.77	1.00	0.99
California mixed conifer	0.56	0.64	0.48	0.62	0.77	0.77	0.80	0.97	0.97
Elm/ash/cottonwood	0.58	0.75	0.51	0.66	0.74	0.77	0.77	1.00	0.99
Aspen/birch	0.43	0.77	0.40	0.48	0.74	0.64	0.60	1.00	0.81
Western oak	0.56	0.76	0.50	0.67	0.80	0.79	0.81	0.98	0.95
Tanoak/laurel	0.59	0.75	0.54	0.68	0.66	0.79	0.77	1.00	0.99

np0015 [a]*Data from Ghimire et al. (2012).*

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propagate a crown fire. Following wildfire, differences in fire mortality between treated (53%) and untreated (97%) forest suggest that fuel reduction treatments can allow for a considerable reduction in the presence of foliage and ladder fuels throughout the stand, though this did not include the effects of direct mortality from the mechanical thinning itself, which would substantially increase overall mortality in the thinned areas.

p0075 The potential for fire to spread vertically to the forest canopy is highly dependent upon the successional stage of the forest stand. As densely stocked stands of shade-intolerant species mature, self-thinning raises the crown height, and the resulting shading discourages the development of ladder fuels, thereby reducing the probability of fire propagation from the ground fuels into the canopy (Odion et al., 2004; Perry et al., 2011). Collins and Stephens (2010) found that stands were most susceptible to high-severity reburn when they were between 17 and 30 years old (also see Chapter 1). Consequently, mature, closed conifer stands can be more resistant to foliage combustion and tree morality than their younger counterparts (Thompson and Spies, 2009, 2010). These findings bear relevance to the commonly held assumption that the probability of high severity fires tends to increase with stand age. Such assumption is often made on the premise that forests accumulate more biomass through time, and thus have more total fuel that could be burned, thereby resulting in fires of higher severity. However, the infrequent occurrence of high-severity wildfires is not necessarily the result of infrequently high amounts of forest fuel availability. For many ecosystems, it is the infrequent occurrence of extreme weather conditions that may lead to a high-severity, foliage-consuming crown fire (Perry et al., 2011).

p0080 Foliage combustion rates may thus be best thought of as a function of fire severity and the vertical strata of the foliage. CFs for grass and forbs range from 0.70 to 0.75 in very-low-/low-severity fires to 1.00 in high-severity fires, whereas the combustion of fuels of small (<7.62 cm diameter at breast height [dbh]) trees and shrubs at a slightly higher vertical strata is slightly less: CFs for low-, medium-, and high-severity fires are 0.44, 0.76, and 0.89 for conifers and 0.50, 0.80, and 1.00 for hardwoods, respectively. Estimated CFs for the foliage of large trees are, as expected, lower than the others because of the vertical distance between foliage and surface fuels, where the majority of combustion takes place. CFs for large (>7.62 cm dbh) foliage in low-, moderate- and high-severity fires are 0.09, 0.27, and 0.69 for conifers and 0.12, 0.29, and 0.58 for hardwoods, respectively (Table 10.1).

s0025 10.4 SOIL COMBUSTION

p0085 Soil represents a considerable fraction of forest carbon, comprising approximately 44% of total forest carbon storage worldwide (Bonan, 2008). Soil carbon storage is usually low among ecosystems with frequent, low-severity fire regimes, such as those found in semiarid ponderosa pine forests. Conversely,

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soil carbon storage can be very high in ecosystems with infrequent (i.e., a mean fire return interval of >200 years) fires. Fires of high intensity and severity typify many forests with infrequent fire regimes. Because of the high magnitude of soil carbon storage in stands with infrequent, high-severity fires, estimates of carbon emissions from wildland fires are highly sensitive to the CF used to estimate the proportion of soil carbon that is consumed. However, estimates of soil carbon combustion are difficult to obtain, particularly in high-severity wildland fires, because of the lack of prefire estimates of soil carbon content.

p0090 The process of soil carbon consumption is dominated by smoldering, as opposed to flaming, combustion. Smoldering combustion is a result of insufficient amounts of oxygen required to support flaming combustion and is most prevalent in organic soils and rotting logs. The combustion of forest soils is highly dependent on the magnitude of the temperatures they are exposed to and the duration of exposure. Agee (1993) suggested that soils can be combusted at temperatures as low as 100 °C, but laboratory-based experiments suggest that significant amounts of soil carbon volatilization require temperatures between 200 °C and 315 °C (Lide, 2004), with peak smoldering temperatures ranging from 300 °C to 600 °C (Rein et al., 2008). Work by Fernández et al. (1997) heated the top 10 cm of soil taken from a *Pinus sylvestris* stand to 150° at a gradually increasing rate (+3 °C min^{-1}), at which point the soil was heated for 30 min thereafter, yet no significant amount of soil carbon combustion was observed. Upon applying the same heating regime at temperatures of 220°, 350°, and 490°C, however, there were significant changes in the content of soil organic matter (i.e., soil carbon). Temperatures of 220°, 350°, and 490°C resulted in losses of 37%, 90%, and nearly 100%, respectively. Others have noted that shorter heating times at 350 °C resulted in a 50% weight loss after only 180 s (Almendros et al., 2003), compared with 90% at 350 °C observed by Fernández et al. (1997). Consequently, exposure to increased temperatures is highly dependent on combustion times and rates of fire spread; the relatively high rates at which fire moves across western North American landscapes, combined with the relatively limited diffusion of oxygen into the relatively nonporous soil profile, limit soil carbon emissions. CFs for soils described by Campbell et al. (2007) were 0.04 for low- and medium-severity fires and 0.08 for high-severity fires (Table 10.1).

p0095 The combustion of soils in boreal forests represents an important exception to the relatively low rates of soil carbon emissions observed in most western US forests. Turetsky et al. (2011) and Kasischke and Hoy (2012) found that the combustion of soil carbon in Alaskan boreal forests can actually constitute the majority of carbon emissions during fires, representing 54-70% of total carbon emissions. Turetsky et al. found that three factors explained most of the variation in the depth of burning/carbon consumption in the surface organic layers of black spruce forests. First, topography was a significant control: Higher fractions of consumption were observed in upland sites compared with lowland sites. Second, season of the fire was also a

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factor: Seasonal thawing of permafrost resulted in drier ground layers as the growing season progressed. Finally, in upland sites, fires that exhibited higher consumption occurred in the early season in years where fires had a large spatial extent compared to those in years where fires had a smaller spatial extent because of drier conditions and more extreme fire behavior.

p0100 Large amounts of biomass with long-term smoldering potential also are found in pocosin shrublands (a type of wetland with deep, sandy, and acidic soils) in the southeastern United States. While pocosin systems can have substantial amounts of combustible fuel contained in deep peat layers, they differ most notably from boreal forests in their lack of both a freeze-thaw cycle and a strong, seasonally sensitive decline in moisture as the growing season progresses. Consumption of fuel beds in these systems is poorly understood, and additional research on moisture dynamics, biogeochemical processes, and combustion is needed (Reardon et al., 2007, 2009).

s0030 10.5 BOLE BIOMASS CONSUMPTION

p0105 While many studies report tree mortality rates, relatively little on the fraction of fire-killed trees that were combusted during wildfire has been reported. In estimates of pyrogenic carbon emissions taken from the Biscuit Fire in 2002, Campbell et al. (2007) found no combustion of bole biomass among large (>7.62 cm dbh) trees, regardless of fire severity (Table 10.1). The lack of combustion for the boles of large trees seems to have been effectively mediated by the combustion of bark, which had CFs of 0.03, 0.06, and 0.20 for conifers and 0.03, 0.11, and 0.22 for hardwoods in low-, medium-, and high-severity fires, respectively. Such a finding is consistent with what is expected of fuels with low surface area-to-volume ratios (Table 10.1).

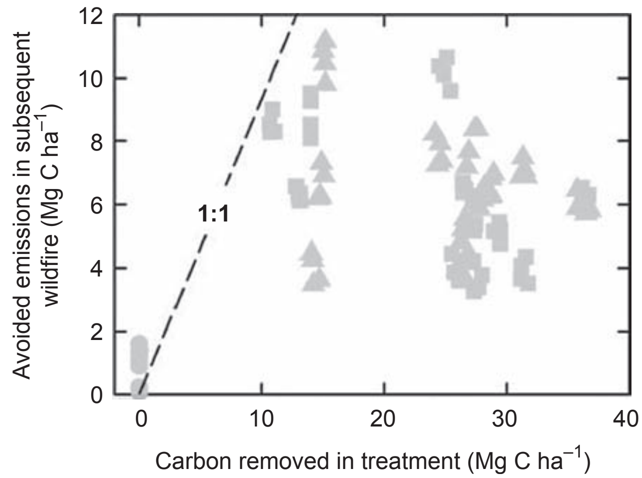
p0110 Bark CFs were much higher for small trees; for low-, medium-, and high-severity fires there were CFs of 0.42, 0.70, and 0.70 for conifers and 0.40, 0.63, and 0.79 for hardwoods, respectively (Table 10.1). As expected, the thinner bark of smaller trees, much of which was combusted, was not effective in protecting the bole biomass from combustion. Estimates of the combustion of the boles of small trees for low-, medium-, and high-severity fires were 0.40, 0.68, and 0.61 for conifers and from 0.40, 0.63, and 0.79 for hardwoods, respectively (Table 10.1). Weighted CFs for all trees, adjusted for the abundance of small tree biomass versus large tree biomass, would be approximately 0.03, 0.07, and 0.08 for low-, medium-, and high-severity fires, respectively (Campbell et al., 2007). Others have used far higher CFs for high-severity fires in modeling studies. An estimated high-severity CF of 0.30 has been used for Siberian forests (Soja et al., 2004), which may be realistic, given the small diameters prevalent in boreal forest stands. Estimates of bole CFs, however, some of which are as high as 0.30 for North American forests (Wiedinmyer et al., 2006), seem to be at odds with those estimated by Campbell et al.

(2007), given the majority of biomass is stored in boles of large trees, none of which is combusted by high-severity fires. Such estimates, if inaccurate, can result in substantial overestimates of pyrogenic carbon emissions because of the considerable stocks of carbon in bole biomass of large trees. Overall, the CFs for total forest biomass (i.e., trees, snags, shrubs, woody fuels, litter, duff, and soil), weighted according to their respective prefire biomass, were 0.13, 0.15, and 0.21 for low-, medium-, and high-severity fires, respectively, in the Biscuit Fire (Campbell et al., 2007) (Table 10.1).

s0035 10.6 FUEL REDUCTION TREATMENTS, CARBON EMISSIONS, AND LONG-TERM CARBON STORAGE

p0115 The application of fuel reduction treatments have become common in many fire-adapted forests throughout the western North America. Such treatments are intended to reduce the severity of fires, primarily out of concern over public safety in fire-prone regions, as well as ~~because of land management agencies that want~~ to minimize ~~widespread mortality in the forests within their jurisdiction~~. ~~Common fuel~~ reduction treatments ~~include~~ understory removal, whereby midstory and understory vegetation ~~is~~ removed through pruning or harvesting. ~~Another fuel reduction treatment is~~ prescribed fire, which reduces surface fuels in order to limit the flame height of a wildfire that might enter the stand. ~~In the field, this~~ is done by removing fuel through prescribed fire or pile burning, both of which reduce the potential magnitude of a wildfire by making it more difficult for a surface fire to ignite the canopy. The timing of prescribed fire can be central to its effectiveness. If performed *after* an understory removal treatment, it may burn any additional residue created by the treatment. ~~Performing~~ prescribed fire under cooler and moisture conditions than those experienced during the fire season is also ideal to avoid the propagation of an unplanned fire. Other fuel reduction treatments involve a partial harvest of overstory trees to limit the potential of fire to spread from crown to crown.

p0120 While such treatments can sometimes be effective in reducing fire severity, if and when fires occur in thinned areas (Rhodes and Baker, 2008), they can come at the expense of carbon storage. The majority of carbon stored in leaves, leaf litter, and duff is typically consumed by high-severity wildfire and often constitutes the majority of the carbon emissions during the a given fire, yet most of the carbon stored in forest biomass (stem wood, branches, and coarse, woody debris) remains unconsumed even by high-severity wildfires. Consequently, fuel removal via forest thinning almost always reduces carbon storage more than the additional carbon that a stand is able to store when made more resistant to wildfire. For this reason, removing large amounts of biomass to reduce the fraction by which other biomass components are consumed via combustion is inefficient (Mitchell et al., 2009). Fuel reduction treatments that involve the removal of overstory biomass (i.e., intermediate-sized and large trees) are, perhaps unsurprisingly, the most inefficient methods of reducing wildfire-related



f0015 **FIGURE 10.2** Simulated effectiveness of various fuel-reduction treatments in reducing future wildfire combustion in a ponderosa pine ~~(*P. ponderosa*)~~ forest. In general, protecting one unit of carbon (C) from wildfire combustion came at the cost of removing approximately three units of carbon in treatment. At the very lowest (least biomass removed) treatment levels, more carbon was protected from combustion than removed during treatment; however, the absolute gains were extremely low. Circles show understory removal, squares show prescribed fire, and triangles show understory removal and prescribed fire. Simulations were run for 800 years with a treatment-return interval of 10 years and a mean fire-return interval of 16 years. Forest structure and growth were modeled to represent mature, semiarid ponderosa pine forest growing in Deschutes, Oregon. Further descriptions of these simulations are given by Mitchell et al. (2009).

carbon losses because they remove large amounts of carbon for only a marginal reduction in expected fire severity (Figure 10.2).

s0040 10.7 INDIRECT SOURCES OF CARBON EMISSIONS

p0125 Our discussion thus far has focused on the *direct* effects of wildfire on carbon emissions as a result of the combustion of live vegetation, dead biomass, and soil organic matter. *Indirect* effects, by contrast, are not the result of the active combustion of biomass or soil organic matter; instead, they result from the long-term decomposition of vegetation killed in wildfire. The magnitude of indirect emissions, and the temporal scales at which they affect the net ecosystem carbon balance, vary with different fire behaviors. Most of the mortality resulting from low-severity fires is limited to understory plants, shrubs, and small trees, which do not typically constitute a significant portion of total stand carbon storage and, by extension, do not represent a significant source of carbon emissions upon decomposition. High-severity fires, by contrast, result in the near-total death of all trees within a stand, including overstory dominants. While the addition of any unburned leaf litter and fine, woody debris from fire-killed trees represent pools with relatively high turnover (10-20 years), a large pool of coarse woody debris (e.g., logs, snags) can be a significant source of carbon emissions (Bond-Lamberty et al., 2003), one that can ~~continue to release carbon~~ for periods of up to, and even exceeding, 100 years (Kashian et al., 2006).

p0130 Fire severity has a significant impact on postdisturbance rates of net primary production and net ecosystem production (NEP). Net primary production is the difference between photosynthesis and autotrophic (i.e., plant) respiration, whereas NEP is a measure of net ecosystem carbon uptake, defined as the difference between photosynthesis and autotrophic respiration plus heterotrophic (i.e., decomposition) respiration. Following a high-severity disturbance, rates of heterotrophic respiration are, for a period of time, far higher than rates of photosynthesis, resulting in negative NEP (Harmon et al., 2011). While indirect sources of carbon emissions following fire can be substantial, particularly following high-severity fire, the postdisturbance regrowth of a new cohort of trees is also a significant contributor to total ecosystem carbon storage and the net ecosystem carbon balance (Figure 10.3).

p0135 The amount of time required for a recently disturbed forest to shift from a source to a sink depends on fire severity, forest type, and local climate. Following high-severity wildfires, forests with low rates of productivity, such as the ponderosa pine forests of the southwestern United States, take relatively longer to make the postfire transition from carbon source to carbon sink (Ghimire et al., 2012). Dore et al. (2008) examined a ponderosa pine forest in northern Arizona 10 years after a stand-replacing fire and found it to be a moderate source of carbon (109 g carbon m^{-2} $year^{-1}$), but they observed a moderate carbon sink (164 g carbon m^{-2} $year^{-1}$) in an unburned stand nearby. The burned stand remained a source of carbon during all months of the year that were measured, even during the growing season in the summer months. Annual ecosystem respiration was 33% lower in the burned stand. The slow recovery of such stands is largely attributed to the climate, whereby cold winters combine with low springtime precipitation to limit gross primary production (GPP), whereas warm summers with periodic precipitation are conducive to respiration-driven losses of soil carbon (Dore et al., 2008). However, this analysis was based on only five plots with a 25 m radius; therefore, some caution regarding broader inferences is appropriate.

p0140 Differences in the postfire carbon balance of uptake were observed in semiarid, mixed-conifer forests of eastern Oregon. Meigs et al. (2009) found that 4-5 years after a mixed-severity fire, areas that burned at low severity were modest net carbon sinks. By contrast, ponderosa pine forests that also were affected by a low-severity fire were carbon neutral in low-severity fire areas. Differences in the recovery time to being a source of carbon once again may be because of differences in productivity; ponderosa pine forests are typically less productive than mixed-conifer forests (Franklin and Dyrness, 1973). Among areas affected by high severity fires, both ponderosa pine and mixed-conifer stands were sources of carbon emissions 4-5 years following fire. Modeled estimates of the postfire transition from carbon source to carbon sink suggest that ~40 years may necessary for low-productivity ponderosa pine forests to shift

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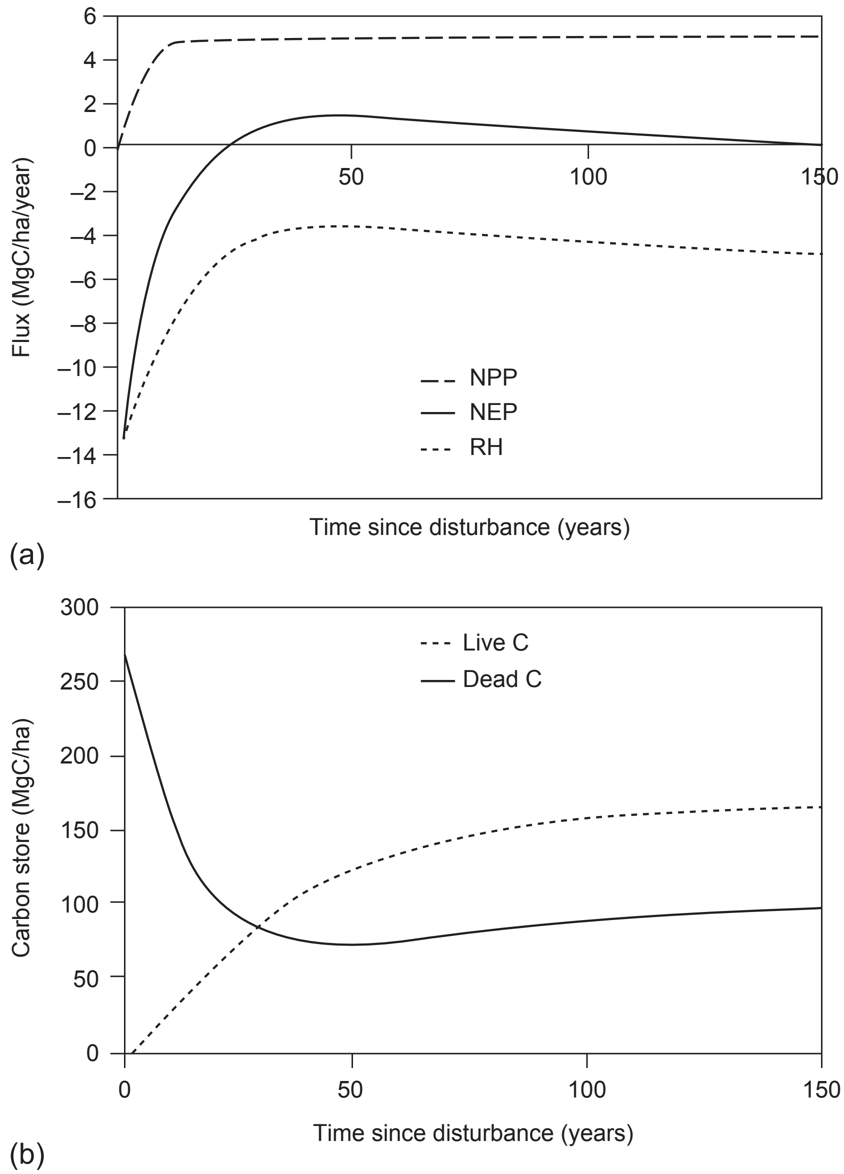

f0020 **FIGURE 10.3** The classic pattern of net primary production (NPP), heterotrophic respiration (RH), and net ecosystem production (NEP) (A) and associated carbon stores (B) following a high-severity disturbance. *(From Harmon et al. (2011).)*

from being a carbon source to a carbon sink (Ghimire et al., 2012), though this analysis did not control for the potentially confounding effect of postfire logging, which is common after high-severity fire in ponderosa pine and mixed-conifer forests (see Chapter 11). Forests with higher rates of productivity, such as coastal range Sitka spruce (*Picea sitchensis*)/western hemlock (*Tsuga* *heterophylla*) forests in the Pacific Northwest, seem to make the postfire transition from carbon source to carbon sink in a shorter amount of time than any other coniferous western forest, potentially in <30 years. Harmon et al. (2011) reviewed the scientific literature on this question for various forest types and concluded that the transition from source to sink following fire sufficiently severe to reset the successional "clock" varied from 14-50 years in forests types characteristic of the Pacific northwestern United States and 5-15 years in boreal forests. High-severity fire rotation intervals are currently several hundred years to more than 1000 years in most mixed-conifer and ponderosa pine forest regions of the western United States, however, and these rates are generally substantially lower than historical rates (see Chapter 1). Thus a long-term spatio-temporal perspective is important to understand more fully the natural disturbance dynamics in these systems (see Chapter 9).

s0045 10.8 CONCLUSIONS

p0145 The majority of carbon stored in montane forest ecosystems of western North America remains unconsumed, even in high-severity wildfires. Large carbon stores in the bole biomass of large forest trees are not consumed, and the substantial proportion of carbon stored in forest soils is only slightly consumed. Most of the carbon emissions in a wildfire are from combustion of litter, duff, and woody debris. In the 2002 Biscuit Fire, CFs for total forest biomass (i.e., trees, snags, shrubs, woody fuels, litter, duff, and soil), weighted according to their respective prefire biomass, were 0.13, 0.15, and 0.21 for low-, medium-, and high-severity fires, respectively. Such factors can be even lower among stands with a higher proportion of carbon storage in bole biomass that likewise remains unconsumed in high-severity wildfires, such as Sitka spruce (*P. sitchensis*)/Western Hemlock (*T. heterophylla*) forests in the coast range of the Pacific Northwest (Smithwick et al., 2002; Mitchell et al., 2009). The application of fuel treatments can be effective in reducing fire severity and carbon emissions, but such treatments come at the cost of a net reduction in carbon storage relative to fire alone (Mitchell et al., 2009).

p0150 Postfire carbon emissions from fire-killed biomass can be substantial for decades following wildfires. Low- or even moderate-severity fires, however, do not necessarily result in a postfire source of carbon released to the atmosphere. High-severity fire temporarily creates a source of postfire carbon emissions as a result of the decomposition of fire-killed biomass, which lessens each year with natural postfire succession of vegetation, transitioning from a carbon source to a carbon sink within 5-50 years, depending on the ecosystem. Rates of postfire recovery are highest among systems with high productivity, whereas high-severity wildfires in forests with low productivity transition from source to sink over a relatively longer timeline, though there are important limitations in the amount and scope of existing studies of these systems. ~~Future~~ research on the relationship between climatic change, disturbance regimes, and postdisturbance successional trajectories may prove to be a crucial step toward projecting the future of pyrogenic carbon emissions in mixed-severity fire regimes.

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Non-Print Items

Abstract

Among the concerns raised by climatic change is the potential for the additional release of carbon dioxide as a result of biomass combustion. Most of the carbon emissions from wildfires are from the combustion of litter, duff, and small woody debris, whereas most, if not all, of the biomass stored in the boles of large trees is not combusted. Consequently, most of the carbon stored in forests remains unconsumed, even by high-severity wildfires. Thus the application of fuel reduction treatments, while sometimes effective in reducing fire severity and carbon emissions, nearly always result in a net reduction in carbon storage. Postfire carbon emissions from the decomposition of fire-killed biomass can continue for decades, but effects of forest regrowth can exceed the losses of carbon from biomass combustion and the decomposition of fire-killed biomass within 5-50 years, depending on the ecosystem.

Keywords: Carbon sequestration; Carbon emissions; Climate change; Fuel reduction treatments; Biscuit Fire; Combustion factor; Carbon dioxide.

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Forest fuel reduction alters fire severity and long-term carbon storage in three Pacific Northwest ecosystems

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Abstract. Two forest management objectives being debated in the context of federally managed landscapes in the U.S. Pacific Northwest involve a perceived trade-off between fire restoration and carbon sequestration. The former strategy would reduce fuel (and therefore C) that has accumulated through a century of fire suppression and exclusion which has led to extreme fire risk in some areas. The latter strategy would manage forests for enhanced C sequestration as a method of reducing atmospheric CO_2 and associated threats from global climate change. We explored the trade-off between these two strategies by employing a forest ecosystem simulation model, STANDCARB, to examine the effects of fuel reduction on fire severity and the resulting long-term C dynamics among three Pacific Northwest ecosystems: the east Cascades ponderosa pine forests, the west Cascades western hemlock–Douglas-fir forests, and the Coast Range western hemlock–Sitka spruce forests. Our simulations indicate that fuel reduction treatments in these ecosystems consistently reduced fire severity. However, reducing the fraction by which C is lost in a wildfire requires the removal of a much greater amount of C, since most of the C stored in forest biomass (stem wood, branches, coarse woody debris) remains unconsumed even by high-severity wildfires. For this reason, all of the fuel reduction treatments simulated for the west Cascades and Coast Range ecosystems as well as most of the treatments simulated for the east Cascades resulted in a reduced mean stand C storage. One suggested method of compensating for such losses in C storage is to utilize C harvested in fuel reduction treatments as biofuels. Our analysis indicates that this will not be an effective strategy in the west Cascades and Coast Range over the next 100 years. We suggest that forest management plans aimed solely at ameliorating increases in atmospheric CO_2 should forgo fuel reduction treatments in these ecosystems, with the possible exception of some east Cascades ponderosa pine stands with uncharacteristic levels of understory fuel accumulation. Balancing a demand for maximal landscape C storage with the demand for reduced wildfire severity will likely require treatments to be applied strategically throughout the landscape rather than indiscriminately treating all stands.

Key words: biofuels; carbon sequestration; fire ecology; fuel reduction treatment; Pacific Northwest, USA; Picea sitchensis; Pinus ponderosa; Pseudotsuga menziesii.

Introduction

Forests of the U.S. Pacific Northwest capture and store large amounts of atmospheric CO_2, and thus help mitigate the continuing climatic changes that result from extensive combustion of fossil fuels. However, wildfire is an integral component to these ecosystems and releases a substantial amount of CO_2 back to the atmosphere via biomass combustion. Some ecosystems have experienced an increase in the amount of CO_2 released due to a century-long policy of fire suppression that has led to increased levels of fuel buildup, resulting in wildfires of uncharacteristic severity. Fuel reduction treatments have been proposed to reduce wildfire severity, but like wildfire, these treatments also reduce the C stored in forests. Our work examines the effects of fuel reduction on wildfire severity and long-term C storage to gauge the strength of the potential trade-off between managing forests for increased C storage and reduced wildfire severity.

Forests have long been referenced as a potential sink for atmospheric CO_2 (Vitousek 1991, Turner et al. 1995, Harmon et al. 1996, Harmon 2001, Smithwick et al. 2002, Pacala and Socolow 2004), and are credited with contributing to much of the current C sink in the coterminous United States (Pacala et al. 2001, Hurtt et al. 2002). This U.S. carbon sink has been estimated to be between 0.30 and 0.58 Pg C/yr for the 1980s, of which between 0.17 Pg C/yr and 0.37 Pg C/yr has been attributed to accumulation by forest ecosystems (Pacala et al. 2001). While the presence of such a large sink has been valuable in mitigating global climate change, a substantial portion of it is due to the development of understory vegetation as a result of a national policy of fire suppression (Pacala et al. 2001, Donovan and Brown 2007). Fire suppression, while capable of incurring

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short-term climate change mitigation benefits by promoting the capture and storage of atmospheric CO_2 by understory vegetation and dead fuels (Houghton et al. 2000, Tilman et al. 2000), has, in part, led to increased and often extreme fire risk in some forests, notably *Pinus ponderosa* forests (Moeur et al. 2005, Donovan and Brown 2007).

Increased C storage usually results in an increased amount of C lost in a wildfire (Fahnestock and Agee 1983, Agee 1993). Many ecosystems show the effects of fire suppression (Schimel et al. 2001, Goodale et al. 2002, Taylor and Skinner 2003), and the potential effects of additional C storage on the severity of future wildfires is substantial. In the *Pinus ponderosa* forests of the east Cascades, for example, understory fuel development is thought to have propagated crown fires that have killed old-growth stands not normally subject to fires of high intensity (Moeur et al. 2005). Various fuel reduction treatments have been recommended for risk-prone forests, particularly a reduction in understory vegetation density, which can reduce the ladder fuels that promote such severe fires (Agee 2002, Brown et al. 2004, Agee and Skinner 2005). While a properly executed reduction in fuels could be successful in reducing forest fire severity and extent, such a treatment may be counterproductive to attempts at utilizing forests for the purpose of long-term C sequestration.

Pacific Northwest forests, particularly those that are on the west side of the Cascade mountain range, are adept at storing large amounts of C. Native long-lived conifers are able to maintain production during the rainy fall and winter months, thereby out-competing shorter-lived deciduous angiosperms with a lower biomass storage capacity (Waring and Franklin 1979). Total C storage potential, or upper bounds, of these ecosystems is estimated to be as high as 829.4 Mg C/ha and 1127.0 Mg C/ha for the western Cascades and Coast Range of Oregon, respectively (Smithwick et al. 2002). Of this high storage capacity for west Cascades and Coast Range forests, 432.8 Mg C/ha and 466.3 Mg C/ha, respectively, are stored in aboveground biomass (Smithwick et al. 2002), a substantial amount of fuel for wildfires.

High amounts of wildfire-caused C loss often reflect high amounts of forest fuel availability prior to the onset of fire. Given the magnitude of such losses, it is clear that the effect of wildfire severity on long-term C dynamics is central to our understanding of the global C cycle. What is not clear is the extent to which repeated fuel removals that are intended to reduce wildfire severity will likewise reduce long-term total ecosystem C storage (TECμ). Fuel reduction treatments require the removal of woody and detrital materials to reduce future wildfire severity. Such treatments can be effective in reducing future wildfire severity, but they likewise involve a reduction in stand-level C storage. If repeated fuel reduction treatments decrease the mean total ecosystem C storage by a quantity that is greater than the difference between the wildfire-caused C loss in an untreated stand and the wildfire-caused C loss in a treated stand, the ecosystem will not have been effectively managed for maximal long-term C storage.

Our goal was to test the extent to which a reduction in forest fuels will affect fire severity and long-term C storage by employing a test of such dynamics at multi-century time scales. Our questions were as follows: (1) To what degree will reductions in fuel load result in decreases in C stores at the stand level? (2) How much C must be removed to make a significant reduction in the amount of C lost in a wildfire? (3) Can forests be managed for both a reduction in fire severity and increased C sequestration, or are these goals mutually exclusive?

Methods

Model description

We conducted our study using an ecosystem simulation model, STANDCARB (Appendix A), that allows for the integration of many forest management practices as well as the ensuing gap dynamics that may result from such practices. STANDCARB is a forest ecosystem simulation model that acts as a hybrid between traditional single-life-form ecosystem models and multi-life-form gap models (Harmon and Marks 2002). The model integrates climate-driven growth and decomposition processes with species-specific rates of senescence and stochastic mortality while incorporating the dynamics of inter- and intraspecific competition that characterize forest gap dynamics. Inter- and intraspecific competition dynamics are accounted for by modeling species-specific responses to solar radiation as a function of each species' light compensation point as well as the amount of solar radiation delineated through the forest canopy to each individual. By incorporating these processes the model can simulate successional changes in population structure and community composition without neglecting the associated changes in ecosystem processes that result from species-specific rates of growth, senescence, mortality, and decomposition.

STANDCARB performs calculations on a monthly time step and can operate at a range of spatial scales by allowing a multi-cell grid to capture multiple spatial extents, as both the size of an individual cell and the number of cells in a given grid can be designated by the user. We used a 20×20 cell matrix for all simulations (400 cells total), with 15×15 m cells for forests of the west Cascades and Coast Range and 12×12 m cells for forests of the east Cascades. Each cell allows for interactions of four distinct vegetation layers, represented as upper canopy trees, lower canopy trees, a species-nonspecific shrub layer, and a species-nonspecific herb layer. Each respective vegetation layer can have up to seven live pools, eight detrital pools, and three stable C pools. For example, the upper and lower tree layers comprise seven live pools: foliage, fine roots, branches, sapwood, heartwood, coarse roots, and heart-rot, all of which are transferred to a detrital pool following mortality. Dead wood is separated into snags and logs to capture the effects of spatial position on microclimate. After detrital materials have undergone significant decomposition, they can contribute material to three increasingly decay-resistant, stable C pools: stable foliage, stable wood, and stable soil. Charcoal is created in both prescribed fires and wildfires and is thereafter placed in a separate pool with high decay resistance. Additional details on the STANDCARB model can be found in Appendix A.

Fire processes

We generated exponential random variables to assign the years of fire occurrence (sensu Van Wagner 1978) based on the literature estimates (see experimental design for citations) of mean fire return intervals (MFRI) for different regions in the U.S. Pacific Northwest. The cumulative distribution for our negative exponential function is given in Eq. 1 where X is a continuous random variable defined for all possible numbers x in the probability function P, and λ represents the inverse of the expected time $E[X]$ for a fire return interval given in Eq. 2:

$$P\{X \leq x\} = \int_0^x \lambda e^{-\lambda x} dx \quad (1)$$

where

$$E[X] = \frac{1}{\lambda}. \quad (2)$$

Fire severities in each year generated by this function are cell specific, as each cell is assigned a weighted fuel index calculated from fuel accumulation within that cell and the respective flammability of each fuel component, the latter of which is derived from estimates of wildfire-caused biomass consumption (see Fahnestock and Agee 1983, Covington and Sackett 1984, Agee 1993). Fires can increase (or decrease) in severity depending on how much the weighted fuel index of a given cell exceeds (or falls short of) the fuel level thresholds for each fire severity class (T_{light}, T_{medium}, T_{high}, and $T_{\max}$), and the probability values for the increase or decrease in fire severity (P_{i} and P_{d}). For example, while the natural fire severity of many stands of the west Cascades can be described as high severity, other stands of the west Cascades have a natural fire severity that can be best described as being of medium severity (~60–80% overstory tree mortality) (Cissel et al. 1999). For these stands, medium-severity fires are scheduled to occur throughout the simulated stand and can increase to a high-severity fire depending on the extent to which the weighted fuel index in a cell exceeds the threshold for a high-severity fire, as greater differences between the fuel index and the fire severity threshold will increase the chance of a change in fire severity. Conversely, medium-severity fires may decrease to a low-severity fire if the fuel index is sufficiently below the threshold for a medium-severity fire. High-severity fires are likely to become medium-severity fires if the weighted fuel index within a given cell falls sufficiently short of the threshold for a high-severity fire, and low-severity fires are likely to become medium severity if the weighted fuel index in a given cell is sufficiently greater than the threshold for a medium-severity fire. Fuel level thresholds were set by monitoring fuel levels in a large series of simulation runs where fires were set at very short intervals to see how low fuel levels needed to be to create a significant decrease in expected fire severity. We note that, like fuel accumulation, the role of regional climate exerts significant influence on fire frequency and severity, and that our model does not attempt to directly model these effects. We suspect that an attempt to model the highly complex role of regional climate data on fine-scale fuel moisture, lightning-based fuel ignition, and wind-driven fire spread adds uncertainties into our model that might undermine the precision and applicability of our modeling exercise. For that reason we incorporated data from extensive fire history studies to approximate the dynamics of fire frequency and severity.

Final calculations for the expected stand fire severity $E[F_{\text{s}}]$ at each fire are performed as follows:

$$E[F_{\text{s}}] = \frac{100}{C} \sum_{i=1}^{n} c_{i(\text{L})} m_{i(\text{L})} + c_{i(\text{M})} m_{i(\text{M})} + c_{i(\text{H})} m_{i(\text{H})} \quad (3)$$

where C is the number of cells in the stand matrix and $c_{i(\text{L})}$, $c_{i(\text{M})}$, and $c_{i(\text{H})}$ are the number of cells with light, medium, and high-severity fires, and $m_{i(\text{L})}$, $m_{i(\text{M})}$, and $m_{i(\text{H})}$represent fixed mortality percentages for canopy tree species for light, medium, and high-severity fires, respectively. This calculation provides an approximation of the number of upper-canopy trees killed in the fire. The resulting expected fire severity calculation $E[F_{\text{s}}]$ is represented on a scale from 0 to 100, where a severity index of 100 indicates that all trees in the simulated stand were killed.

Our approach at modeling the effectiveness of fuel reduction treatments underscores an important trade-off between fuel reduction and long-term ecosystem C storage by incorporating the dynamics of snag creation and decomposition. Repeated fuel reduction treatments may result in a reduction in long-term C storage, but it is possible that if such treatments are effective in reducing tree mortality, they may also offset some of the C losses that would be incurred from the decomposition of snags that would be created in a wildfire of higher severity. STANDCARB accounts for these dynamics by directly linking expected fire severity with a fuel accumulation index that can be altered by fuel reduction treatments while also incorporating the decomposition of snags as well as the time required for each snag to fall following mortality.

Total ecosystem C storage (TEC) is calculated by summing all components of C (live, dead, and stable). For each replicate (i = 1, 2, … 5) and for each period

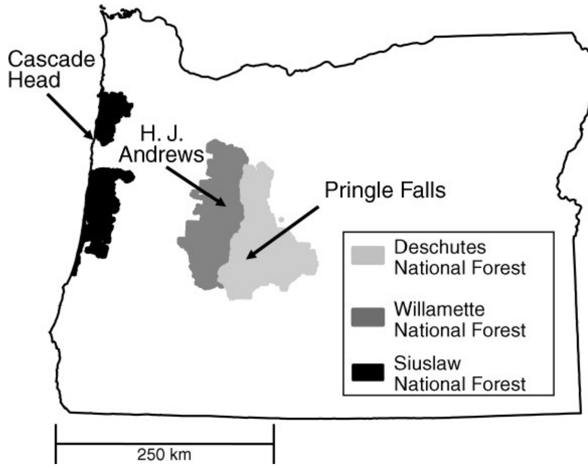

FIG. 1. Site locations in Oregon. Pringle Falls is our representative site for the east Cascades, H. J. Andrews is our representative site for the west Cascades, and Cascade Head is our representative site for the Coast Range.

between fires ($x = 1, 2, \ldots P_i$), the mean total ecosystem C storage (TEC_μ) is calculated by averaging the yearly TEC values ($k = 1, 2, \ldots R_x$).

$$\mathrm{TEC}_{\mu(i,x)} = \frac{1}{R}\sum_{k=1}^{R}\mathrm{TEC}_{(i,x,k)}.$$

Aggregating TEC_μ values in this manner permits the number of TEC_μ values to be the same as the number of $E[F_s]$ values, permitting a PerMANOVA analysis to be performed on $E[F_s]$ and TEC_μ.

Fuel reduction processes

STANDCARB's fire module allows for scheduled prescribed fires of a given severity (light, medium, high) to be simulated in addition to the nonscheduled wildfires generated from the aforementioned exponential random variable function. In addition to simulating the prescribed fire method of fuel reduction, STANDCARB has a harvest module that permits cell-by-cell harvest of trees in either the upper or lower canopy. This module allows the user to simulate understory removal or overstory thinning treatments on a cell-by-cell basis. Harvested materials can be left in the cell as detritus following cutting or can be removed from the forest, allowing the user to incorporate the residual biomass that results from harvesting practices. STANDCARB can also simulate the harvest of dead salvageable materials such as logs or snags that have not decomposed beyond the point of being salvageable.

Site descriptions

We chose the *Pinus ponderosa* stands of the Pringle Falls Experimental Forest as our representative for east Cascades forests (Youngblood et al. 2004). Topography in the east Cascades consists of gentle slopes, with soils derived from aerially deposited dacite pumice. The *Tsuga heterophylla–Pseudotsuga menziesii* stands of the H. J. Andrews Experimental Forest were chosen as our representative of west Cascades forests (Greenland 1994). Topography in the west Cascades consists of slope gradients that range from 20% to 60% with soils that are deep, well-drained dystrochrepts. The *Tsuga heterophylla–Picea sitchensis* stands of the Cascade Head Experimental Forest were chosen as our representative of Coast Range forests. We note that most of the Oregon Coast Range is actually composed of *Tsuga heterophylla–Pseudotsuga menziesii* community types, similar to much of the west Cascades. *Tsuga heterophylla–Picea sitchensis* communities occupy a narrow strip near the coast, due to their higher tolerance for salt spray, higher soil moisture optimum, and lower tolerance for drought compared to forests dominated by *Pseudotsuga menziesii* (Minore 1979), and we incorporate this region in order to gain insight into this highly productive ecosystem. Topography in the Cascade Head Experimental Forest consists of slope gradients of ~10% with soils that are silt loams to silt clay loams derived from marine siltstones. Site locations are shown in Fig. 1 and are located within three of the physiographic regions of Oregon and Washington as designated by Franklin and Dyrness (1988). Additional site data are shown in Table 1.

Experimental design

The effectiveness of forest fuel reduction treatments is often, if not always, inversely related to the time since their implementation. For this reason, our experiment incorporated a factorial blocking design where each ecosystem was subjected to four different frequencies of each fuel reduction treatment. We also recognize the fact that fire return intervals can exhibit substantial variation within a single watershed, particularly those with a high degree of topographic complexity (Agee 1993, Cissel et al. 1999), so we examined two likely fire regimes for each ecosystem. Historic fire return intervals may become unreliable predictors of future fire intervals (Westerling et al. 2006); thus ascertaining the differences in TEC_μ that result from two fire regimes might be a useful metric in gauging C dynamics resulting from fire regimes that may be further altered as a result of continued global climate change.

We based the expected fire return time in Eqs. 1 and 2 on historical fire data for our forests based on the following studies. Bork (1985) estimated a mean fire return interval of 16 years for the east Cascades *Pinus ponderosa* forests, and we also considered a mean fire return interval of 8 years for this system. Cissel et al. (1999) reported mean fire return intervals of 143 and 231 years for forests of medium- and high-severity (stand-replacing) fire regimes, respectively, among the *Tsuga heterophylla–Pseudotsuga menziesii* forests of the west Cascades. Less is known about the fire history of the Coast Range, which consists of *Tsuga heterophylla–Pseudotsuga menziesii* communities in the interior and *Tsuga heterophylla–Picea sitchensis* communities occu-

TABLE 1. Site characteristics (from Smithwick et al. 2002).

Site characteristic	Pringle Falls	H. J. Andrews	Cascade Head
Vegetation	PIPO	TSHE–PSME	TSHE–PISI
Elevation (m)	1359	785	287
Mean annual temperature (°C)	5.5	8.4	8.6
Mean annual precipitation (mm)	544	2001	2536
Soil porosity	sandy loam	loam	loam
Mean C storage potential (Mg C/ha)	183	829	1127

Note: Species codes: PIPO, *Pinus ponderosa*; TSHE, *Tsuga heterophylla*; PSME, *Pseudotsuga menziesii*; PISI, *Picea sitchensis*.

pying a narrow edge of land along the Oregon Coast. Work by Impara (1997) in the interior region of the Coast Range suggested a natural fire return interval (expected fire return time) of 271 years in the *Tsuga heterophylla–Pseudotsuga menziesii* zone, and Long et al. (1998) reported lake-derived charcoal sediment-based estimates of mean fire return interval for the Coast Range forests to be fairly similar, at 230 years. However, the *Tsuga heterophylla–Picea sitchensis* community type dominant in our study area of the Cascade Head Experimental Forest has little resistance to fire, and thus rarely provides a dendrochronological record. We estimated a mean fire return interval of 250 years as one fire return interval for a high-severity fire, derived from interior Coast Range natural fire return interval estimates, and also included another high-severity fire regime with a 500-year mean fire return interval in our analysis.

It is important to note that while the forests of the east Cascades exhibit a significant and visible legacy of effects from a policy of fire suppression, many of the mean fire return intervals for the forests of the west Cascades and Coast Range exceed the period of fire suppression ($\sim$100 years), and these forests in the west Cascades and Coast Range will not necessarily exhibit uncharacteristic levels of fuel accumulation (Brown et al. 2004). However, the potential lack of an uncharacteristic amount of fuel accumulation does not necessarily preclude these forests from future fuel reduction treatments or harvesting; thus we have included these possibilities in our analysis. The frequencies at which fuel reduction treatments are applied were designed to be reflective of literature-derived estimates of each ecosystem's mean fire return intervals, since forest management agencies are urged to perform fuel reduction treatments at a frequency reflective of the fire regimes and ecosystem-specific fuel levels (Franklin and Agee 2003, Dellasala et al. 2004). Treatment frequencies for the Coast Range and west Cascades were 100, 50, 25 years, plus an untreated control group, while treatment frequencies in the east Cascades were 25, 10, and 5 years, and an untreated control group.

We incorporated six different types of fuel reduction treatments largely based on those outlined in Agee (2002), Hessburg and Agee (2003), and Agee and Skinner (2005). Treatments 2–5 were taken directly from the authors' recommendations in these publications, treatment 1 was derived from the same principles used to formulate those recommendations, and treatment 6, clear-cutting, was not recommended in these publications but was incorporated into our analysis because it is a common practice in many Pacific Northwest forests. Treatments 1–4 were applied to all ecosystems, while treatments 5 and 6 were applied only to the west Cascades and Coast Range forests, as such treatments would be unrealistic at the treatment intervals necessary to reduce fire severity in the high-frequency fire regimes of the east Cascades *Pinus ponderosa* forests. Note that these treatments and combinations thereof are not necessarily utilized in each and every ecosystem. Managers of forests on the Oregon Coast, for example, would be unlikely to use prescribed fire as a fuel reduction technique. Our experimental design simply represents the range of all possible treatments that can be utilized for fuel reduction and is applied to all ecosystems purely for the sake of consistency.

1. *Salvage logging (SL)*.—The removal of large woody surface fuels limits the flame length of a wildfire that might enter the stand. Our method of ground fuel reduction entailed a removal of 75% of salvageable large woody materials in the stand. Our definition of salvage logging includes both standing and downed salvageable materials (sensu Lindenmayer and Noss 2006).

2. *Understory removal (UR)*.—Increasing the distance from surface fuels to flammable crown fuels will reduce the probability of canopy ignition. This objective can be accomplished through pruning, prescribed fire, or the removal of small trees. We simulated this treatment in STANDCARB by removing lower canopy trees in all cells.

3. *Prescribed fire (PF)*.—The reduction of surface fuels limits the flame length of a wildfire that might enter the stand. In the field, this is done by removing fuel through prescribed fire or pile burning, both of which reduce the potential magnitude of a wildfire by making it more difficult for a surface fire to ignite the canopy (Scott and Reinhardt 2001). We implemented this treatment in STANDCARB by simulating a prescribed fire at low severity for all cells.

4. *Understory removal and prescribed fire (UR + PF)*.—This treatment is a combination of treatments 2 and 3, where lower canopy trees were removed (treatment 2) before a prescribed fire (treatment 3) the following year for all cells.

5. *Understory removal, overstory thinning, and prescribed fire (UR + OT + PF).*—A reduction in crown density by thinning overstory trees can make crown fire spread less probable (Agee and Skinner 2005) and can reduce potential fuels by decreasing the amount of biomass available for accumulation on the forest floor. Some have suggested that such a treatment will be effective only if used in conjunction with UR and PF (Perry et al. 2004). We simulated this treatment in STANDCARB by removing all lower canopy trees (treatment 2), removing upper canopy trees in 50% of the cells, and then setting a prescribed fire (treatment 3) the following year. This treatment was excluded from the east Cascades forests because it would be unrealistic to apply it at intervals commensurate with the high-frequency fires endemic to that ecosystem.

6. *Understory removal, overstory removal, and prescribed fire (clear-cutting) (UR + OR + PF).*—Clear-cutting is a common silvicultural practice in the forests of the Pacific Northwest, notably on private lands in the Oregon Coast Range (Hobbs et al. 2002), and we included it in our analysis for two ecosystems (west Cascades and Coast Range) simply to gain insight into the effects of this practice on long-term C storage and wildfire severity. We simulated clear-cutting in STANDCARB by removing all upper and lower canopy trees, followed by a prescribed burn the following year. This treatment was excluded from the east Cascades forests because it would be unrealistic to apply it at intervals commensurate with the high-frequency fires endemic to that ecosystem.

7. *Control group.*—Control groups had no treatments performed on them. The only disturbances in these simulations were the same wildfires that occurred in every other simulation with the same MFRI.

In sum, our east Cascades analysis tested the effects of four fuel reduction treatment types, four treatment frequencies, including one control group, and two site mean fire return intervals (MFRI = 8 years, MFRI = 16 years). Our analysis of west Cascades and Coast Range forests tested the effects of six fuel reduction treatment types, four treatment frequencies, including one control group, and two site mean fire return intervals (MFRI = 143 years, MFRI = 230 years for the west Cascades, MFRI = 250 years, MFRI = 500 years for the Coast Range) on expected fire severity and long-term C dynamics. This design resulted in 32 combinations of treatment types for the east Cascades and 48 combinations of treatment types and frequencies for each fire regime in the west Cascades and Coast Range, with each treatment combination in each ecosystem replicated five times.

Biofuel considerations

Future increases in the efficiency of producing biofuels from woody materials may reduce potential trade-offs between managing forests for increased C storage and reduced wildfire severity. Much research is currently underway in the area of lignocellulase-based (as opposed to sugar- or corn-based) biofuels (Schubert 2006). If this area of research yields efficient methods of utilizing woody materials directly as an energy source or indirectly by converting them into biofuels such as ethanol, fuels removed from the forest could be utilized as an energy source and thus act as a substitute for fossil fuels by adding only atmosphere-derived CO_2 back to the atmosphere. However, the conversion of removed forest biomass into biofuels will only be a useful method of offsetting fossil fuel emissions if the amount of C stored in an unmanaged forest is less than the sum of managed stand TEC_{μ}, and the amount of fossil fuel emissions averted by converting removed forest biomass from a stand of identical size into biofuels over the time period considered. We performed an analysis on the extent to which fossil fuel CO_2 emissions can be avoided if we were to use harvested biomass directly for fuel or indirectly for ethanol production. We recognize that many variables need to be considered when calculating the conversion efficiencies of biomass to biofuels, such as the amount of energy required to harvest the materials, inefficiencies in the industrial conversion process, and the differences in efficiencies of various energy sources that exist even after differences in potential energy are accounted for. Rather than attempt to predict the energy expended to harvest the materials, the future of the efficiency of the industrial conversion process, and differences in energy efficiencies, we simply estimated the maximum possible conversion efficiency that can be achieved, given the energy content of these materials. The following procedure was used to estimate the extent to which fossil fuel CO_2 emissions can be avoided by substituting harvested biofuels as an energy source:

1) Estimate the mean annual biomass removal that results from intensive fuel reduction treatments.

2) Calculate the ratio of the amount of potential energy per unit C emissions for biofuels (both woody and ethanol) to the amount of energy per unit C emissions for fossil fuels.

3) Multiply the potential energy ratios by the mean annual quantity of biomass harvested to calculate the mean annual C offset by each biofuel type for each forest.

4) Calculate the number of years necessary for biofuels production to result in an offset of fossil fuel C emissions. This procedure was performed for two land-use histories: managed second-growth forests, and old-growth forests converted to managed second-growth forests.

Calculations for each ecosystem are shown in Appendix B.

Simulation spin-up

STANDCARB was calibrated to standardized silvicultural volume tables for Pacific Northwest stands. We then calibrated it to permanent study plot data from three experimental forests in the region (Fig. 1) to

TABLE 2. Treatment abbreviations.

Treatment abbreviation	Treatment
SL	salvage logging
UR	understory tree removal
PF	prescribed fire
UR + PF	understory tree removal + prescribed fire
UR + PF + OT	understory removal + prescribed fire + overstory thinning
UR + PF + OR	understory removal + prescribed fire + overstory removal

incorporate fuel legacies, which were taken from a 600-year spin-up simulation with fire occurrences generated from the exponential distribution in Eq. 1, where λ was based on each ecosystem's mean fire return interval. Spin-up simulations were run prior to the initiation of each series of fuel reduction treatments, and simulations were run for a total of 800 years for forests of the east Cascades, and a total of 1500 years for simulations of the west Cascades and Coast Range.

Data analysis

We employed a nonparametric multivariate analysis of variance, PerMANOVA (Anderson 2001), to test group-level differences in the effects of fuel reduction frequency and type on mean total ecosystem C storage and expected fire severity. PerMANOVA employs a test statistic for the F ratio that is similar to that of an ANOVA calculated using sum of squares, but unlike an ANOVA, PerMANOVA calculates sums of squares from distances among data points rather than from differences from the mean. PerMANOVA was used instead of a standard MANOVA because it was highly unlikely that our data would meet the assumptions of a parametric MANOVA. PerMANOVA analysis treated fuel reduction treatment type and treatment frequency as fixed factors within each respective fire regime for each ecosystem simulated. The null hypothesis of no treatment effect for different combinations of these factors on TEC_μ and $E[F_s]$ was tested by permuting the data into randomly assigned sample units for each combination of factors so that the number of replicates within each factor combination were fixed. Each of our 12 PerMANOVA tests incorporated 10 000 permutations using a Euclidian distance metric, and multiple pairwise comparison testing for differences among treatment types and treatment frequencies was performed when significant differences were detected (i.e., $P < 0.05$).

RESULTS

Results of the PerMANOVA tests indicate that mean expected fire severity ($E[F_s]$) and mean total ecosystem C storage (TEC_μ) were significantly affected by fuel reduction type ($P < 0.0001$), frequency ($P < 0.0001$), and interactions between type and frequency ($P < 0.0001$) in all three ecosystems. These results were significant for type, frequency, and interaction effects even when clear-cutting was excluded from the analysis for the west Cascades and Coast Range simulations, just as it was a priori for simulations of the east Cascades. When the PerMANOVA was performed on only one of our response variables ($E[F_s]$ or TEC_μ), groupwise comparisons of effects of treatment type showed that the most significant effects of treatment and frequency were related to TEC_μ. TEC_μ was strongly affected by treatment frequency for each fire regime in each ecosystem ($P < 0.0001$) and consistently showed an inverse relationship to the quantity of C removed in a given fuel reduction treatment, and was thus highly related to treatment type. $E[F_s]$, similar to TEC_μ, showed significant relationships with treatment frequency for all three ecosystems ($P < 0.0001$), with statistically significant differences among most treatment types. Boxplots of TEC_μ and $E[F_s]$ for each treatment type in each fire regime for each ecosystem are shown in Appendix C.

Fuel reduction treatments in east Cascades simulations reduced TEC_μ with the exception of one treatment type; UR treatments (see Table 2 for acronym descriptions) in these systems occasionally resulted in additional C storage compared to the control group. These differences were very small (0.6–1.2% increase in TEC_μ) but statistically significant (Student's paired t test, $P < 0.05$) for the treatment return interval of 10 years in the light fire severity regime No. 1 (MFRI = 8 years) and for all treatment return intervals in light fire severity regime No. 2 (MFRI = 16 years). The fuel reduction treatment that reduced TEC_μ the least was SL, which, depending on treatment frequency and fire regime, stored between 93% and 98% of the control group, indicating that there was little salvageable material. UR + PF, depending on treatment frequency and fire regime, resulted in the largest reduction of TEC_μ in east Cascades forests, storing between 69% and 93% of the control group.

Simulations of west Cascades and Coast Range forests showed a decrease in C storage for all treatment types and frequencies. Fuel reduction treatments with the smallest effect on TEC_μ were either SL or UR, which were nearly the same in effect. The treatment that most reduced TEC_μ was UR + OT + PF. Depending on treatment frequency and fire regime, this treatment resulted in C storage of between 50% and 82% of the control group for the west Cascades, and between 65% and 88% of the control group for the Coast Range. Simulations with clear-cutting (UR + OR + PF), depending on application frequency and fire regime, resulted in C storage that was between 22% and 58% of

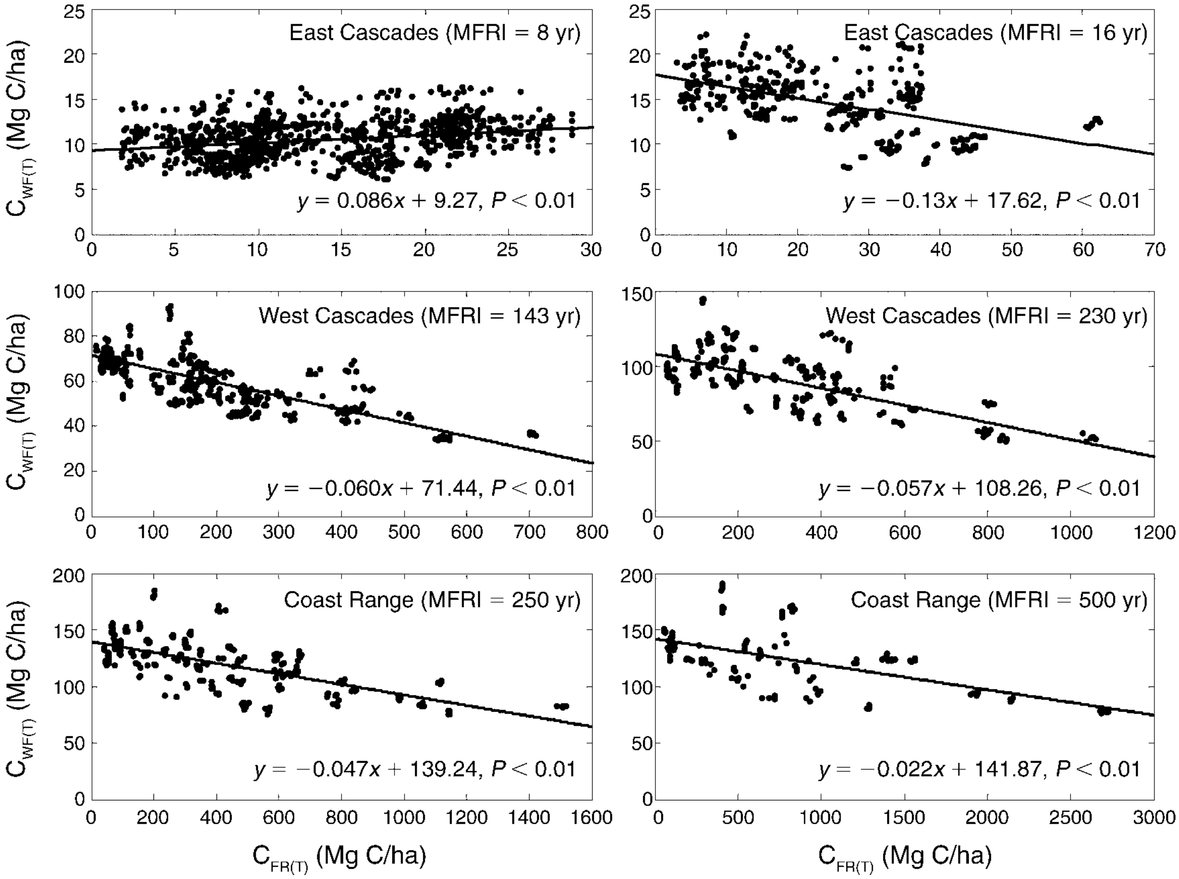

FIG. 2. Scatterplots of C removed in fuel reduction treatments between wildfires $C_{FR(T)}$ (representing fuel reduction [treatment]) and C lost in wildfires $C_{WF(T)}$ for the east Cascades, west Cascades, and Coast Range. Notice the differences in the axes scales. Also note the downward sloping trend for all ecosystems except for the east Cascades where MFRI = 8 years.

the control group for the west Cascades and between 44% and 87% of the control group for the Coast Range.

Similar to TEC_{μ}, $E[F_s]$ was significantly affected by fuel reduction treatments. Fuel reduction treatments were effective in reducing $E[F_s]$ for all simulations. UR treatments had the smallest effect on $E[F_s]$ in the east Cascades simulations and $E[F_s]$ in the east Cascades simulations was most affected by combined UR + PF treatments applied every five years, which reduced $E[F_s]$ by an average of 6.01 units (units range from 0 to 100, see Eq. 3) for stands with an MFRI = 8 years and by 11.08 units for stands with an MFRI = 16 years. In the west Cascades and Coast Range, $E[F_s]$ was least affected by UR treatments, similar to the east Cascades simulations. The most substantial reductions in $E[F_s]$ were exhibited by treatments that removed overstory as well as understory trees, as in treatments UR + OT + PF and UR + OR + PF. In the west Cascades simulations, depending on treatment frequency, $E[F_s]$ was reduced by an average of 11.72–15.68 units where the MFRI = 143 years and by an average of 3.92–26.42 units where the MFRI = 230 years when UR + OT + PF was applied. When UR + OT + PF was applied to the Coast Range, $E[F_s]$ was reduced by an average of 7.06–23.72 units where the MFRI = 250 years and by an average of 1.95–20.62 units where the MFRI = 500 years, depending on treatment frequency. Some UR + OR + PF treatments, when applied at a frequency of 25 years, resulted in $E[F_s]$ that was higher than that seen in UR + OT + PF in spite of lower TEC_{μ} in UR + OT + PF. A result such as this is most likely due to an increased presence of lower canopy tree fuels as a consequence of the increased lower stratum light availability that follows a clear-cut, as lower canopy tree fuels are among the highest weighted fuels in our simulated stands.

Modeled estimates of $E[F_s]$ were reflective of the mean amounts of C lost in a wildfire ($\bar{C}_{WF}$). $\bar{C}_{WF}$ was lower in the stands simulated with fuel reduction treatments compared to the control groups, with the exception of the east Cascades stands subjected to understory removal. Reductions in the amount of C lost in a wildfire, depending on treatment type and frequency, were as much as 50% in the east Cascades, 57% in the west Cascades, and 50% in the Coast Range. In the east Cascades simulations, amounts lost in wildfires were inversely related to the amounts of C removed in an average fire return interval for each ecosystem (Fig. 2), except for the Light Fire Regime No. 1 (MFRI = 8 years). Simulations in this fire regime revealed a slightly increasing amount of C lost in wildfires with increasing amounts removed, though amounts removed were nonetheless larger than the amounts lost in a typical wildfire.

Biofuels

Biofuels cannot offset the reductions in TEC_μ resulting from fuel reduction, at least not over the next 100 years. For example, our simulation results suggest that an undisturbed Coast Range *Tsuga heterophylla–Picea sitchensis* stand (where MFRI = 500 years) has a TEC_μ of 1089 Mg C/ha. By contrast, a Coast Range stand that is subjected to UR + OT + PF every 25 years has a TEC_μ of 757.30 Mg C/ha. Over a typical fire return interval of 450 years (estimated MFRI was 500 years, MFRI generated from the model was 450 years) this stand has 1107 Mg C/ha removed, a forest fuel/biomass production of 2.46 Mg $C{\cdot}ha^{-1}{\cdot}yr^{-1}$, which amounts to emissions of 1.92 Mg $C{\cdot}ha^{-1}{\cdot}yr^{-1}$and 0.96 Mg $C{\cdot}ha^{-1}{\cdot}yr^{-1}$ that can be avoided by substituting biomass and ethanol, respectively, for fossil fuels (see calculations in Appendix B). This means that it would take 169 years for C offsets via solid woody biofuels and 339 years for C offsets via ethanol production before ecosystem processes result in net C storage offsets (see Fig. 3). Converting Coast Range old-growth forest to second-growth forest reduces the amount of time required for atmospheric C offsets to 34 years for biomass and 201 years for ethanol, and like all other biofuel calculations in our analysis, these are assuming a perfect conversion of potential energies. West Cascades *Tsuga heterophylla–Pseudotsuga menziesii* ecosystems (where MFRI = 230 years) that are subjected to UR + OT + PF every 25 years would require 228 years for C offsets using biomass as an offset of fossil-fuel-derived C and 459 years using ethanol. Converting west Cascades old-growth forest to second-growth forest reduces the amount of time required for atmospheric C offsets to 107 years for biomass fuels and 338 years for ethanol. Simulations of east Cascades *Pinus ponderosa* ecosystems had cases where stands treated with UR stored more C than control stands, implying that there is little or no trade-off in managing stands of the east Cascades for both fuel reduction and long-term C storage.

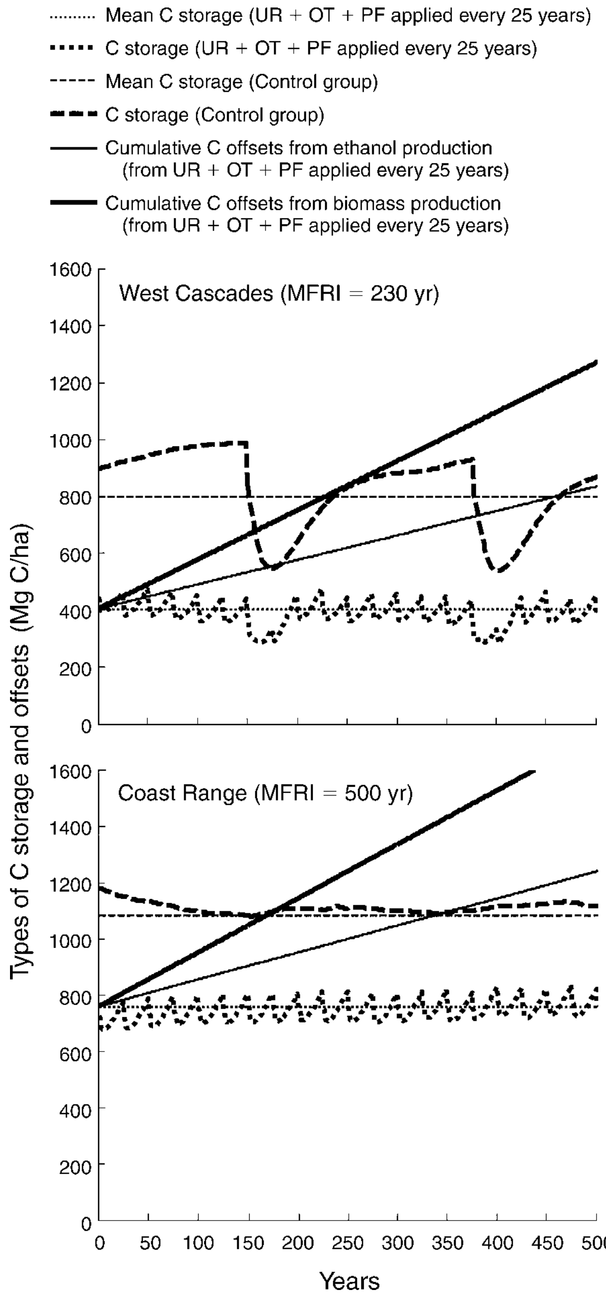

FIG. 3. Time series plots of C storage, mean C storage, and biofuels offsets for control groups and fuel reduction treatment UR + OT + PF (understory removal + overstory thinning + prescribed fire) applied to a second-growth forest every 25 years for the west Cascades and Coast Range. East Cascades simulations were excluded from this plot because there was little or no trade-off incurred in managing these forests for both fuel reduction and C sequestration.

DISCUSSION

We employed an ecosystem simulation model, STANDCARB, to examine the effects of fuel reduction on expected fire severity and long-term C dynamics in three Pacific Northwest ecosystems: the *Pinus ponderosa* forests of the east Cascades, the *Tsuga heterophylla–Pseudotsuga menziesii* forests of the west Cascades, and the *Tsuga heterophylla–Picea sitchensis* forests of the Coast Range. Our fuel reduction treatments for east Cascades forests included salvage logging, understory removal, prescribed fire, and a combination of understory removal and prescribed fire. West Cascades and Coast Range simulations included these treatments as well as a combination of understory removal, overstory thinning, and prescribed fire. We also examined the effects of clear-cutting followed by prescribed fire on expected fire severity and long-term C storage in the west Cascades and Coast Range.

Our results suggest that fuel reduction treatments can be effective in reducing fire severity, a conclusion that is shared by some field studies (Stephens 1998, Pollet and

Omi 2002, Stephens and Moghaddas 2005) and modeling studies (Fulé et al. 2001). However, fuel removal almost always reduces C storage more than the additional C that a stand is able to store when made more resistant to wildfire. Leaves and leaf litter can and do have the majority of their biomass consumed in a high-severity wildfire, but most of the C stored in forest biomass (stem wood, branches, coarse woody debris) remains unconsumed even by high-severity wildfires. For this reason, it is inefficient to remove large amounts of biomass to reduce the fraction by which other biomass components are consumed via combustion. Fuel reduction treatments that involve a removal of overstory biomass are, perhaps unsurprisingly, the most inefficient methods of reducing wildfire-related C losses because they remove large amounts of C for only a marginal reduction in expected fire severity. For example, total biomass removal from fuel reduction treatments over the course of a high-severity fire return interval (MFRI = 230 years) in the west Cascades could exceed 500 Mg C/ha while reducing wildfire-related forest biomass losses by only ~70 Mg C/ha in a given fire (Fig. 2). Coast Range forests could have as much as 2000 Mg C/ha removed over the course of an average fire return interval (MFRI = 500 years), only to reduce wildfire-related biomass combustion by ~80 Mg C/ha (Fig. 2).

East Cascades simulations also showed a trend of decreasing $E[F_s]$ with increasing biomass removal, though a higher TEC_μ was seen in some understory removal treatments compared to control groups. We believe that the removal of highly flammable understory vegetation led to a reduction in overall fire severity that consequently lowered overall biomass combustion, thereby allowing increased overall C storage. Such a result may be indicative of actual behavior under field conditions, but the very low magnitude of the differences between the treated groups and the control group (0.6%–1.2%) suggests caution in assuming that understory removal in this or any ecosystem can be effective in actually increasing long-term C storage. Furthermore, we recognize that the statistically significant differences between the treated and control groups are likely to overestimate the significance of the differences between groups that would occur in the field, as the differences we are detecting are modeled differences rather than differences in field-based estimates. Field-based estimates are more likely to exhibit higher inter- and intrasite variation than modeled estimates, even when modeled estimates incorporate stochastic processes, such as those in STANDCARB. Our general findings, however, are nonetheless consistent with many of the trends revealed by prior field-based research on the effects of fuel reduction on C storage (Tilman et al. 2000), though differences between modeled and field-based estimates are also undoubtedly apparent throughout other comparisons of treated and control stands in our study.

We note an additional difference that may exist between our modeled data and field conditions. Our study was meant to ascertain the long-term average C storage (TEC_μ) and expected fire severities ($E[F_s]$) for different fuel reduction treatment types and application frequencies, a goal not be confused with an assessment of exactly what treatments should be applied at the landscape level in the near future. Such a goal would require site-specific data on the patterns of fuel accumulation that have occurred in lieu of the policies and patterns of fire suppression that have been enacted in the forests of the Coast Range, west Cascades, and east Cascades for over a century. We did not incorporate the highly variable effects of a century-long policy of fire suppression on these ecosystems, as we know of no way to account for such effects in a way that can be usefully extrapolated for all stands in the landscape. *Pinus ponderosa* forests may exhibit the greatest amount of variability in this respect, as they are among the ecosystems that have been most significantly altered as a result of fire suppression (Veblen et al. 2000, Schoennagel et al. 2004, Moeur et al. 2005). Furthermore, additional differences may be present in our estimates of soil C storage for the east Cascades. Our estimates of soil C storage match up very closely with current estimates from the Pringle Falls Experimental Forest, but it is unclear how much our estimates would differ under different fuel reduction treatment types and frequencies. Many understory community types exist in east Cascades *Pinus ponderosa* forests (i.e., *Festuca idahoensis*, *Purshia tridentata*, *Agropyron spicatum*, *Stipa comata*, *Physocarpus malvaceus*, and *Symphoricarpos albus* communities) (Franklin and Dyrness 1988). An alteration of these communities may result from fuel reduction treatments such as understory removal or prescribed fire, leading to a change in the amount and composition of decomposing materials, which can influence long-term belowground C storage (Wardle 2002). Furthermore, there may be an increase in soil C storage resulting from the addition of charcoal to the soil C pool, whether from prescribed fire or wildfire (DeLuca and Aplet 2008).

By contrast, ecosystems with lengthy fire return intervals, such as those of the west Cascades and Coast Range, may not be strongly altered by such a policy, as many stands would not have accumulated uncharacteristic levels of fuel during a time of fire suppression that is substantially less than the mean fire return intervals for these systems. Forests such as these may actually have little or no need for fuel reduction due to their lengthy fire return intervals. Furthermore, fire severity in many forests may be more a function of severe weather events rather than fuel accumulation (Bessie and Johnson 1995, Brown et al. 2004, Schoennagel et al. 2004). Thus, the application of fuel reduction treatments such as understory removal is thought to be unnecessary in such forests and may provide only limited effectiveness (Agee and Huff 1986, Brown et al. 2004). Our results provide additional support for this notion, as they show a minimal effect of understory removal on expected fire severity in these forests, and if in fact climate has far stronger control over fire severity in these forests than fuel abundance, then the small reductions in expected fire severity that we have modeled for these fuel reduction treatments may be even smaller in reality.

We also note that the extent to which fuel reductions in these forests can result in a reduction in fire severity during the extreme climate conditions that lead to broad-scale catastrophic wildfires may be different from the effects shown by our modeling results, and are likely to be an area of significant uncertainty. Fuel reductions, especially overstory thinning treatments, can increase air temperatures near the ground and wind speeds throughout the forest canopy (van Wagtendonk 1996, Agee and Skinner 2005), potentially leading to an increase in fire severity that cannot be accounted for within our particular fire model. In addition to the microclimatic changes that may follow an overstory thinning, logging residues may be present on site following such a procedure, and may potentially nullify the effects of the fuel reduction treatment or may even lead to an increase in fire severity (Stephens 1998). Field-based increases in fire severity that occur in stands subjected to overstory thinning may in fact be an interaction between the fine fuels created by the thinning treatment and the accompanying changes in forest microclimate. These microclimate changes may lead to drier fuels and allow higher wind speeds throughout the stand (Raymond and Peterson 2005). While our model does incorporate the creation of logging residue that follows silvicultural thinning, increases in fire spread and intensity due to interactions between fine fuels and increased wind speed are neglected. However, we note that even if our model is failing to capture these dynamics, our general conclusion that fuel reduction results in a decrease in long-term C storage would then have even stronger support, since the fuel reduction would have caused C loss from the removal of biomass while also *increasing* the amount that is lost in a wildfire.

The amounts of C lost in fuel reduction treatments, whether nearly equal to or greater than our estimates, can be utilized in the production of biofuels. It is clear, however, that an attempt to substitute forest biomass for fossil fuels is not likely to be an effective forest management strategy for the next 100 years. Coast Range *Tsuga heterophylla–Picea sitchensis* ecosystems have some of the highest known amounts of biomass production and storage capacity, yet under the UR + OT + PF treatment a 169-year period is necessary to reach the point at which biomass production will offset C emitted from fossil fuels, and 338 years for ethanol production. Likewise, managed forests in the west Cascades require time scales that are too vast for biofuel alternatives to make a difference over the next 100 years. Even converting old-growth forests in these ecosystems would require at least 33 and 107 years for woody biomass utilization in the Coast Range and west Cascades, respectively, and these figures assume that all possible energy in these fuels can be utilized. Likewise, our ethanol calculations assumed that the maximum theoretical ethanol yield of biomass is realized, which has yet to be done (Schubert 2006); a 70% realization of our maximum yield is a more realistic approximation of contemporary capacities (Galbe and Zacchi 2002).

In addition to these lags, management constraints could preclude any attempt to fully utilize Pacific Northwest forests for their full biofuels production potential. Currently in the Pacific Northwest there are $\sim 3.6 \times 10^6$ ha of forests in need of fuel reduction treatments (Stephens and Ruth 2005), and in 2004 the annual treatment goal for this area was 52 000 ha (1.44%). Unless a significantly larger fuel reduction treatment workforce is employed, it would take 69 years to treat this area once, a period that approximates the effective duration of fire suppression (Stephens and Ruth 2005). The use of SPLATs (strategically placed area treatments) may be necessary to reduce the extent and effects of landscape-level fire (Finney 2001). SPLATs are a system of overlapping area fuel treatments designed to minimize the area burned by high-intensity head fires in diverse terrain. These treatments are costly, and estimates of such treatment costs may be underestimating the expense of fuel reduction in areas with high-density understory tree cohorts that are time consuming to extract and have little monetary value to aid in offsetting removal expenses (Stephens and Ruth 2005). Nevertheless, it is clear that not all of the Pacific Northwest forests that are in need of fuel reduction treatments can be reached, and the use of strategically placed fuel reduction treatments such as SPLATs may represent the best option for a cost-effective reduction in wildfire severity, particularly in areas near the wildland–urban interface. However, the application of strategically placed fuel reduction treatments is unlikely to be a sufficient means in itself toward ecosystem restoration in the forests of the east Cascades. Stand-level ecosystem restoration efforts such as understory removal and prescribed fire may need to be commenced once landscape-level reductions in fire spread risk have been implemented.

Conclusions

Managing forests for the future is a complex issue that necessitates the consideration of multiple spatial and temporal scales and multiple management goals. We explored the trade-offs for managing forests for fuel reduction vs. C storage using an ecosystem simulation model capable of simulating many types of forest management practices. With the possible exception of some xeric ecosystems in the east Cascades, our work suggests that fuel reduction treatments should be forgone if forest ecosystems are to provide maximal amelioration of atmospheric CO_2 over the next 100 years. Much remains to be learned about the effects of forest fuel reduction treatments on fire severity, but our results demonstrate that if fuel reduction treatments are effective in reducing fire severities in the western hemlock–Douglas-fir forests of the west Cascades and the western hemlock–Sitka spruce forests of the Coast Range, it will come at the cost of long-term C storage, even if harvested materials are utilized as biofuels. We agree with the policy recommendations of Stephens and Ruth (2005) that the application of fuel reduction treatments may be essential for ecosystem restoration in forests with uncharacteristic levels of fuel buildup, as is often the case in the xeric forest ecosystems of the east Cascades. However, this is often impractical and may even be counterproductive in ecosystems that do not exhibit uncharacteristic or undesirable levels of fuel accumulation. Ecosystems such as the western hemlock–Douglas-fir forests in the west Cascades and the western hemlock–Sitka spruce forests of the Coast Range may in fact have little sensitivity to forest fuel reduction treatments and may be best utilized for their high C sequestration capacities.

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APPENDIX A

STANDCARB model description (*Ecological Archives* A019-028-A1).

APPENDIX B

Biofuels analysis calculations (*Ecological Archives* A019-028-A2).

APPENDIX C

Carbon storage and fire severity results for each treatment type and frequency (*Ecological Archives* A019-028-A3).

A Statement of Common Ground Regarding the Role of Wildfire in Forested Landscapes of the Western United States

Fire Research Consensus Working Group

Final Report

September 2018

SNAPP Science for Nature and People PARTNERSHIP

Wilburforce Foundation

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Cover photo caption: Night and day on the Pioneer Fire in central Idaho in 2016. How this fire burned and what will happen next reflects the history of fire and fire suppression in this region, as well as land use and changing climate. The Pioneer Fire burned >188,000 acres in 2016, despite active fire management to limit its spread, at a cost of >$100 million. Photo ID_16-08-30 by Kari Greer, Kari Greer Photography, www.wildland-fires.smugmug.com and used with her permission.

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The Steering Committee (Craig Allen, Paul Hessburg, Penelope Morgan, Max Moritz, Dennis Odion, Christopher Topik, and Thomas Veblen) is collectively responsible for the work summarized here, and we adhered to a Project Charter that clarified our goals, roles, and responsibilities. The contributions of Ian McCullough were valuable and substantial enough to merit coauthorship. Excellent project facilitation was provided by Julian Griggs of the Dovetail Consulting Group. The substantial suggestions of Rachel White, science writer/editor for the US Forest Service, were very helpful; additional constructive comments were provided in the USGS internal review process.

Many scientists and managers contributed to this report, either through input to a questionnaire or by providing feedback on interim drafts. Please refer to the online supplemental materials associated with the questionnaire and review process, which include a record of contributors and other documents. The project would not have been possible without the time, effort, and trust of our colleagues, and we are grateful for all contributions.

Executive Summary

For millennia, wildfires have markedly influenced forests and non-forested landscapes of the western United States (US), and they are increasingly seen as having substantial impacts on society and nature. There is growing concern over what kinds and amounts of fire will achieve desirable outcomes and limit harmful effects on people and nature. Moreover, the increasing complexity surrounding cost and management of wildfires suggests that science should play a more prominent role in informing decisions about the need for fire in nature, and the need for society to adapt to the inevitable occurrence of different kinds and amounts of fire and smoke.

Scientists widely view the natural wildfire regime as essential to western US forest ecosystem functioning. However, debates continue over how much low-, moderate-, and high-severity fire is “natural” or desirable in these forests. Ongoing disagreement centers on the characteristics and importance of historical proportions and patch size distributions of low-, moderate-, and high-severity fires of dry, moist, and cold forests, and on the ecological consequences of changing fire-patch patterns and relative abundances. Scientists also debate the relative importance of climate and extreme weather versus fuel as drivers of high-severity fire, as well as the effectiveness and value of fuel treatments for reducing risks of undesired fire effects.

Climate research shows that we should expect shifting future climates in all ecoregions. These expected changes make it difficult for scientists, land managers, and decision-makers to know the degree to which future forest management should be informed by historical conditions. There also is disagreement about how to make western forests more resilient to future disruptions in both climatic and fire regimes. To complicate matters, areas of scientific agreement -- the “common ground” shared by those in the research community -- are poorly articulated. Thus, the focus of the Fire Research Consensus (FRC) project has been to identify common ground among scientists, and provide a summary that can inform management. Land and fire managers are one audience for this report, as are stakeholders and the interested public.

Our analysis, which results from extensive scientific literature reviews and questionnaires sent to western fire scientists and land managers, is summarized in nine key topics:

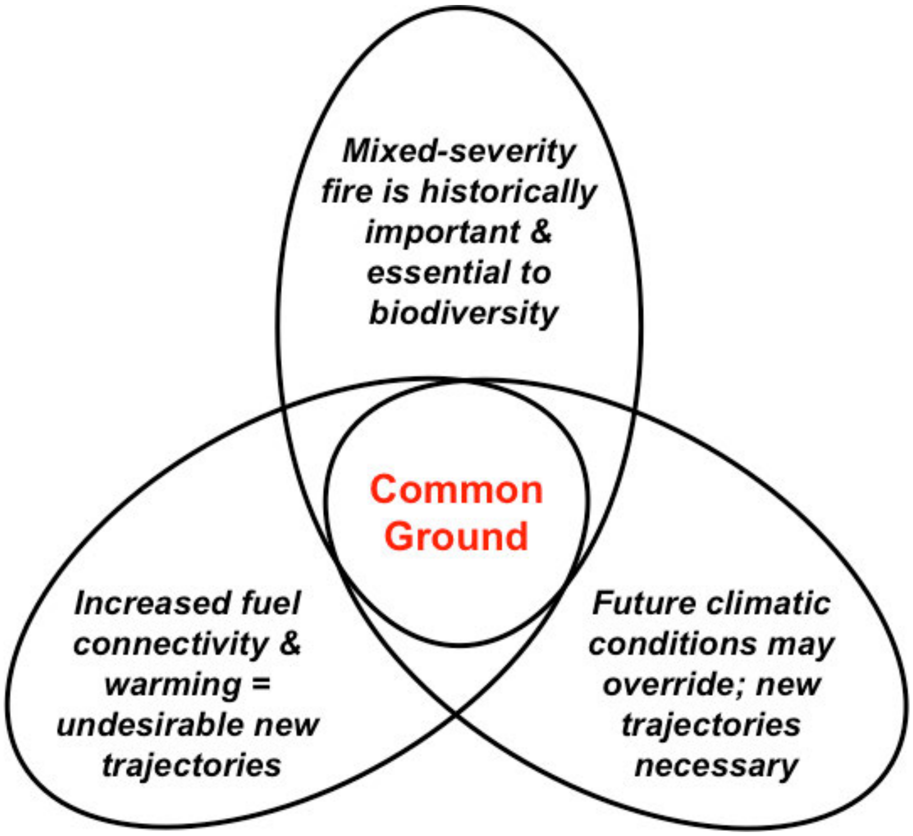
A. Fire history and fire ecology vary with geography.
B. Human impacts and management history vary with geography.
C. Fire is a keystone process, which occurs in almost all western US forest types.
D. Knowledge of historical range of variability (HRV) is useful but does not dictate land management goals.
E. Forest structure, composition, and fuels have changed, affecting burn severity and fire extent.
F. Climate and fuels both influence current fire sizes and their severities.
G. The role of changing climatic conditions is increasingly important.
H. Multiple fire ecology and fire history research approaches can be useful for characterizing fire regimes.
I. Many existing fire management tools and strategies can be useful moving forward.

We found much common ground that will be useful to scientists, managers, citizens, and policy decision-makers. For example, there is wide agreement among scientists that fire is one of the most essential influences on western forests and that more fire is needed on most landscapes, but not all wildfire behavior or extent will do. Fires can produce more positive benefits and fewer negative impacts when they burn with an ecologically appropriate mix and pattern of low, moderate, and high severity. Managers will need assistance and funding to create landscape conditions that favor more desirable fire behavior at broad spatial scales. Note that much societal impact from western wildfires occurs in non-forested landscapes that are not covered in this report, where findings would differ from those reported here for forested landscapes. We summarize additional key points below.

High-severity fire

Respondents disagreed about whether large, high-severity fires have increased to a significant and measurable degree in all forest types *in comparison to historical fire regimes* (i.e., prior to modern fire suppression). There was strong agreement that in dry pine forests at low elevations there has been either an observed increase in high-severity fires or an increase in the potential for fires of elevated severity as the result of increased abundance and connectivity of woody fuels since the late 19th century. There was similar strong agreement about dry mixed-conifer forests in the Inland Northwest, Pacific Southwest, and Inland Southwest (Arizona and New Mexico) that there has been an increase in high-severity fires and an increase in the potential for fires of elevated severity. There was less agreement about the changes in extent, and causes of changes in extent, of high-severity fires in moist mixed-conifer forests. Although there is general agreement that high-severity fires historically played an important role in moist mixed-conifer and cold subalpine forests, there is strong disagreement over the degree of changes in burn severity patch-size distributions and associated successional conditions for these forests between different regions.

Opinions also vary over the consequences of any increases in fire severity. For most dry forests, although there may be some disagreement about trends in burn severity and their causes, there is broad agreement that under current and projected climate, post-fire forest resilience is less than in the past. Some forest habitats, particularly at drier sites, but also in some moist and cold forest sites, show evidence of converting to more flammable non-forest vegetation or less dense forests following recent fires where large patches burn severely, especially if reburned. Reburn potential may depend on the interaction of vegetation, weather, rate of fire spread, time since prior fire, ignitions and fire suppression. Opinions are varied concerning the ecological consequences of departures from historical patterns of fire severity in various mixed conifer and subalpine forests. For example, one viewpoint supports the historical precedence of mixed-severity fire (including relatively large patches of high-severity fire), and the concept that pyrodiversity begets biodiversity. Another viewpoint asserts that increased woody fuel connectivity in combination with a warming climate trend is setting large areas of landscapes on fundamentally new trajectories, with significant undesirable ecological and societal consequences. Still a third viewpoint emphasizes that climatic changes increasingly are of overriding importance, and that new trajectories are unavoidable and thus may be considered desirable in many cases to incrementally foster necessary ecosystem transitions. The figure below characterizes these divergent viewpoints – typical of many areas of disagreement we addressed – and the potential common ground among them.



Uncertainties associated with relative proportions of different burn severities and patch-size distributions combine to cloud key points of consensus that have important management implications. We suggest that resolving many fire science disagreements depends on greater consideration of specific geographical context. This may imply that a narrow range of field experience can limit one's ability to accept findings that depart from that range. A logical way forward is to increase in-depth cross-regional field research experiences of the fire research community. Cross-regional comparisons of top-down and bottom-up determinants of fire activity in similar forest cover types is a fertile area of future research to examine how differences in seasonality, productivity, understory fuels, land use history, and other factors may explain some of the reported geographical differences in historical fire regimes in broadly similar forest types.

There are several reasons for the disagreements about the amount and roles of past higher-severity fire. Both scientists and managers often transfer concepts and findings from one place to another, yet we know that "no one size fits all" for historical fire regimes, even within the same forest type. Likewise, the extent of change in abundance and connectivity of woody fuels varies across forest types and ecoregions. Some of the disagreement derives from use of different scientific approaches. For instance, there is strong debate about the fire regime inferences made from historical and modern tree inventory data, simulation models, and other approaches. We believe that application of diverse research approaches will be useful going forward. Further, multiple approaches will be useful in "triangulating" interpretations for which there is some scientific consensus (see Topic H). We challenge fire scientists who do not share similar perspectives on historical fire regimes in particular ecosystems to engage in civil discourse to better understand the reasons for their disagreement, and to objectively communicate those reasons to managers and other stakeholders. We are heartened by the positive outcomes achieved by some previous attempts when small or large groups work together to find common ground.

The Wildland Urban Interface and Beyond

Respondents strongly agreed on the need for fuel treatments and fire suppression to protect human infrastructure within and adjacent to the wildland urban interface (WUI). There is a strong consensus that preventing undesired human-set fires in the WUI is essential to reducing societal vulnerability. The strategies for managing fire may be different within and adjacent to the WUI than in areas far from the WUI. However, what fire managers do beyond the WUI has implications for fire behavior approaching the WUI, forest resilience, smoke production and its human impacts, water quality, and many other ecosystem services people value.

Fuels management alone, especially if limited to public land, will be insufficient to address the vulnerability of WUI communities to fires. Fuels management will be important for influencing how wildfire behavior will approach the WUI. Thus, policies to make current WUI communities more fire adapted (e.g., implementing current WUI codes) are a critical piece of the puzzle, as are changes in land use policies that influence where and how future WUI areas develop, and the spatial extent and arrangement of managed and wildfire fuel treatments. Controlling human ignitions is important to address fire risk, especially in landscapes where ignitions have the potential to radically increase fire frequency. Communities in fire-prone areas need to learn to live with fire and increase their use of fire and other methods to reduce susceptibility to unacceptable fire damage.

Pattern and Process for Fires in Forest Landscapes

Heterogeneity of fire effects, including the patterns of patches created by fires of all severities, is important to forest resilience to future fires (see Topic E). The scale of the problem is vast, however, so it is likely that the scale of analysis and solutions (e.g., fraction of landscape treated via wildfire use) is also necessarily vast. There are potentially profound implications for forest regeneration, watershed protection, biodiversity, and carbon sequestration if the proportion and spatial pattern of area burned with high-severity fire change. Where wildfires severely burn large areas of forest, local elimination of conifer tree seed sources and reduced tree regeneration under emergent warmer-drier conditions can occur. Large areas of forest are converting to persistent grasslands or shrublands post-fire in some regions. Even relatively small changes in the proportion of large patches can alter system behavior for decades and even centuries. Thus, the patch-size distributions of both forest and non-forest patches are of concern to policy makers, scientists, and managers.

Climate, Fuels, and Implications of Landscape Change

Both fuel and climate are important drivers of fire activity. Increased woody fuel connectivity in combination with a warming climate trend are setting large areas of many landscapes on new trajectories where very large patches burn with high severity. There is agreement that all fire regimes are the product of interactions among varying degrees of top-down climate and weather forcing and bottom-up spatio-temporal controls of local topography and fuels, which reflect legacies of past fires and other agents altering vegetation. In other words, fires respond to interacting influences of climate, weather, fuels, topography, legacies of prior disturbance, and management. The relative importance of these factors varies across landscapes and through time.

While climate is of increasing importance, fuels management is also important. Indeed, fuels are the main landscape characteristic that management can change. But an ecologically and socially appropriate mix of fuel management tools and practices is needed. More flexible management of wildfires and prescribed fires will be useful, depending on local objectives and conditions, to increase the footprint of land areas showing reduced surface and canopy fuel abundance and connectivity. Increased use of prescribed burning combined with thinning will be helpful where forest conditions are not currently manageable via wildfires and prescribed fires alone, and where high certainty about fire perimeter control and fire behavior are key objectives (e.g., adjacent to WUI). Some respondents suggested that accepting a more proactive approach to fire and fuels management on public lands may initially be more expensive, but may reduce overall costs and improve climate change adaptation in the long-term. Other respondents questioned the practicality and effectiveness of fuel treatments under a changing climate. Notably, in their responses, respondents did not integrate the concomitant effects of weather, climate, topography, and fuel abundance.

Decades of research in landscape ecology show that emergent properties have central importance to ecosystems and their pattern and process regulation, whereas many recent studies of climate-driven fire and vegetation change are less focused on local-scale feedbacks and emergent patterns. This difference creates a fundamental problem in linking climate change and landscape ecology research. Climate models assume that top-down climate covariates drive temperature, precipitation, and solar radiation conditions. Landscape ecology research shows that those top-down inputs can be highly modified by meso- and fine-scale bottom-up environmental controls to produce emergent climatic conditions that are strictly speaking neither the top-down or bottom-up inputs, but are influenced by these inputs. Climatic forcing alone poorly explains the shifts in landscape patterns because lagged patterns of historical disturbances continue to influence emergent patterns, under all but the most extreme events. The path forward to more effective projection of future fire and landscape change includes better integration of feedbacks from landscape ecological models into climate-driven models of future fire and landscape change. Broad-scale studies are still needed to tease apart the roles of changing climate and changes in fuels in the observed trends in frequency of large fires.

Effective Management will Depend on Both Science and Trust

Our understanding of historical fire regimes can inform decision-making; indeed, such evidence-based decision-making can build trust. While history does not provide precise prescriptions for managing landscapes, it does offer precautionary principles. Adaptive resilience for the future will require applying what we learn from history to some future range of variability, where fires burn and ecosystems respond in both similar and different ways.

At the same time, fire science points to complex patterns that vary with local conditions. Unique ranges of vegetation and fuel patterns are the result of interactions among regional climate, topography, landforms, geology, and biotic communities of an area, along with associated meso- to fine-scale pattern heterogeneity. Thus, no single solution, such as logging or limiting all logging, will accomplish desired objectives in all forests. Further, any management, including no intervention, has consequences, so all decisions need monitoring to evaluate the assumptions of management. Effective monitoring can improve knowledge, and through collective learning can build common understanding and trust.

Fire management can become more proactive and strategic. Existing tools, such as mechanical fuel treatments, prescribed fire, prevention of accidentally-ignited human fires, and managing wildfires, will all be useful, but adaptation and mitigation responses to climate change and changing fire activity will require using these tools in strategic ways to fit area-specific goals. Some past disagreements about fire and fuel management strategies may be due to lack of clarity about specific goals, such as resident and firefighter safety, cost reduction, biodiversity issues, and ecosystem resilience under a changing climate.

The timing of fires is important, particularly in the context of a changing climate. While recognizing that wildfire seasons are long and getting longer, we must also take advantage of the milder fire weather and associated effects of fires in the “shoulder seasons.” Managers may find that both less-aggressive fire suppression and expanded use of managed wildfire under relatively moderate weather conditions can aid them where reducing the vulnerability of people and natural resources to fires is the objective. Managing wildfires may be one important way to achieve relatively widespread vegetation change at the spatial scales and in the short timeframe needed.

One of the grand challenges of fire management is balancing the reality that wildfires will occur and are needed by western forest ecosystems, yet people, property, and economies need protection from the adverse effects of fire. Another grand and fairly urgent challenge is discovering the tipping points of transformative change for various forest landscapes in their respective geographies, where large, high severity fires (regardless of whether they are considered unprecedented or not) may tip forest ecosystems into persistent non-forest states by constraining tree regeneration opportunities. Particularly as climate changes, we also need a deeper understanding of which landscapes may not be able to sustain forests in the future and how fast such transitions are likely to occur. It is clear that our western history of substantial forest fire activity will continue, one way or another -- many fires will occur in the future and some will be large. Ultimately, we must find ways to both sustainably use and live with fires that are well-adapted to both ecosystem and societal needs of local landscapes.

Introduction

Wildfires have, for millennia, markedly influenced forests and non-forested landscapes of the western United States (US), and they are increasingly seen as having substantial impacts on society and nature, even though less area burns in many forests than burned historically. Informed planning and fire management preparations and responses are thus becoming more important, with lives, property, government expenditures, biodiversity, and ecosystem services at stake. Federal and state firefighting costs now routinely exceed available funds, which are then either borrowed or permanently taken from funds that would ordinarily support resource management activities.

At the same time, climate research shows that we should expect to experience future climates in many ecoregions that will, to varying degrees, differ from those of the recent past. These expected changes make it more difficult for scientists, land managers, and decision makers to know the degree to which future forest management and wildfire policy should be informed by the past. The increasing complexity surrounding cost and management of wildfires suggests that science might play a prominent role in informing decisions about the need for fire in nature, and the need for society to adapt to fire.

Scientists widely view fire as a normal part of ecosystem functioning and one of the most essential influences on forests of the western US. They also recognize that fire directly affects the health and wellbeing of people living near fire-prone landscapes, influencing the water, wildlife, recreation, forest products, and other aesthetic and spiritual benefits these landscapes provide. However, scientific debates continue over several important fire-related topics, including: how much low-, moderate-, and high-severity fire[1] is “natural” or desirable in varied forests across the western US; the relative importance of climatic versus fuel factors as drivers of high-severity fire; and the effectiveness and value of fuel treatments for reducing risks of undesired fire effects.

There is also apparent disagreement about how to make these forests more resilient to future disruptions in both climatic and fire regimes. In many policy and management arenas—from national forest policy to state, county and Tribal-level management—debates about wildfire have sometimes slowed effective integration of research into public policy, and hindered informed planning and management. Much is clearly at stake.

[1] Low, moderate, and high burn severity are usually defined in the western USA by level of mortality of overstory trees or shrubs within individual fires. Low-severity burns are often surface fires with scattered tree torching, where most trees survive (e.g., <20% mortality), while high-severity burns are often stand-replacing fires that kill >70% of the overstory trees (derived from some mix of surface and crown fire behavior). Moderate-severity fires include areas with intermediate levels of overstory mortality (20-70% of basal area or canopy cover of a given patch) from fire. All low-, moderate- and high-severity fire regimes result in intermixed patches of burned and unburned vegetation, but the scale of patchiness differs. Note that we use moderate for intermediate effects of individual fires and mixed for fire regimes. From Agee (1993) Fire ecology of Pacific Northwest forests. Island Press.

Overview: Purpose and Scope of this Report

This report summarizes work of the Fire Research Consensus (FRC) project, which formed to provide insights for scientists, land managers, and human communities with respect to recent controversies over the role of low-, moderate-, and high-severity fires in western US forests. The goal has been to clarify agreements, disagreements, research needs, and possible management implications of scientific common ground. Our hope is that stakeholder groups will avoid the selective use of particular scientific papers to argue for their particular ends. Instead, they will be able to point to key shared assumptions, common understandings considering the entire body of fire science literature, and terminology to support decision-making in constructive ways. This should facilitate better awareness and application of existing and future scientific findings. In particular, land and fire managers are a key audience for this report, as are other stakeholders and the interested public engaged in discussions about land management. Future work is needed to more directly emphasize fire-related research needs and open scientific questions.

We acknowledge that public land management agencies are charged by society to make management decisions, and take associated actions on those lands. Actions are constrained and focused by existing laws, land use plans, policies, and pertinent Acts. We further acknowledge that management agencies are required to accomplish annually-funded land management targets, which can be at odds with some societally-held land-use values, or other landscape and resource management goals. Our focus on areas of broad scientific agreement within the fire science community is intended to make the application of fire science more useful to land management agencies and lawmakers, but it does not, and cannot, resolve diverse social, economic, philosophical, and political debates about preferred land use values across a spectrum of ideologies and management methods. These societal debates play out through broader public conversations and decision-making processes that are only partly informed by fire science. Key roles of fire science are to provide high-quality information to support high-quality societal conversations and decision-making about land (and fire) management, and to assist in monitoring outcomes. In our implications comments at the close of each major section, we provide case examples of how areas of agreement might be considered in the development of management applications.

As a prelude to more in-depth coverage in the report, our analysis can be summarized to nine key topics:

A. Fire history and fire ecology vary with geography.
B. Human impacts and management history vary with geography.
C. Fire is a keystone process, which occurs in almost all western US forest types.
D. Knowledge of historical range of variability (HRV) is useful but does not dictate land management goals.
E. Forest structure, composition, and fuels have changed, affecting burn severity and fire extent.
F. Climate and fuels both influence current fire sizes and their severities.
G. The role of changing climatic conditions is increasingly important.
H. Multiple fire ecology and fire history research approaches can be useful for characterizing fire regimes.
I. Many existing fire management tools and strategies can be useful moving forward.

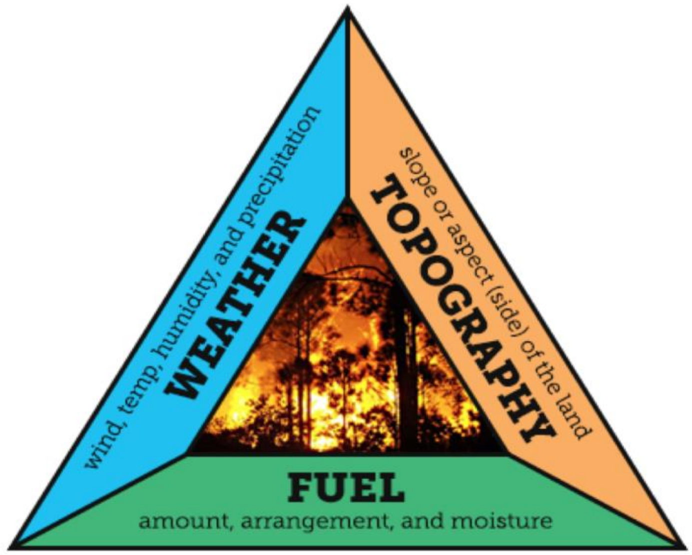
Given the intertwined nature of these topics, there is repetition of themes among some of the material presented. The FRC steering committee believes that the summaries derived from this work are representative of current fire science and can usefully inform fire and land management. It is our intent that in the future, land managers and community leaders will be able to better understand, and more accurately and precisely communicate, the need for fire in the environment and how to better prepare for its impacts. Further, the goals and priorities for fuel and climate change adaptation treatments will be better understood, such that responses to them are less polarized. Scientists will have a clearer picture of the key research questions that underpin current debates. Instead of a focus on disagreements, a deeper appreciation of the research that is agreed upon will allow us all to be more deliberate and proactive when thinking about and managing wildfire environments in the West.

Note that much societal impact from western wildfires occurs in non-forested landscapes that are not covered in this report, where findings would differ from those reported here for forested landscapes.

Fundamental Principles vs. "Common Ground"

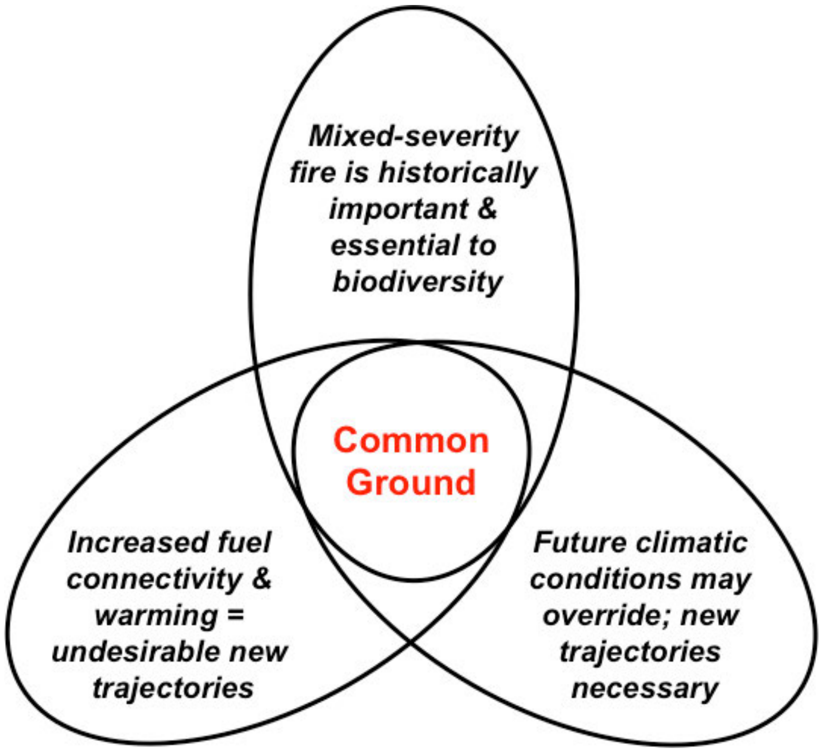
At the outset, we acknowledge some core scientific principles that are widely accepted by those engaged in all sides of these debates. One example is the idea that wildfire is inevitable, and it is a process essential to all western forest ecosystems. Wide agreement therefore exists about the extensive benefits of fire, even if this agreement may not be shared outside of the research community. The notion that fire is an essential ecological process was universally shared and was a guiding principle of most questionnaire respondents.

Another key example is the set of factors making up what is considered the "fire behavior triangle" shown below. This construct was developed by scientists to capture the physical and chemical principles that govern fire behavior, namely characteristics of 1) fuel, 2) weather, and 3) topography in affecting a given fire's rates of spread, flame lengths, and intensities. There is also extensive agreement about there being trade-offs in the relative importance of these factors, such as the influence of fuel characteristics in some instances diminishing in more extreme topographic settings (e.g., steeper slopes) and weather conditions (e.g., higher wind speeds, lower humidities). There is natural variation in how different factors intersect in space and time, resulting in often complex dynamics and only semi-predictable outcomes. Even so, certain relationships are predictable enough at finer scales to be useful for models of fire spread and crown fire initiation; broad-scale simulation of fire behavior patterns is also possible, although with known limitations.



Fire Behavior Triangle

In the context of our project, the fundamental science that underpins the study of fire is not what we mean by “common ground” shared among disagreeing groups. Here instead we are referring to areas of agreement, or the overlap in perspectives, that emerge when debates over a given issue are deconstructed. As a hypothetical but realistic example, consider the Venn diagram below, which represents three partly overlapping views about possible causes and consequences of increases in high-severity fires. There is evidence that some forest habitats, particularly at drier sites, are converting to non-forest vegetation or less-dense forests following recent fires, where large and severely burned patches are created. Conversely, afforestation has occurred in some forest types as a result of fire suppression, which can reduce fire intensity and spread, compared to some non-forest vegetation. Opinions are varied concerning departures from historical patterns of fire severity in various mixed-conifer and subalpine forests, as well as their ecological consequences. One viewpoint supports the historical precedence of mixed-severity fire (including relatively large patches of high-severity fire), and the concept that pyrodiversity begets biodiversity. Another viewpoint asserts that increased woody fuel connectivity in combination with a warming climate trend is setting large areas of landscapes on fundamentally new trajectories, with significant undesirable ecological and societal consequences. Still a third viewpoint emphasizes that climatic changes increasingly are of overriding importance, and that new trajectories are unavoidable and thus may be considered desirable in many cases to incrementally foster necessary ecosystem transitions.



In the realm of public discourse, these three perspectives might be reduced to simplistic and utterly contrasting sound bites, spanning the following extremes:

- Fuel treatments are urgently needed across nearly all forests.
- Fuel treatments should be focused around communities and plantations; hazard reduction elsewhere is futile.
- There is high uncertainty about where and when fuel treatments are beneficial.

Regardless of public perception, there is still a solid scientific basis for each of the three perspectives shown in the example above, and much can be learned by examining the common ground of their intersection. We explore the common ground of these and other such areas of overlap in divergent scientific perspectives in this document.

Philosophical and Contextual Issues

At times, differences in perspective may be linked to whether one's research emphasizes fire effects on tree survival, residual vegetation structure, or fire effects on overall ecosystem function and biodiversity. Frustrations and value judgements about management activities and their impacts on public lands have also contributed to differing scientific perspectives about possible paths forward. Scientist and public mistrust of past and current management on some public lands is one of the largest impediments to forward progress, and yet most discussions focus on improving fire science rather than improving trust. Fire scientists, ecologists, and land managers need to better understand how science has been used in the past to justify various management actions, and how various breaches of trust have affected adoption of modern scientific findings. Such trust can be rebuilt with monitoring and stakeholder engagement in land management decision making.

There was wide agreement among questionnaire respondents that fire science often gets overly simplified in the media, even when more nuanced views may be held among scientists doing the research. Sometimes simplification links back to early narratives and research findings, which may then be inappropriately applied by others beyond their original context. An example of this is the notion that climate change will universally increase fire frequencies and severities, despite growing evidence of more complex outcomes. In other cases, scientists, journalists, policy makers, land managers, NGOs, or politicians may simplify stories to increase their clarity or impact, or to deliver specific messages to the public, and these stories are then carried forth as "debates."

Many respondents also recognize the need for better terminology and conceptualizations of fire regimes[2], both for communicating with the public, and for use among scientists. To some extent, imprecision or ambiguity of terms and concepts may be partly responsible for certain debates in the fire literature. For example, numerous respondents commented on how fire regimes have been oversimplified as fitting into one of the three broad classes of low-, mixed-, or high-severity. Imprecision or lack of agreement on objective classification of fire regimes conflates with actual disagreements over the interpretation of fire history evidence. Although disagreements are not simply based on semantics, poor semantics contribute to confusion. In addition, scientific interpretations of fire regimes made at specific spatial and temporal scales are sometimes fraught with unspecified assumptions, imprecision, or error in the scope of the inferences made.

Additionally, respondents often had differing priorities for, and definitions of, "restoration" and "resilience," generally reflecting the plurality of these definitions and priorities in modern society. For example, how important is the past to understanding and planning for the future? What exactly is being restored, to what benchmark, and for what purposes? What ecological and social values support any intended restoration? Underpinning each of these questions are differing perspectives about the importance of historical ecology based on differences in human social values, even on the part of scientists.

Restoration[3] and resilience[4] are often identified as goals of forest management, and yet the enabling legislation and funding sources of different land management agencies actually

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[2] A fire regime is the pattern, frequency, fire size, spatial complexity, and severity of fires over space and time. Fire regimes are characterized based on fire frequency (how often fires occur), intensity (amount of heat released at the flaming front), severity (both soil and tree mortality effects), type (ground, surface, crown), size, spatial pattern (including patch size distribution) and seasonality. Ground fires burn organic matter in the soil. Surface fires burn leaf litter, fallen branches, and plants on and near the soil surface. Crown fires burn through to the top layer of trees and shrubs. From Morgan et al. (2001) Mapping fire regimes across time and space: understanding coarse and fine-scale fire patterns. International Journal of Wildland Fire, 10(4): 329-342.

[3] Restoration is "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (Society for Ecological Restoration International Science & Policy Working Group [2004] Primer on Ecological Restoration. www.ser.org). Also see Hessburg et al. (2015) Restoring fire-prone forest landscapes: Seven core principles. Landscape Ecology 30(10): 1805-1835.

[4] Resilience: The capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity, and feedbacks (Society for Ecological Restoration www.ser.org). See also Schoennagel et al. (2017) Adapt to more wildfire in western North American forests as climate changes. Proceedings of the National Academy of Sciences. 114(18): 4582-4590.

dictate how restoration and resilience are defined and implemented. Therefore, even if common ground might exist on the need for fire to play a more natural or culturally central role, there can be widely varying opinions about what to do, and varying options as to how to make that happen, legislatively and administratively.

Not surprisingly, there were differing opinions about tradeoffs between human social values and the ecological benefits of fire. For example, smoke from wildfires or prescribed fires is a great concern that can have important influences on how various fire treatments are applied. Reconciling these varied opinions and the associated trade-offs was not in the purview of this effort. Views on forest restoration and ecosystem resilience are thus embedded in this larger context of other human social values, which greatly adds to the complexity of consensus-building and informed decision-making.

Looking forward, assessments of the effectiveness of fire management under climate warming will provide important results, ideally through science-based monitoring and management actions that are intentionally adapted by lessons learned. Most fire scientists and managers agree that fuel treatments can affect fire behavior, though effectiveness can vary with weather, treatment type, location, and time since treatment. Clearly, wildfire researchers recognize the importance of both extreme weather and fuel conditions on fire behavior. However, some respondents suggested that policy makers are unaware of uncertainties associated with attaining fire mitigation goals in the face of more frequent extreme-fire weather, but the management requirement to address such goals persists. Fire managers look to fire science for clear answers about methods, and their reasonable application, because planning and implementing actions in response to climate change, forest restoration, and other needs are essential to their mission.

A number of respondents lamented the time and energy devoted to disagreements over the interpretation of fire history in forest management debates. They suggested that the real challenge is to face the reality of a changing climate and changing fuels by considering the effectiveness of fire mitigation strategies (both old and new). There was also wide agreement on the need for land-use strategies that reduce societal and resource vulnerability to negative consequences of wildfire and climate change, while providing for the essential role and many benefits of fire in forests. We acknowledge that this is an example of fire scientists pointing to a need for stronger engagement with social scientists.

Numerous western fire scientists, when asked, chose not to participate in this survey, and others reluctantly participated. Several cited previous unproductive and unprofessional interactions in the context of debating fire science and related land management issues. Some questioned the motives of researchers not sharing compatible viewpoints on fire issues. Quite a few, including individuals from all sides of the debate, expressed frustrations with the peer review process of some mainstream ecology and forest science publications, and the resulting contradictory messages conveyed to land managers. The FRC Project Steering Committee is well aware of deep division within a portion of the fire research community that is impeding healthy, productive scientific debate. It is beyond the scope of the FRC to examine the non-scientific bases of these conflicts. Instead, our focus has been to identify the common ground shared among a majority of fire scientists on key issues, and to provide a summary that can support informed management decision-making going forward.

Methods and Data

We considered the entire extent of the scientific literature and views of scientists relating to fire research in western US forests. The Steering Committee is committed to inclusion of the full range of scientific perspectives reflected in the questionnaire responses and in the peer-reviewed literature. To facilitate a broad scope of input, we invited responses to a multi-part questionnaire from scientists from many different geographic areas and scientific perspectives. Invited respondents were those who had "*published significant primary research on fire occurrence and fire effects on ecosystem attributes in forests of the western US prior to intensive management, or in areas with limited active management such as large wilderness areas*."

This invitation criterion filtered out potentially important scientific contributions (e.g., those focusing on Native American use of fire, wildlife, post-European settlement periods, ecosystem resilience, and climate change adaptation) from the initial questionnaire. However, a broader range of scientific perspectives was included when the draft common ground statement was distributed for external review. After several rounds of invitation, 77 researchers were contacted, which yielded 36 respondents, including steering committee members. We believe that the depth and geographical breadth of responses were sufficient to identify key areas of agreement and disagreement among fire scientists.

Individual questions in the questionnaire were often intentionally structured as false dichotomies. Using this mechanism, we intended to generate thoughtful responses that would include details as to why a respondent might agree or disagree with the framing of a given issue. While this approach worked overall, it was clearly frustrating to some, and even appeared to a few as evidence of inherent bias in the process.

Between November 1st and 4th, 2016, our steering committee convened a workshop to summarize and organize responses to the questionnaire. Due to great variation in the nature of the questions and how much respondents tended to use literature citations in their responses, we opted not to incorporate citations throughout this report; doing so in a consistent manner was simply seen as intractable. In addition, our common ground document draws upon our own experience and critical reading of the literature, also without the use of citations. As an archived supplement to this report, however, we list citations that were used by respondents in supplemental online materials; any future refereed publications derived from this work will incorporate citations. Note that an exception to this approach is our inclusion of citations in a relatively small number of definitional footnotes throughout the report.

An external evaluation of the completeness and tone of our common ground statement was undertaken in June of 2017. For scientific perspectives, we invited 100 researchers for feedback on our draft statement; this group was larger than the original 77 invitees, to include a broader range of expertise. We received feedback from 36 individuals, not including the FRC steering committee. We also invited review and comments on the draft and its usefulness from 60 land managers and other stakeholders, 22 of whom provided feedback. To the best of our ability, we then integrated the feedback we received into this final document.

Forest Type Classifications

We based our discussion on three broad forest types in the western US, which we refer to as 1) dry pine and/or dry mixed-conifer (*aka*, dry forests), 2) moist mixed-conifer (moist forests), and 3) cold subalpine (cold forests). These are broad terms that are used colloquially to generalize forest types across the western US. Within particular regions, these terms can be crosswalked to classifications that are commonly used by land managers and in peer-reviewed literature.

To the extent that it may be helpful for cross-regional communication and possible generalizations we provided some examples of forest types covered in the questionnaire based on the US National Vegetation Classification (US NVC).

Dry forests, including for example:

- Central Rocky Mountain Dry Forest Macrogroup M501 (1.B.2.Nb.2 *Pinus ponderosa var. ponderosa - Pseudotsuga menziesii - Pinus flexilis*)
- Southern Rocky Mountain Forest & Woodland Group G228 (1.B.2.Nb.1.b *Pinus ponderosa*)

Moist forests, including for example:

- Central Rocky Mountain Mesic Lower Montane Forest Macrogroup M500 (1.B.2.Nb.3 *Tsuga heterophylla - Abies grandis - Larix occidentalis*)
- Central Rocky Mountain Forest Group (1.B.2.Nb.3.c *Abies grandis - Pseudotsuga menziesii* East Cascades Forest Group)
- Mesic Southern Rocky Mountain Forest Group G225 (*Abies concolor - Picea pungens - Pseudotsuga menziesii)*
- Vancouverian Lowland & Montane Forest Macrogroup M023 (*Calocedrus decurrens – Pinus jeffreyi – Abies concolor* var*. lowiana* Forest Macrogroup (exclude *Pseudotsuga macrocarpa –Quercus chrysolepis*)

Cold subalpine forests, including for example:

- Rocky Mountain Subalpine-High Montane Conifer Forest (1.B.2.Nb.5 *Abies lasiocarpa - Picea engelmannii - Pinus albicaulis)*
- California Red Fir - Mountain Hemlock - Sierra Lodgepole Pine Forest (1.B.2.Nd.4 *Abies magnifica - Tsuga mertensiana - Pinus contorta* var. *murrayana)*

Topic A. Fire history and fire ecology vary across geography

Common Ground

Key points of common ground among respondents to the questionnaire include:

- Generalized models of historical fire regimes vary by ecoregion and forest type.
- Even within the same ecoregion and forest type, there is variation in historical fire regimes among differing environmental gradients.
- There are many different historical fire regimes throughout the western US, and a single model cannot represent this variation (i.e., one size does not fit all).
- Historically, some degree of low-, moderate-, and high-severity fire has occurred in all forest types, but in substantially different proportions and patch size distributions at different locations.
- Classification of historical fire regimes according to forest types can be coarse; thus, failure to recognize variation of historical fire regimes *within* forest types can lead to overgeneralization and oversimplification of landscape conditions.

Respondents strongly emphasized how geographical context is critical in understanding and characterizing past, present, and future fire regimes. Many respondents commented that their responses were dependent on geographical context, or they simply noted that the geography under consideration is important. Respondents described numerous examples of how fire regimes vary at a broad scale across large gradients from warm-dry to cool-wet habitats.[5] Within the fire research community, there is essentially unanimous agreement that historical fire regimes differed fundamentally among strongly contrasting forest types such as low-elevation dry pine forests (mainly involving relatively frequent surface fires) versus cool to cold subalpine forests (mainly involving relatively infrequent high-severity fires), so that a one-size-fits-all approach clearly should not apply to management discussions. The spatial and temporal scales at which generalizations about natural or cultural fire regimes are valid vary, and can be uncertain or as yet poorly researched, which may be an important explanation for some disagreements about fire history among researchers, and appropriate management goals among practitioners. In these latter cases, managers with a need to make progress toward agency goals may inappropriately apply knowledge gained from different but related systems, or from expert panels.

A majority of respondents agreed that any singular characterization of fire regimes and how they have been altered by modern land-use practices—at the scale of the western US—is clearly inappropriate. For example, only at the scale of an ecoregion can we estimate patch size distributions of low-, moderate-, and high-severity fires of any particular forest type. However, individual landscapes within ecoregions do not show the full variability extant within an ecoregion. Neither is it always appropriate to simply assign fire regimes by forest type. Within an ecoregion, gradients of climate and vegetation attributes are well understood as determinants of fire regimes and their variation. Most significantly, broad-scale spatial variability of fire regimes results from broad spatial variability in long-term climate, annual weather,

[5] The existence of major differences in fire regimes in strongly contrasting ecosystems such as low-elevation, dry pine forests and high-elevation cold forests is relatively non-controversial and well documented in the literature (e.g., Schoennagel et al. 2004; Hessburg et al. 2007, Perry et al. 2011).

environmental and topographic conditions suitable for burning, *and* variability in amounts and spatial continuity of fuels, the nature of fuels, (e.g., forest vs. shrub vs. grass vegetation), and in the history of prior fires.

Many respondents emphasized that the commonly applied classification of fire regimes as "low-, mixed- , or high-severity" adequately describes dominance but inadequately describes variation in fire regimes across the western US (see Topic C). Whereas low- and high-severity fires are at least theoretically well understood endpoints of a continuum, a broad, poorly defined "mixed" category is the source of much confusion and misunderstanding. For example, "mixed-severity" is used to describe both the temporal variability in fire effects over multiple fire events at one site, and the spatial variation in burn severity within a single fire. Even in the case of the two extremes of low-and high-severity, respondents noted that there is often some degree of variability, with under-appreciated ecological impacts. However, there is agreement among respondents that all fire regimes are the product of interactions between varying degrees of top-down climate and weather-forcing and of bottom-up spatio-temporal controls of topography and local fuel, that reflect legacies of past fires and other agents altering vegetation, and hence fuel properties (see also Topic E).

Some respondents questioned whether commonly used vegetation classification schemes are a suitable basis for generalizing about fire regimes, and expressed that known geographic variation in fire regimes within forest types argues for improved forest and fire regime classifications. Many noted that broad classifications such as "dry forest" encompass substantial amounts of variability in historical, current, and future fire regimes, making generalizations at the level of an entire forest type suspect. Numerous respondents emphasized that variability in historical fire regimes within a broad forest type often reflects dominant influences of neighboring forest types and their associated fire regimes.

Areas of Divergence

Key areas of divergent opinion among respondents included:

- The relative proportions of different historical fire severities in particular geographical areas.
- The relative importance of extreme weather events to historical burn severity.
- Desirable proportions of low-, moderate-, and high-severity fire in the future.

Respondents disagreed about the relative proportions of different severities of historical fires for some of the *same geographical areas of study*. While this is a key source of debate, it is noteworthy that most studies conducted in the same study areas find qualitative similarities in historical fire regimes. Some studies stress the quantitative differences in the proportions of a study area interpreted as fitting into various classes of burn severity. Most commonly, such disagreement involves potential inferences from different types and scales of evidence of past fire, or past vegetation attributes. For example, tree-ring evidence sometimes supports conclusions that contrast with those derived from landscape-scale inventory and monitoring data using different sampling frames (see Topic H). Yet these different types of evidence of past fire sometimes also yield overlapping or even similar estimates of past fire activity.

In other cases, disagreements about proportions of low-, moderate-, and high-severity fire are based on findings from studies conducted in one area that were applied to another. In other words, some respondents assumed transferability of research findings across ecoregions, based on similarity of forest type. In certain instances, this may be true, but in others it may be inaccurate. Respondents expressed a fairly high degree of consensus about historical fire regimes within particular forest types and ecoregions. Few fire history researchers have significant field experience in more than one ecoregion, but a few of those with cross-regional experiences articulated support for the occurrence of contrasting fire regimes in similar dry, moist, and cold forest types among differing ecoregions. Certainly a narrow range of field experience can limit the ability to interpret and accept findings that differ from one's own experience.

Respondents exhibited a wide range of opinions, explicit or implied, about the potential importance of extreme weather events in overriding historical fire behavior and burn severities (see Topic F). Respondents noted that historical fires in some areas were mostly low-severity, but some high-severity events were also evident in tree-ring records incorporating stand ages, tree growth changes, and tree mortality dates, consistent with other evidence. Most others emphasized the greater frequency and extent of low-severity events and their role in reducing fuel quantities, creating fuel-limited systems or open canopy forests. Some respondents stressed the importance of long-lasting ecological effects from infrequent, moderate- or high-severity fires in the same study areas. Respondents who emphasized the longer time scales of charcoal records noted that most areas of predominantly low-severity fires also showed some incidence of moderate- or high-severity fire over longer time frames. However, the spatial imprecision of those longer charcoal records relative to particular forest types and their location makes these insights difficult to interpret. Some respondents related the occurrence of high-severity fires to extreme climate/weather conditions (both past and present), whereas other literature stresses fuel accumulation or both climate and fuel as the main explanations for high-severity fire.

Determining what proportions and patterns of various burn severities[6] may be desirable in the future is a question that goes far beyond the information available from either fire history research or elicited in our questionnaire. What is desirable will be based on fire's expected influence on ecosystem goods and services that are valued by people, and the social acceptability of those influences. Thus, the predominant viewpoint among land managers and policy makers is that wherever feasible, fire and fuels management should promote the fuel and successional conditions that will support the natural fire regime going forward. In areas such as wilderness, where commodity production is not a management objective, the goals are much the same. Regardless of the management allocation, heterogeneity of fire effects, including the pattern of patches created by fires and other disturbances, is important to forest resilience to future fires (see Topic E).

Respondents exhibited a wide range of opinions about desirable future proportions of burn severity. Some stressed that fire and forest managers often propose treatments designed to reduce future potential for large areas burned with high severity. In contrast, others explicitly stated the benefits of high-severity fire, generally stressing its role in providing habitats for certain wildlife species, forest successional heterogeneity, and biodiversity. Proponents of this latter viewpoint stressed recognition and agreement that allowing high-severity fires in the Wildland-Urban Interface (WUI)[7] was not socially acceptable. Some respondents noted a

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[6] Burn severity is ecological change due to fire, often characterized within the first year or more after fire. In contrast, fire severity refers to effects during the fire. From 1) Morgan et al. (2014). Challenges of assessing fire and burn severity using field measures, remote sensing and modelling. *International Journal of Wildland Fire 23*(8):1045-1060, and 2) Keeley (2009) Fire intensity, fire severity and burn severity: a brief review and suggested usage. *International Journal of Wildland Fire 18*(1): 116-126.

[7] Wildland Urban Interface (WUI): The area where structures and other human development meet or intermingle with undeveloped wildland or vegetative fuels. From NWCG glossary of wildland fire terms (https://www.nwcg.gov/glossary-of-wildland-fire-terminology, accessed 8 May 2017).

paradigm shift from a prevalent view in the 1990s that the only acceptable or “good fire” is a low-severity fire, to a growing viewpoint stressing the benefits of some level of moderate- and high-severity fires, as well as the need for societal adaptation to “managing wildfire” and “living with fire.” Many respondents stressed the importance of different management objectives in different settings (e.g., remote areas versus the WUI, general forest versus wilderness management), and of the clearly different historical fire regimes in low-elevation dry mixed-conifer forests versus cold subalpine forests.

Implications

Managers and scientists alike are challenged with overcoming the tendency to simplify historical fire regimes across and within ecoregions and forest types. While managing for the inherent complexity of fire regimes can be daunting and painstaking work, the resulting patterns and effects on processes provide important and compensating benefits. There is no single model of historical fire regimes applicable to all forest types and ecoregions. Managers should exercise care when applying scientific understanding developed in different landscapes, and recognize that this may result in erroneous scientific underpinnings and failure to meet objectives. Thus, management decisions are generally best-informed by area-specific understanding of fire ecology, which in some cases may require new partnerships between managers and researchers, both in implementation and monitoring. Scientists must clarify the importance of place when characterizing and presenting knowledge about historical fire regimes, and would benefit by sharing methodological approaches and collaborating across ecoregions. Stakeholders—from the general public to land managers to society at large—must wrestle with and decide what future proportion and pattern of burn severity might be desirable in each locality, both for the ecosystem, and for the people who live nearby and depend upon their services. Bearing this in mind, stakeholders will need to discuss the ability of various management prescriptions to achieve their desired changes, the social cost and acceptability of the changes, and alternative approaches to accomplishing them (see Topic I).

A logical way forward is to increase cross-regional and in-depth field research experiences within the fire research community. Improved collaboration across research groups, defined geographically or by previous narratives, can overcome some of the current atmosphere of deep distrust and interpersonal acrimony. Cross-regional comparisons of top-down and bottom-up determinants of fire activity is a fertile area of future research, which can examine how differences in seasonality, productivity, surface and canopy fuels, climatic differences, and other factors may explain some of the reported geographical differences in historical fire regimes in broadly-similar forest types. Likewise, inter-regional comparisons of various land-use practices by Native Americans and EuroAmerican settlers would improve our understanding of how these practices have contributed to past and present geographic differences in fire regimes.

Agencies like the US Forest Service and Bureau of Land Management, by virtue of their enabling legislation and Congressionally appropriated annual budgets, are legally required to manage for improved fuel and fire behavior conditions. Actions that can effectively treat large areas over a short period of time often suffer from an oversimplified understanding of the desired conditions. Because there are strong relationships among spatial patterns of surface and canopy fuels, seral stages, expected burn severity patterns, and onsite climate and fire weather conditions, care must be taken to avoid oversimplifying those patterns for the sake of simply reducing expected wildfire severity. Such oversimplifications can have profound effects on habitat patterns resulting from all burn severities, and their spatial complexity and connectivity. Thus, in each geographic area, managers must seek to obtain a clear understanding of the historical spatial patterns of surface and canopy fuels, and of seral stages through focused study and reconstruction of those conditions. Further, they should use modern climate change evaluation tools to assess how these historical patterns would be altered under the 21st century climate anticipated for that area. This larger understanding would enable managers to then consider conditions in this larger context, and develop landscape prescriptions to make the needed adjustments. Tools to be applied would be those that matched the land allocation and the specific needs for change.

Topic B. Human impacts and management history vary with geography

Common Ground

Key points of common ground among respondents to the questionnaire included:

- The influence of humans directly on fire ignitions and suppressions as well as on landscape drivers of fire activity is ubiquitous and important.
- Impacts of humans vary through time, and are not uniform geographically.
- Human influences are pronounced in dry, moist, and cold forests, but impacts vary.
- The role of human ignitions on wildfire prevalence and severity varies markedly in western forests.
- Climate change is a human impact and a strong driver of fire occurrence and effects.

Respondents to the questionnaire strongly emphasized that fire suppression[8], despite its widespread effects, was not the only human activity profoundly affecting fire regimes and fire-prone ecosystems. Most respondents mentioned other activities as influential in altering fire regimes, such as domestic livestock grazing, logging (selective, post-fire, and clearcut), diverse types of anthropogenic ignitions, mining, overly generalized reforestation practices, invasive plants and animals, road and rail construction, and land-use or development changes. Respondents also spoke to the decimation of Native American communities through the introduction of human diseases, and later marshalling onto reservations, which significantly reduced ignitions and cultural fire uses by native aboriginal people. Human impacts vary with degree of access via roads and trails, but even remote areas have been influenced by people. For example, selective and clear-cutting timber harvests have widely affected dry and moist mixed-conifer forests, where the favored commercial species principally grew.

Many respondents noted that the impacts of these different activities are known to have varied over space and time, posing difficulties for generalized characterizations of human impacts over broad geographical areas or forest types. In other words, there was strong consensus that geographical context matters, and this influences the local assortment of human impacts. Some respondents noted that wilderness areas and actively managed forests often have had different human use histories, including Native American influences, and therefore different trajectories. Wilderness areas, along with some national parks and large roadless areas, offer examples of potentially different human influences, and related opportunities for both research and management. A few respondents elaborated on similarities

[8] Fire suppression is the act of extinguishing or fighting fires. Fire exclusion has partially eliminated fires from the landscape using fire suppression and other land uses, such as grazing, settling in valleys, road and railroad building, and agricultural conversion of most native grasslands.

between the effects of some human activities (e.g., fewer fires may be due to active fire suppression, reduced Native American ignitions, and/or grazing that removed surface fuels) in certain ecosystems, while most noted that dense recruitment of shade-tolerant species has been a direct result of nearly-ubiquitous fire suppression efforts, or logging of large, fire-tolerant trees across many western ecosystems.

A notable point of common ground among many respondents and a chord that was detected throughout the literature was that human impacts have been most detectable and pronounced in dry and many moist mixed-conifer forests, where the most commercially desirable species were logged. This logging, along with fire suppression, has resulted in generally altered and often more-homogenous forest compositions and structures. Such homogenization of forests is often due to harvest of larger and older trees and species (like western white pine, sugar pine, western larch, Douglas-fir, ponderosa and Jeffrey pines) followed by regeneration of higher density, young shade-tolerant forests (of grand fir, white fir, subalpine fir, Douglas-fir, red fir, and incense cedar, or mixes of these species), or due to fire exclusion and a variety of other related mechanisms. Less agreement exists on the degree and causes of homogenization with regard to cold subalpine forests. Regardless, this relative consensus about where human impacts have been most pronounced hopefully provides a stepping-stone for further discussion and common ground.

Many also recognized that climate change at broad scales is a dominant human influence affecting fires and fire effects in all ecosystems. We address it here and in sections F and G because it is the one common denominator affecting all forest types and all fire regimes.

Areas of Divergence

Key areas of divergence of opinion among respondents included:

- The general applicability of “thinning and prescribed burning remedies” to offset human influences.
- The significance of human impacts on forest successional conditions in moist and cold forests.

The questionnaire was intended to elicit a wide variety of responses about the generalized applicability of forest thinning and prescribed burning techniques, in response to changes in fire regimes and forest successional and fuel properties that have occurred across different forest types, and in different geographic locations. These topics might have been better separated, which could have made the areas of agreement and disagreement more distinct. Regardless, there was a general pattern among respondents, based on whether they viewed the fire regime of the forest in question as more driven by fuels versus weather and climate (see also Topic F).

For low-elevation ponderosa pine forests and woodlands and to a lesser extent in dry mixed-conifer forests, respondents generally viewed thinning and prescribed burning to have wide utility, both for ecological and social reasons. However, some asserted that, even where such activities may be useful and justified, their effects may be better accomplished primarily through wildfire.

While a majority of respondents agreed with the statement that cold subalpine forests have been little affected by fire suppression, many studies highlight that human impacts on forest successional conditions have been significant in dry, moist, and cold forests in ecoregions of the northern Rockies, Inland Northwest, Pacific Southwest, and Inland Southwest. In particular, there is evidence in these ecoregions that once-complex cold subalpine forest patchworks composed of early, mid, and late-seral forest conditions have been simplified by extensive timber harvesting, fire exclusion and fire suppression, but also to a lesser degree by livestock grazing of the often widespread wet and dry meadows, and road development.

Implications

There is general consensus that human impacts vary widely across western US forests in terms of type of activity and associated ecosystem effects. Although some human activities had similar influences on many forest ecosystems, failure to recognize the heterogeneity of human impacts can lead to overly generalized prescriptions for forest restoration and management. Thus, there is likely no one-size-fits-all management or restoration approach—available to all conditions—due to the importance of locally-coupled human-natural histories, and current social or political considerations. Most fire scientists assume prehistoric Native American influences on fire and forests to have been relatively widespread, but to varying degrees in different landscapes and habitats. However, more clarity is needed about differences in how Native American and more modern human influences shaped forests of today.

The importance of local context in the management of fire-prone landscapes underscores the need to move away from oversimplified narratives that encourage application of fire research beyond its original scope of inference. Nonetheless, a widespread challenge facing land managers is the need to make forest management decisions in the substantial areas of landscape where fire-vegetation history research has not been conducted; this is a major future research need. General agreement about drier forests being the most impacted by human activities could provide a path forward among those disagreeing about the extent of high-severity fire in these ecosystems. Human impacts have been pronounced but with different effects and implications for moist and cold subalpine forests. Additional studies of landscape changes, and of vegetation response to fires and fuel treatments in these forest types, will inform discussions about forest landscape restoration and management.

To apply knowledge of the relative human impacts on local vegetation conditions, managers need to develop a clear understanding of the specific impacts geographically, their period of influence, and some understanding of their relative strength (also see Topic D). Important human impacts to date include:

- domestic livestock grazing, period of grazing, and density and types of animals grazed;
- introduction of non-native plants or animals, their distribution, and influence on herbivory and the local fire regime;
- wildfire suppression, including the number, locations, and timing of wildfires suppressed;
- timber harvest, type of timber harvest, and frequency of harvesting;
- presence of roads and railroads, their density, and the period of road impacts;
- historical frequency of Native American burning and time since that burning ceased;
- other changes in patterns and trends of anthropogenic (e.g., recent EuroAmerican) and natural (lightning) fire ignitions;
- conversion to cropland, exurban, or urban development, other conditions.

Research shows that the presence or absence of even a single one of these human influences can have profound effects on the resulting vegetation and fire behavior conditions. For example, the absence of timber harvest in some studied wilderness areas reveals significant differences in species composition and tree density in comparison with harvested locations growing in similar climatic conditions and forest types. Knowledge of the local human impacts, their period, and relative intensity can help guide the selection of areas needing and not needing restorative treatments, and it can aid in the selection of appropriate management tools.

Topic C. Fire is a keystone process[9] that occurs in almost all western US forest types

Common Ground

Key points of common ground among respondents to the questionnaire included:

- Low-, moderate-, and high-severity fires historically occurred in nearly all forest types.
- Fires of all severities play important ecological roles.
- Since nearly all western US forests are significantly fire-influenced, fire is a key driver of ecosystem patterns and processes.
- Burn severity patterns and resulting successional and fuel bed conditions have changed due to human activities in most forest types.
- In many western forests, a period of fire exclusion persists, reflecting successful passive and active suppression of the vast majority of ignitions (95-98%) over the past century.

There was consensus among respondents that various combinations of low-, moderate-, and high-severity fire occur in nearly all western US forest types, and associated agreement that fires of all severities play important ecological roles in each forest type. Unsurprisingly, there is also consensus that fire has been, is, and will continue to be an essential ecosystem process across nearly all western US forest types. A key challenge for researchers has been to estimate the proportions of fires that could be classified into one of the three commonly-used descriptive severity classes (low, moderate, high), and how those proportions may have changed over time.

An increasing emphasis in fire research conducted over the past 20 years has specifically aimed at estimating proportions of areas historically affected by low-, moderate-, or high-severity fires, but there remain uncertainties about the actual variability of burn severity historically. Some of this uncertainty is due to methodological limitations, especially in the case of high- and moderate-severity fires, where much of the evidence of past fires is destroyed. This renders fire history studies that exclusively use fire scars less useful under these conditions. However, much progress has been made in recent years by combining fire-scar data with extensive tree age data, tree growth release data, and data on tree mortality events, to provide a more nuanced understanding of the history of fire effects. In addition, aerial photographic reconstructions were employed in the interior Columbia Basin and East-side Forest Health Assessment studies, and these have provided expanded insights into the proportion of patches burned with low-, moderate-, and high-severity fires of those studied ecoregions across the 20th century (Topic H).

[9] A keystone process is one upon which other species and processes in an ecosystem largely depend, such that if it were removed or significantly altered, the ecosystem would change drastically.

Varying degrees of increased continuity of forest in all forest types (i.e., loss of early seral grass- and shrublands, and sparse woodlands and savannas) have been observed with implications for increased vulnerability to larger and more continuous crown fire disturbances, particularly in combination with successful suppression of all but the largest fires. A highly promising area of current research is the integration of dendroecological studies with the existing aerial photographic reconstructions currently covering millions of hectares across the northern Rockies and Inland Northwest. Recent research focusing on proportion of area affected by various burn severities and the emergent patterns represents an important improvement over the former focus almost exclusively on past fire frequencies.

There also was consensus that in many western US forests, there has been dramatically less fire activity over the last century than in prior centuries and millennia, tied to intense and pervasive societal efforts to actively suppress and exclude wildfires. Respondents broadly agreed that patterns of fire occurrence have changed in relation to historical patterns, especially in many dry forests, but also in some other forest types and locations. This is a response to changes in climate and/or fuel properties, recognizing that both extreme fire weather and combustible fuels have always existed to some degree (see Topic F).

There are many existing studies of fire history based on stand-origin mapping over study areas of many tens of thousands of hectares. However, a commonly held view in the fire science community is that even larger areas (i.e., many hundreds of thousands of hectares) are required for effective analyses—combining multi-century fire history data with landscape ecological approaches—to understand past fire patterns and simulation of future fire patterns. A fertile area of future research is analysis of large regional and local landscape historical patterns and patch size distributions of burn severity, and how these varied with topography, climate, prior disturbance, and other influences. Such research is needed because inferences are generally drawn from historical fire frequency, rather than pattern analysis.

Areas of Divergence

Key areas of divergent opinion among respondents included:

- Relative proportions of low-, moderate-, and high-severity fire within western US forests historically.
- Magnitude of changes in fire frequency, severity, sizes, and their consequences for various forest types since the 19th century.
- Magnitude of recent changes in forest patterns relative to historical conditions.
- The urgency, scale and overall need for various active and passive management options.

Key areas of divergent perspective among respondents centered on the relative importance of the various fire attributes that everyone agreed were generally important. For example, whereas numerically dominant perspectives can be identified, there was no consensus about the historical proportions and sizes of differing burn severity classes in some forest types, nor agreement about the magnitude of changes in fire frequencies, severities, and sizes; thus changes in the absolute significance and relative importance of different fire regimes in various landscapes is still debated. It is noteworthy that spatial reconstructions of historical proportions and sizes of differing burn severity classes in various forest types are relatively lacking in the literature for some ecoregions, which is likely a key reason for divergent opinions on this topic.

In particular, perspectives on historical patterns and changes in the occurrence and effects of both low-, moderate-, and high-severity fires in dry and moist mixed-conifer forests were a key area of divergence, with most respondents concerned over the negative effects from historical fire suppression, resultant fuel accumulation, and recent increases in high-severity fire. These observations contrasted with some respondents who highlighted climate and extreme fire weather over fuel accumulation as the main driver of high-severity fire, debated the historical relative importance of low- versus high-severity fire, and emphasized the ecological values and importance of past and present high-severity fires in all forest types, but less so in the driest forest types. Notably, respondents did not try to integrate the concomitant effects of weather, climate, topography, and fuel abundance.

We note that many studies use climate covariates to predict trends in annual area burned. These studies generally do not include fuel covariates, and lacking any evidence of the contribution of fuels covariates, conclude that weather and climate drive area burned. More important are area burned by severity class and changes in patch size distributions of severity classes, which lead to changes in patch size distributions of successional conditions. The lack of data on potential changes in the role of fuels may have fostered disagreements regarding the relative urgency and risks of various active (fuel treatment) versus passive (wildfire only treatment, suppression of human ignitions) management options, the appropriate locations and scale of desirable management actions, and the desirability and trade-offs among alternative forest fire management goals and actions.

Implications

Uncertainties associated with relative proportions of different burn severities and patch-size distributions combine to cloud key points of consensus that have important management implications. There is consensus that various combinations of low-, moderate-, and high-severity fire are important to ecological processes in almost all western US forests. Likewise, there is consensus that these combinations of burn severity, and their variability over space and time, contribute to seral stage pattern and complexity, and the future flammability of the landscape. Therefore, given that landscape patterns of successional and fuel conditions aid in controlling and are to a large extent controlled by fire, and that ecosystem function is altered in the absence of fire, the recent reduction of fire activity in many areas has important ecological implications. Managers are open to using fire on the landscape, but they often are unable to use fire alone. They have intimate knowledge of their landscapes and fuel characteristics, and many acres are not amenable to fire-only prescriptions. Managers wish to use combinations of tools, as is appropriate to the fuel conditions and the land management allocations, to restore more natural patterns of burn severity and of successional conditions that will support them down the road. They can use biophysical and topographic templates to tailor desired treatment patch sizes and intensities to their landscapes. And, they will have to accept some uncertainty about the effectiveness of their fire mitigation procedures under different future climates.

Public land managers throughout the western US are concerned with calibrating fire regimes in many forest types. Central to this idea of calibration is geographically pertinent knowledge of historical patch size distributions of seral stages, burn severity patches, and patterns of lifeform and physiognomic conditions. Nevertheless, paleo studies of fire covering multiple centuries to millennia show significant variability in area burned so that expectation of a long-term stationarity in fire patch sizes is unrealistic. Despite the likely lack of long-term stationarity, these landscape conditions and their variability contribute to the patterns and variability of fire regimes. Specific geographic knowledge of these conditions is often lacking and instead managers often apply knowledge of related or nearby systems, often with less than adequate precision. To learn how to better calibrate relative proportions of each burn severity and patch size distributions, managers should work closely with fire and landscape ecology researchers to improve their local characterizations of these historical conditions. Future proportions of low-, moderate-, and high-severity fire will depend strongly on local context, which includes the HRV, societal and political objectives, prior land-uses, climate and weather, topography, vegetation, and other factors.

Topic D. Knowledge of historical range of variability (HRV) is useful but does not dictate land management goals

Common Ground

Key points of common ground among respondents to the questionnaire included:

- Knowledge of the HRV provides essential context for discussion of land management decisions but it does not set management targets.
- There is no single model of the HRV of forest successional and fuel conditions and fire effects that can be applied across the western US.
- Because the HRV differed greatly from place to place, HRV findings from one area may or may not have relevance to another.
- Understanding the determinants of the HRV is useful in assessing future ecosystem responses to climate change and land-use practices.
- Although appropriate time frames of the HRV are often difficult to define, time frames must be specified for the HRV of particular attributes.
- Deep understanding of the HRV may require application of multiple research methods (see Topic H).

The HRV refers to the variation of ecological conditions and processes over spatial and temporal scales that are essential for understanding current ecosystem conditions[10] and their current departures. While historical patterns of fire and associated vegetation patterns are often the focus of HRV studies, comprehensive HRV studies also examine historical variability of many other factors including climate, impacts of forest insects and pathogens, and land uses. Interpretations of changes in fire regimes may thus be related to numerous potential drivers. These interpretations require consideration of climate variability as well as a broad range of land-use practices such as grazing, logging, mining, and management explicitly aimed at altering fire activity.

The HRV describes *a body of knowledge about historical conditions* without any explicit prescription for how that body of knowledge should be applied. In the sense of understanding how current landscape conditions reflect effects of historical biophysical processes and past human impacts, the HRV provides essential insights for how processes create and maintain spatial patterns of forest and non-forest conditions, and how those patterns in turn drive the processes of interest. Examples of the utility of HRV knowledge include understanding of how

[10] This definition and application of HRV is taken from: Hayward et al. (2012) Challenges in the application of historical range of variation to conservation and land management. Chapter 3 in: Wiens et al. (eds). Historical Environmental Variation in Conservation and Natural Resource Management. P. 32-45. Wiley-Blackwell.

past climate change and land-use impacts have affected modern landscape pattern and structure. Teasing apart the effects of land-use impacts such as grazing, logging, and/or fire exclusion on forest conditions from the effects of climatic variation on wildfire activity and forest conditions requires historical ecological understanding.

The respondents' comments reflected a strong agreement among scientists that knowledge of the HRV provides essential insights for decision-making in land management, in the context of current and future ecosystem responses to climate change. Hypotheses about climatic drivers of future ecological change can be developed and tested with HRV data covering a range of time frames.

Retrospective studies of fire are essential for developing a mechanistic understanding of disturbance-mediated ecological changes, including those driven by climate variability, which in turn supports the development of simulation models of future landscape dynamics driven by climate change. Some respondents stressed relatively abrupt or extreme changes in both historical and modern ecosystem conditions under climate variability as a basis for expecting future "surprises" in ecosystem conditions in the face of climate change. Other respondents suggested that future vegetation predictions from regional and global change models are still crude, particularly if those predictions do not consider fire feedbacks from altered fuel complexes and patchworks, and do not represent adequate advances in understanding sufficient to warrant reduced consideration of the HRV of any geographic area.

Respondents emphasized that knowledge of past natural variability is an essential reference for evaluating impacts of modern land-use practices such as grazing, fire suppression, and logging on current ecosystem conditions and processes. They noted the continuing challenge of distinguishing among the relative effects of past logging or grazing from effects of active fire suppression.

Many respondents stressed that the insights synthesized in an HRV assessment are intended to inform discussions of potential management goals that incorporate social values for decision-making. The value judgments involved in a deliberative decision-making process are improved by knowledge of HRV, but adoption of management goals is not dictated by environmental history.

Areas of Divergence

Key areas of divergent opinion among respondents included:

- In practice and in communicating with the public, static representations of the HRV often continue to be inappropriately emphasized.
- The applicability of HRV knowledge from well-studied regions to similar but less studied forest types in other geographical regions.

The areas of divergence reflected in comments of both survey respondents and in broader discussions with stakeholders appear to reflect different views on how HRV information should be applied to management decision-making. HRV studies are increasingly viewed as scientific and analytical tools useful in decision-making, not as the management goal. In that context, some stakeholders and fire scientists assume that the primary purpose of an HRV study is to reconstruct a set of vegetation parameters (e.g., tree sizes, stand densities, tree spatial patterns) as representing past "natural" conditions and suitable "reference conditions." Other fire scientists stress that such reconstructions may only be "snapshots" in time in the sense that their relevance is time dependent, for example possibly depicting conditions that may have existed ephemerally, but are not fully representative of the range of ecosystem conditions over a longer time period. Still others have shown that HRV conditions, when reflected via a space-for-time substitution sampling methodology, can adequately reflect historically extant variation in forest spatial patterns as reconstructed or simulated by state-transition models. These responses highlighted the importance of comparing alternative methods and time periods that may be used to predict or reconstruct variability of an HRV.

Many respondents emphasized that oversimplified models of the HRV are often applied indiscriminately across a diversity of landscapes so that actual ranges of variability are underappreciated. Numerous respondents identified cases where oversimplified models of HRV did not apply either to an entire study area or were inappropriately applied to landscapes where the model had not been tested through sufficient data collection, independent calibration, or observation. Some respondents noted that divergent views of the HRV reflected the transfer of general models and interpretations from regions that had been well studied, to regions lacking any similar studies that might highlight differences related to unique geography. This is often done based on the assumption that an HRV should be similar in broadly defined cover types.

Implications

The HRV is most useful as a guide to management. Although the HRV can provide invaluable insights about how various processes and patterns interacted in the past, each HRV is but one reference range – it can vary widely across different locations and temporal scales. Managers should exercise caution when applying HRV information collected in other landscapes, recognizing that there is no single HRV model that can be generalized across the entire western US or generally to certain forest types. Despite debates about specific methods and applications of HRV, there was widespread agreement that understanding the climatic, land use, and other determinants of past fire activity and fire effects is useful in assessing future ecosystem responses to climate change and land-use practices.

One of the difficulties facing public land managers is their concern about how to address climate change, wildfire area burned, and burn severity predictions for the mid-21st century, given the high uncertainty associated with those projections, especially projections of future vegetation and lifeform changes, which are thought to be some of the most uncertain. This uncertainty forces managers to generally lean on HRV predictions to hedge their bets going forward. Nonetheless, managers have tools to estimate near-future precipitation, water deficit, plant-available water, and evapotranspiration conditions over the next few decades, and these estimates can be used to condition their understanding of desired forest successional, lifeform, and fuel patterns, and patch-size distributions in light of HRV estimates.

Topic E. Forest structure, composition, and fuels have changed, affecting burn severity and fire extent

Common Ground

Key points of common ground among respondents to the questionnaire included:

- Historical landscape and disturbance ecology strongly influence fuel patterns and legacies of live and dead forests.
- Forest structure and composition have been homogenized in many places by timber harvest, fire suppression, grazing, mining, road-building and other activities.
- Fire behavior is patchy in space and time, and resulting patch-size distributions are important to understanding its effects on the landscape.
- Landscape patch configuration (heterogeneity) is important and is a key determinant of fire regimes, fire behavior and ecosystem function; not every configuration will do.
- Several spatial scales and types of vegetation and fuel heterogeneity exist, and each scale has important and different ecological functions.

In the western US, historical patterns of forest structure, composition, and fuels—collectively making up successional conditions—resulted from recurring wildfire, insect, disease, and weather disturbances that kill trees and regenerate forests. Through time, wildfires repeatedly affected most western forests. Burn severity varied with seasonal weather, previous fires and regional climatic conditions, but also topographic, biotic, and geomorphic conditions. Burn severity patches occurred in predictable frequency-size distributions, which captured the spatio-temporal variability of disturbance and effects on local and regional successional patterns. Within this historical context, respondents generally agreed that fires were prevalent and greatly influenced forests, though fire frequencies and effects varied. Further, respondents all agreed that this historical ecology needs to be incorporated into our understanding and management of forest landscapes.

Respondents identified a number of recent studies showing that successional patterns of many western US forests have been altered by 20th-century management. Management actions included timber harvests, wildfire suppression, domestic livestock grazing, mining, and road and railroad building, which generally fragmented successional patchiness, increased forest area and density, and created novel successional and fuel patterns. Chief among these changes was increased abundance and connectivity of dense, multi-layered young forests, with greater proportions capable of supporting crown fire. However, the degree of these changes has varied across forest types and ecoregions. There was general agreement that these changes occurred in many western ecoregions, especially in the dry ponderosa pine, Jeffrey pine, and in some dry mixed-conifer forests (see Topic B).

Several respondents commented on patch and landscape-level feedbacks, noting that landscape-level feedbacks mediated the frequency-size distributions of future low-, moderate-, and high-severity fire, whereas patch-level feedbacks influenced the likelihood of low- and moderate-severity fires. Prior fires were likely complex patchworks of already burned and recovering vegetation, which increased or decreased the size and severity of future disturbances.

Respondents noted that reconstructed historical landscape patterns, fire history studies, and simulation studies show how landscape successional and fuel patterns and their variability may have supported particular historical fire regimes. Unique ranges of vegetation and fuels patterns were the result of interactions among regional climate, topography, landforms, geology, and biotic communities of an area, along with associated meso- to fine-scale pattern heterogeneity. This pattern of heterogeneity was unique and important to facilitating local variation in burn severity patterns, habitat patterns, and was of central importance at all spatial scales.

Areas of Divergence

Key areas of divergent opinion among respondents included:

- The extent to which future fires and forests are constrained by forest and landscape legacies.
- Importance of bottom-up versus top-down variables in fire regimes.
- The relative amount of forest structural change of an area (e.g., increased density and more complex tree layering leading to increased vertical continuity of fuels that can propagate fire upward).
- Costs and benefits of fuel treatments at necessary spatial and temporal scales.

Respondents disagreed about the extent to which structural change and successional forest patterns have been altered by 20th-century management, as well as the relevance of these legacies for future fire regimes. For example, large landscape assessments in the Inland Northwest showed that the increased abundance and connectivity of dense, multi-layered young to intermediate aged forests, with high crown-fire potential, has occurred in dry, moist, and cold forests. In cold subalpine forests this has occurred via the elimination of formerly complex early-, mid-, and late-seral forest patchworks. In dry and moist forests in the Inland Northwest, this has occurred via increased area of forest (as meadows, sparse woodlands, and some shrub vegetation has been encroached upon by forests), and increased density of a once more-complex patchwork of open and closed canopy forests. In contrast to these patterns, respondents and the peer-reviewed literature for the Colorado Front Range, for example, agreed that for the lower elevation areas of dry ponderosa pine forests there has been a substantial increase in woody fuel connectivity. However, respondents noted that the peer-reviewed literature demonstrates a much smaller shift towards increased woody fuel connectivity in mid-elevation dry mixed-conifer forests and even less in the cold subalpine forests. These respondents noted that for dry mixed-conifer forests of the upper montane zone, abundant research does not support a pattern of significant shift towards a higher percentage of the landscape capable of supporting crown fires today in comparison with historical fire regimes, which also included moderate- and high-severity fires.

Overall, divergence of perspectives on the degree of change in vegetation structure and fire potential often reflects studies conducted in similar forest types but different geographical regions, although in other cases, there are fundamental disagreements over the validity or interpretation of evidence for the same landscape using different methods.

Another area of divergence can be traced to a lack of dialogue and theory integration between climate and landscape ecology researchers. A significant body of landscape ecology research shows that “emergent” properties have central importance to ecosystems and their pattern and process regulation, whereas climate scientists are less focused on local-scale feedbacks and emergent patterns. This creates a fundamental problem in linking climate change and landscape ecology research. Climate models assume that top-down climate covariates drive temperature, precipitation, and solar radiation conditions. Landscape ecology research shows that those top-down inputs can be highly modified by meso- and fine-scale bottom-up environmental controls to produce climatic conditions that are strictly speaking neither the top-down or bottom-up inputs, but are influenced by these inputs. Until the processes that produce such emergence are incorporated into downscaled climate modeling, and until landscape ecology studies incorporate the full suite of realistic climate futures, these uncertainties will remain a problem in applying climate change science to landscapes and their restoration.

Implications

There is consensus that landscape pattern, which is influenced by vegetation, topography, climate, and past fire disturbances, is nearly always an important mediator of fire size and burn severity. A variety of management and land-use activities have altered western US forest landscapes at multiple spatial scales, and essentially created a new landscape template for 21st century fire regimes. Successional and fuel patterns will influence future fires, including size and burn severity of patches. When historical patterns are unknown, efforts to create locally representative reconstructions may be needed.

Forest structure, composition, and fuels have changed to varying degrees in different areas, and in some forest types there is broadly shared common ground that these changes are affecting burn severity and fire extent. While changes observed in some dry forests became a prime motivator for agencies to act, and for Congress to focus financing on restorative actions, there is less common ground about the degree of these changes West-wide in other forest types. However, informed dialogue among scientists and managers, and in some cases additional research, can help to improve common understanding concerning the degree of change and appropriate restorative action for other forest types. Monitoring and adaptive management are needed, especially where reconstructions of representative historical patterns and predictions of future patterns are hard to come by. This is a prime opportunity for scientists to work closely with managers in support of resilience-oriented management. In these cases, a significant monitoring component will facilitate learning. Information gained may be used to initiate restoration of forest structure, composition, and fuels, using the tools that best fit the circumstances. Because simply applying the best available science will not always be sufficient to gain assent from stakeholders and interested parties, collaborative dialogue that factors in local social values and emphases tempered by that science may provide an adequate way forward.

Topic F. Climate and fuels both influence current fire sizes and their severities

Common Ground

Key points of common ground among respondents to the questionnaire included:

- Climate and weather are now and will continue to be primary drivers of fire size and annual area burned.
- Surface and canopy fuels are important drivers of burn severity.

Global and regional climates vary over centuries, decades, and between years, including conspicuous oscillations between the relative dominance of warm-dry versus cool-moist weather patterns. As recently as the late 20th century, a sizable portion of the ecological literature assumed relative stationarity in climate, but increasingly abundant and diverse lines of evidence overwhelmingly demonstrate that the Earth's climate, and that of its many ecoregions, has constantly varied over multiple time scales.

Changes at decadal, centennial, and longer time scales have the potential to redefine biophysical settings. Hence, maps of plant associations, environments, existing and potential vegetation, and physiognomic types are now all seen as shifting patchworks. In landscape ecology, this is an accepted view and is wholly consistent with its body of theory. However, in forest, plant, and rangeland ecology, this view of shifting environmental or biophysical settings has stretched thinking for many practitioners and researchers. Relating projected climate changes to anticipated changes in forest fuel conditions and fire regimes adds further complexity (see Topic G).

Operating within this broader context of changing climate and landscapes, respondents agreed that woody fuel quantity, arrangement, and moisture are important to both the current flammability of western US landscapes, and to the ecological effects of fires. Changes in fuels along with topography drive changes in energy release, fireline intensity, flame length, burn severity, and emissions. Respondents agreed that widespread increases in the area that is forested and in the fuel quantity and vertical and horizontal fuel continuity in many ecoregions and forest types have increased the likelihood of large forest fires and higher burn severities via increased likelihood of crown-fire initiation and spread.

Regional climatic variability and extremes also influence wildfire size and burn severity. Based on the last several decades of research, respondents noted that annual, decadal, and multidecadal climate variability has always been important to fire size, and annual area burned. Respondents also agreed that the largest fires have always been driven by extreme fire weather, and they will continue to be. However, within large historical fires, including those burned under extreme conditions, burn severity was often patchy in response to topography and vegetation (i.e., fuels) conditions. The result was variably-sized patches of low, moderate, and high severity within burn perimeters. These patchy burned areas have changed into the 20th and 21st centuries, and more areas are being burned under high severity than is often typical for the forest types. While this view is supported by many respondents and published studies, there are other studies that question its generality. For example, some research based on historical aerial photography in the northern Rockies on burn area and severity from the 1880s to the early 2000s showed that over this long record, the proportion burned with high severity did not increase, despite extensive area burned in recent decades. Likewise, studies based on satellite imagery, while generally showing trends of increasing burn area since 1984 across the western US, do not show increases in burn severity for all ecoregions or even in a majority of regions. However, we note that in pre-1900 low-severity regime landscapes of the southwestern US and low-elevation Colorado Front Range ponderosa pine ecosystems, the most spatially extensive fire years and likely the largest fires occurred in dry years that followed one or more wet years, which apparently supported buildup and broad-scale continuity of fire-spreading fine surface fuels. Smaller fire sizes and low- and moderate-severity fires are generally associated with milder fire weather and moderating climate conditions.

What has changed most significantly since about 1985 is the frequency of large fires in association with warming temperatures and drought. While some of the increase in the frequency of large fires is expected from increased woody fuel continuity, broad-scale studies based on robust research designs are still needed to tease apart the roles of changing climate and changes in fuels in the observed trends in frequency of large fires. Some respondents argued that the loss of the patchwork created by the historically superabundant small fire-affected patches also has contributed to larger patch sizes of recent forests, and in fact this is a key focus of much current research. In many forests, not just dry mixed-conifer forests, some respondents also noted that fire suppression has resulted in loss of the most numerous smaller and most extensive (in some landscapes) lower-severity fires, which has removed an historical resilience mechanism that once had regulated the frequency and severity of the largest fires by controlling fire growth. Expectations under projections of continued climatic warming include more effective fuel drying during years or seasons of reduced precipitation, as well as more extreme short-term events such as heat waves, driving extreme fire activity. This coupling has the ability to significantly alter the size distribution and burn severity of burned patches and functioning of affected landscapes, including their future physiognomic types[11] and patterns of species composition. What is apparently most important is that increasingly extreme fire weather is increasing the frequency of large and severe fires, and quite small increases in the frequency and extent of large high-severity fire patches can result in tipping points for ecosystems.

These points of common ground coincide with increasing evidence that when recent wildfires severely burn large areas of forest, local elimination of conifer tree seed sources and reduced tree regeneration under emergent warmer-drier conditions can occur. As a result, large areas of forest increasingly are converting to persistent grasslands or shrublands post fire in some regions.

Areas of Divergence

Key areas of divergent opinion among respondents included:

- With respect to current fire regimes, the relative importance of landscape changes in vegetation and fuel properties in comparison with weather and climatic changes.
- The degree to which the frequency of large, high-severity fires and large, severely burned patches within fires has increased, and over what time frames.
- The extent to which landscape tipping points have been reached as a result of high-severity fires.

[11] Examples of physiognomic types include evergreen broadleaf forest, deciduous broadleaf forest, evergreen needle-leaf forest, deciduous needle-leaf forest, grasslands, shrublands. From: 1) Kuchler (1949) A Physiognomic Classification of Vegetation. Annals of the Association of American Geographers, 39(3), 201-210; 2) Box (1981) Predicting physiognomic vegetation types with climate variables. Vegetatio 45: 127-139.

One core area of divergent opinion is the relative importance of landscape change to current fire regimes. Empirical research in some landscapes shows that landscape abundance and horizontal and vertical continuity of woody surface and canopy fuels has increased in many western US ecoregions, which when combined with empirical and modeling research on fire behavior, supports an inference of increased fire intensity, longer flame lengths, increased crown-fire ignition and spread potential, and burn severity (i.e., *fuels affect fire behavior and burn severity*).

On the other hand, much recent research concludes that trends in annual area burned or in numbers of large fires are explained by weather and climatic influences on fuel availability. In these latter studies, drought and related time series are used to predict annual area burned. Models generally show fair to good prediction of a positive climate involvement (i.e., *climate drives the recent increase in area burned*). However, more complex statistical models that show multi-way and multi-scale interactions among fuel properties, fire weather, topography, and climatic predictors of fire extent and burn severity are needed.

A critical limitation on this front has been the lack of quantitative data, for some ecoregions, on changing fuel properties geographically and by forest type. Currently, in some ecoregions, we know more about how area burned and fire extent are influenced by climate than how the ecological effects of fires are affected by both changing climate and fuels. We also know that burn severity varies with fire weather, topography, vegetation, and time since fire (or other disturbances), even when large fires are burning under relatively extreme weather. However, there are few studies that show the relative contributions of each of these factors and climate together to burn severity. Recent reports of increasing burn severity for some ecosystem types are mostly, but not entirely, limited to the 1984-present period, due to the limited temporal depth of Monitoring Trends in Burn Severity (www.MTBS.gov) data. In addition, some respondents were concerned about adequate validation of the MTBS data for that period.

A second core area of disagreement hinges on the degree to which the frequency of large, high-severity fires and large, severely burned patches within fires has increased, and how this differs for dry, moist, and cold forest types. Many respondents believe that the frequency of large fires has increased in association with *both* climatic warming and increased woody fuel abundance and continuity, but as noted, broad-scale analyses of the relative contributions of climate parameters versus altered fuels to observed fire trends remains an important research challenge. Nevertheless, for landscapes with documented large-scale increases in woody fuel connectivity, there is a widely shared concern that increased abundance of large high-severity wildfires has expanded the potential for creating broad-scale shifts in dominant physiognomic types.

Implications

There is broad agreement that both climate and fuels are critical regulators of fire regimes in western US forests. In extreme weather, fires are likely to be large and severe, and managers should be mindful that extreme fire weather is expected to become increasingly common in the 21st century. Under milder conditions, however, fire behavior is mediated by complex interactions among climate, weather, topography, vegetation type, and fuel properties that vary spatially due to successional patch structure and patch size distributions. Further, prior fires (both managed and wild) can alter the extent, burn severity, and patch size distribution of subsequent fires depending on time since fire, topography, climate, and other factors.

In many, but not all, portions of the West (including the Inland Northwest and Pacific Southwest, Colorado Front Range, and monsoonal Southwest), scientists and managers have a reasonably large range of studies documenting changes in forest fuel and seral stage patterns of interior forest types, especially those leading to altered fire regimes. It is likely that restoration activities that seek to reduce fuels and restore successional conditions and their altered spatial patterns can be adequately informed, in particular if appropriate attention is paid to the differences in forest type and habitat.

Topic G. The role of changing climatic conditions is increasingly important

Common Ground

Key points of common ground among the respondents to the questionnaire included:

- Climate variability is a key driver of historical and current fire regimes, with distinctive historical patterns of climatic drivers of fire activity evident in different landscapes.
- The western US has recently been affected by a rapidly warming climate, characterized by reduced snowpack, earlier springs, longer fire seasons, hotter droughts, and more frequent periods of extreme fire weather.
- Recent trends in many western forest regions of more large fires and more area burned are linked to recent climatic trends of hotter droughts and longer, more severe fire seasons.
- Projected climate changes toward substantially hotter and drier conditions in the western US are expected to become increasingly significant drivers of amplified forest fire activity and severity; associated climatic interactions with vegetation and fuel conditions will also increase in significance.
- Climate changes, along with other anthropogenic drivers of global change, affect many vital climate-driven forest processes that will interact with changes in fire activity.

Questionnaire respondents noted that climate variability is now accepted as a driver of both historical and current fire regimes in all western US forests. Distinctive historical patterns of fire activity—driven by periods of hot and dry climate—are evident and well-documented in numerous western US landscapes (see Topic F). This important consensus coincides with the broader scientific consensus that the current western US climate has trended hotter and effectively drier in recent decades. This hotter and drier climate has fostered reduced winter snowpacks, milder winters, earlier springs, more rain-on-snow events, longer fire seasons (at times 40 to 80 days longer), drier fuels, and more instances of extreme fire weather—all generally consistent with regional model projections of future climatic change. Some western ecoregions now have nearly year-round fire seasons.

Consensus also emerged from the questionnaire that these recent climatic trends are linked to changes in fire activity since about 1980-85, contributing to larger fires, more area burned, and more moderate- and high-severity fire in some western US forests. Projected future climate changes toward progressively drier fuels and more extreme fire weather conditions in the western US are expected to amplify forest fire size and area burned. Proportion of high-severity fire may follow different trends as burn severity is more affected by topography, vegetation, and fuel beds, and less by climate than area burned (see discussion about the relative importance of fuel treatments; Topic E). We note that climate and fire weather largely determine the moisture content of vegetation and surface fuels, which has a strong effect on the availability of fuels to burn, energy released by the fuel complex, and resulting flame length, fireline intensity, and smoke emissions. Given projected climate warming and drying in the West, current forest fuel accumulations will be reduced through time by anticipated increases in fire activity (although surface fuel loads typically spike within a decade as standing post-fire snags [i.e., dead boles and branches] fall down amidst diverse vegetation regrowth), by constraints on forest regrowth under a hotter and drier climate, and by forest transitions to non-forest vegetation over increasingly large areas. In some of these areas, afforestation due to lack of fire has occurred, which reduces vegetation flammability and rate of spread. Anticipated future changes in forest fire activity and fire effects ultimately will be modulated by these feedbacks among fire, fuels, vegetation succession, and climate.

Respondents also indicated that emerging climatic changes also widely increase tree physiological stress, and adversely affect tree regeneration, growth and mortality losses, and associated insect and disease outbreaks. Thus, ongoing and future climate-induced changes in forest extent, forest fire extent, severity, and effects must be understood in relation to these additional biotic, abiotic, and anthropogenic factors.

Areas of Divergence

Key areas of divergent opinion among respondents included:

- There remains a divergence of opinion over the relative contributions of climate change and fuel accumulation to current patterns and trends of wildfire activity.
- Effectiveness of fuel treatments under projected climate futures and associated more extreme fire weather.

All respondents agreed that climate change is occurring and likely to continue. The main divergence among respondents involved perceptions of the relative importance of climatic versus fuel factors as drivers of changing fire activity, both now and in the future. This basic divergence in perspectives emerged repeatedly in questionnaire responses, as noted in Topics E and F, despite a general lack of scholarly work to explore joint contributions of climate and fuel to fire extent and burn severity.

This divergence in perspectives about the relative importance of climatic versus fuel factors as drivers of changing fire activity also extends to a related divergence in views on the effectiveness of fuel treatments under projected climate futures and associated more extreme fire weather; this area of divergence is presented under Topic I.

Implications

There is wide agreement that climate has long been a principal regulator of wildfire activity and therefore there is broad consensus that climate change via decreased fuel moisture and more extreme fire weather will considerably impact future wildfire activity. There is also wide agreement that fuels are a principal regulator of wildfire activity and fire effects. Divergent opinions emerge with respect to the relative importance of climate and fuel accumulation. Looking ahead, managers should expect climate change to create conditions of declining favorability to historically dominant forest communities, including warmer droughts, reduced snowpack and other phenomena. These general climatic trends are likely to be conducive to longer fire seasons and greater fire activity in the 21st century. Increasingly extensive vegetation transitions to more drought-tolerant and better fire-adapted species and/or lifeforms are anticipated. Although fuel properties directly influence fire behavior and fire effects, managers require in-depth knowledge of all determinants of fire behavior, including expected climate-related effects on fuel moisture and vegetation and other ecological changes, to determine the extent of possible feedbacks with climate change.

Anticipated changes in western US wildland fire activity have the potential to disruptively challenge the sustainability of historical forest ecosystems and our linked human societies. We expect that a broad range of fire-related adaptation measures will be considered in many western forest landscapes, ranging from increased regulation of human land use activities (e.g., disincentives for exurban development, building codes, seasonal recreation restrictions), implementation of diverse vegetation treatments (including managed wildfire, prescribed burning, and strategically-placed mechanical treatments), to management of forest stand structures, tree species compositions, and genetic variability, in order to foster resilience to growing drought stresses and associated disturbances (fire, insect outbreaks, tree regeneration failures). We expect increased societal attention and preference for such adaptation efforts in order to increase the likelihood of favorable forest adjustments to increasingly novel climate and other emerging environmental stresses.

Topic H. Multiple fire ecology and fire history research approaches can be useful to characterizing fire regimes

Common Ground

Key points of common ground among the respondents to the questionnaire included:

- It is desirable to use multiple methods to reconstruct historical fire regimes. More can be learned using multiple approaches and considering data from diverse temporal and spatial scales.
- Integrating and interpreting findings derived from diverse methods, data sources, and different scales of inquiry can be challenging.

The interpretation of any research evidence and the scope of related inferences is limited by scaling and sampling concerns associated with the methods, and these limitations apply to all research methods. Respondents to our survey strongly agreed with the statement that "*New and important insights should be possible through studies that use and compare alternative sources of data, and results may be used to examine fire history and fire effects in the same study areas.*" Respondents disagreed with the statement, "*Even if we find many different study areas where alternative sources of data are available, there are too many uncertainties or incompatibilities among them to make such comparisons useful.*" Thus, respondents recognized the high potential value of using and considering multiple approaches, data sets, and scales of observation to more robustly assess historical fire regimes. Broadly speaking, this reflects widely-accepted scientific views on the general benefits of using multiple lines of evidence when possible, with increased confidence in conclusions when most results are in agreement.

For this project, we decided to focus on the evidence regarding fire regimes of recent centuries, although substantial paleoecological research using sedimentary charcoal and pollen data has been essential in expanding our understanding of long-term variations in fire regimes. All methods for reconstructing historical fire regimes are necessarily indirect. They may include, but are not limited to, interpreting evidence of past fires or the extent of fire-dependent ecosystems from historical documents, land surveys, aerial photographic reconstructions, fire-scar and growth-release data from tree rings, tree age and death dates from tree-ring data, climatic data linked with past fires, charcoal and pollen deposits, current characteristics of stands (i.e., structure, species, and stand age distribution), fire perimeter mapping, historical timber survey data, and use of statistical distributions for modeling stand-replacing fire. In addition to utilizing multiple methods, the use of clear and shared terminology is needed for effectively combining research approaches to characterize fire regimes. Similarly, the use of diverse archaeological, anthropological, and cultural resource research methods that address the extent and impact of aboriginal fire uses in landscapes can provide useful information in support of restoring culturally important landscapes and their fire-maintained cultural resources.

Respondents noted that multiple methods enhance the potential of inferring the severity and other ecological effects of past fire events, which is central to current debates about the relative proportions of fires of different severity in the past. There are diverse examples where western fire researchers have used multiple methods to characterize historical fire regimes. Commonly, there is general agreement among studies about characteristics of historical fire regimes, particularly for ecosystem types that have had a history dominated by either low-severity fires (e.g., leaving scars but not killing many adult trees) or high-severity fires (killing many adult trees).

In recent decades, we have increased our learning about the strengths and weaknesses of diverse methods and data sources for analyzing high-severity fire, and also the scope of spatial and temporal inference limits for reconstructing historical fire regimes and forest conditions in varied western US landscapes. A particular challenge has been elucidating historical spatial patterns, such as patch sizes, shapes and arrangement. Much of this expanded insight has come on the heels of examining relationships between documented fire histories and associated forest successional or cohort conditions. In particular, further developing studies that cross-walk dendroecological fire histories with aerial photo interpretation and cohort age structure analyses offer much promise. These methods too can be combined with simulation studies that may offer additional insights. Respondents recognized that a more productive approach to multi-methods analysis might be for research laboratories that specialize in one method or another to collaboratively join their strengths in designing, implementing, interpreting, and documenting results of such research through joint work in multiple landscapes.

Areas of Divergence

The areas of divergence in opinion among respondents included:

- The introduction of new methods for reconstructing historical fire regimes has, in recent years, resulted in unresolved debates regarding the limits and usefulness of some new and old methods.

  There currently is significant debate about the validity and thus utility of some new approaches using historical (General Land Office, GLO) and current (USFS Forest Inventory and Analysis, FIA) land and timber survey data to infer the amount of high-severity fire, forest species composition, and the density and age structure of historical forests. Similarly, extrapolating from historical tree-ring and fire-scar point data across much larger areas has been a topic of some debate, but the disagreements are quite different. In the former case, disagreements center around the usefulness of the land survey data to the ends applied. This results from doubts regarding differences in interpretations of historical fire regimes based on tree-ring or other data versus historical land survey data. In some cases these differences are large but in other cases the percentages of a landscape classified as having an historical fire regime of mainly low-severity versus mixed (or higher) severity fire are relatively slight. The validity of reconstructing historical forest conditions and fire regimes in particular from all types of historical land or timber survey data has been critiqued. Such scrutiny of the validity of methods is a normal part of the scientific process, and highlights the need for continued research based on cross-validation from multiple types of data and methods.

Implications

The use of multiple methods for characterizing historical fire regimes, combined with increasingly clear and shared terminology, can improve our understanding of HRV patterns and processes in western forests. However, there can be significant challenges associated with bringing together evidence about historical fire regimes from differing methods and data sources. Each line of evidence has a different scope of spatial and temporal inference, and issues about the nature of the data captured in each sample. In addition, there is substantial skepticism about the utility of some methods for HRV reconstruction purposes, which will have to be resolved. Nonetheless, one new frontier of fire ecology research is the exploration of multi-method approaches by collaborating labs toward more-nuanced understandings of diverse fire regimes. For example, in mixed-severity fire regime forests, by combining time series derived from diverse dendroecological data sources (e.g., fire scars, death dates of trees, establishment of postfire cohorts, growth releases on surviving trees), land survey data, aerial photographic interpretations of successional and past fire severity conditions, landscape panoramic photos, and simulation modeling, stronger inferences may be possible about the ecological effects of past fire events.

Topic I. Many existing fire management tools and strategies can be useful for managing fire going forward

Common Ground

Key points of common ground among the respondents to the questionnaire included:

- Many tools can be useful to fire managers for reducing human vulnerability to fires and increasing ecosystem resilience.
- Managed wildfire is underutilized but viable ecologically and socially in many areas.
- Managing fuels is important and fuels are one contributing factor that can be influenced through management.
- Thinning alone without managing the resulting fuels increases surface fuels and does not mimic many of the ecological effects of fire.
- Firefighter and citizen safety, degree of smoke production, financial costs, and effective scales of treatment must all be considered.
- Land-use and financial incentives could be used to reduce human vulnerability to wildfires in and near the WUI.

Many respondents stressed that a wide variety of tools and policies can be useful to increase forest resilience and reduce human vulnerability to future fires. Suppressing fires to protect highly valued resources is important, but managers need a full suite of active and passive management strategies and tools because different management situations often call for different approaches. There was strong support for managing wildfires to accomplish resource benefits and also support for prescribed burning[12]. We agree. However, there was very little discussion of how and where wildland fire use can be effectively implemented to foster desirable patch size distributions, particularly where climate and forest conditions have changed, and surface and canopy fuels have accumulated over the period of fire suppression. Broad-scale landscape planning for wildland fire use will be essential to better understand special circumstances and clear opportunities for its use.

Wildland or prescribed fire use can be effectively complemented with fire suppression strategies and with thinning to reduce vertically and horizontally continuous fuels that contribute to fire hazard. Tools such as the Wildland Fire Decision Support System are used to make effective fire management decisions considering landscape conditions, jurisdictions, fire weather, values at risk and local management objectives.

Increasing education and outreach, managing post-fire to reduce soil erosion potential where values are at risk, decommissioning roads, creating snags where they are in short

[12] Wildfires are ignited by people or lightning. They may be suppressed, either aggressively or with more limited efforts, depending on management objectives, values at risk, costs, firefighter risk, and other factors. Managed fires are those that achieve resource objectives. They are monitored and parts may be actively suppressed while other parts are managed with less aggressive suppression. Prescribed fires are ignited by management actions under certain, predetermined conditions to meet specific objectives, such as reducing hazardous fuels, improving habitat, managing cultural resources, firefighter training, fire behavior experiments, or restoring forests. Prescribed fires are nearly always conducted under written, approved plans.

supply, and other tools can further help accomplish management objectives, while protecting people and property from fire and fire effects. Other strategies for helping communities become more fire-adapted include altering residential development in highly fire-prone environments, and making existing homes safer from wildfires. Land-use (e.g., applying the national WUI building codes proactively and retroactively, zoning to concentrate development in lower fire danger environments) and financial incentives (e.g., tax, insurance, mortgage restrictions, fees, assistance with fuel treatments around homes and towns, support to mitigate structure ignition vulnerabilities) could be used to reduce vulnerability to wildfires in and near the WUI.

Certainly, fire managers must consider financial costs, firefighter safety, public safety, and smoke. These and other societal and operational management constraints vary geographically, so managers must look for opportunities to adapt and use multifaceted strategies. There was widespread agreement among respondents that the suitability of different tools is highly context specific. In discussing strategies, both the often significant beneficial and detrimental consequences of taking no action must be considered.

There is strong consensus that more fire is needed on the landscape, but not all wildfire behavior or extent will do. Managers need assistance and funding to create landscape conditions that favor more desirable fire behavior at spatial scales and extents that can make a difference to current conditions.

Respondents generally indicated that the scale of landscape change in western US forests is quite broad, and that it could be difficult to overcome (i.e., a high level of landscape inertia), especially with the current level of defunding of public land management agencies. The cost of fire suppression has risen from 17 to nearly 60% of the entire Forest Service budget in the last 25 years, greatly limiting the financial capacity of the agency for proactive work at any meaningful scale. Treatments need to be of sufficient scale and pattern to be effective at restoring patch-size distributions of low-, moderate-, and high-severity fire, and at reducing what is seen by many as an increasing risk of unusually large, high-severity patches within fires. Although fuel treatments can be prioritized across very large landscapes to be potentially effective in managing wildfires to accomplish resource benefits, such treatments must be designed consistently with other ecological and management goals including riparian corridors, habitat for listed species, and the like. Restorative treatments likely need to occur at the scale of the landscape changes to change current fire regime conditions. Due to widespread existing habitat reserve commitments, opportunities for strategically allocated treatments are substantially limited.

Given the profound influence of the type and amount of fuel on fire behavior (Topic F), the type, location, timing, frequency, and maintenance of fuel treatments[13] will all influence their effectiveness. Forest thinning is one commonly applied fuel treatment. Most cutting methods that are applied to reduce future burn severity are thinning treatments where emphasis is on removal of the less fire-resistant trees (usually the smaller ones especially of shade tolerant species). The intensity of thinning determines the amount of branches and tree tops (slash) left behind. There is consensus that follow-up burn treatments of this slash are critical, however, this can be logistically and financially challenging because of highly restrictive smoke management policies. Post-harvest slash burning typically involves burning of piled slash concentrations, and in some cases, broadcast burning of remaining fuels. Prescribed burning

[13] Fuels treatments and fuels management include planned prescribed burns, mechanical treatments such as mastication or thinning, and silvicultural treatments and other treatments designed to change or reduce wildland fuel quantity and arrangement, the intensity of future fires, and increase the ease of fire suppression.

can also be done independently of thinning to reduce surface and ladder fuels, and to reintroduce more natural fire to the ecosystem. There are often significant constraints to this sort of prescribed burning though. For example, where surface fuels are too abundant, and where tree density and layering are significant, burn-only treatments are difficult to execute with any certainty. Burn-only treatments are also highly influenced by favorable fire weather (moderate conditions are best to accomplish goals), availability of fire crews, and smoke management restrictions. Respondents support efforts to overcome these roadblocks so that more fire can be reintroduced. Additional prescribed burn considerations for managers include improving public support for them, helping to design fuel treatments that mimic historical fires, using more fire during the fire season, enlarging the number, size, and positive effects of burns, and decreasing undesired effects of slash burning. Prescribed burns also consume less fuel and are far smaller than large wildfires, thus they produce far less smoke and smoke exposure to the particle sizes that are most harmful to human health.

Many respondents noted that managed wildfire is underutilized. It can be a viable tool ecologically, despite operational constraints. Ideally, this will result in more area burned under less than extreme weather conditions, and more moderate-severity fire effects resulting in heterogeneity that can be more consistent with both the historical range of variability and long-term management goals, if only for altering where and how future fires burn. Practically, there are large areas where mechanical thinning is neither allowed nor feasible, for example in wilderness and roadless areas. Managing wildfire may be useful there for reducing fuel quantity and altering vegetation composition and heterogeneity consistent with management objectives and enabling policy.

Wildfires can sometimes be managed at less cost and less risk to firefighters in areas where other fuel treatments are neither feasible nor desirable. Advanced planning is needed, as is accepting long-term risk and smoke when such fires burn for many days. Public lands are sometimes mapped into zones designated for particular management, included allowing fire. Challenges include societal constraints (e.g., smoke, fear of fire, concerns about shifts in weather, and distrust of managers and scientists) and operational constraints (e.g., costs, long-term risks, timing, and suitable weather). Smoke from fires poses human health hazards and visibility issues. Despite best efforts, some managed wildfires will not go as planned. The biggest challenges are the expanding area of WUI, public and political perceptions of fire and smoke, and unpredictable changes in fire weather. When homes burn, the fear of wildfires and their smoke often fuels political support for aggressive fire suppression, which reinforces the current predicament. But there are beneficial aspects of wildfire smoke too. For example, in northwestern California, Mid-Klamath Basin tribes recognize benefits of canyon smoke inversions for reflecting direct sunlight, and cooling air and river temperatures that can benefit native salmonids. Smoke is also a naturally occurring fumigant that reduces nut, seed, and acorn infestations by forest insects, and along with fire, facilitates seed germination of some native plants if smoke occurs during the natural fire season.

Managing wildfires to accomplish resource benefits may be one important way to achieve relatively widespread vegetation change at the spatial scales and in the short time frame needed to make a difference in the short-term. Depending on the situation, this will typically require strategically pretreating a portion of the landscape using prescribed fire—sometimes coupled with thinning—to reduce vertical and horizontal continuity of fuels, and to anchor managed wildfire or prescribed burning treatments. Such strategies can help manage risks and help society be more comfortable with less aggressive fire suppression, especially in or near the WUI. In remote locations far from the WUI and most vulnerable infrastructure, fairly typical mixes (for the fire regime of interest) of low-, moderate-, and high-severity fires may be a desirable and achievable outcome that is compatible with forest resilience, despite the many challenges managers face in managing wildfires.

Where there is a high concentration of values at risk and sensitive human populations within the WUI, aggressive fire suppression and fuel treatments may be the only socially acceptable strategies. In these situations, managing forests abutting the WUI with thinning and prescribed or pile burning, and aggressive fire suppression, will be appropriate.

Areas of Divergence

Key areas of divergent opinion among respondents included:

- Appropriate locations, scale, and effectiveness of thinning aimed at reducing fire hazard.
- The scale of thinning that can be feasibly and repeatedly implemented relative to the scale of the need.
- Advantages and disadvantages of managed wildfires, including acceptable levels of risk.

Communities often feel a strong sense of urgency and need for hope in the face of threats from wildfires. For areas distant from the WUI and municipal watersheds, some respondents disagreed with the degree of urgency and scale of need for thinning and prescribed burning.

Another area of divergent opinion is the role that forest restoration and fuel treatments might play in charting a new course for forest resilience, especially where public lands are considered. Arguments for restorative treatments range from concerns that prescribed burning or other treatments are needed where the condition of many forests before wildfires can result in undesirable burn-severity and patch-size distributions, to the belief that treatments are not restorative because large areas can only be effectively restored by proactively working with wildfires that are assumed to be “natural.” A range of other arguments falls somewhere on this continuum. One challenge is that fuel treatments may not be performed at a necessary pace and scale, especially when coupled with operational maintenance costs over time. Another view is that fuel treatments are less important in areas that would have experienced some degree of high-severity fire, and these areas may have been widespread and have greater positive influence on biodiversity and wildlife than is currently understood. This is underscored by strong opposition to partial to complete post-fire logging (salvage) of fire-killed trees, because some snag forests provide valuable habitat, and there are concerns about ecological integrity. The crux of this disagreement is whether the dead trees are most useful for their commercial or ecological values. Another view shows that some areas of high-severity fire tend to burn again at high severity, and that efforts to treat fuels and re-create more varied successional and fuels mosaics can help break this cycle. Yet another view asserts that in some instances, burned forests would benefit from removing dead smaller trees that could constitute critical reburn fuels.

Some scientists’ opinions diverge regarding the relative importance of climate and weather (fuel moisture and availability to burn), and fuel quantity (Topic F). For example, for some, there is more acceptance of the utility of fuel treatments within dry forests than in cold subalpine forests. Further, many scientists differed on the scale of treatment needed to influence high-severity fire at landscape scales because of questions about treatment effectiveness given the large amount of fire that burns under extreme weather conditions. Indeed, most current large wildfires are not even finding fuel treatments at the current low level of application. There can also be problems with non-native invasive species increasing in abundance following thinning and/or fire, particularly in lower-elevation forests. Many disagreed on the extent to which levels of high-severity fire and landscapes have changed, and the degree to which fuel treatments far from the WUI are a net benefit. The degree of divergence differs by forest type and landscape context, with stronger agreement about landscape change for dry mixed-conifer forests, and less for landscapes dominated by cold subalpine forests. Another argument by some for not actively managing landscapes outside the WUI is that even where there is strong support for treatments, the cost and difficulty of implementing and maintaining existing treatments may already be too great for society to absorb; this consideration is beyond the scope of this report. Of course, societal cost and practicality must be considered in the context of potential loss of forests, fire-adapted biota and other resource and social values. Further, reduced reliance on fuel treatments might imply an increased use of managed wildfire, but there is currently no consensus framework for weighing the costs and benefits of managed wildfire.

Some divergent opinions derive from establishing forest treatment targets, especially when those targets are not yet socially acceptable. For instance, thinning from below (removing many smaller, fire-intolerant trees while leaving older, larger trees) will reduce fire hazard under many circumstances, and can be a first step in ecological restoration treatments in many dry mixed-conifer forests. However, these treatments must be followed by prescribed burning, and a certain amount of smoke production, to reduce fuels and potentially restore ecosystem processes in the short-term. In some cold subalpine forests, however, where fires are more often limited by weather than by fuels, fuel treatments beyond the WUI may be relatively ineffective when and where fires spread by long-range spotting.

Some respondents noted that current landscape conditions reflect suppression of most fires, effects of past logging, and land uses that have often resulted in landscapes that are more homogeneous fuel-wise, notwithstanding widespread fragmentation by roads. Some respondents argued that these more-homogenous landscapes are more vulnerable now to very large patches burned with high-severity fire relative to historical conditions. Others saw less divergence between present and past high-severity fire potential (see areas of divergence Topics B, E). Careful analysis is needed in each unique geographic location.

Another challenge is that treating large areas is difficult when there is strong level of distrust. Collaboration with diverse groups has in many cases strengthened trust, especially when treatment approaches have been altered through a consensus-building process.

Implications

Managers seeking to reduce human vulnerability to wildfire and enhance forest ecosystem resilience should have available to them a flexible set of management options that includes suppression, thinning and other fuel treatments, prescribed burns and managed wildfires, as well as broad education on both the essential roles of fire and on prevention of undesired human-caused fires. The uses of these various tools should depend highly on management priorities and local context, including vegetation structure and composition, legacies of past fuel treatments and land use, the historical range of variability, presence of houses and resources people highly value, and acceptance by people. In the future, prescribed fire and managed wildfire will be useful to increase or maintain forest structural heterogeneity and restore associated ecosystem processes. Fires can limit the extent and severity of subsequent fires.

Fires respond to interacting influences of climate, weather, fuels, topography, legacies of prior disturbances, and management. The relative importance of these factors varies across landscapes and through time. Those who express that increasingly extreme fire weather with climate change will increasingly override the importance of fuels argue that fuel treatments should be focused around the WUI with limited fuel treatment elsewhere. Their logic is that direct protection of human assets is the top priority on which to focus, and that fuel conditions are less important as fire weather becomes more extreme. Those who emphasize the importance of fuels to fire behavior urge strategic fuels management in both WUI and non-WUI forest landscapes, using a variety of tools and prescriptions as needed across dry, moist, and cold forest types. They also assert that fuels are the main landscape characteristic that management can change. Where fuels and vegetation patterns have changed to foster more contagious fire spread, fires will be widespread and often large when fire weather and fuel moisture are conducive, particularly where grass fuels are continuous.

Going forward, monitoring is important to assure that fire management supports long-term vegetation management goals, particularly in the context of climate change, or to modify management to better align it with goals. We need to learn where fuel treatments are effective under different environmental conditions and where they are not, and then we must adapt management informed by monitoring. Scientist-manager partnerships could be particularly useful here to develop useful monitoring frameworks.

Where managed wildfire is not socially acceptable, more aggressive fire suppression and fuel treatments will be appropriate, along with prescribed burning. Many wildfires will occur and some will be large. With thoughtful management, we may be able to influence their severity and spatial extent under many but not all fire weather conditions.

To address future challenges in the face of expanding WUI, longer fire seasons, and altered forest conditions, managers need many different tools to balance ecosystem needs, costs, risk to firefighters and the need to protect people and property from fire. Federal fire policy allows this flexibility, and fire managers need it if they are to reduce societal vulnerability to fire and smoke, while also limiting costs and risks to fire personnel, and managing for ecosystem values. Addressing the vulnerability of the WUI depends on other approaches as well. Thus, policies to make current WUI communities more fire adapted are critical, as are changes in land use policies that influence where and how future WUI areas develop.

Conclusions

We found much common ground that will be useful to scientists, managers, and others for moving forward. There is wide agreement among scientists that fire is one of the most essential and pervasive influences on the forests of the western US. Further, fires can produce more positive benefits and fewer negative impacts when they burn with an ecologically appropriate mix of low, moderate, and high severity, and in patch size distributions that reflect the natural variability of fire behavior and fire effects.

Many questionnaire respondents suggested that the real challenge is to face the twin realities of increased abundance and connectivity of woody fuels and a changing climate. Not surprising, there were differing opinions about trade-offs between social values and the ecological benefits of fire. For example, smoke from wildfires or prescribed fires is a great concern that can have important influences on how various fire treatments are applied. There was wide agreement on the need for land management strategies that reduce societal and ecosystem vulnerability to negative consequences of wildfire, while providing for the essential role and benefits of fire in forests.

Areas of agreement outnumbered areas of disagreement. Respondents agreed that geographical context is very influential and that human impacts vary, and therefore there is no single one-size-fits-all management prescription. There was strong support for utility of the historical range of variability (HRV) for fostering understanding of how and why ecosystems have changed, and how they respond to fires of varying severity. Despite rapidly changing conditions, HRV will continue to be useful as a guide, but not a prescription for future landscapes. From HRV we can learn how ecosystems respond to wildfires, and the HRV of landscapes and ecosystems forewarns about ecosystem capacities and limitations in response to varied climate and disturbance drivers. As a guide for managing future landscapes, history does not provide precise prescriptions, but does offer precautionary principles. We fully recognize that adaptive resilience for the future will require applying what we learn from history to some future range of variability, where fires burn and ecosystems respond in both similar and different ways. There was strong support for prescribed burning, coupling thinning with prescribed burning, and for managing wildfires to accomplish resources objectives. This is common ground.

Forest structure, composition, and fuels have all changed, affecting burn severity and successional patch size distributions. Climate and fuels together with topography will influence future fires and their effects. There was consensus that many fire management tools and strategies will be useful moving forward, and no tools should be excluded.

We challenge managers and scientists to overcome the tendency to oversimplify historical fire regimes across and within ecoregions and forest types: there is no single model of historical fire regimes. Managers should exercise caution when applying scientific understanding developed in different landscapes and recognize that this may result in erroneous scientific underpinnings and failure to meet local objectives. Rapidly changing circumstances suggest that future management should be highly adaptive, incorporating learning from what works well and poorly. To adopt an adaptive management stance though, managers will need to engage in ongoing monitoring to detect and learn more about the best and poorest methods and outcomes. Scientists can work with managers in these practices, and such partnerships could provide a potent resource for managers. Scientists must also clarify the importance of place when characterizing and presenting knowledge about historical fire regimes, and scientists and managers would both benefit from sharing methodological approaches and collaborating across ecoregions. Scientists and managers should work together with science communication experts to create training and reference materials that capture appropriate levels of simplification and complexity.

Broader discussions center on issues where there is less common ground, including:

High-severity fire

There was strong common ground that for dry pine and some dry mixed-conifer forests, there has been either an observed increase in high-severity fires or an increase in the potential for fires of elevated severity. These changes have occurred as the result of increased area and density of forests, and increased connectivity of woody fuels since the late 19th century. In contrast, for cold subalpine forests the majority of survey respondents agree with the statement that these forests have been less affected by fire suppression. Yet, large-scale landscape assessments of forest spatial patterns in the Inland Pacific Northwest show that recent (i.e., post 1984) patterns of high-severity fire and changes in patch size distributions in many moist mixed-conifer and cold, subalpine forests reflect significant departures from longer-term patterns linked to both climate, increased forest area, and increased density, layering, and connectivity of these forests. Expanded woody fuel connectivity is a result of synchronized successional conditions as a consequence of fire suppression and fire exclusion. Suppression of wildfires in moist and cold forests has yielded much lower prevalence of early seral conditions, and increased connectivity of mid- and late seral conditions, which has concomitantly increased landscape connectivity of conditions that are conducive to initiation and spread of crown fires. Conclusions differ for some cold subalpine forests in the Southern Rockies based on published studies and survey responses. Unfortunately, we don't have similar information on landscape change across moist and cold subalpine forests in some other ecoregions. Reasons for these different perspectives on the degree of long-term change in the extent of higher severity fire are varied and complex. Some of the variability in scientific perspectives is empirically attributable to geographical differences in the factors determining historical and modern fire regimes. Others reflect disagreements over methods of examining changes in fire regimes and the interpretation of the evidence of past higher severity fire. Still others reflect the goal of influencing management.

Dry mixed-conifer forests (including areas once dominated by pure or nearly pure ponderosa pine), moist forests, and cold forests have all changed in recent decades. The degree of change is not the same everywhere, yet fires interacting with climate and current forest conditions have the potential to create very large patches and a relatively high proportion of areas burned with high severity. This has implications for post-fire tree regeneration (without which forests convert to non-forest), soil burn severity, and related erosion and watershed change, and other ecosystem services valued by society and affected by varying plant successional processes and trajectories. Non-forest vegetation may be maintained by fire where it is not suppressed; even in the absence of fire, however, forests may not regenerate if seed sources are not available. In some areas, forests have increased and non-forest decreased, due to fire suppression.

Both fuel and climate are important as increased woody fuel connectivity in combination with a warming climate trend is setting large areas of landscapes on fundamentally new trajectories where very large patches burn with high severity. Climate is of increasing importance.

Empirical and simulation studies and landscape ecology theory suggest that even small increases in the frequency of the largest high-severity patches can have a semi-permanent influence on future local and regional landscape habitat configurations and wildfire frequency, severity, and spatial extent. Thus, individual fire events can change the broad-scale resistance of the landscape to future wildfires. How these scenarios will play out under continued warming and more fires is highly uncertain.

We suggest that resolving many disagreements depends on greater consideration of specific geographical context. A logical suggestion is to increase in-depth cross-regional field research experiences of the fire research community. Cross-regional comparisons of top-down and bottom-up determinants of fire activity in similar forest cover types is a fertile area of future research to examine how differences in seasonality, productivity, understory fuels, land use history, and other factors may explain some of the reported geographical differences in historical fire regimes in broadly similar forest types. Likewise, systematic regional comparison of the timing and nature of land-use practices by Native Americans and European settlers on fire regimes would improve our understanding of how changes in anthropogenic ignitions, fire exclusion, logging, ranching, mining, and landscape fragmentation may have contributed to geographic differences in historical fire regimes.

There are several reasons for the disagreements about patterns of past fire severity. First, both scientists and managers often uncritically transfer concepts and findings from one place to another (see Topic A). We know that fire effects at a point depend to some degree on the surrounding landscape and forest composition. Some of the disagreement derives from debates over the relative utility and validity of different scientific methods; nonetheless we believe that application of diverse research approaches is useful in HRV reconstructions. We challenge fire scientists who do not share similar perspectives on historical fire regimes in particular ecosystems to engage in civil discourse to better understand the reasons for their disagreement and to objectively communicate those reasons to managers and other stakeholders. We are heartened by the positive outcomes achieved by some previous attempts when small or large groups work together to find common ground.[14]

WUI and beyond

Respondents strongly agreed on the need for fuel treatments and aggressive fire suppression within and adjacent to the WUI. The strategies for managing wildfire will be quite different within and adjacent to the WUI than in areas far from the WUI. However, what fire managers do beyond the WUI has implications for forest resilience, smoke production and its human impacts, water quality, and many other ecosystem services people value.

Fuels management alone, especially if limited to public land, will not be sufficient to address the vulnerability of WUI communities to fires. Fuels management will be important for influencing the resilience of the future forest, and for influencing the behavior of wildfires that approach the WUI. Thus, policies to make current WUI communities more fire adapted (e.g., implementing current WUI codes) are a critical piece of the puzzle, as are changes in land use

[14] See: 1) Kaufmann et al. (2006) Historical fire regimes in ponderosa pine forests of the Colorado Front Range, and recommendations for ecological restoration and fuels management. *Front Range Fuels Treatment Partnership Roundtable, findings of the Ecology Workgroup.* 2) Romme, et al. (2009) Historical and modern disturbance regimes, stand structures, and landscape dynamics in pinon–juniper vegetation of the western United States. *Rangeland Ecology & Management* 62(3): 203-222. 3) Baker et al. (2017) The landscapes they are a-changin' – Severe 19th-century fires, spatial complexity, and natural recovery in historical landscapes on the Uncompahgre Plateau. Colorado Forest Restoration Institute.

policies that influence where and how future WUI areas might develop, and the spatial extent and arrangement of managed and wildfire fuel treatments.

Pattern and Process for Fires in Forest Landscapes

Heterogeneity of fire effects, including the pattern of patches created by fires and other disturbances, is important to forest resilience to future fires (see Topic E). There are potentially profound implications for forest function and carbon sequestration if the proportion of area burned with high-severity fire changes. Even more importantly, as the patch size distribution changes greatly, particularly with respect to the proportion and size of the largest patches burned with high severity, there are multiple ecological consequences. For instance, the proximity of seed sources from surviving trees of fire tolerant species can affect forest regeneration, and the future flammability of the forest. Wildlife habitat use will change, both for those species dependent on hiding and thermal cover adjacent to more open areas and for those thriving in recently burned forest openings. Similarly, the fire refugia that many species depend upon to bridge fire disturbances will all be greatly affected. Further, soil erosion potential is often higher when large patches burn with high severity.

Fires are essential to ecosystem function. Largely missing from many western landscapes are the historically most numerous small- and medium-sized fires that burn under less-than-extreme conditions of weather and fuels. Even when they don’t burn much area individually, such fires cumulatively shape landscape heterogeneity, the resistance of the landscape to wildfire growth, the frequency of large fires, and landscape capacity to respond to future large fires. Simulation modeling in forested landscapes suggests that even relatively small changes in the proportion of large patches alters system behavior. Thus, the patch-size distribution of low-, moderate-, and high-severity fires is of prime concern to policy makers, scientists, and managers.

Climate, Fuels, and the Implications of Landscape Change

Because fuels, weather, and topography dictate fire behavior, fuels management is important to efforts to mitigate fire behavior. However, mechanical treatment of fuels alone is not enough. Thinning without follow-up prescribed burning will typically worsen the problem. More flexible and extensive management of wildfires and prescribed fires will be essential, depending on local objectives and conditions, to increase the footprint of land areas showing reduced surface and canopy fuel abundance and connectivity. More extensive use of prescribed burning combined with thinning will be helpful, where forest fuel conditions (both surface and canopy fuels) are not currently manageable via wildfires and prescribed fires alone. The influence of prior fires on the extent and severity of subsequent fires, even when those fires burn under extreme weather and fuel moisture conditions, is a reflection of the importance of fuels. In some areas, forest conditions are such that some manipulation of fuels is needed so that key ecosystem elements are not lost in extreme fires. Many respondents accept that a proactive approach to fire and fuels management on public lands will reduce overall costs and improve climate change adaptation in the long-term. Some respondents questioned the practicality and effectiveness of fuel treatments under a changing climate; however the literature is clear that fuel reductions reduce flame length and fireline intensity, which reduce the likelihood of high-severity crown fires. Sound, science-based monitoring needs to be coupled with adaptive management to provide locally appropriate stewardship of our forests.

Decades of research in landscape ecology show that emergent properties have central importance to ecosystems and their pattern and process regulation, whereas many recent studies of climate-driven fire and vegetation change are less focused on local-scale feedbacks and emergent patterns. This creates a fundamental problem in linking climate change and landscape ecology research. Climate models assume that top-down climate covariates drive temperature, precipitation, and solar radiation conditions. Landscape ecology research shows that those top-down inputs can be highly modified by meso- and fine-scale bottom-up environmental controls to produce emergent climatic conditions that are strictly speaking neither the top-down or bottom-up inputs, but are influenced by these inputs. Climatic forcing alone poorly explains the shifts in landscape patterns because lagged patterns of historical disturbances continue to influence emergent patterns, under all but the most extreme events. The path forward to more effective projection of future fire and landscape change includes better integration of feedbacks from landscape ecological models into climate-driven models of future fire and landscape change.

In many landscapes, the increased abundance and connectivity of forests and fuels is favoring larger fires, and larger patches burned with higher severity. This is widely shared common ground for ponderosa pine forests and in some ecoregions is also applicable to dry mixed-conifer forests. This view is also commonly applied to moist and cold forests in the Inland Northwest, Pacific Southwest, Northern Rockies, and the Inland Southwest, whereas much less increase in fuel connectivity is believed to have occurred in the cold forests of the Southern Rockies. Regardless of uncertainties about departures from historical landscape conditions, there is a coherent argument based on first principles of fire spread that increasing forest patch heterogeneity could foster resilience to future fires, even as the climate changes. Thus, encouraging heterogeneity at various scales and in various processes is important for biodiversity, reducing connectivity of woody fuels, and increasing resilience with future climate change.

Effective management will depend on both science and trust

Our understanding of historical fire regimes can inform decision-making; indeed, such evidence-based decision-making can build trust. However, fire science points to complex patterns that vary with local conditions, so no single solution, such as logging or limiting all logging, will accomplish desired objectives in all forests. Further, no intervention also has consequences, so all decisions need monitoring to support the assumptions of management. Effective monitoring can improve knowledge, and through collective learning can build common understanding and trust.

Fire management can become more proactive and strategic. Existing tools, such as mechanical fuel treatments, prescribed fire, prevention of accidentally-ignited human fires, and managing wildfires will all be useful, but adaptation and mitigation responses to climate change and changing fire activity will require using these tools in strategic ways to fit area-specific goals. Some past disagreements about fire and fuel management strategies may be due to lack of clarity about specific goals, such as resident and firefighter safety, cost reduction, biodiversity issues, and ecosystem resilience under a changing climate.

The timing of fires is important, particularly in the context of a changing climate. While recognizing that wildfire seasons are long and getting longer, we must also take advantage of the milder fire weather and associated effects of fires in the “shoulder seasons.” Managers may find that both less-aggressive fire suppression and expanded use of managed wildfire under relatively moderate weather conditions can aid them where reducing the vulnerability of people and natural resources to fires is the objective.

One of the grand challenges of fire management is balancing the reality that wildfires will occur and are needed by western forest ecosystems, yet people, property, and economies need protection from the adverse effects of fire. Another grand and fairly urgent challenge is discovering the tipping points of transformative change for various forest landscapes in their respective geographies, where large, high-severity fires (regardless of whether they are considered unprecedented or not) may tip forest ecosystems into persistent non-forest states by constraining tree regeneration opportunities. Particularly as climate changes, we also need a deeper understanding of which landscapes may not be able to sustain forests in the future and how fast such transitions are likely to occur. It is clear that our western history of substantial forest fire activity will continue, one way or another: many fires will occur in the future and some will be large. Ultimately, we must find ways to sustainably use and live with fires that are well-adapted to both ecosystem and societal needs of local landscapes.

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ECOSYSTEMS

Compounded Perturbations Yield Ecological Surprises

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ABSTRACT

All species have evolved in the presence of disturbance, and thus are in a sense matched to the recurrence pattern of the perturbations. Consequently, disturbances within the typical range, even at the extreme of that range as defined by large, infrequent disturbances (LIDs), usually result in little long-term change to the system's fundamental character. We argue that more serious ecological consequences result from compounded perturbations within the normative recovery time of the community in question. We consider both physically based disturbance (for example, storm, volcanic eruption, and forest fire) and biologically based disturbance of populations, such as overharvesting, invasion, and disease, and their interactions. Dispersal capability and measures of generation time or age to first reproduction of the species of interest seem to be the important metrics for scaling the size and frequency of disturbances among different types of ecosystems. We develop six scenarios that describe communities that have been subjected to multiple perturbations, either simultaneously or at a rate faster than the rate of recovery, and appear to have entered new domains or "ecological surprises." In some cases, three or more disturbances seem to have been required to initiate the changed state. We argue that in a world of ever-more-pervasive anthropogenic impacts on natural communities coupled with the increasing certainty of global change, compounded perturbations and ecological surprises will become more common. Understanding these ecological synergisms will be basic to environmental management decisions of the 21st century.

Key words: altered community states; dispersal; multiple disturbances; recovery intervals; scaling disturbances.

INTRODUCTION

All natural assemblages are perturbed by both physical and biological forces. These agents of change occur with different intensities, frequencies, and spatial distributions. Some essentially scour the landscape, resetting the successional clock to time zero. More commonly, disturbances leave a residual assemblage that provides a legacy on which subsequent patterns build. We consider the range of single perturbations, from small-scale/frequent disturbances to large/infrequent catastrophes, to be central to much traditional ecology; such directional or cyclical changes stimulated the development of ecology's first paradigm, succession (Cowles 1899; Clements 1905, 1916). A century of accumulated detail on the interplay between pattern and process has provided descriptors for the nature of successional change and system-dependent rates of recovery. There are few surprises embedded here: depending on the time frame of interest, species arrive and depart, canopies or other structures develop, and the system "recovers," converging on the predisturbance state at a rate reflecting the spatial extent and intensity of the disruptive forces. Such patterns have been extensively reviewed (Pickett and White 1985), and variation in recovery dynamics can be attributed to different processes acting independently or in concert (Drury and Nisbet 1973; Con-

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nell and Slatyer 1977). Even large, infrequent disturbances (LIDs) do not appear to override the biotic mechanisms that structure eventual recovery. For example, the 1988 Yellowstone National Park fire, which burned 36% of the park and was an order of magnitude larger than comparable large, infrequent fires, has to date generated no ecological surprises: "the postfire ecosystems are shaping up to be essentially the same as those that prospered before the flames" (Stone 1998: 1527). We argue that cycles of disruption and recovery are the usual state of affairs and submit that rapidly compounded perturbations have more serious implications for long-term alterations of community state, occasionally or even often generating a different assemblage of species.

Physical agents of change are well documented and described by such terms as windstorm, landslide, forest fire, flood, hurricane, and volcanic eruption. Many of these are primarily of terrestrial importance and leave their signature on landscapes as sites with recognizable boundaries and measurable shapes and areas. Biologically based counterparts—clear-cutting of terrestrial forests and trawling on the ocean floor—generate similar map properties. Populations are also subject to biological disturbances that vary from slight to catastrophic. Although these may lack spatially discrete boundaries, their implications for community structure can be at least as profound. Here we combine, when appropriate, biological disturbances like pestilence, population eruptions, invasions, and overharvesting with the more traditional physical forms of disturbance. In so doing, we add an animal and therefore a trophic dimension to a subject traditionally dominated by plant ecologists.

Figure 1 is a heuristic portrayal of our approach. In the top panel, a single large disturbance is followed by eventual return to some baseline condition at which point the assemblage can be considered "recovered." The diverse literature on succession is primarily concerned with this pattern and its underlying mechanisms. The following panels identify our focus. In the middle panel, two large disturbances are shown to occur nearly simultaneously or in close progression. We believe that recovery, if possible, is often substantially delayed under such conditions, and we provide examples below. In the bottom panel, a major disturbance is superimposed on an assemblage already maintained in an altered state, usually by anthropogenic processes. Current examples could include populations depressed by persistent overfishing, whole systems altered by chronic pollutants, or the developing impacts of climate change, such as the apparent increases in frequency and intensity of major storms

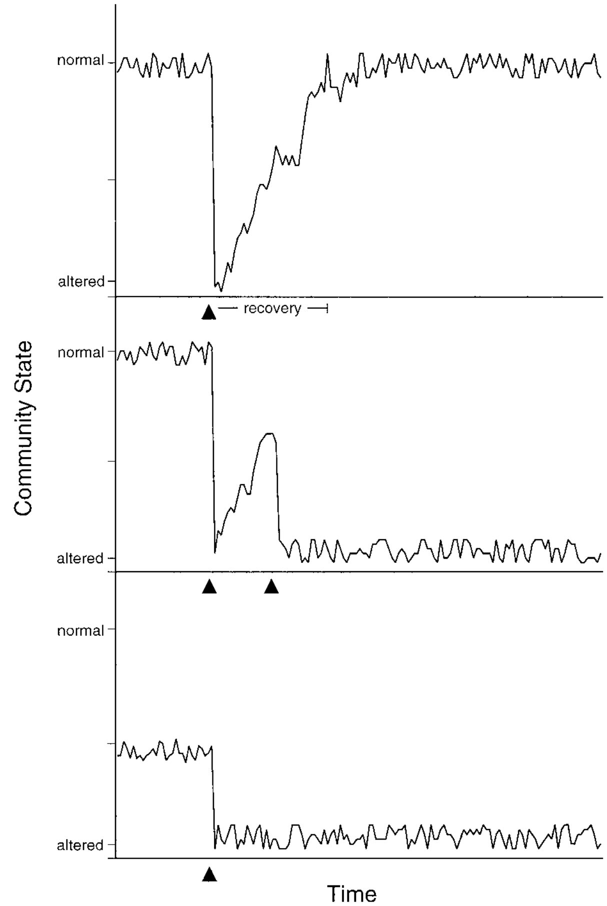

Figure 1. Schematic representation of the effects of large, infrequent disturbances (LIDs) on community state. Top, A normal community is subjected to a single LID and subsequently recovers. Middle, A normal community undergoes a second (or multiple) disturbance(s) before recovery from the first is completed; the combined effects lead to long-term alteration in community state. Bottom, A major disturbance is superimposed on an assemblage already already altered by anthropogenic processes or disease; again the combination of stresses leads to long-term alteration of comunity state. Arrowheads mark the disturbances.

and other climate extremes with increasing temperatures [for example, see Flavin (1996)]. Jansson and Velner (1995: 332), for example, suggest that in the Baltic Sea, a brackish body of water with minimal connection to the North Sea, which has been heavily impacted by eutrophication and toxic inputs, "pollution has reached the point where damage may be irreversible."

The scenarios discussed next include systems that appear, albeit temporarily, to have entered a new ecological domain; that is, they have not recovered. They share two features in common. First, all have been subjected to large (based on duration or spatial magnitude) and severe (quantified as a major mortality event) perturbations that may be physical or biological in origin. Second, these have occurred either simultaneously or in a sequence rapid enough that recovery from the single pulse has not significantly progressed. In some instances, three or more pulses appear to be necessary to initiate the changed state. Another way to describe our concern is to recognize that the community effect of compounded perturbations is multiplicative, not additive. If true and general, ecological surprises should be increasingly commonplace, prediction of recovery rates and trajectories less certain, and management more difficult.

Quantifying Disturbances and Their Frequencies

We adopt the definition of disturbance used by White and Pickett (1985: 7): "A disturbance is any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment." Pickett and White (1985) make the point that *disturbances* span three orders of magnitude of time (years) and ten of space (m^2). Disturbances, while causing spatially identifiable mortality to some species, usually provide open or invadable space for others, often renewing resources in the process. Sometimes, they can be identified by the resultant patchiness, recognizable by shape, size, postdisturbance internal composition, and spatial distribution. These patterns also have a dynamic, especially frequency of formation and rate of return toward the predisturbance state. Turner and colleagues (1997) suggest that it is also necessary to recognize the roles that individuals or species surviving a disturbance event can play in mitigating the event's impact. They suggest that only events characterized by few "residuals" be considered as large. We concur.

Disturbances can mean high mortality, often death of all individuals in the disturbance area. LIDs have different meanings for different ecosystems. For example, a large disturbance in the tidal zone may be on the order of tens of square meters, whereas in a forest it may be thousands. Hence, some scaling relationship must be used to ensure that these terms have equivalent meanings between systems. If, for example, we choose population dynamics as the processes of interest, disturbance size and frequency could be scaled by birth, death, or immigration (dispersal). Dispersal seems a particularly significant scaling metric for it governs the rate of recolonization of the disturbed site. Thus, ruderal (fugitive) species are typically both early invaders and excellent dispersers. Greene and Johnson (1989) applied a scaling metric involving seed terminal velocity, height of seed release, and mean horizontal wind speed to make dispersal comparable between species with different characteristics. For instance, ashfall accompanying the eruption of Mount St. Helens in 1980 greatly reduced many insect and spider populations. Because adult female leafhoppers (*Errhomus*) lack long-distance dispersal, they were slow to return to preeruption abundances in contrast with spiders (Showalter 1985). Similarly, on rocky marine shores, the poorly dispersing brown alga *Postelsia* can be driven locally extinct (Paine 1988 and unpublished), whereas local extinction is highly improbable for the associated barnacle *Balanus glandula,* whose larvae can traverse hundreds of kilometers. Thus, defining when a disturbance is large depends on the interplay between the spatial magnitude of the area disrupted and the dispersal (reinvasion) ability of the species of interest. Similarly, the frequency of disturbance could be scaled by some measures of generation time or age of first reproduction. In contrast, scaling on the basis of size of the dominant organisms does not appear to be a useful metric; for example, consider giant kelp (*Macrocystis pyrifera*) and terrestrial trees: sizes are comparable but time scales for age to first reproduction and life spans differ by orders of magnitude.

Ecological evidence seems to indicate that most LIDs do not override the biotic mechanisms governing species composition: in many disturbances, the postdisturbance composition is similar to the predisturbance composition (Turner and others 1997). This result might have been anticipated if size and frequency of disturbance had been scaled in terms of dispersal distances and generation times instead of the quantity of hectares and years as usually used.

Two other kinds of disturbances lack spatially explicit features but can have equally significant consequences: (a) Populations can be thinned commercially or reduced to mere vestiges of their original abundances by disease. For example, the majority of commercial fish stocks in US and Canadian coastal waters have been overexploited or are currently at maximum sustainable yield levels (NOAA 1993). Rinderpest decimated herds of African ungulates, especially buffalo (Sinclair 1977). The community changes resulting from density shifts of dominating species, often of high trophic status, can be extensive. They are a biologically based disturbance and often leave no immediate spatial signature. (b) Global climate change, a more cosmic form of disturbance, will surely have a substantial though currently unquantified and debated impact. It will provide a background of change in which both physical (for example, fire) and biological (for instance, harvesting and disease) disruptive forces will operate, and it might become the dominant influence on community structure and change in the coming decades despite its subtlety.

When do changes in community composition occur or under what conditions can they be anticipated? We believe that such dramatic shifts are most likely when both the spatial extent and especially frequency of disturbance are at the extremes of normal expectations. Multiple, usually sequential occurrences of these extreme and rare events can produce alternative stable states, that is, abnormal conditions or ecological surprises that defy the norm. The following scenarios describe communities that have been subjected to multiple perturbations and appear to have entered (or be facing) new domains.

Compounded Perturbations in Ecological Time

El Niños, Storms, and Kelp Bed Recovery

El Niño–Southern Oscillation (ENSO) events are large-scale oscillations of the tropical Pacific Ocean–atmosphere system with far-reaching climatic and economic impacts. The 1982–83 El Niño was widely considered the strongest of the century and had a corresponding impact on forests of giant kelp along the coasts of Alta and Baja California. Winter 1982–83 was the most severe storm season in many decades, as atmospheric teleconnections linked to the warming of the eastern equatorial Pacific Ocean affected the Aleutian low-pressure center, generating a large number of severe storms from an unusual southerly direction. These storms devastated kelp forests throughout the range of *Macrocystis pyrifera,* an economically valuable kelp. Anomalous poleward flow of warm, oligotrophic waters that rendered upwelling ineffective led to nutrient depletion, massive loss of kelp biomass, and extensive summer mortality in the southern half of *M. pyrifera*'s range. The severity of the warm-water effects was related to latitude. In South America, anomalies were as high as 11°C during 1982–83; there was mass mortality of *M. integrifolia* and associated animal populations in Peru and northern Chile (Dayton and Tegner 1990). In the southernmost part of the range in Baja California, giant kelp went extinct in some areas and site preemption by lower standing kelps prevented its recovery after the ENSO.

In the San Diego region, the combined effects of the storms (ENSOs can be storm free) and the 4°–5°C warm-water anomalies constituted the most severe disturbance of a giant kelp forest community ever documented, yet recovery was rapid once conditions returned to normal (Tegner and Dayton 1987; Dayton and others 1992). At the other end of the range in central California, the intensity and duration of the warm event were smaller and conditions remained within the suitable range for kelp [reviewed by Tegner and Dayton (1987), Dayton and Tegner (1990), and Dayton and others (1992)]. Kelps are well adapted to winter-storm disturbance, with correspondingly timed reproduction, spore dispersal tied to water movement, and success of the propagules a function of open space. The more problematic warm-water effects result from the severity and duration of the events and probably the frequency, as well.

ENSOs are natural climate variations to which communities have been subjected for at least hundreds of years, but there are major questions about whether the intensity and/or frequency of these events may change as a result of global warming (Trenberth 1995). The observational record indicates that ENSO events have changed in frequency and intensity in the past century, but the frequency of strong events appears unchanged back to 1625 (Enfield 1988). Coupled ocean–atmosphere general circulation models find that ENSOs will continue to exist in a warmer world, but have yet to address frequency and intensity. Trenberth (1995) suggests that because these events have the effect of creating droughts and floods in different parts of the world and global warming tends to enhance the hydrological cycle, there is a real prospect that future ENSOs will be accompanied by more severe droughts and floods. For Northern Hemisphere kelp forests, the question may be whether ENSO thermal additions to already warmer water conditions [for example, see Roemmich and McGowan (1995)] in the future can push kelps beyond the range of recovery, especially in the southern end of the range.

Climatic Extremes and Exotic Species in San Francisco Bay

San Francisco Bay, at the mouth of rivers draining 40% of California, is considered to be the major estuary in the United States most modified by human activity (Nichols and others 1986). Many alterations of the ecology and the bay, such as loss of habitats, water-quality changes, introduced species, and excessive freshwater diversion, date to the 19th century, yet recent disturbances have led to profound changes. In late 1986, the euryhaline Asian bivalve mollusc *Potamocorbula amurensis* was first sampled in San Francisco Bay (Carlton and others

1990; Nichols and others 1990). Within 2 years, it had spread throughout the estuary, on all sediment types and water depths, and reached densities at some sites exceeding 10,000 m^{-2}. This invasion almost certainly resulted from the discharge of seawater ballast from cargo vessels.

Two years of climatic extremes apparently contributed to this remarkable population explosion (Nichols and others 1990). Before *P. amurensis* was discovered, the benthic community in Suisun Bay (northern region of the bay) varied predictably with river inflow: years of normal or high flow were characterized by brackish or freshwater species, and years of low flow by estuarine species. The end of the 1984–85 dry event was marked by an extreme but short-lived flood that eliminated the estuarine species. Thus, when *P. amurensis* was introduced, the Suisun Bay region was inhabited by a disturbed and depauperate community that may have contributed to the initial success of the invader. The invader's timing after the flood guaranteed it months to exploit the available space before the dry-period species would return, and by 1988 the near absence of the dry-period community demonstrated how well *P. amurensis* had displaced the former community. The ability of the invader to live in low-salinity water suggests that it will not be displaced with the return of normal river flow and that the benthic community is permanently altered (Nichols and others 1990).

Carlton and colleagues (1990) predicted significant community changes as this abundant consumer, competitor, disturber, and prey altered the interactive trophic webs in San Francisco Bay; these are beginning to unfold. Within a year, chlorophyll concentration and adult abundance of three common copepod species had declined by 53%–95%; these values persisted through 5 years of study (Kimmerer and others 1994). Before 1987, chlorophyll concentration varied with river flow; after mid-1987, it remained low despite variations in flow. The effect on copepods appears to be via direct clam predation on nauplii; egg production was not affected. Estimates of clam clearance rates are consistent with the reduction in copepod abundance. Although it may be premature to forecast permanent changes in the zooplankton populations, Kimmerer and colleagues (1994) voice serious concern: several species of fish that pass their larval lives in the upper estuary are also in serious decline. Moyle and coworkers (1992) list five species, including those in valuable sport and commercial fisheries, in which poor first-year classes correlate with reduced freshwater outflow, presumably because of decreased survival of larvae and juveniles. A sixth, the delta smelt, which is federally listed as threatened, feeds primarily on copepods, has a narrowly defined habitat in the mixing zone between fresh and salt waters, is an annual species very sensitive to environmental fluctuations, and has declined in concert with increasing water diversions since 1984 (Moyle and others 1992). Thus, the compounded effects of two major disturbances—one biological (the successful establishment of a nonindigenous species) and the other physical (drought followed by an extreme flood)—have initiated sweeping and probably permanent changes in ecosystem structure.

Boreal Forest Wildfires, Forest Fragmentation, and Logging

Fire frequency in the boreal forest is primarily controlled by large-scale climate processes, specifically persistent midtropospheric anomalies that at the surface are expressed as blocking high pressures (Schroeder and others 1964; Newark 1975; Street and Birch 1986; Flannigan and Harrington 1988; Johnson and Wowchuk 1993). In this century, however, agricultural settlement in the southern fringe of the boreal forest and logging further north have resulted in a new multiple-disturbance regime that has caused significant changes in forest composition.

In the last three centuries, fire frequency in the boreal forest has changed several times, each time associated with large-scale climate changes (Johnson 1992). The changes in the early 1700s and the 1800s were a result of the Little Ice Age (Grove 1988). In the boreal forest, this period had more frequent fires than the periods before or since (Bergeron and Achambault 1993). These changes in fire frequency appear to be associated with increased numbers and length of persistent midtropospheric anomalies. Years with large areas burned are known to have more sequences of days during the fire season with warmer, drier weather, which are associated with upper-level ridges. These persistent upper-level ridges over the boreal forest are *teleconnected* (spatially and temporally correlated) to upper-level troughs in the North Pacific and eastern North America. This teleconnection is called the Pacific North America pattern (Rogers 1981; Wallace and Gutzler 1981; Knox and Lawford 1990; Johnson and Wowchuk 1993). Similar patterns have been described in the southwestern United States as a result of ENSOs (Swetnam and Betancourt 1990).

The transition periods between different fire frequencies, for example, at the end of the Little Ice Age, seem to have been periods in which fires occurred more erratically for a decade or more. Often these periods were marked by clusters of years with very large areas burned and many persistent upper-level ridges. One could speculate that the large areas burned in the boreal forest since the 1980s are a result of one of these transition periods, perhaps related to global warming. However, more understanding of these transient processes is required before anything definitive can be said.

These climate-driven changes in fire frequency have generally been part of the ecosystem dynamics of the boreal forest for millennia. In this century, two new classes of large-scale disturbances were added to the climate-driven fire frequency. These anthropogenic forces were agricultural settlement in the southern boreal forest and logging.

Homesteading in the early 1900s led to progressive clearance of forest in the southern fringe of the boreal forest in western Canada (Vanderhill 1958). The effect on the forest north of the fringe was to increase the frequency of fires above that of the natural lightning fire regime. The increase was due to escaped clearance fires spreading from the settlement areas north up to 50–60 km into the unsettled forest. The result of this major increase in fires meant that trees that required longer time to reach sexual maturity, did not have serotinous cones, had little or no vegetative reproduction potential, were greatly reduced in abundance, and in many areas were locally exterminated (Weir 1996). For example, white spruce (*Picea glauca*) and balsam fir (*Abies balsamea*) both became relatively unimportant trees in many forests and trembling aspen (*Populus tremuloides*) significantly increased in abundance. Also, change from a mixed-wood (conifer–deciduous) to primarily deciduous forest has caused many other changes in plant and animal species (Weir 1996). At the same time that this forest was being subjected to an increase in the frequency of fire, high-grade logging (cutting of only large trees) of the white spruce was further reducing this dominant boreal species.

The southern boreal forest today has a significantly different composition than it did a century ago. This change has been due to multiple perturbations: a natural lightning fire regime augmented by settlement fires spreading from adjacent areas and logging.

Early Succession and Exotic Species

Volcanic eruptions clearly embrace the concept of disturbance, either by presenting new landscapes and initiating primary successional processes, or by altering preexisting ones via ashfalls, pyroclastic scorching, and the like. The end result of the recovery/regeneration process seems fairly predictable: Turner and colleagues (1997) compare the assembly of the plant community on Mount St. Helens (WA) following its 1980 eruption with other single large infrequent disasters. On this barren landscape, some degree of successional uncertainty may well characterize the early stages of the recovery process. Morris and Wood (1989) found in experiments on lupine, a nitrogen-fixing pioneer species, evidence that two other invaders could be either facilitated or inhibited. Such stage dependency complicates the successional process; it probably does not alter the ultimate community composition, although insufficient time has elapsed since the eruption to evaluate the consequences of these initial uncertainties.

The Hawaiian Islands are also of volcanic origin and, despite being earth's most isolated archipelago, have been invaded by 4600 exotic plants, 86 of which represent serious threats to the native ecosystem (Vitousek 1990). The successful invasion of a nitrogen-fixing exotic (*Myrica faya*) on the slopes of Hawaii Volcanos National Park provides a classic example. A 1959 eruption deposited 1–2 m of ash on the native vegetation, thinning it substantially. *Myrica* invaded and initiated a series of changes including the identity of the dominant tree, nutrient cycling, and productivity. For instance, Woodward and coworkers (1990) showed that native Hawaiian birds, while visiting *Myrica,* rarely feed on its fruit. Nonnative species visited, fed, and effectively dispersed viable seeds. *Myrica* itself is fecund: Vitousek and Walker (1989) estimate the seed rain at 4.6 million/ha under 21 adult *Myrica*/ha. In addition, the mean adult growth of these invaders is approximately 15 times that of a native tree (Vitousek and Walker 1989). *Myrica* is quadrupling the input of soil nitrogen (Vitousek 1990); earthworms are 2–8 times as dense under it than under native vegetation, which will change litter-processing dynamics and the rate of accumulation of soil organic matter (Aplet 1990).

As Vitousek (1990; Vitousek and Walker 1989) has demonstrated, the changes in whole ecosystem-level function are substantial. One major speculation is that when, or if, *Myrica* is replaced during primary succession, it will be replaced by another exotic. The competitively aggressive strawberry guava is a likely candidate species. Because, in general, sites with more fertile soils—higher concentrations of soil nitrogen in a system where N is a limiting resource—are conducive to invasion, "nitrogen fixation by *Myrica* will ultimately favor invasion by a broader range of exotic species" (Vitousek and Walker 1989: 261). Finally, as these authors note, invasion changes the ground rules governing coexistence of native assemblages recovering from or responding to disturbance. The problem is of great pragmatic importance to conservation biology: it further signals the existence of surprises at ecosystem levels when disturbances, in this case volcanism and biological invasion, are compounded.

Hypoxia in the Northern Gulf of Mexico

Oxygen depletion, long known from confined bodies of water, such as basins, fjords, bays, and estuaries, is increasingly reported from coastal ocean environments (Boesch and Rabalais 1991; Diaz and Rosenberg 1995). This may take the form of anoxia, where dissolved oxygen concentrations are essentially zero and hydrogen sulfide (toxic to metazoans) is detectable, or more commonly hypoxia, where oxygen concentrations on the sea floor are reduced to levels low enough to cause severe stress and mass mortality of benthic and water-column fauna. Varying with the severity of the oxygen depletion, effects on the biota range from avoidance of the affected area, to mortality of more sensitive taxa such as crustaceans and echinoderms, to emergence of the redox potential discontinuity from the sediments, a condition where only chemoautotrophic bacteria can live. Diaz and Rosenberg (1995) report that no other variables of such ecological importance to coastal marine ecosystems have changed so drastically in such a short period of time. Increasing evidence of oxygen depletion in coastal ecosystems is associated with anthropogenic eutrophication and, when eutrophication is coupled with adverse meteorological and/or hydrodynamic events, hypoxic events increase in frequency and intensity.

The inner and middle continental shelf of the northern Gulf of Mexico from the Mississippi River Delta to Texas is the largest and most severely affected area in North America subject to seasonal hypoxia [operationally defined by dissolved oxygen levels less than 2 mg/L or less than 1.4 ml/L (Rabalais and others 1991)]. From 1985 to 1988, hypoxic waters were found from April to October, from 5 to 60 m water depth, from 5 to 60 km offshore, extended up to 20 m above the bottom, and covered up to 9500 km^2. Hypoxia in this region is coincident with strong, salinity-controlled stratification during the warmer months of the year, which restricts reoxygenation of the bottom waters. Large quantities of decomposing phytoplankton biomass fueling intense water-column and benthic respiration rates enhance the effects of stratification on oxygen depletion. Although hypoxia did not cause extensive mortalities on the northern Gulf of Mexico shelf until 1978, it has occurred almost annually since it was first discovered in 1973 (Diaz and Rosenberg 1995). Severity and extent vary interannually with river flow, shelf circulation, and tropical storm mixing (Rabalais and others 1991).

The importance of the extent and duration of the hypoxia on the Louisiana shelf relates to fishery landings from this state, which constitute 28% of the US total (Rabalais and others 1991). Abundances of finfish, shrimp, and swimming crabs are severely depressed in hypoxic areas, and the period of oxygen stress includes critical life-history periods of several commercially important species. The macrofauna either succumb or move to avoid stressful conditions; typically, demersal species have been observed high off the bottom where mortality due to predation is undoubtedly high (Boesch and Rabalais 1991). When hypoxia persists, only highly resistant taxa such as some polychaetes and nematodes survive. Posthypoxia community dynamics depend on the extent of the mortalities, age of affected populations, timing with respect to availability of recruits, size of the affected area relative to short-dispersal recruits, frequency of hypoxic stress, and degree of organic carbon buildup. Because of the recurrent nature of hypoxia in the northern Gulf of Mexico, there are few large or long-lived sedentary species, and the benthic community is dominated by opportunistic species characteristic of an early successional state (Boesch and Rabalais 1991; Diaz and Rosenberg 1995). Intensified commercial fishing on the continental shelf in the 1970s and 1980s has been accompanied by alarming declines in the estimated sizes of remaining fish stocks; although overexploitation is clearly important, Darnell (1992) suggests that habitat deterioration is affecting both nursery areas and food chains for commercial species.

The Mississippi River and its distributaries drain 40% of the coterminous United States. Nitrogen concentrations in the rivers, the major source of "new" nutrients to the offshore phytoplankton, have more than doubled since the mid-1950s (Rabalais and others 1991). Turner and Rabalais (1994) recently demonstrated a close coupling between riverine loading and phytoplankton production; changes in biologically bound silica in the sediments below the river plume were virtually coincident with increases in nitrogen loading. After major flooding in 1993, the Gulf of Mexico hypoxic zone doubled to 18,000 km^2 and has not shrunk much since (Kaiser 1996). On 12–13 August 1996, winds forced oxygen-depleted water from the offshore dead zone below the mouth of the Mississippi River close to shore. This caused a "jubilee" along 36 km of Louisiana coastline, a condition where shrimps, crab, and finfish crowd close to shore to escape the oxygen-deficient water—highly increasing their susceptibility to fishing (Buck 1996). Kaiser (1996) reports that the US federal government may be finally waking up to warnings that the Gulf of Mexico hypoxic or "dead zone" may be one of the nation's worst ecological problems. A multiagency committee has been convened to discuss the problem and recommend management steps, such as voluntary reductions in fertilizer use in the Midwest. Again, compounded perturbations—coastal eutrophication exacerbated by extreme meteorological events—have produced an altered community state.

Phase Shifts in Jamaican Coral Reefs

Like all communities, coral reefs are subject to occasional intense natural disturbances: plagues of the starfish *Acanthaster* (Birkeland 1982), hurricane devastation (Woodly and others 1981), and high temperature stress (Gates 1990). Long-term studies of reef structure along a depth gradient at Discovery Bay, Jamaica, provide a clear and sobering view of phase shifts in assemblage structure associated with a compounded series of severe disturbances.

Baseline data at the main site exist from the 1950s (Goreau 1959), and the site has been repeatedly examined quantitatively since then (Andres and Witman 1995). A sequence of events initiated in the early 1980s, but set against a subtler background of increasing coastal pollution and extreme harvesting pressures on herbivorous fishes, has led to what Hughes (1994) calls "large scale degradation" and vividly portrays as a phase shift in community structure. Depending on the depth at which corals are sampled, percent cover has decreased from 30%–60% in 1977 (Huston 1985) to approximately 5% at depths less than 30 m in 1992 (Andres and Witman 1995). Conversely, algal cover, accounting for less than 5% cover in 1982, comprised approximately 70% cover in 1992. As a result of algal preemption of space, larval recruitment of all corals failed after 1984 (Hughes 1994).

The compounded disturbances—two major hurricanes (Allen in 1980 and Gilbert in 1988) and mass mortality of an ecologically significant grazer, the sea urchin *Diadema,* from 1982 to 1984—occurred well within the normal recovery interval of reefs. Hughes (1994) suggests that reef regeneration was initiated shortly after Hurricane Allen despite reduced grazing fish populations. The *Diadema* dieback delivered the coup de grâce and the recovery trend was reversed. Andres and Witman (1995) suggest that Hurricane Gilbert only slowed the developing domination of algae and therefore failed to enhance coral recovery. Furthermore, these authors imply that if urchin and fish populations had retained their pre-Allen levels, coral recovery and domination of the benthos would have occurred. With continued depression of herbivore populations, recovery of the coral assemblage is not forseeable.

In this case, a variety of disturbances, occurring rapidly relative to coral regeneration capacity, have yielded a novel community state—one that would not have developed if the disturbance events had occurred individually at intervals appropriate for reef recovery.

Discussion

A world of ever-more-pervasive anthropogenic impacts on natural communities coupled with the increasing certainty of climatic response to human activities suggests that compounded perturbations and ecological surprises will become more common. The frequency of disturbances is often scaled in terms of severity and return intervals: for example, the 30-year flood or the 100-year storm. The large number of record weather events in the news in recent years raises the issue of climate change and its interactions with LIDs. We reiterate our belief, and concern, that ecologists must refocus their interest from the ordinary (for instance, recovery sequences between normally spaced disturbances) to the extraordinary (for example, rapid sequential disturbances occurring against a background of increasing climate change).

We do not question the capacity for single LIDs to distort ecosystem structure and functioning temporarily. They obviously do, and merit investigation in their own right. On the other hand, the ecological literature is replete with underappreciated studies of compounded disturbances. Here we identify two more to suggest their ubiquity. Zedler and colleagues (1983) describe the interaction of fire and an annual grass planted to control erosion. Their conclusion that (p. 817) "We doubt if traditional climax-oriented successional theories can be of much use in predicting the outcome" resonates with our theme. In addition, several of the case histories developed in Gunderson and coworkers (1995) suggest that maladaptive management actually decreases ecosystem resilience, thus increasing susceptibility to subsequent disturbances, perhaps even facilitating disturbances and thwarting effective management.

The 1995 Intergovernmental Panel on Climate Change report (IPCC) (Houghton and others 1996), the international consensus summary of climate science, reviews the low probability that observed changes could be caused by natural factors alone, the average rate of warming for the 21st century that is likely to be warmer than any seen in the last 10,000 years, and projections for sea-level rise. Globally, existing data do not offer statistical evidence that extreme weather events or climate variability have increased in the 20th century, although there is clear evidence on regional scales. The IPCC review of model results finds agreement on predictions of general warming with regional variability and an enhanced global mean hydrological cycle. Regional forecasts are more problematic, but the incidence of floods, droughts, fires, and heatwaves is expected to increase in some areas and decrease in others as temperatures rise (Houghton and others 1996).

Our scenarios were chosen to identify the range of possibilities and the resulting ecological surprises. We have made no attempt to be encyclopedic. Rather, our list is characterized by events with blurred or no discernible borders, except as dictated by local geography (San Francisco Bay, for example) or study at a specific location (for example, Jamaica). Missing entirely are those great disturbances with sharp spatial boundaries as often characterize tornado paths or intertidal mussel-bed destruction. Missing also are references to the human condition, for instance, the interplay between malnutrition and disease. The growing evidence for unusual meteorological events leading to outbreaks of disease infectious among humans, such as hantavirus, cholera, and plague (Linden 1996), provides numerous examples. In one well-studied case, Colwell (1996) relates the massive cholera pandemic that struck South America in 1991 to the transport of zooplankton in a freighter's ballast water to the coast of Peru at the onset of an El Niño event. The El Niño brought rain and the influx of nutrients from land as well as warm sea surface temperatures, factors associated with the initiation of plankton blooms. This El Niño event, which lasted from 1990 to June 1995 and is the longest on record since 1882, may also be associated with the cholera bacterium remaining endemic in the region (Colwell 1996).

Present and future global climate-change effects on the environment may be debatable, but this is not the case for increasing human impacts: population growth, urbanization, deforestation, eutrophication, overfishing, loss of habitat, and so on. In ironic contrast with many political leaders, the insurance industry has recognized the significance of the nexus of these issues. Faced with rapidly escalating insured costs for increasing numbers of severe catastrophes over the last decade, that industry is realizing that it must demand political action to protect the climate to prevent its own financial ruin (Flavin 1996; Munich Re 1996). Our perspective is that societies and ecologists must begin to prepare themselves for novel and unanticipated consequences of previously well-understood phenomena. Jamaican coral reefs may not recover, Hawaiian volcanic slopes may develop a novel flora, and San Francisco Bay appears to be acquiring a new and not necessarily desirable invertebrate assemblage, in the process losing native species. Global warming may accelerate the effects of oxygen deficiency and enlarge affected areas in the Gulf of Mexico. These altered and possibly alternative states (Lewontin 1969; Sutherland 1974) may or may not be persistent (stable). On the other hand, mounting evidence suggests that sequential, large-scale disruption of the current state will make these altered states the ecological reality of the future. Understanding the role of compounded disturbances, some natural and others of anthropogenic origin, will be basic to environmental management decisions in the 21st century.

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Western Watersheds Project

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What's Eating the Pando Clone?

Two Weeks of Cattle Grazing Decimates the Understory of Pando and Adjacent Aspen Groves

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What's Eating the Pando Clone?

Two Weeks of Cattle Grazing Decimates the Understory of Pando and Adjacent Aspen Groves

by Jonathan B. Ratner,[1] Erik M. Molvar,[1] Tristan K. Meek,[1] and John G. Carter[2]

EXECUTIVE SUMMARY

The Pando Clone is an aspen grove on the Fishlake National Forest in south-central Utah that was heralded in 1992 as the world's largest single living organism. Adult trees that are joined by a single rootstock and share identical chromosomes comprise the Pando Clone and, like many aspen groves across the West, it has suffered for a number of years from die-back and failure to regenerate new shoots to replace the aging adult trees for a number of years.

The U.S. Forest Service created fenced exclosures to protect a portion of the Pando Clone from herbivory (browsing - the consumption of woody growth - by mule deer and domestic cattle), and initiated some small-scale vegetation treatments. Aspen regeneration has responded inside the exclosures in both treated and untreated areas, while outside the exclosures, on public lands leased for livestock grazing, regeneration failure and die-backs continue to plague the Pando Clone as well as other aspen groves subjected to the same pattern of livestock and mule deer hebivory, and die-back continues.

Western Watersheds Project initiated a one-year monitoring project in order to quantify ungulate use in the area, using stationary motion-sensing cameras to quantify by species the use of the area and document levels of herbivory by both domestic cattle and mule deer over the 2018 growing season in the unfenced portions of the Pando Clone and in adjacent aspen groves. At our monitoring sites, we documented 4.5 times the amount of cattle use herbivory in two weeks than the mule deer use over six months. Forage utilization by mule deer prior to the onset of livestock grazing was unobservable, while forage utilization by livestock (plus mule deer) during the 2 weeks of cattle grazing consumed 70 to 90 percent of the understory vegetation's annual production.

Cattle have a greater propensity to consume aspen sprouts in autumn, when the nutritional quality of other understory vegetation declines, and the virtual elimination of understory vegetation by this high intensity livestock use may also cause mule deer to switch to aspen shoots, further amplifying the impacts. Our results show that the brief but intense cattle grazing appears to be a major contributor to the decline of the Pando Clone, as well as other aspen groves in the immediate vicinity, in addition to the much lighter continuous herbivory by mule deer. Based on comparisons of the exclosures with the area open to both livestock and mule deer that this high level of use in the unfenced areas effectively eliminates regeneration. A previous study (Rogers and McAvoy 2018) attributed the failure of the Pando Clone to regenerate solely to mule deer, but our results indicate that cattle are also having a major impact on understory vegetation. Our results suggest that livestock herbivory may be having a synergistic interaction with mule deer foraging to suppress aspen sprout growth, and that trampling of soils by livestock may also play a role in depressed aspen recruitment in unfenced portions of the Pando Clone and adjacent aspen stands.

Based on our results, we recommend removal of livestock from the Pando Clone area to protect this globally significant organism, and also recommend that livestock be removed from public land pastures elsewhere where aspen groves show signs of depressed regeneration.

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Introduction

Aspen (*Populus tremuloides*) stands are found across the interior West from southern Arizona to the Canadian Rockies, and typically occur in montane or sage steppe environments, often in association with abundant soil moisture. Aspen groves range along a spectrum from fire-dependent transitional communities that regenerate through periodic fires to stable communities that do not require fire for persistence (Shinneman et al. 2013). Reproduction via seeds happens most commonly in conjunction with severe disturbance such as fire (Long and Mock 2012). More frequently, aspens reproduce by sending up new shoots, or "suckers," from the existing rootstock, and the resulting aspen grove may persist for thousands of years (Jones and DeByle 1985a).

Aspen groves are frequently clones, where a single root system sends up hundreds or even thousands of individual stems (Barnes 1975), with each "tree" being a genetically identical surface expression of one large organism connected through its common root network. Gardner (2013) found that areas with high clonal diversity in aspens occurred in areas with a more frequent fire history, while areas with low aspen clonal diversity, often larger clones, corresponded to areas with less frequent fires. Clones may be long-lived; Kay (2003) hypothesized that aspen clones in north-central Nevada have maintained their presence for thousands of years via vegetative regeneration. As aspen trees age (i.e., exceed 100 years of age), they generally produce relatively few suckers (O'Brien et al. 2010).

For the purpose of clarity, it is useful to define some terms that will be used throughout this report. Aspens growing from a common rootstock are called *ramets*, a term that encompasses fully-grown adult ramets ("trees") as well as immature, regenerating trees rising as *adventitious shoots* ("shoots" or "suckers" in this report). Both new shoots with terminal buds and branches that have lateral buds can be referred to as "stems." The term "seedling" is used in this report exclusively to refer to young aspens growing from a seed, and excludes young aspens growing from adventitious buds on an existing rootstock. Aspen reproduction can be sexual ("seeding") or asexual ("suckering") from buds on the root system. Aspen recruitment occurs when young plants grow above the upper browse level of large herbivores. Clusters of aspen are referred to as "groves." Where such clusters are comprised of genetically identical trees joined by a common root system they are called "clones" and represent a single organism with many adult trees, sometimes thousands. "Regeneration" occurs when the recruitment of aspen suckers replaces the die-off of adult trees.

Aspens commonly grow where soil moisture is relatively abundant. However, in forested areas, sites containing aspens may be wetter simply because they transpire less water into the atmosphere than do conifers (Jones and DeByle 1985b). Aspen groves often contribute more water to drainage systems than do coniferous trees because they transpire only during the part of the year when they have leaves (versus year-round transpiration for conifers) and collect more snow in the understory than do conifers (DeByle 1985c). Aspens also have chlorophyll in their stems, and can photosynthesize throughout winter when leaves are absent (Grant and Mitton 2010). Presumably, water loss is minimal during winter when leaves are absent.

Aspen groves are hotspots of biodiversity and a number of bird species appear to be dependent on aspen habitats. Aspen groves harbored the greatest number of native species (45) of any habitat type in Grand Staircase – Escalante National Monument (Bashkin et al. 2003). Red-naped sapsucker, black-capped chickadee, house wren, warbling vireo, and northern saw-whet owl are closely associated with aspen woodlands (Hejl et al.,1996). Loose and Anderson (1995) found that 30 of 33 woodpecker nests in their Sierra Madre study area were found in aspens, and among these, there was a significant preference for large, old trees. According to Winternitz and Cahn (1983), 40% of species that inhabit aspen are cavity nesters, with a significant preference for large trees over 100 years old and trees infected by heartrot fungus. Heartrot-infected aspens are easier for birds and mammals to hollow out to create cavity nests. Aspens also are of critical importance as a food source for beavers (Williges 1946).

Jones and DeByle (1985c) compiled a thorough analysis of the role of fire in aspen ecology. According to these biologists, almost all even-aged aspen stands in the western U.S. appear to be the result of severe fire, presumably in coniferous forests. In Yellowstone National Park, Romme and Knight (1982) found that fire suppression has led to denser coniferous forests, a decrease in aspen, and an increase in sagebrush in meadow areas. Forest fires can foster aspen regeneration because fallen timber provides refugia for aspen seedlings to escape browsing by ungulates (Ripple and Larsen, 2001).

Strong aspen regeneration typically occurs even after severe burns. An abundance of aspen woodlands in the coniferous forest zone often indicates the prevalence of past stand-replacement fires. But Fornwalt and Smith (2000) noted that old, multi-storied aspen stands can maintain their productivity over time and are in many cases self-perpetuating. Thus, previous assumptions that aspen stands require periodic disturbance to maintain themselves are not universally true, and some stands (particularly old, multi-story stands) perpetuate themselves in the absence of any management treatment.

Although aspen habitats are viewed as valuable grazing resources by the livestock industry, these areas are very sensitive to overgrazing. Meuggler (1985b) reported that heavy grazing by domestic sheep can turn the rich and diverse herbaceous understory that occurs in ungrazed stands into a depauperate understory of grasses. In aspen stands that are overgrazed, invading, unpalatable plants can form a stable grazing disclimax (an unnatural, disturbed plant association that can persist indefinitely), reducing the wildlife habitat value of the grove (Mueggler 1985a). In addition, in older stands, heavy livestock grazing can prevent regeneration and speed the decline of the aspen stand itself (DeByle 1985a). Cole (1993) found that aspen-forb communities are highly susceptible to trampling damage even from human foot traffic. The physical trampling of nests and habitat degradation associated with livestock grazing can be detrimental to ground-nesting birds that prefer aspen habitats, such as the hermit thrush, junco, white-crowned and Lincoln's sparrows, veery, ovenbird, and nighthawk (DeByle 1985b). Because livestock grazing is currently permitted on more than 232 million acres of federal public land in the western United States (Beschta et al. 2013), the potential for ecological damage from livestock grazing is widespread.

The Pando Clone

The name "Pando" comes from the Latin "to spread." Kemperman and Barnes (1976) originally proposed the Pando Clone as a single living thing covering approximately 108 acres in area and made up of approximately 47,000 *ramets*, or stems. Grant et al. (1992) concluded that the Pando Clone represents the largest single organism in the world, with an areal extent of approximately 106 acres (43 ha) and an estimated weight of more than 6,600 tons (6 million kg). The Pando Clone was confirmed through genetic testing to be a single massive organism by DeWoody et al. (2008). Some of the trees in the Pando clone show triploid chromosome patterns (in effect, possessing a third set of chromosomes), and these individuals have a competitive advantage over diploid stems in terms of height and diameter growth (DeRose et al. 2015). This gives these stems an advantage in the 'self-thinning' stage of stand development, when only the most fit stems survive to attain tree form.

The age of the Pando Clone is a matter of substantial scientific debate. Kemperman and Barnes (1976) hypothesized that the Pando Clone was originally established more than 8,000 to 10,000 years before present. Aspen clones in this southern, unglaciated portion of the species' range, including Utah, can be unusually large and of much greater age

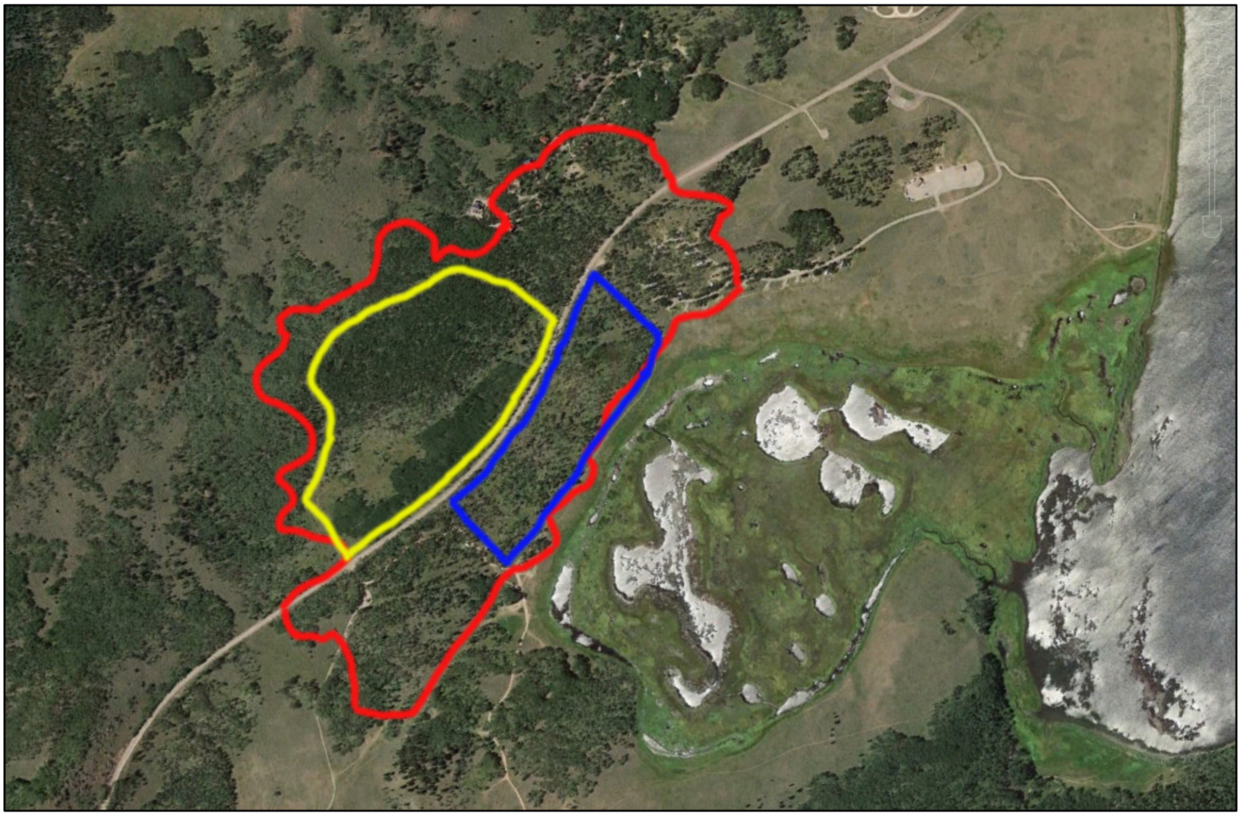

Figure 1. The boundary of the Pando Clone, in red (after Kemperman and Barnes 1976), showing the 2013 exclosure fence (in blue) which more successfully excludes mule deer, and the 2014 exclosure fence (in yellow) which has been less successful in excluding mule deer. State Highway 25 can be seen bisecting the Pando Clone, and Fish Lake appears at the right of the image. Image courtesy Google Earth.

(Barnes 1975). Mock et al. (2008) identified a number of other genetically distinct aspen clones along the fringes of Pando, and hypothesized that the relatively few mutational variants within the Pando Clone may indicate a much less ancient age for Pando. DeRose et al. (2015) hypothesized that the Pando Clone initially became established as recently as the 1880s. However, this conclusion is based on core sampling of existing trees; it is unlikely that the 108-acre root system of the Pando Clone arose spontaneously in a single year or two; indeed it is far likelier that the clone spread gradually over a long span of years. Thus, the definitive overall age of the Pando Clone remains undetermined at this time. Grant and Mitton (2010) estimated the age of the Pando Clone at 80,000 years.

On the Fishlake National Forest, where the Pando Clone grows, aspen cover has declined by 60% from historic levels (Wooley et al. 2008). Fragmentation of the Pando Clone stand itself is currently occurring, due to browsing by herbivores suppressing sapling recruitment, rural real estate development, and a fungal infection called sooty-bark canker (DeWoody et al. 2008). As a result, sapling recruitment in unfenced portions of the Pando Clone is failing to replace aging adult trees. According to Rogers and Gale (2017: 9), "Judging from the near-complete lack of recent recruitment (> 2 m height) and mid-story aspen throughout the study area, it has been many years, likely even decades, since this amount of stand renewal [0.5 ramets per overstory tree] has taken place at Pando." As overstory trees continue to die without replacement by sucker recruitment, the overall size of the Pando clone ultimately will shrink (*id.*). Mule deer and cattle affect the Pando Clone and its ability to regenerate through browsing adventitious suckers and trampling.

Elk do not appear to be affecting the Pando Clone. According to Rogers and Gale (2017: 11, internal citation omitted), "While elk browsing of aspen is a serious concern regionally, we did not see elk or record their scat at Pando." Rogers and McAvoy (2018) reported that "[e]lk sign is evident in the broader area" and used that as a basis for asserting that elk were presently accessing the Pando Clone, but documented no elk sightings and no elk scat during the course of their study.

In 2012, the Forest Service issued a decision to fence 67 acres (22 ha) of the Pando Clone's 108-acre (43-ha) extent (see Figure 1) to prevent herbivory from domestic and wild ungulates and authorized some small-scale, experimental cutting inside the exclosures to stimulate suckering (USFS 2012). The exclosures were built of 8-foot tall woven wire topped with a barbed-wire strand. One exclosure of 17 acres (7 ha) was constructed in 2013 to the east of State Highway 25, and it appears to mostly exclude both mule deer and livestock (Rogers and Gale 2017, Rogers and McAvoy 2018), although Coles-Ritchie documented deer sign and evidence of browsing inside this exclosure. Aspen recruitment is progressing well inside the Pando Clone ungulate exclosure, even though the presence of mule deer has been documented inside the exclosure (Coles-Ritchie 2018). A second exclosure of 37 acres (15 ha) was constructed in 2014 to the west of the highway, incorporating a 22-year-old section of fence, and mule deer appear to be able to enter this fenced exclosure (Rogers and McAvoy 2018). Rogers and Gale (2017) found that the portions of the Pando Clone that had been fenced to exclude large herbivores showed a positive response in terms of regeneration (irrespective of cutting treatments), while the remaining 52 acres (21 ha) of the Pando Clone outside the exclosure showed no improvement. Rogers and Gale (2017) found that fencing alone resulted in an average of 550 regenerating suckers per acre inside the 2013 exclosure, a level sufficient for stand replacement according to earlier scientific findings (Mueggler 1989).

Aspen Declines

The regeneration problems experienced by the Pando clone mirror widespread declines in aspen regeneration, both on the Fishlake National Forest and throughout the West. In addition to the gradual replacement of aspen woodlands through the invasion of conifers in certain areas, aspen die-offs also occur in the absence of conifer encroachment. These die-offs can eliminate adult stems within a period of two years, and are often accompanied by an absence of sapling recruitment (Bartos 2008). On Cedar Mountain in south-central Utah, aspen stands showed depressed sucker recruitment and almost one-fifth showed crown dieback greater than 20% (Rogers et al. 2010). Evans (2010) found that drought weakened aspen on Cedar Mountain, Utah, making them more susceptible to a long-term decline that reduced the area of aspen woodland by 24% over a 23-year span. Many aspen stands in northern Nevada are in poor condition and have not regenerated in more than 100 years, due primarily to heavy livestock browsing (Kay 2003). Brown (1995) attributed the decline of aspen in eastern Oregon and Washington to intensive grazing and fire exclusion. Fairweather et al. (2007) documented a sudden decline of aspens on the Coconino National Forest in Arizona following a severe frost event, followed by a severe drought and an outbreak of tent caterpillars. Smooth brome, an invasive perennial grass, may affect aspen suckering (O'Brien et al. 2010). Overall, multiple factors can contribute to the decline of adult aspens, but reproduction through suckering typically occurs unless it is suppressed be herbivory by non-native livestock or by native herbivores such as deer and elk.

While the gradual decline of aspen groves over time may be widespread, aspen die-offs also occur that eliminate adult stems within a period of two years, with an absence of sapling recruitment (Bartos 2008). Sudden Aspen Death syndrome is associated with aspens at high altitude under water stress (Worrall et al. 2010). The decline of the Pando Clone appears to be of the more gradual variety, rather than Sudden Aspen Death syndrome.

Aspens most commonly reproduce adult stems via suckering from the rootstock; its seeds, while abundant, are short-lived and have demanding germination requirements (Schier 1981, Kay 2003, Long and Mock 2012). As a result, seedling establishment typically occurs only during extremely wet summers (Jones and DeByle 1985b).

Schier (1975) described the dynamics of sucker production as governed by apical dominance, a phenomenon whereby hormones from the terminal buds of aboveground stems (auxins) inhibit hormones in the root system that stimulate sucker growth (cytokinins). When disturbance of the stems reduces the flow of auxins, the cytokinins can initiate the regenerative process. However, when aboveground stems weaken and die, the root system dies back due to a lack of photosynthate being furnished to the roots. Schier (1976) suggested that sucker regeneration is proportional to above-ground disturbance, citing examples from clearcut logging studies where the number of suckers generated is proportional to the number of stems removed by logging. Where suckering is suppressed by ungulate browsing, the die-off of adult aspens can result in areal reductions in aspen habitats across the landscape.

Shepperd (2001) proposed hormonal stimulation, a proper growth environment, and sapling protection as the three elements of an aspen regeneration triangle. Natural disturbances such as fire can stimulate suckering and regenerate aspen stands, but if livestock are not excluded from the aspen grove for several years following fire, their browsing can severely suppress sucker growth (Kay 2003).

The Role of Herbivory in Aspen Declines

Heavy ungulate browsing over extended time periods can cause regeneration failure over spans of many decades, resulting in an even-aged grove of older trees that is less resilient to drought and other stressors (Lindroth and St. Clair 2013). In the Book Cliffs of northeastern Utah, Rogers and Mittanck (2014) found that only three of 77 aspen stands (less than 4%) contained adequate levels of recruitment to perpetuate the stand, due substantially to browsing by wild and domestic herbivores. Herbivory by both domestic livestock (sheep and cattle) and wild ungulates (deer and elk) can suppress aspen shoot recruitment, and thus impair regeneration.

In some circumstances, large herbivore grazing and/or browsing in aspen stands may not put significant pressure on aspen reproduction. For example, Beck and Peek (2005) found only a 3% dietary overlap between spring and summer mule deer and cattle diets in aspen stands, with deer preferring browse and cattle preferring grasses and forbs, and also found that elk and cattle did not have significantly different diets. However, this study found that all of the herbivores studied had a 0% dietary consumption of aspen, with the exception of spring diets in one of three years, which showed <1% aspen contribution to the elk diet. Mower and Smith (1989) found that elk and mule deer diets in northern Utah were quite similar, and although shrubs made up a significant component of both, aspens were not noted as a significant component of the diet. Notwithstanding the preference of native and domestic ungulates for other forage plants, overbrowsing of aspen shoots to the point of regeneration failure is widespread.

Aspens have defensive compounds – phenolic glycosides and tannins – that provide adequate defense against insects and mammalian herbivores when browsing is light, but which is an inadequate defense under heavy browsing, which results in high levels of damage to the trees (Lindroth and St. Clair 2013). Elk may respond negatively to increasing phenolic glycoside content (Wooley et al. 2008). However, the scientific consensus is that while the tannins and phenolic glycosides present in aspens evolved as a defense against herbivory, they are insufficient to prevent browsing by either domestic or wild ungulates.

Deer and Elk Browsing

Aspen communities often are an important source of protein for mule deer in summer, whereas Utah serviceberry and big sagebrush communities may only provide maintenance amounts of protein (Austin and Urness 1985). This dietary advantage of aspen communities may contribute to mule deer preference for them. Severe browsing by elk and deer virtually eliminated sapling recruitment during an aspen die-off in northern Arizona (Fairweather et al. 2007). Additionally, population irruptions of mule deer on the Kaibab Plateau of northern Arizona in the 1920s had, between 1953 and 1962, completely suppressed aspen recruitment on the Kaibab Plateau of northern Arizona (Binkley et al. 2006). In Michigan, Randall and Walters (2011) found that increasing densities of white-tailed deer in aspen stands suppressed suckering and reduced forb density and species richness.

Livestock grazing in aspen groves may come at a cost for resident mule deer. Loft et al. (1991) suggested that as a result of livestock grazing, displacement of mule deer from these habitats and expansion of deer home ranges resulted in a lowering of inclusive fitness for mule deer. According to Loft et al. (1991: 22, internal citation omitted), "Once aspen stands had been occupied by cattle for a few weeks, there was little forage or hiding cover available, and deer essentially quit using the habitat." These studies indicate the likelihood that forage removal by cattle or domestic sheep can alter mule deer habitat selection and/or diet choices.

The likelihood of suppressed aspen regeneration from concentrated elk browsing appears to be greater than for mule deer. Compton (1974) found that elk in the Sierra Madre Range in Wyoming concentrated their summer use in subalpine parks, and found heavy autumn use in aspen cover types. Beck et al. (1997) reported that aspen made up 10% of elk summer diet, versus 3% of domestic sheep summer diet, in north-central Utah. Elk foraging on winter ranges has been shown to depress growth and prevent reproduction of aspen in Rocky Mountain National Park (Baker et al. 1997, Suzuki et al. 1999, Binkley 2008). Aspen are likely to be suppressed where elk density exceeds four elk per square kilometer (Painter et al. 2018). Elk also damage aspens by browsing new shoots, rubbing flexible saplings with their antlers, and by gnawing tree bark to get at the phloem underneath (Fairweather et al. 2007).

In the absence of large native predators, elk can suppress aspen sapling recruitment (Binkley 2008, Beschta and Ripple 2009). Ripple and Larsen (2000) found that due to heavy browsing by elk, following removal of wolves, only 5% of the current overstory aspen in the Northern Range of Yellowstone National Park originated after 1921. Painter et al. (2018) found that the percentage of aspen suckers browsed annually in Yellowstone National Park was 80-100% in 1997-98, decreasing to 30-60% in 2011-15 after the re-establishment of a wolf population. Elk shifted their habitat use and herbivory intensity away from Yellowstone National Park and toward the lower Madison River Valley in response to increasing wolf populations in the Park (Painter et al. 2018). However, in some localities within Yellowstone National Park, elk densities have remained high enough to continue to suppress aspen suckering (*id.*). White et al. (1998) found that elk browsing suppressed aspen recruitment in Canadian Rockies national parks, except under conditions when elk densities were reduced by wolves. There is now a broad scientific consensus that the absence of large native predators can result in depressed recruitment of aspen and other woody species (Beschta and Ripple 2009, Painter et al. 2018).

Browsing Pressure from Domestic Livestock

Livestock often concentrate their grazing activity in aspen groves due to the availability of shade and preferred understory forage species. In the Sierra Nevada mountains, cattle utilized meadow riparian and aspen habitats most strongly, selecting them over other habitats (Loft et al. 1991). According to Kay (2003: 41), "Even on allotments where livestock use has been controlled, aspen stands near water may still be in poor ecological condition because cattle tend to concentrate in those areas." Bailey et al. (1990: 213) found that cattle impacts on aspen are so severe that livestock can be used as a means to suppress aspen reproduction, stating "Overgrazing is generally considered to be detrimental to range stability and productivity over the longer term, but short duration heavy grazing may have a place in forage establishment and control of woody species." These researchers (*id.*: 214) recommended, "Clearly, for immediate control of aspen suckers, top removal or defoliation must be timed similarly to the late grazing treatment in this study. However, aspen suckers are suitable forage for cattle provided they are maintained within reach."

Beschta et al. (2014) found that aspen recruitment rates plummeted in the late 1800s with the onset of cattle grazing on the lands that would become Hart Mountain National Wildlife Refuge in southern Oregon, and increased by an order of magnitude after livestock were removed in 1990. These researchers attributed the decline of aspen groves on Hart Mountain to top-down forcing by cattle browsing, which suppressed aspen sapling recruitment, rather than climate changes. On Monroe Mountain in south-central Utah, Bartos and Campbell (1998a) provided photographic documentation of the effects of livestock preventing aspen regeneration using fenceline contrasts of a previously burned and logged area which remained barren in the presence of livestock and failed to regenerate. Across the fence, on habitats accessible to native herbivores but where livestock were excluded, dense regeneration was evident. Alexander (1995) documented that trampling by livestock broke 40% of aspen samples under both moderate and heavy grazing in his Alberta study; trampling caused damage in the form of basal scars that were present on 25% of surviving aspen saplings. By the second spring of cattle grazing, aspen sapling mortality in this study was 25%, 70%, and 89% for the ungrazed, moderately grazed, and heavily grazed sites, respectively.

Cattle selection for aspen shoots differs by season. According to Kay (2003: 32), "Year-long or season-long grazing is particularly detrimental to aspen, while early-season or dormant-season use may allow aspen to successfully regenerate." According to Jones et al. (2011: 629), "Aspen suckers received no early-growing season use by cattle but received the heaviest late-growing season use of all three vegetation types. Utilization was the same for all vegetation types at mid-growing season. Mean late-growing season use of aspen suckers was greater than 60%, and some stands received 100% use." Jones et al. (2011: 630) observed, "By mid-growing season, the quality of meadow and aspen understory vegetation approached minimum nutritional levels required for cattle." Alexander (1995) found that aspen suckers that have not yet begun to lignify, or become woody (i.e., one-year-old suckers), are a palatable forage for cattle, while two-year-old suckers were "not readily used" by cattle.

Even moderate levels of livestock grazing can suppress aspen regeneration. Alexander (1995) found that moderate and heavy grazing by cattle were equally effective at preventing aspen regeneration, with both moderate and heavy grazing both had a significant negative effect on understory biomass production in aspen stands.

Methods

We quantified ungulate use of the Pando Clone area with two motion-triggered cameras (Cameras 2 and 3) that were placed in portions of the Pando Clone outside the fenced exclosures, and two cameras that were placed in neighboring aspen groves (Cameras 1 and 4) subjected to the same pattern of livestock and mule deer herbivory. The cameras were sited in areas open to grazing and browsing by both domestic livestock and wild ungulates. The cameras were set to take photographs of all motion-triggered events separated by at least 1 minute. Cameras were installed on May 11th, 2018 and retrieved on November 22nd, 2018 to record herbivore activity throughout the growing season. The cameras were more sensitive to motion than expected. As a result, two of the cameras (Cameras 1 and 4, the cameras sited in neighboring aspen groves) ran out of battery power well before the end of the monitoring period, and therefore failed to record photographs during the livestock grazing period. These cameras, when remaining operational throughout the summer and into the fall, provide useful comparisons of forage utilization during cattle-free and cattle grazing periods, but could not be used to compare animal unit equivalents between deer and cattle due to the absence of livestock records.

After retrieval, the photographs were individually examined and the counts of ungulates were tallied for each camera. In order to more accurately compare total use by ungulate species, use was calculated based on body size/forage consumption by the Animal Units Equivalents (AUEs). A literature search found a range of estimates for mule deer, ranging from 0.2 (Pratt and Rasmussen 2001, NRCS 2003) to 0.17 (Ogle and Brazee 2009). For our calculations, we used 6 deer per 1 cattle animal unit (AU) (0.167), which is conservative being based on a 1,000-pound cow with calf. Cattle weights have increased significantly over the last 40 years with current average slaughter weight is presently 1,382 pounds (628 Kg) as of December 2017, (NASS 2018). We graphed the AUE data by week to display use over the monitoring period. It should be noted that the ratio of six deer per cow greatly underestimates the difference. A 1,382-pound (628 kg) cow consumes 3% of its body weight per day, or 41.6 lbs (18.9 Kg) (Ogle and Brazee, 2009). A 150-pound (68 kg) mule deer consumes 1.5 kg/day (UWSP 2019). This current information indicates a mature cow consumes 12.6 times the forage demand of a mule deer. However, we used the lower value to provide a conservative comparison.

We created time lapse videos of the photographs from each camera to help visualize conditions and herbivore use throughout the growing season.

The Interagency Landscape Appearance Method

This method's descriptions classify forage utilization into the following Herbaceous Utilization classes (USFS 1993; *see also* BLM 1996):

1. No Use (0-5%). The rangeland shows no evidence of grazing use; or the rangeland has the appearance of negligible grazing.

2. Slight (6-20%) The rangeland has the appearance of very light grazing. The key herbaceous forage plants may be topped or slightly used. Current seedstalks and young plants of key herbaceous species are little disturbed.

3. Light (21-40%) The rangeland may be topped, skimmed, or grazed in patches. The low-value herbaceous plants are ungrazed and 60 to 80 percent of the number of current seedstalks of key herbaceous species remain intact. Most young plants are undamaged.

4. Moderate (41-60%) The rangeland appears entirely covered as uniformly as natural features and facilities will allow. Fifteen to 25 percent of the number of current seedstalks of forage plants are utilized. (Moderate use does not imply proper use.)

5. Heavy (61-80%) The rangeland has the appearance of complete search. Key herbaceous species are almost completely utilized with less than 10 percent of the current seedstalks remaining. Shoots of rhizomatous grasses are missing. More than 10 percent of the number of low-value herbaceous forage plants have been utilized.

6. Severe (81-100%) The rangeland has a mown appearance and there are indications of repeated coverage. There is no evidence of reproduction or current seedstalks for key herbaceous species. Key herbaceous forage species are completely utilized. The remaining stubble of preferred grasses is grazed to the soil surface.

These videos can be accessed at https://www.westernwatersheds.org/pando-clone-time-lapse/. We also took photographs along the 2013 exclosure fence to document the contrasting rates of regeneration within and outside the exclosure.

Utilization of vegetation by herbivores was estimated using the interagency Landscape Appearance Method descriptions, an estimation procedure used on the Fishlake National Forest (USDI Technical Reference 1734.3), see accompanying box. Utilization was estimated at the photo location on the day prior to livestock entry, 7 days after livestock entry (half of the livestock use period) and again after livestock removal.

Results

The exclosures constructed by the Forest Service in 2013 and 2014 within portions of the Pando Clone provide a clear contrast between natural recovery rates inside the exclosures with the effects of this heavy to severe level of utilization outside the exclosures. The exclosures were built of 8-foot tall woven wire topped with a barbed-wire strand. Figures 2 and 3 are taken from the same location, with one looking into the interior of the 2013 exclosure and the other looking into the grazed allotment, and area used by both deer and cattle.

From the ongoing aspen recovery that has occurred since the exclosures were constructed in 2013 and 2014, and the complete lack of any recruitment of aspen sprouts occurring outside the exclosures, it is clear that current management outside the exclosures prevents the regeneration of the Pando Clone in areas open to livestock grazing. Inside the 2013 exclosure fence, we found successful aspen recruitment is occurring irrespective of any mule deer that may have found a way to enter the exclosure area.

Camera 1 recorded from May 11th, 2018 through August 13th, 2018, prior to the onset

Figure 2. A view inside the 2013 exclosure. Note abundant regeneration 8-12 feet tall after 5 years of exclusion. June 10th, 2019.

Figure 3 (above). Looking into an unfenced portion of the Pando Clone from the same location with no regeneration. June 10th, 2019.

Figure 4 (below). - Fenceline contrast with abundant regeneration inside the 2013 exclosure (left) and no regeneration occurring outside (right). June 10th, 2019.

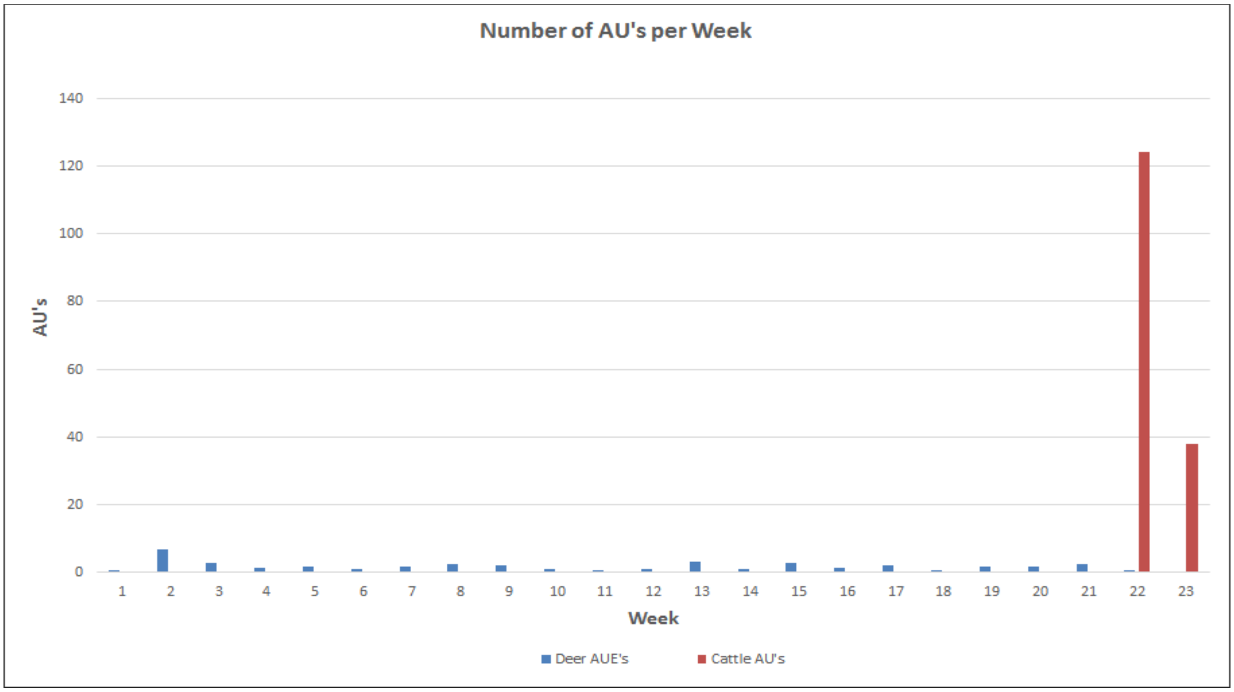

Figure 5. Camera 2, within an unfenced portion of the Pando Clone, deer versus cattle Animal Units by week.

of livestock grazing. Camera 2 recorded from May 11th, 2018 through October 9th, 2018.

Camera 3 recorded from May 11th, 2018 through November 22nd, 2018. Camera 4 recorded from May 11th, 2018 through September 22nd, 2018, prior to the onset of livestock grazing. The livestock use period began on October 4th and ended October 16th for a total of 13 days, during which domestic cattle were the type of livestock present in the project area. The area under study received use by mule deer (*Odocoileus hemonius*) throughout the monitoring period.

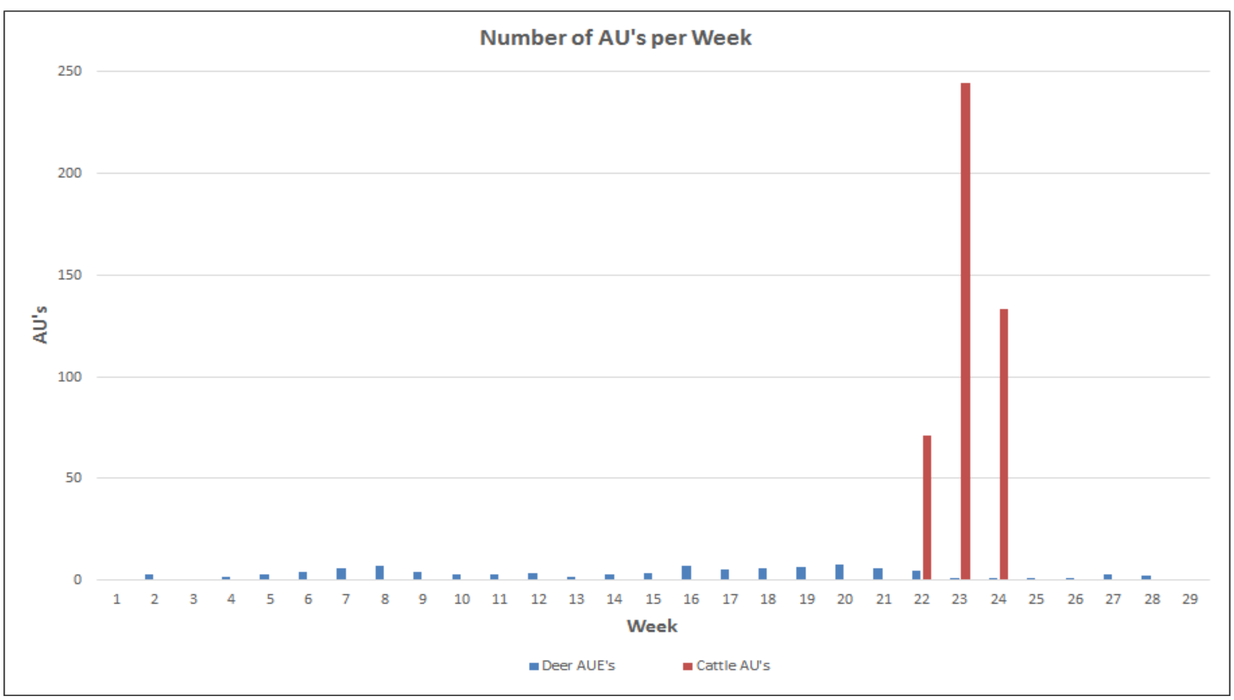

Figure 6. Camera 3, within an unfenced portion of the Pando Clone deer versus cattle Animal Units by week.

Rogers and McAvoy (2018) reported that "[e]lk sign is evident in the broader area" and used this as a basis for asserting that elk might presently be accessing the area. We documented no elk sightings in the Pando Clone, but Camera 1 recorded four elk in one instance in an adjacent aspen grove.

For Camera 1, motion from grass moving in the wind depleted the power supply by August 13th, 2018, so only forage utilization observations could be made. For Camera 2, deer use during the 6-month period totaled 42 AUE's, while cattle use during the 6 days (slightly less than 50% of the cattle use period) totaled 162 AU's. For Camera 3, deer use during the 6-month period totaled 101 AUE's, while cattle use during the 13 days totaled 448 AU's. For Camera 4, motion from grass moving in the wind depleted the power supply by September 27th, 2018, so only utilization observations could be made. On average, the index for animal use documented for cameras that lasted into the livestock grazing season was found to be four times higher for cattle during the 13 days of livestock grazing than for mule deer over the course of the entire growing season. Camera 4, on the eastern shore of Fish Lake, documented a similar result.

During the months prior to the arrival of livestock all cameras documented no observable utilization of the vegetation, whereas within 7 days after the arrival of livestock utilization was in the "heavy" category (61-80% utilization) for Cameras 2 and 3, inside the Pando Clone. After livestock removal, use was in the upper "heavy" to mid "severe" (81-100%) categories at all four sites (see Figures 10, 16, 22, 27, 28, and 29).

Figure 10 shows conditions following livestock removal for Camera 1, in an aspen grove adjacent to Pando. Based on the descriptions in the Landscape Appearance Method this would fit in the upper end of the "heavy" (61-80%) category. Within the Pando Clone, patterns of herbivory by mule deer and livestock were essentially identical to Pando's genetically distinct neighboring groves. For Camera 2, livestock use was near the upper end of the "heavy" category by day 7, with significant utilization on rabbitbrush (*Chrysothamnus* sp.), which has low palatability (see Figure 15). By the time livestock were removed, forage utilization levels, based on

Figure 7. Camera 1 (in an aspen grove immediately adjacent to the southeast corner of the Pando Clone) at deployment. Note mountain lion.

Figure 8 (above). Camera 1, mid-June.

Figure 9 (below). Camera 1, mid-summer.

Figure 10 (above). Camera 1 location on November 22nd, 2018, after livestock removal. Forage utilization shown here is in the upper end of the "heavy" (61-80%) category.

Figure 11 (below). Camera 2, within the Pando Clone, at deployment.

Figure 12 (above). Camera 2, inside the Pando Clone, in mid-June.

Figure 13 (below). Camera 2 in late summer.

Figure 14 (above). Camera 2 just prior to livestock entry.

Figure 15 (below). Camera 2 after 7 days of livestock use.

Figure 16 (above). Camera 2 location on November 22nd, 2018, taken in the general direction the remote camera had been pointed, after livestock removal. Forage utilization shown here is in the mid to upper end of the "severe" (81-100%) category.

Figure 17 (below). Camera 3 (within the southeastern edge of the Pando Clone) at deployment.

Figure 18 (above). Camera 3 in mid-June.

Figure 19 (below). Camera 3, mid-summer.

Figure 20 (above). Camera 3 just before livestock entry. Note for reference the two large bunchgrasses on the left and the scattered fallen limbs on the ground.

Figure 21 (below). Camera 3 after 7 days of livestock use.

Figure 22 (above). Camera 3 after livestock removal. Forage utilization shown here fits in the upper end of the "heavy" (61-80%) category.

Figure 23 (below). Camera 4, above the eastern shore of Fish Lake, at deployment.

Figure 24 (above). Camera 4 in mid-June.

Figure 25 (below). Camera 4, mid-summer.

Figure 26 (above). Camera 4 just before livestock entry.

Figure 27 (below). Camera 4 location, taken in the general direction of the remote camera, on November 22nd, 2018 after livestock removal. Based on the descriptions in the Landscape Appearance Method this level of herbivory fits in the upper end of the "heavy" (61-80%) category.

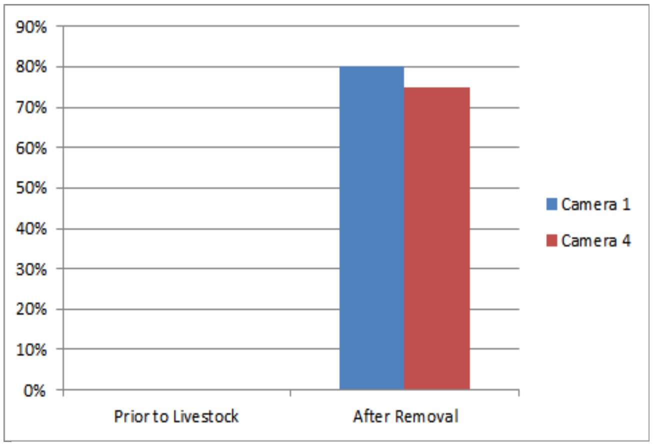

Figure 28. Forage utilization levels before livestock entry and after removal for Cameras 1 and 4, in aspen groves adjacent to the Pando Clone.

the descriptions in the Landscape Appearance Method grazing levels shown by Camera 2 this would fit in the mid to upper end of the "severe" (81-100%) category. By day 7 of livestock use documented by Camera 3 (see Figure 21), the large bunchgrasses had been completely grazed and only a small fraction of the seedheads remained. Note the difference in visibility of the fallen branches at ground level between Figures 20 and 21. Figure 22 shows conditions for Camera 3 following livestock removal. Based on the descriptions in the Landscape Appearance Method this would fit in the upper end of the "heavy" (61-80%) category. By the time livestock were removed, forage utilization levels shown by Camera 2 (see Figure 16) would fit in the mid to upper end of the "severe" (81-100%)

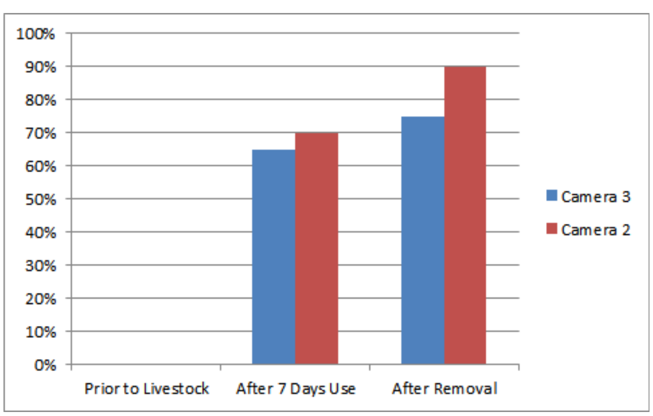

Figure 29. Forage utilization levels before livestock entry and after removal for Cameras 2 and 3, sited within the Pando Clone.

down to the same level as the rest of the forage base.

While we hoped to document direct herbivory by deer and/or cattle on aspen sprouts with the remote cameras, in fact we were unable to document any aspen sprouts at all during the growing season period over which our cameras were deployed. This is consistent with the findings of Rogers and Gale (2017), who also reported essentially no aspen sprouts outside the exclosure fence. Thus, like Rogers and McAvoy (2018), we are unable to measure direct herbivory of aspen by either mule deer or cattle, and are left with making inferences from indirect measures (in the case of this monitoring report, overall forage consumption and animal use). The level of trampling by cattle appears to be heavy in all locations we monitored.

Our findings support the conclusions of Loft et al. (1991), that the presence of livestock results in habitat abandonment by deer. Deer use dropped to nearly zero after the arrival of livestock and only returned after livestock removal and then at much lower levels than prior to livestock entry.

Discussion

We documented levels of herbivory by mule deer that were too light to quantify throughout the summer, measured by the Landscape Appearance Method used by the Forest Service to estimate forage utilization. This was followed by heavy to severe understory utilization by cattle that virtually eliminated understory vegetation during the 14-day period in October when cattle were turned out both in unfenced portions of the Pando Clone, and in neighboring aspen groves subjected to the same pattern of livestock grazing. The level of livestock forage utilization we documented (70 to 90%) was consistent with heavy grazing as defined by Alexander (1995), who classified 73% forage utilization by cattle as "heavy" and found this level – entailing the browsing of 95% of aspen saplings – to be sufficient to suppress aspen regeneration (see Figures 28 and 29). This is supported by scientific observations at Pando itself. Rogers and Gale (2017: 11) concluded, “A key message, then, is that while we cannot state unequivocally that there are ‘too many’ herbivores at Pando, we do know that there are too many for current conditions.”

These heavy to extreme levels of forage utilization exceed the Forest Service allowable utilization level of 50% (USDA 2018). In addition, these levels are much greater than the 25% level supported by leading range scientists (Galt et al 2000).

By quantifying the ungulate use of the Pando area and tracking utilization over the study period, our data and analysis demonstrates, based on Animal Unit Equivalents, that more than 4 times the animal use occurs in the unfenced portion of the Pando Clone and in neighboring aspen groves from livestock than for mule deer. Nearly all of the observable forage utilization in the understories of aspen groves in this area during the monitoring period was the result of livestock. According to Rogers and Gale (2017: 6),

> we counted only one mule deer scat pile, but 219 cattle deposits in Year 1. In Year 2, we counted no scat piles of any species within the fence, but 72 cattle and five deer piles outside the exclosure. By Year 3, cattle deposits were 64 and deer scat was 14, all outside the exclosure.

Our results are consistent with these findings.

Our findings contrast with Rogers and McAvoy 2018, which concluded that mule deer are the primary factor in regeneration failure in the Pando Clone. The Rogers and McAvoy study used “browse level, and feces counts as a surrogate for ungulate presence.” Its analysis identified deer presence (indexed by density of pellet groups) as the key factor relating to failure of aspen sprouts to recruit. Cattle presence as indexed by feces was negatively related to both recruitment and aspen density but was not identified as a major factor by this exploratory analysis. It is troubling that while pellet groups were negatively related to aspen regeneration in the Rogers and McAvoy study, browse level was not a significant factor. Browsing of aspen saplings would presumably be the direct means by which either mule deer or cattle would directly affect sapling survival and recruitment.

In addition, Rogers and McAvoy’s identification of cattle concentration as an unimportant factor in aspen recovery runs contrary to earlier findings that aspen recruitment is lowest in portions of the Pando Clone accessible to livestock, and higher in fenced areas, whether these are accessible to mule deer or not (Rogers and Gale 2017, Coles-Ritchie et al. 2018). Rogers and McAvoy (2018) concluded that deer were the cause of regeneration failure. But in their analysis of regeneration, the 2014 exclosure was accessible to deer but not cattle, and had a browse level of 24%, while in the unfenced area, where both deer and cattle were present, the browse level was 55%. Furthermore, aspen recruitment was highest in the 2014 exclosure (1,204 stems/ha) in the presence of deer and lowest in the 2013 exclosure from which both deer and livestock were absent, further muddying this conclusion.

The season of livestock grazing can also have a major impact on regeneration. Livestock show greater preference for browsing aspen shoots in autumn than in spring (Fitzgerald et al. 1986). Aspen suckers have higher nutritional quality than other forage types throughout the year, but cattle focus their foraging on meadow and understory vegetation in early and late summer, increasing utilization of aspen suckers only late in the growing season when other forage types were of low nutritional quality and depleted by grazing (Jones et al. 2011). However, experimentally browsed aspens showed greater growth when browsed in the autumn than when browsed in early or late summer (Jones et al. 2009). Balancing aspen’s greater resilience to livestock grazing in fall with the far greater tendency of cattle to select aspen browse at this same time of year thus becomes critical.

Late-season grazing by cattle (just before leaf drop) is the most effective season for cattle grazing to suppress aspen regeneration, and livestock grazing during this time of year can eliminate aspen seedling recovery after six consecutive seasons of grazing post-fire (Bailey et al. 1990). Jones et al. (2011) recommended avoiding late-season grazing by cattle in aspen stands to minimize browsing on aspens, and recommended that mid- and late-season grazing by cattle not occur in consecutive years. Jones et al. (2011) recommended avoiding late-season grazing by cattle in aspen stands to minimize browsing on aspens, and recommended that mid- and late-season grazing by cattle not occur in consecutive years. In the case of the Pando Clone, livestock use this pasture in the fall every year, at the very time of year when the greatest selection by cattle for aspen shoots occurs. In this case, cattle were turned out in the Pando Clone in early autumn, precisely the season when the tendency of cattle to browse on aspen saplings would be expected to be greatest based on the science.

In our monitoring, we found livestock use in the "heavy" to "severe" categories that would result in complete use on any aspen suckers that had emerged. Our cameras were unable to detect aspen sprouts – or either mule deer or cattle herbivory on them – but the end result was that aspen sprouts were virtually completely suppressed outside the exclosure fences, based on the absence of any aspen sprouts visible in our photographs. This finding is consistent with other reports documenting little or no aspen recruitment outside exclosure fences that prevent grazing by livestock (but do not always prevent access by mule deer).

Cattle grazing in parts of the Pando Clone outside the exclosure, and in neighboring unfenced aspen groves, may also have a synergistic effect with the herbivory by native mule deer, resulting in impacts to aspen recruitment that may be greater than simply adding the two types of impact together. Wild herbivores may be drawn to ungrazed areas where livestock have been excluded (O'Brien et al. 2010). Aspen habitats are preferred by mule deer when cattle are absent, but preference declines under moderate to heavy grazing to the point where deer use aspen habitats roughly in proportion to their availability (Loft et al. 1991). Mueggler and Bartos (1977) studied an exclosure accessible to deer but not livestock in which production of forbs, or broad-leaf understory herbs, occurred inside the exclosure. This abundance of forage likely concentrated deer foraging activity inside the exclosure, to the detriment of aspen suckers, which failed to survive to reach tree status between 1905 and 1934, based on subsequent tree-ring analysis.

Austin and Urness (1985) reported that aspen proportion in mule deer summer diets ranged from 0.2 – 3%, but increased to 9% in September. The heavy level of understory utilization by cattle in the unfenced parts of the Pando Clone and in nearby aspen stands (70-90% as found in our study) during a time of year when deer intrinsically increased their herbivory on aspen saplings may, through competition, further increase mule deer browsing on aspen shoots by leaving behind few alternative sources of forage.

Kay and Bartos (2000) studied exclosures on the Dixie and Fishlake National Forests that excluded deer and livestock both, or livestock only. Complete failure of new regeneration occurred in the presence of both livestock and deer herbivory outside the exclosures at 4 of the 5 sites where portions of the exclosures prevented access by both deer and livestock, and at 3 of the 8 sites having livestock-only exclosures new regeneration failed in areas where the livestock were excluded. Kay and Bartos found that excluding livestock and/or native herbivores increased recruitment of aspen saplings in the 2-meter to 5-centimeter diameter-at-breast-height range, with an average of 4,474 surviving aspen ramets under livestock and cervid exclusion, 2,498 ramets surviving by excluding livestock only, and an average of 1,012 surviving ramets outside the exclosures, where aspens were subject to herbivory by cattle, sheep, deer, and/or elk. Rogers and Gale (2017) documented a more than fourfold increase in aspen regeneration inside the Pando Clone's fenced exclosure compared with outside.

In this monitoring project, we found little visual evidence of aspen recruitment outside the exclosure fences, indicating either that aspen sprouts were browsed away prior to the onset of the growing season for grasses, or that deer and/or livestock herbivory was eliminating them prior to the point at which they would become visible to the camera. Given the extreme level of understory herbivory by cattle during the 13-day grazing period that we recorded in 2018, it is entirely possible that mule deer returning to the Pando Clone following cattle grazing would have found little understory forage, increasing the likelihood of 100% utilization of aspen sprouts that emerged prior to the onset of the following season. In this way, the overgrazing by cattle that we recorded within unfenced portions of the Pando Clone may be interacting with herbivory by mule deer to eliminate aspen recruitment outside the ungulate exclosures.

Bailey et al. (1990: 214) found fall cattle grazing to be an effective tool for eliminating aspen regeneration:

> Suckers defoliated by grazing in August, late in the growing season, were nearly eliminated after only 1 defoliation (FitzGerald and Bailey 1984) whereas suckers defoliated earlier in the season continued to regenerate and took 7 years to decline to 7% of original stem densities…. Schier (1976) indicated that repeated removal of tops and consequent initiation and growth of new suckers leads to a gradual depletion of nonstructural carbohydrates in the roots. Exhaustion of carbohydrates by annually repeated destruction of growing points appears to take from 6 to 8 years…. Clearly, for immediate control of aspen suckers, top removal or defoliation must be timed similarly to the late grazing treatment in this study.

These authors conclude by stating, "If the first priority is to nearly eradicate regenerating aspen suckers, then late season, short duration heavy grazing should be applied." Unfortunately, this is exactly what is happening within the unfenced Pando Clone and surrounding aspen groves.

Trampling damage by ungulates has often been implicated as a potentially significant cause of aspen regeneration failure (Schier 1981, DeByle 1990, Brown 1995). With regard to cattle, Weatherill et al. (1969: 5) concluded that "[c]onsumption reduces photosynthesis, trampling may break stems and leaves, while soil compaction can injure root systems and decrease soil aeration and water holding capacity." While Dockrill et al. (2004: 261) found that damage from cattle due to direct browsing and trampling damage killed individual aspen sprouts, these researchers concluded that "[h]igh mortality among stems without observed injuries might have been indirectly associated with cattle damage resulting from soil compaction, reduced root oxygen and subsurface severing of lateral roots." Because adventitious buds forming on lateral roots are the genesis of aspen sprouts, and because the level of trampling by cattle appears to be substantial based on our monitoring, more detailed study of the effect of trampling by livestock on the roots, adventitious buds, and initiation of suckering in the Pando Clone is necessary prior to concluding that herbivory by deer or livestock (or some synergistic combination of the two) is primarily responsible for the failure of sprout recruitment outside fenced exclosures.

Livestock appear to have the heavier impact than mule deer on aspen regeneration, based on exclosure studies that differentially exclude cattle and wild cervids. Based on a study of 30 grazing exclosures in aspen habitats in Nevada, Kay (2003: vi) stated,

> The [declining] status and trend of aspen communities in north-central Nevada, however, is not related to climatic variation, fire suppression, or browsing by mule deer. Instead, the condition of individual aspen communities is related to past and present levels of livestock grazing. That

> is, aspen is declining throughout most of north-central Nevada due to repeated browsing of aspen suckers by cattle and/or domestic sheep – repeated browsing eliminates sucker height growth, which prevents their maturation into aspen saplings and trees. Without stem replacement, aspen clones are consigned to extinction.

Livestock in mountain ranges of central Nevada contributed to poor aspen clone condition, and grazing by sheep and cattle accounted for 99.5% of the grazing pressure based on feces counts (Kay 2001).

While mule deer have been implicated as the cause of regeneration failure in the Pando Clone (Rogers and McAvoy 2018), the bulk of science thus far published (reviewed herein) does not necessarily support this conclusion, and our own monitoring photos show quite clearly that cattle, rather than mule deer, are having the heaviest impact on understory vegetation in the Pando Clone and on the understories of neighboring aspen groves.

Recommendations

We recommend eliminating livestock grazing during all seasons for the entire Pando Clone, and for aspen habitats generally, livestock should be removed if aspens are experiencing regeneration failure. This should be done until aspen regeneration is above browse height, and will require periodic repetition to prevent future aspen sprout suppression. Kay (2003) recommended fencing critical aspen stands or restricting livestock to only early-season grazing. According to Beschta et al. (2014: 36, internal citations omitted), "Our results indicate that for areas grazed by livestock and where aspen recruitment is either absent or occurring at low levels, implementing strategies that eliminate or minimize the effects of livestock herbivory may be needed. Given the vast amount of public land annually utilized by domestic ungulates and the large losses in aspen those lands have experienced to date, reducing livestock grazing effects within and across ecoregions may be required for attaining ecological restoration of herbivore-altered plant communities." According to Alexander (1995: 120), "even though aspen sucker density was still high after two years [cattle] grazing, it was the author's opinion that if the grazing treatments were continued, the prognosis for successful aspen forest regeneration would be poor."

Mechanical treatments such as coppice logging do not appear to be warranted in the Pando Clone based on the science. Aspen stands can reach high densities without stagnating because they are self-thinning (DeByle 1984). Thus, the thinning or logging of aspen stands is unwarranted from a silvicultural perspective. Bird species richness increases with aspen patch size (Johns 1993), suggesting that fragmenting aspen stands into progressively smaller patches through clearcutting may lead to a loss of bird diversity. In the Pando Clone, coppice logging of aspens might also inadvertently cause a loss of genetic diversity by completing the dominance of triploid aspens (DeRose et al. 2015). The successful regeneration of aspen saplings inside the Pando Clone's exclosure fence in the absence of mechanical treatments is proof positive that mechanical interventions are unnecessary.

The idea of eliminating grazers from aspen stands struggling to reproduce is not a new concept. Mueggler (1989) recommended protecting aspen groves with exclosures where the stand is heavily grazed or browsed. According to Shepperd (2001: 363), "Fencing is the only guaranteed means of directly protecting sprouts from browsing animals." O'Brien et al. (2010: 28) recommended, "In situations where the relative impact of domestic livestock versus wildlife has not been determined, a livestock exclusion fence alone (followed with monitoring) may be a reasonable first choice."

The significant role of cattle grazing in the Pando Clone has been acknowledged by scientific researchers. According to Rogers and Gale (2017: 11), "While we know that mule deer are responsible for a portion of aspen sucker browsing, cattle reduction and exclosure seem to also play an important role as evidenced by the combination of scat counts, browse levels, and overall regeneration response inside and outside our study area."

At a minimum, the existing exclosures should be expanded to encompass the entire perimeter of the Pando Clone, plus a quarter-mile buffer to allow for expansion, and livestock grazing should cease in this area. A better solution would be to permanently close the Dry Ponds pasture and any other pasture that encompasses the Pando Clone, to livestock grazing. Further research is needed to determine thresholds at which mule deer and/or cattle density reduce aspen recruitment below self-sustaining levels, and the degree to which soil trampling by livestock contributes to sprout suppression and root damage in aspen clones.

Aspens and mule deer have been evolving together for thousands of years. In light of our findings that heavy cattle utilization of aspen understories in the unfenced portions of the Pando Clone and in neighboring aspen stands, and the likelihood that this heavy level of grazing could work synergistically with mule deer browsing to suppress aspen regeneration, previous hypotheses that mule deer browsing alone is responsible for the decline of Pando Clone sucker establishment appear highly unlikely. Taken together, the evidence brought forward thus far suggests that livestock grazing and/or trampling may be the critical factor(s) tipping browsing pressure over the threshold at which aspen regeneration begins to fail. Removing livestock grazing from the pastures south of Fish Lake and measuring suckering and recruitment for a period of 5 years would be a logical method to determine whether the primary driver of the failure to recruit is deer or livestock.

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forest policy

Implementing the 2012 Forest Planning Rule: Best Available Scientific Information in Forest Planning Assessments

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National forests and grasslands in the United States are governed by land and resource management plans that should be updated every 15 years to reflect changing social, economic, and environmental conditions and to address new priorities. A new forest planning rule finalized in 2012 introduces new planning approaches and requirements, and several forests have completed the forest assessment phase of their planning process. Using document analysis and interview data, we analyzed four completed forest assessments to gain insights into early forest planning efforts under the 2012 rule. We found that forest assessments address the required topics, although the organization and depth of treatment varies across cases; government sources and academic publishers are relied on most often as sources of scientific information; and approaches to best available scientific information rely on peer-reviewed information, agency technical reports and syntheses, and personal expertise and judgement.

Keywords: early adopter, expertise, US Forest Service

Management of the 154 national forests and 20 grasslands in the United States is governed by land and resource management plans (also called forest plans), as required by the National Forest Management Act of 1976 (NFMA; 16 U.S.C. 1604). The forest plan functions as a guiding document that outlines goals, objectives, and strategies for management of the unit. Periodically, the rule related to forest planning is revised to reflect societal changes, new approaches and technologies, and scientific discoveries. For many years the US Forest Service (USFS), which manages the system of national forests and grasslands, has operated under a planning rule finalized in 1982 (47 FR 43026) despite several efforts (2000, 2005, and 2008) to revise and improve the rule (Schultz et al. 2013). A new planning rule issued in April 2012 (77 FR 21161) introduces several significant changes, including a renewed emphasis on collaboration, improved transparency, and a strengthened role for public involvement throughout the planning process. Of interest for our study is the requirement to use the best available scientific information (BASI) to inform the assessment, plan revision decisions, and monitoring program.

To date, little research has addressed implementation of the 2012 planning rule. Schultz et al. (2013) examined approaches to wildlife conservation planning under the new rule, raising concerns regarding potential extirpation of species. Another study analyzed public participation processes in 12 national forests (University of Montana 2015), and Schembra (2013) explored the role of standards and guidelines and how they are used in planning activities. Forest planning under the 2012 rule consists of three phases (assessment, plan development, and monitoring). The assessment phase is important, as it assembles relevant scientific information that planners will rely on to make decisions on forest management in the plan development phase. Our study contributes to this growing body of knowledge by examining the assessment phase of the forest planning process.

Eight "early adopter" national forests, along with several other forests, are currently developing their forest plans using the 2012

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rule. These forests were designated as early adopters because they provide important benefits, had strong existing collaborative networks in place, and needed to revise their forest plans (USDA Forest Service 2012a). The eight early adopter forests are: Cibola (NM), Chugach (AK), El Yunque (PR), Nez Perce and Clearwater (ID), and three forests that are coordinating planning on a regional basis: Inyo, Sequoia, and Sierra (CA).

Although implementation is still in early stages, several of the early adopter forests have completed their forest assessments and draft forest plans, which presents an opportunity to study implementation of the planning process under the new rule. One forest (the Francis Marion in SC) has completed the full plan revision process as of this writing. We examined four forests that have completed their assessments, including three forests identified by the agency as early adopters and one forest that is keeping pace with this group. The study explored three questions: 1) What does the 2012 planning rule require regarding the structure, content, and process for forest assessments? 2) How have forests implemented the directives related to forest assessments under the 2012 planning rule? 3) How are forests approaching the requirement for the use of best available scientific information in their assessments?

Forest Planning under the 2012 Rule

The 2012 planning rule suggests an adaptive approach to forest planning, instructing managers to 1) *assess* forest conditions; 2) *revise or amend* plans if the assessment indicates a need for change; and 3) *monitor* plan implementation (36 CFR 219.5). The process is cyclical, with monitoring data feeding back into the assessment of conditions in the management unit (USDA Forest Service 2012b). During the assessment phase, planners are expected to "rapidly evaluate existing information about relevant ecological, economic, and social conditions, trends, and sustainability, and their relationship to the land management plan within the context of the broader landscape" (36 CFR 219.5(a)(1)). The second phase of the planning process is plan development, amendment, or revision, where planners use the results of the assessment to establish a need for change and generate planning alternatives (36 CFR 219.5(a)(2)), and the public has the greatest opportunity for input. The plan development phase includes environmental impact assessment, public input, and plan publication (36 CFR 219.5(a)(2)). The third phase (monitoring) is an opportunity to track and measure management effectiveness over time (36 CFR 219.5(a)(3)). The planning *process* under the 2012 rule is similar to the process specified under the 1982 rule, but differs in terms of the specific elements required for the *assessment* (2012 rule) and the *analysis of the management situation* (the assessment's counterpart in the 1982 rule).

We focused our study on the assessment phase of the planning process. The assessment phase is important because it requires the forest to assemble and synthesize the most recent, relevant, and highest-quality science on social, ecological, and economic conditions to inform the plan development. Not only does this provide planners an opportunity to evaluate changes in biophysical and socio-economic conditions based on the latest monitoring data, it also represents a chance to reflect on new concepts, models, and methods that result in new scientific information about the local forest environment. Under the 2012 planning rule, the assessment phase identifies existing conditions, trends, risks, uncertainties, and information gaps that are relevant to land and resource management issues in the unit (36 CFR 219.5–219.6). In the assessment phase, the planning unit is not required to generate new studies or information, but is expected to obtain pre-existing information that is publicly available or voluntarily provided (36 CFR 219.6). Information can come from government and nongovernment sources, and the rule instructs the Forest Supervisor to provide opportunities for stakeholders to provide information for the assessment (36 CFR 219.6). The primary product of the assessment phase is an assessment document that evaluates existing information for 15 specific topic areas (Figure 1). Although the general topic areas are mandated by the 2012 rule, the Forest Supervisor has discretion to determine the scope, scale, and timing of the assessment, assuming the other requirements in the planning rule are followed (36 CFR 219.6).

Role of Science in Natural Resource Management

Historically, natural resource management in the United States was guided by the idea of scientific management and Progressive-era approaches (Taylor 1896). In particular, Samuel Hays's "gospel of efficiency" relied on a rational and scientific method of making decisions through a single, central authority. The thought was to avoid conflict via a scientific approach to social and economic issues (Hays 1959, p. 267). The US Forest Service exemplifies the approach of technical rationality and empirical science as the basis for sound resource management practices (Wellman 1987; Kaufman 1960). Foresters and natural resource managers

Management and Policy Implications

Although implementation of the US Forest Service's 2012 planning rule is still in the early stages, several national forests have completed the assessment phase and moved on to the next phase of forest planning. Our analysis of forest assessments from several "early adopter" forests illustrates that forest planners are making serious efforts to address required topics and rely on the best available scientific information. Assessment reports were disproportionately heavy in science related to terrestrial and aquatic ecosystems, and more limited in treatment of infrastructure, land ownership and access patterns, cultural heritage, and areas of tribal importance. Ensuring that assessment teams include broad and diverse disciplinary experts will help address this challenge, recognizing that some forests may not have access to necessary disciplinary specialists. It is also possible that some of the topics (e.g., ecosystem services, tribal and cultural resources, land status and use patterns) simply do not have as much relevant and available information as other topics. Assessment teams may want to consider additional ways to interact with scientists and others to create functioning communities of practice related to science exchange for forest planning. In the same way, agency scientists may consider forging new and enduring relationships with planners and managers that could generate new science that is of immediate relevance. We found similarities across all forests in the most common approaches to identifying BASI in addition to other approaches such as data sharing meetings, a wiki review site, and requests for a science synthesis. Information from non-peer-reviewed sources was more difficult for planners to assess and evaluate. Sharing best practices, along with revised guidance for planning rule implementation, may help national forest planners improve the utility, efficiency, and quality of forest assessments.

- Terrestrial ecosystems, aquatic ecosystems, and watersheds
- Air, soil, and water resources and quality
- System drivers, including ecological processes, disturbance regimes, and stressors
- Baseline carbon stocks
- Threatened, endangered, proposed and candidate species; potential species of concern
- Social, cultural, and economic conditions
- Benefits people obtain from the planning area (ecosystem services)
- Multiple uses and their contributions to economies
- Recreation settings, opportunities, and access, and scenic character
- Renewable and nonrenewable energy and mineral resources
- Infrastructure (recreational facilities, transportation and utility corridors)
- Areas of tribal importance
- Cultural and historic resources and uses
- Land status, ownership, use, and access patterns
- Existing designated areas including wilderness and wild and scenic rivers; need and opportunity for new designations

Figure 1. Topics for forest plan assessments (36 CFR 219.6)

are expected to incorporate state-of-the-art scientific knowledge to manage public lands (Lachapelle et al. 2003). However, the role of science in natural resource decision-making has become much more complex (Mills and Clark 2001). Recent literature acknowledges that no important policy issue or decision is purely technical, that established practices are problematic, and that politics are unavoidable (Brunner et al. 2005). In spite of this, numerous policies reflect the scientific management paradigm in their calls for best available science.

In the United States, many policies and statutes contain references to best available science, including the Marine Mammal Protection Act of 1972, the Endangered Species Act of 1973, and the Safe Drinking Water Act of 1974. Despite references to the concept of best available science, these policies do not include specific definitions of its properties, standards, or practical application in the decision-making process (Doremus 2004; Smallwood et al. 1999), leading to different definitions of what it means. Ryder et al. (2010) identify attributes of best available science from published literature that span topics such as endangered species legislation, protection of conservation areas, forest management, water resource management, and ocean fisheries. The paper highlights the diversity of attributes assigned to best available science, and demonstrates that no single attribute is common to all studies, suggesting that best available science is context specific (Ryder et al. 2010). Moreover, as Lowell and Kelly (2016) observe, the ability to use best available science may be inhibited by institutional constraints within particular agencies limited by time or organizational capacity. Other literature has attempted to assign descriptors to the concept. For example, “best” often connotes scientific information with the greatest degree of excellence and authenticity based on sound logic (Moghissi et al. 2010), or that there is no better scientific information, and suggests the use of the most relevant and contemporary data and methods (National Research Council 2004). “Available” connotes scientific information that is accessible and attainable (Moghissi et al. 2010), or that decisions can be consistent with the scientific information that is available even though data gaps exist (National Research Council 2004). “Science or Scientific information” is defined as knowledge that emerges from a process of observation, identification, description, and testing of explanatory hypotheses about fundamental principles that govern cause-and-effect (National Research Council 2004). The National Research Council report includes guidelines for effectively using best available science, including concepts of relevance, inclusiveness, objectivity, transparency and openness, timeliness, and peer review. Finally, Charnley et al. (2017) analyzed a science synthesis for three national forests and suggest criteria for evaluating “best available *social* science,” which may be different from the criteria used to evaluate best available biophysical science.

A key aspect of the 2012 planning rule is that it requires the planning process to draw on the best available scientific information (36 CFR 219.3). The preamble to the planning rule notes that there is a range of information that can be considered BASI, stating:

> In some circumstances, the BASI would be that which is developed using the scientific method, which includes clearly stated questions, well-designed investigations and logically analyzed results, documented clearly and subjected to peer review. However, in other circumstances the BASI for the matter under consideration may be information from analyses of data obtained from a local area, or studies to address a specific question in one area. In other circumstances, the BASI also could be the result of expert opinion, panel consensus, or observations, as long as the responsible official has a reasonable basis for relying on that scientific information as the best available. (77 FR 21192 [April 9, 2012])

Planning Directives are agency guidance documents that direct implementation of rules such as the 2012 planning rule, and directives for assessments are in Chapter 10 of the Land Management Planning Handbook (USDA Forest Service 2015a). The definition of BASI is contained in the “zero code” chapter of the handbook and specifies three primary criteria for determining BASI: accuracy, reliability, and relevance (FSH 1909.12.07.12), in addition to referencing the Data Quality Act (PL 106–554) for guidance on evaluating available information (Figure 2). Available is defined as information that currently exists in a form useful for the planning process without further data collection, modification, or validation (FSH 1909.07.01).

The directives also provide guidance regarding sources of scientific information. The sources mentioned in the guidance include peer-reviewed articles, scientific assessments, other scientific information (expert opinion, panel consensus, inventories, or observational data), data prepared and managed by the Forest Service

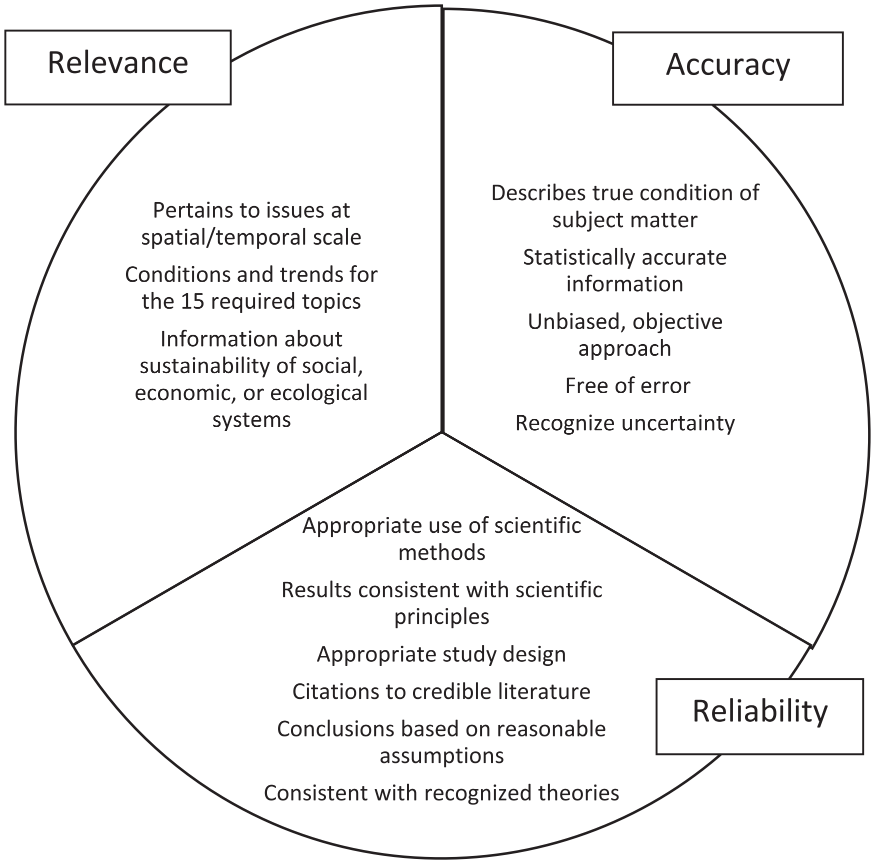

Figure 2. Criteria for determining best available scientific information (BASI). Source: Forest Service Handbook 1909.12.07.12

or other federal agencies, information prepared by universities, national research networks, and other reputable scientific organizations, and data or information from public and governmental participation (FSH 1909.12.07.13).

At the US Forest Service, two regional science synthesis efforts were initiated to assist forest planners in identifying BASI for their assessments. The first synthesis included the Sierra Nevada, southern Cascades, and Modoc plateau areas of California, and informed plan revisions on three national forests (Long et al. 2014). The second synthesis is currently underway as part of the Northwest Forest Plan area planning process, which covers 17 national forests and five Bureau of Land Management units across parts of the Cascade and coastal ranges of Washington, Oregon, and northern California. Once drafted, the synthesis report underwent independent third-party peer review, in addition to public review, and is currently under revision (Spies et al. 2017). Science synthesis efforts represent a noteworthy approach to developing BASI for use in forest assessments, creating a role for public engagement, and for employing a bioregional approach to assembling the latest science for use by multiple forests.

Methods

We used an exploratory case study approach to examine four national forest planning units that were revising their forest plans under the 2012 rule. Information on the USFS website helped us determine the planning status of each national forest as of spring 2015. The primary selection criterion was completion of the assessment process by spring 2015. We also strove to select national forests from different regions. Based on these criteria, we selected the Chugach National Forest (Alaska), Cibola National Forest (New Mexico), Inyo National Forest (California), and the Nantahala and Pisgah National Forests (North Carolina). Table 1 displays characteristics of each national forest planning unit in our sample.

Our research approach relied on content analysis of documents and interview data. We began by conducting a chapter-by-chapter analysis of each forest's assessment report to identify and characterize the information presented. We recorded page counts for each of the 15 assessment topics specified in the 2012 rule. In some cases, the chapters directly aligned with the required topics (Figure 1). In other cases, we had to make a more subjective characterization of the chapter contents. We also noted and analyzed any references to the use of best available science.

Second, as part of the document review, we analyzed data sources used in the assessment. For each assessment report, we identified all of the items cited in the reference section. We then coded each cited item according to the type of publishing entity and the type of document. Every cited item was placed in one category for each coding exercise. For each cited item, we determined the appropriate categories by examining the information in the citation entry and (when necessary) directly reviewing the item or gathering information on the publishing entity. We grouped publishing entities into five types: government; non-government; scientific, scholarly, or peer-reviewed; universities; and unknown or other (Table 2). This categorization approximates the rigor of scientific review, but there is overlap in categories. Most scholarly journals require a double-blind peer-review process, where reviewers and authors are

Table 1. Characteristics of national forests in the study.

Management unit(s)	Geography	Total acreage* (millions of acres)	Notes on use and resources	Designated early adopter?	Most recent previous plan revision	Notes on current plan revision
Chugach National Forest Alaska Region (R10)	Southcentral Alaska: major geographic areas are Cooper River, Prince William Sound, and eastern Kenai Peninsula	6.26	Subsistence, timber, recreation, mining. Human use concentrated in Kenai area. Very limited road coverage and use in other areas. Habitat for all 5 Pacific salmon species	Yes	2002	Managed by a planning team housed within unit
Cibola National Forest Southwest Region (R3)	West-Central New Mexico: Eight noncontiguous parcels organized around distinct mountainous areas known as "sky islands"	2.11	Recreation, timber, cultural heritage, range. Surrounding region experiencing population growth and demographic changes. Pinyon-juniper & ponderosa pine are predominate vegetation types	Yes	1985	Managed by a planning team housed within unit. Does not include 4 associated national grasslands
Inyo National Forest Pacific Southwest Region (R5)	Eastern California & West Nevada: Two noncontiguous parcels at intersection of Sierra Nevada, Great Basin, and Mojave Desert areas	2.07	Water supply, hydroelectricity, recreation, timber, range. Nearly 47% of total area is wilderness. Focus on wildland fire management. Substantial variation in vegetation type, habitat, and elevation	Yes	1988	One of three early adopters in R5. Coordination through a regional planning team, with separate planning teams for each unit. Each unit releases its own assessment & forest plan. Joint EIS for 3 units
Nantahala & Pisgah National Forests Southern Region (R8)	Western North Carolina: Blue Ridge region of Appalachian Mountains	2.48	Timber, recreation, cultural/historical heritage, water development. Located in Blue Ridge National Heritage Area. Hardwood forest with high species diversity	No	1987	Both units will use same revised plan. Managed by planning team housed at NF in NC headquarters

*Total acreage includes NFS-owned land and acreage under other ownership within each unit. Source: USDA Forest Service 2015b.

Table 2. Categories for coding type of publishing entity.

Publishing entity	Description of coding criteria
Government	Federal, tribal, state, or local governments in the United States; foreign governments; international intergovernmental groups such as the United Nations and affiliates. Includes peer-reviewed and non-peer-reviewed materials
Non-government	Materials not published by a government agency, university, or peer-reviewed entity. Includes businesses, consulting firms, and advocacy groups
Scientific scholarly or peer reviewed	Associations, societies, journal publishers, university presses, or other entities that produce peer-reviewed scientific or scholarly material
Universities	Materials from universities that may or may not be subject to rigorous academic peer review. Includes university or college departments, programs, laboratories, and centers, and theses and dissertations from universities
Unknown or other	News organizations or other undefined groups; disposition of publisher could not be determined

unknown to each other. University and government agency scientific documents often require peer review, but the level of rigor of the review may be variable. It was not possible to discern the level or type of peer review or scientific rigor for each category.

For the type of document, we sorted the references into 12 categories: academic book; non-academic book; conference proceeding; correspondence; database; scientific journal; news; technical report; statute or regulation; thesis or dissertation; website; and unknown (Table 3).

Our final data collection activity was qualitative interviewing with members of the planning teams at three of the forests in our study. We conducted nine semi-structured interviews (nine people in total; three interviews each from three forests). Unfortunately, we were not able to recruit interview participants from the Cibola planning effort. Potential interview participants were identified through the list of preparers included in each assessment document. Interviewees were subject matter experts who had contributed material to the assessment reports, along with planning staff officers or coordinators. Interview questions explored the overall structure of the assessment process, the role of the planning directives, the overall organization of the forests' plan revision efforts, and approaches to identification and use of best available science. Interviews were audio-recorded, transcribed, and analyzed using content analysis with a coding framework developed by the study team. Content analysis is a method that uses codes, or labels that assign meaning to descriptive or inferential data collected during a study (Miles et al. 2014). The codes are used to retrieve and organize similar data and aid the researcher in relating data to research questions, theoretical concepts, and themes (Araujo 1995; Miles et al. 2014).

Results

We present results of our analysis in three sections: 1) required topics; 2) sources and types of information; and 3) identifying and using BASI.

Required topics in the forest assessment

The number and percent of pages devoted to each required topic is presented in Table 4. We did not include introductory front matter in the page counts. A 0* entry means that the assessment report did not

Table 3. Categories for coding type of document.

Document type	Description of coding criteria
Academic book	An item printed, bound, distributed as a book, or released as an e-book by a peer-reviewed/scholarly entity
Non-academic book	An item printed, bound, distributed as a book, or released as an e-book by an entity whose primary orientation is not peer reviewed/scholarly
Conference proceeding	Papers, abstracts, and talks presented at a conference and published in a conference proceeding collection
Correspondence	Letters or emails written by individuals of any affiliation
Database	Raw data or data analysis tools/software; online databases
Scientific journal	A peer-reviewed article in a scholarly journal
News	Articles in newspapers (print or online) and news magazines
Technical report	Technical and research reports, white papers, policy papers, fact sheets, briefings
Statute, regulation, and planning documents	Federal, state, or local laws and rules; EISs; management plans; strategic plans
Thesis or dissertation	Advanced degree projects and papers
Website	One or more webpages on a non-database website, including encyclopedias with narrative entries
Unknown	The type of document could not be discerned

Table 4. Page counts and percentages of total pages for 15 required assessment topics.

		Number of pages (pct. of total pages in report)				Pct.
Topic #	Assessment topics (per 36 CFR 219.6)	Chugach	Cibola	Inyo	N&P	Avg.
1	Terrestrial ecosystems, aquatic ecosystems, and watersheds	66 (22.9%)	51.5 (11.2%)	38.5 (21.0%)	29 (15.7%)	**17.7**
2	Air, soil and water resources and quality	17 (5.9%)	88 (19.2%)	9 (4.9%)	19 (10.3%)	**10.1**
3	System drivers (processes, disturbance regimes, and stressors)	40 (13.9%)	21 (4.6%)	15 (8.2%)	7 (3.8%)	**7.6**
4	Baseline carbon stocks	7 (2.4%)	6 (1.3%)	4 (2.2%)	7 (3.8%)	**2.4**
5	Threatened, endangered, candidate species; potential species of conservation concern	12 (4.2%)	36 (7.9%)	24 (13.1%)	4 (2.2%)	**6.8**
6	Social, cultural, and economic conditions	21 (7.3%)	71 (15.5%)	14 (7.7%)	8 (4.3%)	**8.7**
7	Benefits obtained by people (ecosystem services)	49 (17.0%)	0* (0.0%)	2.5 (1.4%)	4 (2.2%)	**5.1**
8	Multiple uses and their contributions to economies	0* (0.0%)	26 (5.7%)	15 (8.2%)	17 (9.2%)	**5.8**
9	Recreation settings, opportunities, and access, and scenic character	29 (10.0%)	39 (8.5%)	15.5 (8.5%)	21 (11.4%)	**9.6**
10	Renewable and nonrenewable energy and mineral resources	17 (5.9%)	18 (3.9%)	3.5 (1.9%)	8 (4.3%)	**4.0**
11	Infrastructure	2 (0.7%)	12 (2.6%)	9.5 (5.2%)	10 (5.4%)	**3.5**
12	Areas of tribal importance	2 (0.7%)	13 (2.8%)	4.5 (2.5%)	3 (1.6%)	**1.9**
13	Cultural and historical resources and uses	3.5 (1.2%)	40 (8.7%)	7 (3.8%)	23 (12.4%)	**6.6**
14	Land status and ownership, use, and access patterns	8 (2.8%)	17 (3.7%)	7 (3.8%)	9 (4.9%)	**3.8**
15	Designated areas, potential/need for new designations	15 (5.2%)	20 (4.4%)	14 (7.7%)	16 (8.7%)	**6.5**
	TOTAL	**288.5**	**458.5**	**183**	**185**	**100**

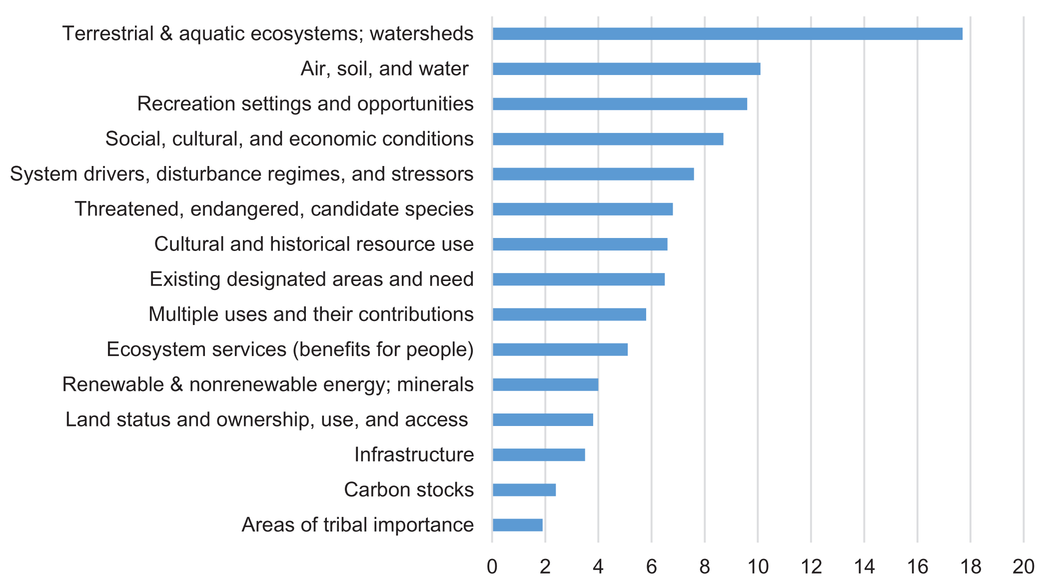

Figure 3. Average percentage of pages devoted to each topic in each forest assessment for all forests combined

have any pages that were specifically devoted to the topic, but references to the topic were instead interspersed throughout the report and it was too difficult to separate them from other topic page counts.

Two of the national forests (Inyo and Nantahala-Pisgah) published assessment reports that consisted of 15 chapters that directly reflected each of the required topics. Meanwhile, the Chugach and Cibola took a different approach; some of the chapter topics aligned with the topic requirements in the 2012 rule, but other required topics were broken up and distributed among multiple chapters. For example, the Chugach had one chapter for areas of tribal importance and one chapter for land status and ownership, but divided the terrestrial and aquatic ecosystems and watersheds

Table 5. Percent allocation of predominant topics among four forest assessments.

Rank	Chugach topics	Pct.	Cibola topics	Pct.	Inyo topics	Pct.	N&P topics	Pct.
1	Terrestrial and aquatic ecosystems	23%	Air, soil, and water	19%	Terrestrial and aquatic ecosystems	21%	Terrestrial and aquatic ecosystems	16%
2	Benefits obtained by people (ecosystem services)	17%	Social, cultural, and economic conditions	16%	Threatened and endangered species	13%	Cultural and historic resources	12%
3	System drivers, disturbance regimes, and stressors	14%	Terrestrial and aquatic ecosystems	11%	Recreation settings and opportunities	9%	Recreation settings and opportunities	11%
4	Recreation settings and opportunities	10%	Cultural and historic resources	9%	System drivers, disturbance regimes, and stressors	8%	Air, soil and water	10%
5	Social, cultural, and economic conditions	7%	Recreation settings and opportunities	9%	Multiple uses	8%	Multiple uses	9%
Total		71%		63%		59%		59%

into five chapters, one each for watersheds, fish, wetlands, vegetation, and wildlife, and these chapters were integrated with material discussing soils and carbon stocks. Two forests did not have any pages specifically devoted to one required topic each (benefits obtained by people for the Cibola, and multiple uses for the Chugach), but these subjects were still referenced in the context of the other topics.

For all four assessments combined, the required topic with the largest average percentage of pages was terrestrial and aquatic ecosystems and watersheds (17.7%), followed by air, soil, and water resources (10.1%) and recreation opportunities (9.6%) (Figure 3).

Terrestrial and aquatic ecosystems and watersheds comprised the largest section of the assessment for three of the four forests. Air, soil, and water was especially prominent for the Cibola National Forest, and all of the forest assessments covered recreation evenly. In contrast, the three required topics with the smallest page counts, on average, were areas of tribal importance (1.9%), carbon stocks (2.4%), and infrastructure (3.4%). Benefits obtained by people (ecosystem services) had the most variable coverage, with one of the shortest sections for three of the four forest assessments, but the second longest topic for the Chugach National Forest. In all four assessment documents, benefits obtained by people were mentioned throughout the document in sentences or paragraphs at too fine a scale for this analysis to count.

We found some variation among the forest assessments in terms of the extent to which a forest focused on a particular topic (Table 5).

For the Chugach National Forest, the top five topics comprised more than 70% of the assessment, with the bulk emphasizing terrestrial and aquatic ecosystems, which reflects the importance of salmon habitat. The Chugach was the only forest to emphasize ecosystem services as a predominant framework to capture benefits obtained by people. However, other forests may have captured this topic under the category of multiple uses. Disturbance regimes (fire and invasive species) were also important for the Chugach. The Cibola National Forest was unique in their emphasis on air, soil, and water as well as social, cultural, and economic conditions and cultural and historic sites. Because water access is very important in the southwest, the predominance of this topic is not surprising. For the Inyo National Forest, the topic of threatened and endangered species was prominent, while topics related to recreation and disturbance regimes (fire, invasive species, and other ecosystem stressors) were also important. Meanwhile, cultural and historical resources were prominent in the Nantahala and Pisgah National Forests, along with recreation.

Although the 2012 rule provides a list of 15 distinct required topics, these topics overlap and are not discussed in complete isolation from one another. As we found in our analysis, it is difficult to discuss multiple uses without also discussing benefits obtained by people; air, soil, and water resources; recreation; and terrestrial and aquatic ecosystems and watersheds. In our analysis, we often found that an assessment chapter devoted to a required topic also contained information that closely resembled material discussed elsewhere. In particular, we found the chapters on multiple uses and benefits obtained by people to be largely redundant, given the other topics that were also included in the report.

Sources and types of information in the forest assessment

To understand the sources and types of information used in the assessments, we conducted a systematic examination and tally of citations by publication source and type. Overall, government sources were the most commonly cited information source (51.8%), followed by scientific scholarly publications (30.7%) (Table 6).

Table 6. Citations based on information source for forest assessments.

	Count (Percent)				
Publishing entity	**Chugach**	**Cibola**	**Inyo**	**Nantahala & Pisgah**	**TOTAL (Mean)**
Government	239 (53.6%)	159 (49.8%)	131 (49.8%)	109 (54.0%)	**638 (51.8%)**
Scientific scholarly or peer reviewed	155 (34.8%)	82 (25.7%)	82 (31.2%)	63 (31.2%)	**382 (30.7%)**
Non-government	21 (4.7%)	39 (12.2%)	24 (9.1%)	18 (8.9%)	**102 (8.7%)**
Universities	30 (6.7%)	39 (12.2%)	19 (7.2%)	11 (5.5%)	**99 (7.9%)**
Unknown or other	1 (0.2%)	0 (0%)	7 (2.7%)	1 (0.5%)	**9 (0.9%)**
TOTAL	446	319	263	202	**1230**

Table 7. Citations based on document type for forest assessments.

	Count (Percent)				
Document type	Chugach	Cibola	Inyo	Nantahala & Pisgah	**TOTAL**
Technical report	174 (39.0%)	121 (37.9%)	108 (41.1%)	73 (36.1%)	**476 (38.5%)**
Scientific journal article	129 (28.9%)	47 (14.7%)	63 (24.0%)	48 (23.8%)	**287 (22.8%)**
Academic book	28 (6.3%)	36 (11.3%)	20 (7.6%)	15 (7.4%)	**99 (8.2%)**
Statute, regulation, or planning document	43 (9.6%)	26 (8.2%)	23 (8.8%)	12 (5.9%)	**104 (8.1%)**
Website	33 (7.4%)	42 (13.2%)	3 (1.1%)	13 (6.4%)	**91 (7.0%)**
Database	17 (3.8%)	25 (7.8%)	17 (6.5%)	18 (8.9%)	**77 (6.8%)**
Conference proceeding	10 (2.2%)	3 (0.9%)	6 (2.3%)	18 (8.9%)	**37 (3.6%)**
Non-academic book	4 (0.9%)	9 (2.8%)	10 (3.8%)	0 (0.0%)	**23 (1.9%)**
Correspondence	0 (0.0%)	7 (2.2%)	5 (1.9%)	4 (2.0%)	**16 (1.5%)**
Thesis or dissertation	8 (1.8%)	2 (0.6%)	2 (0.8%)	1 (0.5%)	**13 (0.9%)**
News	0 (0.0%)	1 (0.3%)	3 (1.1%)	0 (0.0%)	**4 (0.4%)**
Unknown	0 (0.0%)	0 (0.0%)	3 (1.1%)	0 (0.0%)	**3 (0.3%)**
TOTAL	446 (100.0%)	319 (100.0%)	263 (100.0%)	202 (100.0%)	**1230 (100.0%)**

A large portion of the government sources included US Forest Service publications (average of 28%), which were more commonly cited than other federal government sources (average of 12%) or state and local governments (average of 11%). Some variation exists among the forests in our sample, but the trends were consistent in terms of reliance on government sources and scholarly peer-reviewed publishers for the majority of citations (82.5% combined average for both categories). The Chugach relied to a greater degree on scholarly publications than other forests. The Cibola had the highest proportion from non-governmental organizations and trade groups (12.2%). The Inyo and the Nantahala and Pisgah mirrored the group average.

Next, we explored citations by the type of document referenced. We found that technical reports were the most common type of document cited in the assessments, with an average of 38.5% (Table 7).

The technical report classification is broad and includes technical and scientific reports, policy briefings, white papers, and other types of information (sometimes referred to as gray literature). All four forests were consistent in the ratio of technical reports cited. The second most common document type was the scientific journal article, with an average of 23%, although the Cibola assessment featured far fewer than the other forests. All of the forests cited a wide variety of regulations, statutes, and planning documents, (e.g., water quality regulations, county comprehensive plans, environmental impact statements, state resource management plans, and forest plans). The Cibola assessment featured the greatest variety of document types, relying on websites and academic books more than the other forests. The Nantahala and Pisgah assessment relied more heavily on conference proceedings. The least commonly cited document types, on average, were news articles (0.4%), theses or dissertations (0.9%), and correspondence (1.5%). Although there is a separate category for websites, documents in many of the other categories were readily available online.

Table 8. Approaches to identifying and using BASI from interview data.

BASI approach	Chugach	Nantahala/ Pisgah	Inyo
Literature review (e.g. Google Scholar for scholarly literature)	x	x	x
Forest Service reports, monitoring data	x	x	x
Personal expertise/training/judgement	x	x	x
Existing dataset/database	x		x
Nearby Forest Service research station		x	x
Nearby university		x	
Host data sharing meeting (partners and stakeholders)		x	
Meet with scientists		x	
Post draft documents on wiki site for public review/editing			x
Other public review opportunity		x	
Gray ("non-peer-reviewed") literature, traditional knowledge			x

Identifying and using best available scientific information in the forest assessment

In interviews, respondents were asked how they identified and obtained BASI for their assessment. Table 8 displays the different approaches used by three of the four forests.

Literature reviews and searches, Forest Service reports and datasets, and personal scientific expertise were mentioned by all nine respondents as primary ways that they identified and obtained BASI. Literature reviews focused on identifying peer-reviewed journals, conference proceedings, or agency reports. Existing datasets and nearby Forest Service research stations and universities were also relied upon. The Sierra Nevada science synthesis effort, which informed the Inyo National Forest assessment, took nearly 18 months to complete (Long et al. 2014). The Inyo also posted draft documents on a wiki site for public review and editing. All nine interviewees stated that their assessment team used the Draft Planning Directives, but also mentioned that the directives were not clear, save for the focus on organizing around the 15 topics. No respondent mentioned specific guidance beyond the draft directives on how to identify BASI. The final directives do specifically address the definition of BASI, as discussed above (Figure 2). Gray literature and traditional knowledge presented challenges, as it at times conflicted with peer-reviewed information. Two respondents mentioned that they wanted to incorporate this type of information, but were unsure how to do so.

Assessments must document what information was determined to be BASI, explain the basis for that determination, and explain how the information was applied to the issues considered (36 CFR

219.3). Our analysis of the assessment documents reveals that all documents discuss the use of high-quality and valid scientific information, citing criteria such as clearly defined and well-developed methodology; standardized methodology; logical conclusions; and reasonable inferences (Chugach National Forest 2014; Inyo National Forest 2014; Nantahala and Pisgah National Forests 2014; Cibola National Forest and National Grasslands 2015). The assessments for all forests mention their reliance on information relevant to their specific forests and issues. Only the Nantahala-Pisgah assessment presented a hierarchy of information sources, with peer-reviewed journal articles the highest, followed by government documents and reports, monitoring datasets, theses and dissertations from universities, and expert opinion where facts were not known through the other sources.

Discussion

The 2012 forest planning rule requires that each national forest or grassland conduct a scientific assessment to guide plan development. We found that assessment reports were disproportionately heavy in science related to terrestrial and aquatic ecosystems, and more limited in treatment of infrastructure, land ownership and access patterns, cultural heritage, and areas of tribal importance. Recreation was the only topic to receive consistent attention across all four forests, although the topic was overshadowed by terrestrial and aquatic ecosystems. We may only speculate about why terrestrial and aquatic ecosystem information was the most prevalent in all four forests, but it is consistent with agency administrative hiring practices since the 1980s that have emphasized recruitment of ecologists, biologists, and other biophysical scientists, compared to social scientists, for example (Thomas and Mohai 1995). The abundance of agency specialists in these topic areas may reinforce the relative importance of terrestrial and aquatic ecosystems compared to other topic areas, such as recreation, social science, or cultural resource management. This has been confirmed by a national assessment of interdisciplinary planning team composition (Cerveny et al. 2011). Ensuring that assessment teams include broad and diverse disciplinary experts will help address this challenge, recognizing that some forests may not have access to necessary disciplinary specialists. It is also possible that some of the topics (e.g., ecosystem services, tribal and cultural resources, land status and use patterns) simply do not have as much relevant and available information as other topics.

The benefits obtained by people (ecosystem services) topic received little or no explicit coverage in all but one assessment. The limited coverage of ecosystem services may make sense because it was not even considered an area of research until the late 1990s, so there would be less existing information on certain important ecosystem service topics (e.g., pollination, stormwater attenuation, medicinal resources, and spiritual and historical significance) compared to recreation, threatened and endangered species, and other traditional assessment topics (Blahna et al. 2017). Previously, "forest benefits to people" were considered elements of "multiple use" and planners might have addressed these benefits under the "multiple use" topic. Ecosystem services (ES) are often categorized into four classes: provisioning, regulating, cultural, and supporting. Timber, recreation, wildlife, and other traditional forest planning topics all fall into one of these four classes. Another reason for lack of coverage of ecosystem services may be that planners could not differentiate the normal assessment topics from the ecosystem service classes. Efforts to help planning team members understand ecosystem services approaches and how they can be used to inform the planning process may be warranted, and the rule's current requirement for only using existing data in assessments may need to be revisited (Blahna et al. 2017). For example, implementation teams working on ecosystem services may consider the benefits of providing specific tools, frameworks, and guidelines for integrating ecosystem services models into the forest planning process. In addition, critical issues and topics (e.g., newly listed threatened or endangered species, or changing recreation behaviors) that forest plans need to address may change from one planning cycle to the next.

The specific required topics may not be universally appropriate for every planning unit. Planners felt obligated to address all 15 topics, but the lack of coverage for some topics suggests that the topic was not deemed relevant or meaningful for their plan, there was no available data on the topic, or it was unclear how the topics could be covered. Variability in application of the directives, and acknowledgment of local context and conditions, is consistent with the overall Forest Service approach toward decentralized decision-making (Kaufman 1960; Tipple and Wellman 1991; Koontz 2007) and localized interpretation by planning teams, similar to "street-level" bureaucrats who create de facto policy through everyday practice (Sabatier et al. 1995; Lipsky 2010; Trusty and Cerveny 2012). Kaufman (1960) observes the traditional Forest Service practice of maintaining control of heterogeneous and geographically dispersed management units by issuing centralized directives that provide parameters (or "side boards") within which line officers have some leeway to make decisions. This tendency toward uniformity and "pre-formed" decisions may result in some inefficiencies and omissions. The implied obligation to cover all 15 topics may have resulted in some assessments that distract from the most important management issues for the unit. This will be especially important during the next stage of planning—revision or amendment—where the assessment data will be used to analyze different management scenarios. Approaches for identifying and analyzing the most relevant assessment data that address the key environmental problems or social conflicts that confront each planning unit will be needed (Blahna et al. 2017). This is especially important for topics like human benefits (ecosystem services) and multiple uses, which cut across all of the other topical areas and are not as easily categorized in assessments. Recent efforts to engage the public in science synthesis efforts in support of forest planning suggest that there may be an important role for the public to help prioritize forest assessment topics.

The most common sources of information were government sources, followed by scholarly academic sources. Many of the agency sources were peer-reviewed scientific studies, which appear to be especially useful because of the topical specificity or geographic focus (relevance). Although not all technical reports are peer reviewed, they may be more accessible and usable compared to scholarly journal articles, which may require planning team members to interpret the findings and make inferences for relevance to local conditions. This finding is consistent with previous research examining the information needs and sources of Forest Service fire managers (Ryan and Cerveny 2011) and recreation managers (Ryan and Cerveny 2010). Fire managers relied heavily on agency information sources. Although managers in the study noted the availability of high-quality, relevant information, they faced significant barriers in terms of time, funding,

and personnel to access and use that information. Similarly, recreation managers also relied on agency information sources, but indicated strong preferences for enhanced interactions with agency scientists, including collaborative research, conferences, and a desire for agency researchers to reach out more directly to managers to ensure their research was relevant and useful. With regard to forest assessments, engagement with scientists is particularly important for topics where little research is available. Assessment teams may want to consider additional ways to interact with scientists and others to create functioning communities of practice related to science exchange for forest planning. In the same way, agency scientists may consider forging new and enduring relationships with planners and managers that could generate new science that is of immediate relevance.

The 2012 planning rule and its directives provide criteria for BASI, and we found similarities across all forests in the most common approaches to identifying BASI, in addition to other approaches, such as data sharing meetings, a wiki review site, and requests for a science synthesis. Information from non-peer-reviewed sources was more difficult for planners to assess and evaluate, and it is not clear how this information was incorporated into each assessment. Teams may not have the capacity to separately evaluate and assess the many different types and sources of information, and so they rely on hierarchical ranking approaches (peer-reviewed sources being highest rank) to streamline the evaluation. Planning teams clearly value peer-reviewed and agency-generated information, and it may be that they are simply identifying information that is "available" and using the "best" of that based on their judgments. This may result in situations where the science expertise on each team could influence BASI decisions. As discussed above, consideration of the makeup and membership of the assessment team is important here, as well as increased transparency regarding the process for determining science relevance and quality.

Conclusion

Implementation of the US Forest Service 2012 planning rule is still in its early stages. Our study illustrates that forest planners use a variety of approaches to address required topics, and do rely on BASI as they develop their forest assessments. While each national forest assessment included the 15 required topics, we found considerable variation in coverage, which suggests that planners may emphasize topics most relevant to their forest, or that variation exists in terms of what science or planning team expertise is available or deemed desirable. The predominance of science related to terrestrial and aquatic ecosystems in the assessments compared to other topics warrants further inquiry in order to learn whether this asymmetry is based on policy, availability of information, existing expertise, or other factors. Efforts to include the public in the process of prioritizing topics for the assessments could also be evaluated. The reliance on government sources for scientific information suggests that agency-supported science is either more accessible or more relevant to the planning team. It also suggests that there may be benefits to bolstering "communities of practice" for key topical areas covered by forest assessments that bring together university and agency scientists with managers.

The appearance of science in an assessment report is important, but the actual *use* of science in planning may be more important. Although our findings are not generalizable to all national forests, they do provide an understanding of plan assessment activities for those in the early phases of forest planning, whose efforts are likely to inform and influence other national forests. Our goal was to provide an early glimpse of plan revision efforts in order to highlight important lessons learned and create a foundation for future research. For example, do planners find that the required topics provide useful guidance for developing their assessments? How can planners become more confident in knowing what BASI is, and how to identify and use it? Is additional guidance needed for incorporation of traditional knowledge and other information? Of particular interest is whether the "science synthesis" information is useful to forest planners in addressing their forest assessment needs, given the significant agency resources devoted to developing science syntheses. Finally, how is information from the assessment used in forest plan revision (development and selection of management options) and monitoring efforts? While draft environmental impact assessment (EIS) reports are available in various stages, as of this writing only one final Record of Decision (ROD) has been issued for a forest plan undergoing revision under the 2012 rule. Thus, it remains to be seen how scientific information will be incorporated in development of alternatives, impact statements, and final management decisions.

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LETTER

How will climate change affect wildland fire severity in the western US?

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Supplementary material for this article is available online

Abstract

Fire regime characteristics in North America are expected to change over the next several decades as a result of anthropogenic climate change. Although some fire regime characteristics (e.g., area burned and fire season length) are relatively well-studied in the context of a changing climate, fire severity has received less attention. In this study, we used observed data from 1984 to 2012 for the western United States (US) to build a statistical model of fire severity as a function of climate. We then applied this model to several ($n = 20$) climate change projections representing mid-century (2040–2069) conditions under the RCP 8.5 scenario. Model predictions suggest widespread reduction in fire severity for large portions of the western US. However, our model implicitly incorporates climate-induced changes in vegetation type, fuel load, and fire frequency. As such, our predictions are best interpreted as a *potential* reduction in fire severity, a potential that may not be realized due human-induced disequilibrium between plant communities and climate. Consequently, to realize the reductions in fire severity predicted in this study, land managers in the western US could facilitate the transition of plant communities towards a state of equilibrium with the emerging climate through means such as active restoration treatments (e.g., mechanical thinning and prescribed fire) and passive restoration strategies like managed natural fire (under suitable weather conditions). Resisting changes in vegetation composition and fuel load via activities such as aggressive fire suppression will amplify disequilibrium conditions and will likely result in increased fire severity in future decades because fuel loads will increase as the climate warms and fire danger becomes more extreme. The results of our study provide insights to the pros and cons of resisting or facilitating change in vegetation composition and fuel load in the context of a changing climate.

Introduction

Fire regimes in North America are expected to change over the next several decades as a result of anthropogenic climate change (Dale *et al* 2001). Fire activity (i.e., annual area burned and fire frequency) is expected to increase in many regions (Krawchuk *et al* 2009, Littell *et al* 2010) and new research shows that fire seasons are now starting earlier and ending later compared to previous decades (Jolly *et al* 2015). However, the effect of climate change on one very important fire regime characteristic—*fire severity*—is not well-studied or understood (Flannigan *et al* 2009, Hessl 2011). In the context of this paper, we define severity as the degree of fire-induced change to vegetation and soils one year post-fire (Key and Benson 2006, Miller and Thode 2007). For example, a stand-replacing fire in upper-elevation conifer forest is considered high severity because the site has drastically changed one year post-fire compared to pre-fire conditions, whereas a surface fire in a grass-dominated ecosystem is considered low severity because the vegetation is nearly fully recovered one-year post fire.

The severity at which a site burns influences vegetation response and successional trajectory (Barrett *et al* 2011), faunal response (Smucker *et al* 2005), carbon emissions (Ghimire *et al* 2012), and erosion rates and sedimentation (Benavides-Solorio and MacDonald 2005). Furthermore, human safety and infrastructure are influenced by the severity at which a site burns (Miller and Ager 2013), and management responses to fire and allocation of firefighting resources are also influenced by the expected fire severity (e.g., Calkin *et al* 2011). As such, there is a need to better understand how fire severity will respond to a changing climate (e.g., Miller *et al* 2009).

At fine temporal scales, fire severity depends on factors that are highly variable over time, such as fire spread rate and direction (e.g., heading versus backing fire) and weather (Finney 2005, Birch *et al* 2015). At broader temporal scales, however, climate (in terms of climatic normals) is a major influence through its interactive effect on productivity (and hence amount of biomass) and moisture availability (i.e., wet versus dry ecosystems) (Parks *et al* 2014b, Whitman *et al* 2015). Consequently, because fire regimes are intrinsically defined by the characteristics of fires that occur over extended periods of time (years to centuries) (Morgan *et al* 2001), evaluations of fire severity over gradients of observed and predicted climatic normals allows for a formal assessment of how fire severity may respond to climate change.

We seek to quantify how fire severity in the contiguous western United States (US) (hereafter the ‘western US’) may respond to climate change. We use statistical relationships between observed climatic normals and fire severity (Parks *et al* 2014b, Kane *et al* 2015) to conduct a formal evaluation of future fire severity patterns. Because the relationship between climate and fire regimes is known to be weak in areas of high human impact (Parks *et al* 2014b), we used data from areas with low anthropogenic influence to build a statistical model of fire severity as a function of climatic normals over the 1984–2012 time period. We then predicted contemporary (1984–2012) and future (mid-century; 2040–2069) fire severity using climate data from numerous global climate models (GCMs) for the western US. As far as we know, this study is the first to examine how fire severity may respond to a changing climate over such a broad spatial extent. The results of this study will advance our understanding of fire regimes in the western US in the context of a changing climate and will assist policy makers and land managers to better manage for resilient landscapes.

Methods

Consistent with major fire severity mapping efforts (Key and Benson 2006, Eidenshink *et al* 2007), we define fire severity as the degree of fire-induced change to vegetation and soils. We built a statistical model of fire severity as a function of climate by first partitioning our study area (the western US; figures 1(a) and (b)) into 500 km^2 hexagonal polygons (i.e., ‘hexels’). Within each hexel, we summarized fire severity using the delta normalized burn ratio (dNBR) (Key and Benson 2006), a satellite index (resolution: 30 m) that differences pre- and post-fire Landsat TM, ETM+, and OLI images and has a high correspondence to field-based measures of severity such as the composite burn index (CBI; $R^2 \geqslant 0.65$) (van Wagtendonk *et al* 2004, Parks *et al* 2014a). The CBI is a post-fire assessment in which individual rating factors in each of several vertically arranged strata (soil and rock, litter and surface fuels, low herbs and shrubs, tall shrubs, and trees) are assessed on a continuous 0–3 scale indicating the magnitude of fire effects. A rating of 0 reflects no change due to fire, whereas 3 reflects the highest degree of change. Factors assessed include soil char, surface fuel consumption, vegetation mortality, and scorching of trees. Ratings are averaged for each stratum and then across all strata to arrive at an overall CBI rating for an entire plot. The CBI indicates that, as dNBR values increase, there is generally an increase in char and scorched/blackened vegetation and a decrease in moisture content and vegetative cover (Key and Benson 2006). Measurements of fire severity (dNBR and CBI) are generally conducted one year after fire, so any regrowth that occurs within one year will result in reduced severity compared to assessments conducted immediately post-fire; this is particularly relevant for species that recover quickly after fire (e.g., resprouting shrubs, grasses).

Fire severity (i.e., dNBR) data were obtained from the Monitoring Trends in Burn Severity project (Eidenshink *et al* 2007) for all fires $\geqslant$400 ha for the 1984–2012 time period. Raw dNBR values obtained from MTBS were adjusted using the ‘dNBR offset’ (Key 2006), which accounts for differences due to phenology or precipitation between the pre- and post-fire images by subtracting the average dNBR of pixels outside the burn perimeter. This adjustment can be important when comparing severity among fires (Parks *et al* 2014a). A mean dNBR was calculated using all pixels of all fires that intersected each 500 km^2 hexel; pixels classified as nonfuel were excluded in the calculation of the mean. We square-root transformed mean dNBR values to linearize the relationship to the CBI (figure S1).

We summarized climate normals within each hexel using five variables with known links to fire regimes (e.g., Littell and Gwozdz 2011, Abatzoglou and Kolden 2013, Parks *et al* 2015b): actual evapotranspiration (AET), water deficit (WD), annual

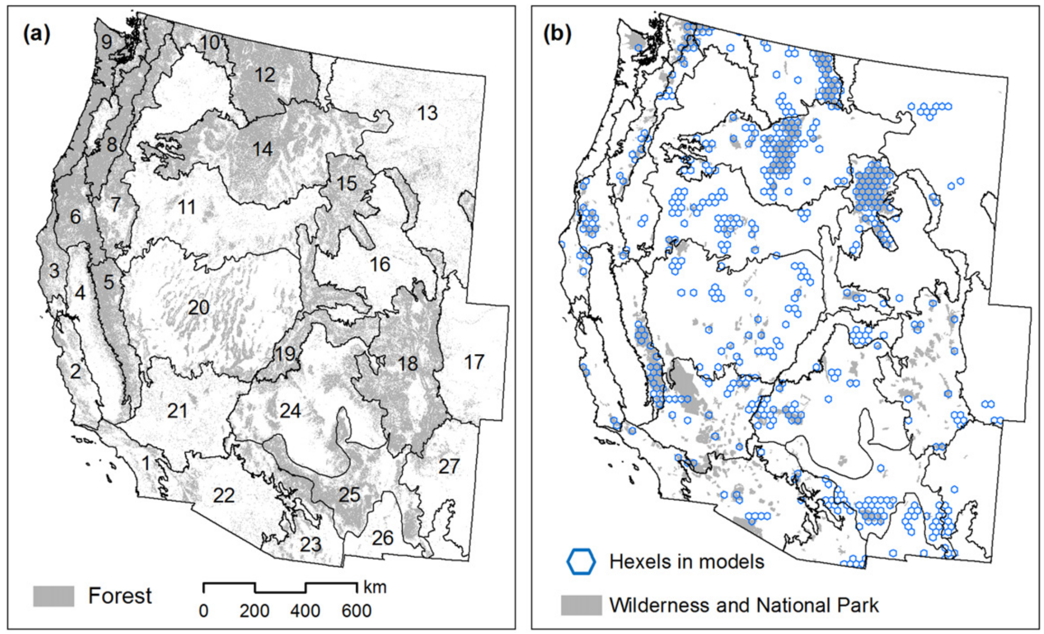

Figure 1. Study area of the western US for which we predicted changes in fire severity under a future climate. Map showing ecoregion boundaries (The Nature Conservancy 2009) and forested areas (in gray) (a) and showing designated wilderness areas and national parks (in gray) as well as the 544 hexels (in blue) used to build the models of fire severity as a function of climate (b). Ecoregion names and boundaries provided for context.

precipitation (PPT), soil moisture (SMO), and snow water equivalent (SWE). Gridded monthly temperature and PPT data were obtained from the parameter-elevation regression on independent slopes model (PRISM; Daly *et al* 2002), which uses weather station data and physiographic factors to map climate at a spatial resolution of ~800 m. In addition, daily and sub-daily surface meteorological variables (~4 km resolution) describing temperature, humidity, winds, solar radiation, and precipitation were produced following Abatzoglou (2013). These data were collectively used to compute climatic water balance following Dobrowski *et al* (2013) to estimate AET, SWE, SMO, and WD. This water balance model operates on a monthly time-step and accounts for atmospheric demand (via the Penman–Monteith equation), soil water storage, and includes the effect of temperature and radiation on snow hydrology via a snow melt model. Each variable was averaged within each hexel for the years 1984–2012, thereby matching the years of the fire severity data. We similarly summarized these five climate variables representing mid-21st century (2040–2069) conditions using 20 global climate models (GCMs) for the RCP8.5 emissions scenario (table S1). These tables were statistically downscaled to the same grid as observed data using the multivariate adapted constructed analogs approach (Abatzoglou and Brown 2012).

Because the relationship between climate and fire is weaker in landscapes that are highly influenced by humans (Parks *et al* 2014b), we built our model using data from a subset of hexels with low human influence (figure 1(b)). We selected only those hexels that were comprised of at least 50% designated wilderness or national park or had an average 'human footprint' (Leu *et al* 2008) ⩽ 2.5 (on a scale of 1–10). We further limited our dataset to include only those hexels with at least 400 ha of total burned area from 1984 to 2012. These selection criteria resulted in 544 hexels that, despite representing a small proportion of our study area (8.7%), are climatically representative of much of the western US, with the notable exception of the wet regions of the Pacific Northwest (figure S2).

Using data from the subset of 544 hexels, we modeled fire severity (dNBR) as a function of contemporary climate (1984–2012) using boosted regression trees (BRT) ('gbm' package) in the R statistical environment (R Development Core Team 2007). BRT is a nonparametric machine-learning approach that does not require *a priori* model specification or test of hypothesis (De'ath 2007). The BRT algorithm fits the best possible model to the data structure, including complex interactions among variables. It does so by building a large number of regression trees, whereby, through a forward stage-wise model-fitting process, each term represents a small tree built on the weighted residuals of the previous tree. The stage-wise procedure reduces bias, whereas variance is decreased through model averaging. The BRT method also employs 'bagging', the use of a random subset of samples, which typically improves model predictions. Comparisons to other modeling techniques indicate that BRT models consistently produce robust predictive estimates (Elith *et al* 2006). We followed the recommendations of Elith *et al* (2008) for selecting BRT options; we set the bagging fraction to 0.5, learning rate to 0.005, and tree complexity to three. We used a custom script from Elith *et al* (2008) to determine the necessary number of trees, thereby reducing the potential for overfitting. We evaluated the model fit using the (a) correlation between predicted and observed fire severity and (b) ten-fold cross-validated correlation between predicted and observed fire severity.

We used the model to predict contemporary (1984–2012) fire severity (dNBR) for all hexels in the western US. However, interpreting dNBR and changes in dNBR under a changing climate is challenging because dNBR units have no direct ecological interpretation. As such, we rescaled these predictions to correspond to the ecologically relevant composite burn index (hereafter 'inferred CBI') that ranges from 0 to 3 (Key and Benson 2006): the lowest predicted severity was given an inferred CBI of 0.1, which is the threshold for 'unchanged' (Miller and Thode 2007), and the highest predicted severity was given an inferred CBI of 3.0. We were then able to infer the CBI of all remaining predictions because the square-root transformation of dNBR linearized the relationship to CBI (figure S1). Consequently, we generated a map representing the inferred CBI for the western US under contemporary climate.

We then predicted fire severity for the mid-21st century (2040–2069) as projected by each GCM using the BRT model. We inferred CBI as previously described using the linear relationship between dNBR and CBI of the observed predictions to make the inferences. Note that the predictions for all hexels in the western US were 'clamped' to avoid predicting outside of the observed range of severity values; all predictions >3 and <0.1 were given values of 3.0 and 0.1, respectively. For each BRT prediction (one for each GCM), we then quantified the predicted change in fire severity by subtracting the inferred CBI of contemporary climate from the inferred CBI of mid-21st century climate. We summarized the results by generating maps of (1) contemporary fire severity, (2) predicted mid-21st century fire severity (averaged over 20 GCMs) and, (3) the average change (for all 20 GCMs) in fire severity (i.e., inferred CBI) between contemporary and mid-century time periods.

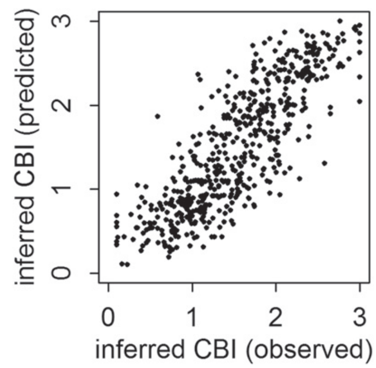

Figure 2. Plot showing predicted versus observed inferred CBI. Inferred CBI of observed data was calculated using similar methods to that of the inferred CBI of predicted data but were bounded by the 0.5 and 99.5 percentile dNBR to ensure extreme values did not overly influence this interpretation. $R^2 = 0.64$.

Results

The correlation between predicted and observed dNBR among the 544 hexels was 0.80 and the cross-validated correlation was 0.72. A plot showing predicted versus observed inferred CBI also indicates a good fit ($R^2 = 0.64$; figure 2). Water deficit and PPT were the most influential variables (relative influence = 41.5% and 29.8%, respectively) (figure 3(a)). Fire severity generally decreased with WD and increased with PPT (figures 3(b) and (c)). The map of predicted contemporary (1984–2012) fire severity indicates that cooler and wetter forested ecoregions (e.g., Pacific Northwest, Northern Rocky Mountains, and Southern Rocky Mountains) experience more high severity fire (inferred CBI $\geqslant 2.25$) compared to warmer and drier forested ecoregions (e.g., Arizona - New Mexico Mountains) (figure 4(a)). Non-forested ecoregions for the most part experience fairly low fire severity (inferred CBI < 1.25). The map of mid-21st century fire severity shows a similar pattern in that the cooler and/or wetter regions generally have higher severity than elsewhere (figure 4(b)), but for the most part, fire severity is predicted to decrease over much of the western US (figure 4(c)). The results of current, future, and predicted changes in fire severity are strikingly similar when we measured fire severity using a relativized metric (the relativized burn ratio; RBR) (Parks *et al* 2014a) instead of dNBR (figure S3).

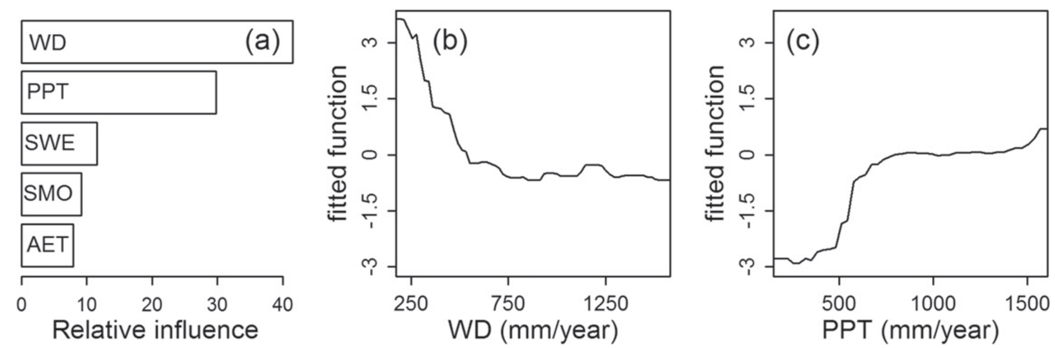

Figure 3. Variable importance in the BRT model (a) and partial dependence plots showing the relationship between dNBR and the two most influential variables (WD and PPT) (b), (c). Note that the partial dependence plots do not reflect interactions between variables and therefore simplify the relationships.

Discussion

Our models based on contemporary fire–climate relationships predict a widespread reduction in fire severity for large portions of the western US by the mid-21st century. Only a very small proportion of the western US is predicted to experience an increase in severity. Our prediction contrasts with those based on the direct influence of climate on fuel moisture and associated fire danger indices that occur at seasonal time scales (Fried *et al* 2004, Nitschke and Innes 2008). Our use of broad-scale climate as a proxy for vegetation composition and fuel load instead emphasizes the indirect influence that climate has on fire regimes (Miller and Urban 1999, Higuera *et al* 2014). Specifically, the predicted decrease in fire severity can be attributed to climatic conditions associated with higher WDs (figures 5(a) and (b)), lower productivity, and less burnable biomass (Zhao and Running 2010, Stegen *et al* 2011).

Our approach and findings are based on an implicit assumption that vegetation composition and fuel load will track changes in climate. Indeed, this is a common assumption that underlies numerous climate change studies, including those that use distribution models to project shifts in habitat ranges (Engler *et al* 2011) and fire activity (Krawchuk *et al* 2009, Moritz *et al* 2012). Specifically, our predictions of overall lower fire severity implicitly assume that vegetation composition and burnable biomass will reflect lower productivity associated with warmer and drier climates (e.g., increased WD; figure 5(b)). As such, our predictions are best interpreted as a *potential* reduction in fire severity, a potential that may not be realized where there is disequilibrium between climate and vegetation. Disequilibrium dynamics are the result of many factors and signals that directional changes in climate may not result in immediate changes in vegetation composition and fuel load (Sprugel 1991, Svenning and Sandel 2013). For example, leading-edge disequilibrium can arise when species are dispersal limited or don't reach reproductive maturity for many years (Svenning and Sandel 2013). Trailing-edge disequilibrium can arise because some species are long-lived and have deep roots, thereby facilitating survival and persistence under substantial inter-annual and decadal fluctuations in climate even though seedlings of the same species are unable to survive (Grubb 1977, Jackson *et al* 2009). To compound this, human-induced disequilibrium has also substantially affected most ecosystems in the western US (and globally) (Parks *et al* 2015b), in that natural disturbances such as fire have been excluded by factors such as livestock grazing, fire suppression, and landscape fragmentation (Marlon *et al* 2008). Both climate- and human-induced disequilibrium underlie present-day concerns about restoration of fire-adapted ecosystems after a century of fire exclusion (Stephens *et al* 2013, Hessburg *et al* 2015).

Consequently, our predictions are more likely to hold up in the presence of an active disturbance regime that catalyzes climatically driven changes in vegetation composition and fuel load (Flannigan *et al* 2000, Turner 2010). Disturbance catalysts are critical components for maintaining a dynamic equilibrium between vegetation and climate and appear to already be occurring with increasing frequency in some regions. For example, many studies have concluded that fire activity has increased in recent years (Westerling *et al* 2006, Kelly *et al* 2013) and widespread tree mortality has been attributed to drought and insect outbreaks (Allen *et al* 2010, Bentz *et al* 2010). In areas recently affected by these disturbances, the post-fire species and vegetation densities may be more tailored to the emerging climate (Overpeck *et al* 1990, Millar *et al* 2007). Although generally considered undesirable, disturbance-facilitated conversions from forest to non-forest vegetation are likely to occur in some situations (Stephens *et al* 2013, Coop *et al* in press), especially when compounded by human-induced disequilibrium.

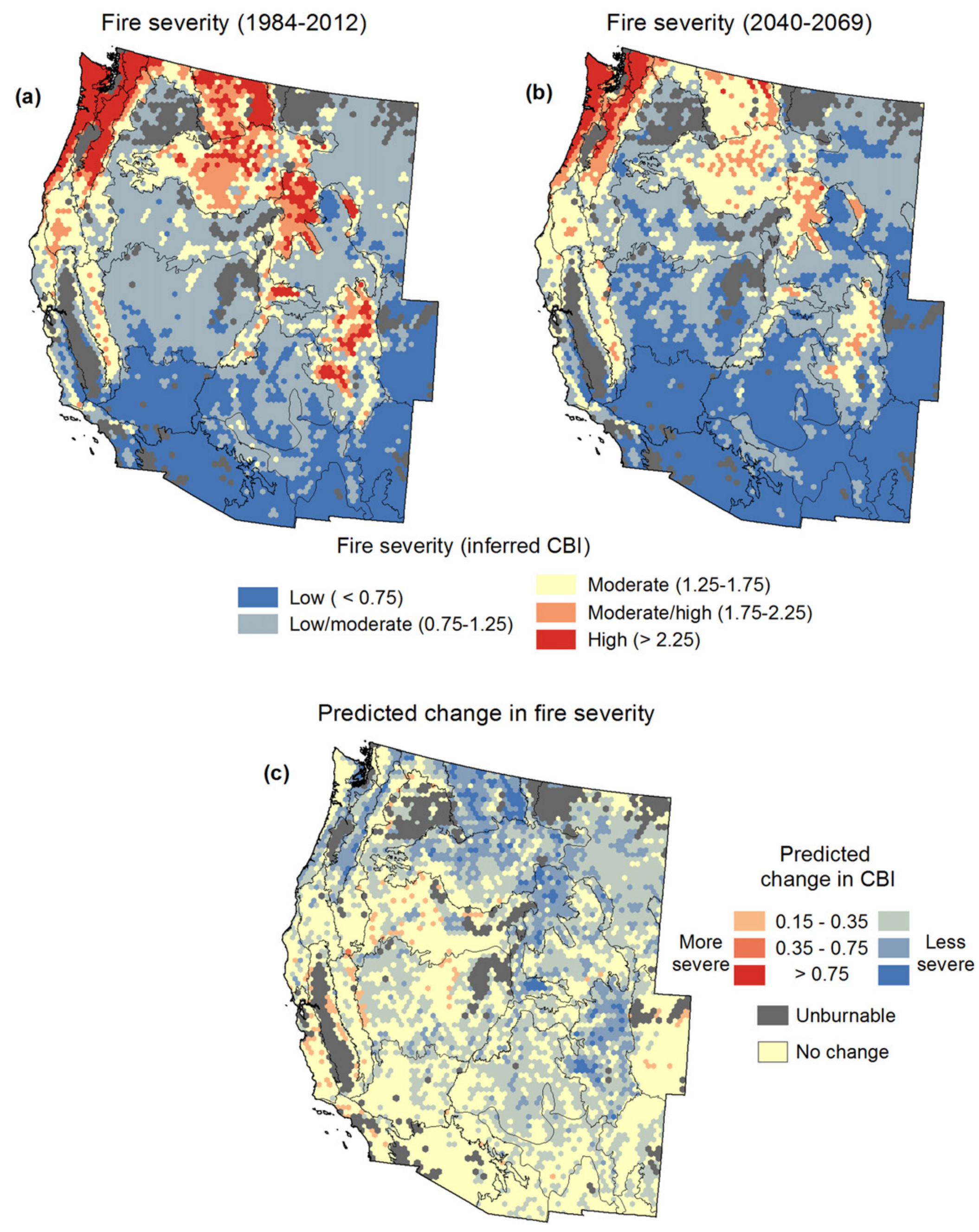

Figure 4. Predicted fire severity under observed (a) and mid-century climate (b). Mean change in fire severity among the 20 predictions (one prediction for each GCM) (c).

Most forested regions in the western US are currently experiencing a 'fire deficit' (Marlon *et al* 2012, Parks *et al* 2015b) because human activities and infrastructure (e.g., fire suppression and roads) exclude fire as an important disturbance agent. Consequently, human-induced disequilibrium between vegetation and climate, coupled with a changing climate, has important implications for future fire severity. We posit that such amplified disequilibrium will likely result in *increased* fire severity in future decades as fuel loads increase, fire seasons lengthen, and fire danger becomes more extreme (Collins 2014, Jolly *et al* 2015). This supposition is consistent with the findings of other studies that found a climate-induced increase in fire severity when assuming static vegetation (Fried *et al* 2004, Nitschke and Innes 2008). Continuing to resist catalysts of vegetation change only increases the probability of undesirable effects given that fire is inevitable (North *et al* 2009, Calkin *et al* 2015). An alternative to this unsustainable cycle is to actively facilitate transition of ecosystems to conditions that are more suited to the future climate by means of managed wildland fire or other restoration treatments (Millar *et al* 2007).

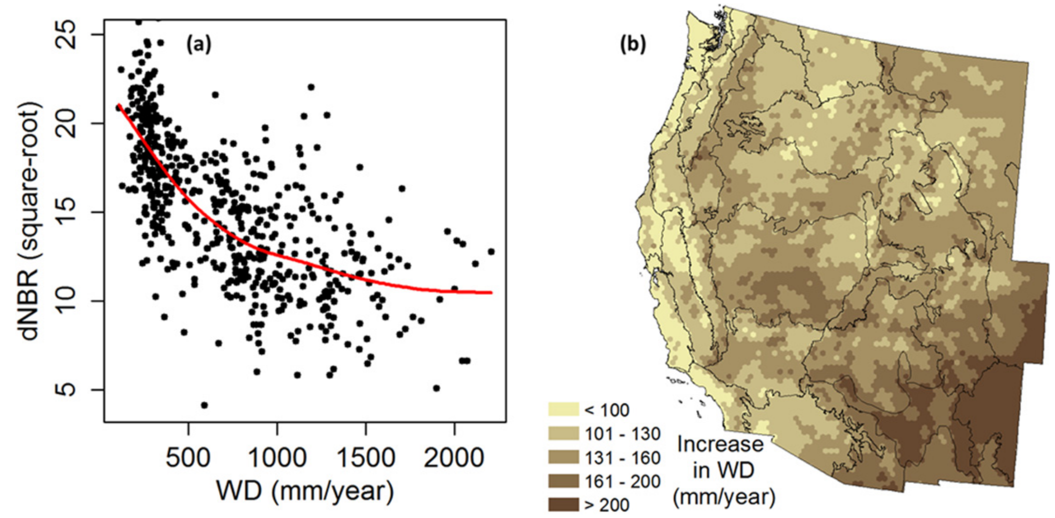

Figure 5. Plot of observed fire severity as a function of observed (1984–2012) water deficit (WD; the most influential variable in the BRT model) for the 544 hexels used to build the model (a). The red line shows the model fit according to a generalized additive model. Map of predicted increase in WD from contemporary to mid-century (2040–2069) climatic conditions (b); values depict the multi-model average change between time periods. According to this simple relationship, increased WD due to climate change will result in decreased fire severity. Note that the relationship flattens out which suggests a weaker response in dry ecosystems.

Our study complements and expands our understanding of controls on fire regimes and how they may respond to a changing climate in the western US. Specifically, predicted increases in fire activity (Littell *et al* 2010, Moritz *et al* 2012) imply that less biomass will be able to accumulate between successive fires, resulting in less biomass available for combustion and a reduction in fire severity. Furthermore, predicted increases in WD (figure 5(b)) are expected to increase water stress and decrease productivity in the generally water-limited western US (Chen *et al* 2010, Williams *et al* 2013), ultimately reducing the amount of biomass available to burn and resultant fire severity. It should be noted, however, that temperature-limited ecosystems (i.e., alpine environments) will likely experience an increase in productivity (and fire severity) under a warmer climate (Grimm *et al* 2013, Goulden and Bales 2014).

Our study relied on observed and predicted climatic normals (i.e., multi-decadal averages) to predict potential changes in fire severity. This is in contrast to other climate change fire studies that used annually or seasonally resolved climate (observed and GCM projections) and fire data to make predictions of potential changes in *fire activity* (i.e., fire frequency or area burned) (Littell *et al* 2010, Stavros *et al* 2014). The latter approach is often used because of the noted importance of climatic extremes on fire regimes (e.g., Westerling *et al* 2006). Although we could have built our model of fire severity using annually resolved data, we posit, for the purpose of predicting future fire severity, using long term averages (e.g., 1984–2012) is more appropriate for at least three reasons. First, although several studies have shown that fire severity responds to annual, seasonal, or daily variability in climate or weather, the relative influence of this variability can be fairly weak (Dillon *et al* 2011, Birch *et al* 2015). This is in contrast to broad temporal scales where the relationship between fire severity and climate has been found to be much stronger (Parks *et al* 2014b, Kane *et al* 2015). Second, because models built at a fine temporal resolution are more focused on the direct influence of climatic variability on fire weather and fuel moisture, they generally fail to incorporate climate- or fire-induced changes in vegetation composition or fuel load (Allen *et al* 2010, Parks *et al* 2015a). We suggest that predictions based on climatic normals implicitly incorporate such changes (Kelly and Goulden 2008, Marlon *et al* 2009). Lastly, GCMs may not adequately simulate annual climatic variability and thus are better suited for predicting long term trends (Stoner *et al* 2009).

Our model used broad scale data and the predictions of widespread reduced fire severity under future climate should be interpreted accordingly. For example, fire severity and climate vary at scales finer than the spatial resolution of the hexel used in this study (Schoennagel *et al* 2004). As such, our analysis does not likely capture finer-scale changes in fire severity that could occur. For example, in alpine environments where localized upward shifts in treeline under a warmer climate are expected to contribute to increases in biomass (Higuera *et al* 2014), fire severity might be expected to increase. Although our model of fire severity (dNBR) as a function of climate performed reasonably well (see section Results), we acknowledge that further error may be introduced due to error in the relationship between CBI and dNBR. However, we posit that the improved ecological interpretation attained by converting dNBR to CBI outweighs any increased error in our predictions.

Our measure of fire severity relied on dNBR (a unitless ratio) and CBI (a composite rating) and, consequently, there is no definable unit of measurement (e.g., grams of carbon consumed m^{-2}). Instead we infer changes in CBI, which integrates several strata (e.g., soil and shrubs) and scales severity from 0 to 3. This is admittedly a somewhat vague framework for assessing potential changes in fire severity, but takes advantage of the widespread availability of satellite-inferred metrics of fire severity and their documented correlation to the CBI. We suggest future research efforts involving fire severity and climate change aim to use more definitive and quantitative units of measurement. On a similar note, fire severity has ecological significance beyond what can be inferred from dNBR and is the result of many complex physical, biological, and ecological factors (Morgan *et al* 2014). For example, in ecosystems that are ill-adapted to fire (e.g., the Mojave Desert), dNBR values may be irrelevant, as any and all fires might be considered 'severe' (Brooks and Matchett 2006). Accordingly, although we used dNBR and CBI as a convenient and standardized way to assess fire severity, predictions for some ecoregions should be carefully interpreted.

Our model does not consider plant physiological responses to a CO_2 enriched atmosphere (e.g., improved water use efficiency and plant productivity) that could lead to increases in fire severity (Drake *et al* 1997, Keenan *et al* 2013). Given that today's atmospheric CO_2 concentration is the highest it's been for at least 650 000 years (Siegenthaler *et al* 2005), this could be a particularly important consideration for extreme water limited ecosystems such as grasslands, where woody plant encroachment could cause changes in biomass amount and structure (Morgan *et al* 2007, Norby and Zak 2011). Consequently, other research approaches using tools such as dynamic global vegetation models may predict different outcomes (Thonicke *et al* 2001).

Although we relied on data from protected areas and other areas of low human influence and thus underrepresented certain climatic environments (see Batllori *et al* 2014), these data represent a surprisingly broad range of ecosystem types in the western US ranging from warm desert (Death Valley National Park (NP) to dry conifer forest (Gila Wilderness) to cold forest (Yellowstone NP) (figure S2). As such, we suggest that under-represented climates have only a marginal effect on our results (see figure S2). Indeed, our analysis (figure S2) indicates that the data we used to build the model adequately represents the climates of most of the western US with the most notable exception being those in the Pacific Northwest where fires were historically and are currently infrequent (Agee 1993).

Conclusions

Our study predicts an overall decrease in fire severity for much of the western US by mid-century (2040–2069) due to changing climatic conditions. These predictions are best interpreted as *potential* decreases in severity that may not be realized unless vegetation composition and fuel load change in parallel with climate. Disequilibrium between plant communities and climate will only escalate, particularly in forested areas, unless natural disturbances and management activities (i.e., prescribed fire and restoration treatments) act as catalysts of vegetation change and push plant communities towards a state of equilibrium with climate. A high degree of disequilibrium between plant communities and climate is generally considered undesirable because the result may be an uncharacteristically severe wildland fire that causes abrupt ecosystem state shifts from, for example, forest to non-forest vegetation (e.g., Coop *et al* 2016).

Our findings support a passive management approach to ecosystem restoration (Arno *et al* 2000), whereby natural disturbance regimes are used to facilitate the transition of plant communities towards a state of equilibrium with the emerging climate. Active restoration treatments may also aid in facilitating these changes in certain situations (Millar *et al* 2007, Stephens *et al* 2010), but the current pace and scale of such treatments is insufficient to make a meaningful impact across the vast forested regions of the western US (North *et al* 2012). In addition, legal (e.g., designated wilderness) and logistical constraints (e.g., steep slopes) make certain activities (mechanical thinning) infeasible across a large proportion of land in the western US (North *et al* 2014). Achieving landscape resilience in a changing climate will likely require increased use of managed wildland fire, especially when weather conditions are not extreme (North *et al* 2015), and in fact, resisting change via activities such as aggressive fire suppression may be counter-productive in the long-run (Calkin *et al* 2015). As such, the results of this study provide insights to policy makers and land managers in the western US as to the pros and cons of resisting or facilitating change in vegetation composition and fuel load in the context of a changing climate.

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California Spotted Owl, Songbird, and Small Mammal Responses to Landscape Fuel Treatments

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A principal challenge of federal forest management has been maintaining and improving habitat for sensitive species in forests adapted to frequent, low- to moderate-intensity fire regimes that have become increasingly vulnerable to uncharacteristically severe wildfires. To enhance forest resilience, a coordinated landscape fuel network was installed in the northern Sierra Nevada, which reduced the potential for hazardous fire, despite constraints for wildlife protection that limited the extent and intensity of treatments. Small mammal and songbird communities were largely unaffected by this landscape strategy, but the number of California spotted owl territories declined. The effects on owls could have been mitigated by increasing the spatial heterogeneity of fuel treatments and by using more prescribed fire or managed wildfire to better mimic historic vegetation patterns and processes. More landscape-scale experimentation with strategies that conserve key wildlife species while also improving forest resiliency is needed, especially in response to continued warming climates.

Keywords: adaptive management, mixed conifer, restoration, Sierra Nevada, wildlife conservation

The role of wildfire in many of the world's forests that are adapted to frequent, low- to moderate-intensity fire regimes has been altered through fire exclusion, timber harvesting, livestock grazing, and urbanization (Agee and Skinner 2005, Collins et al. 2010). In the western United States, these land-use practices have affected forest structure and species composition, increasing surface fuel loads, tree density, the dominance of shade-tolerant tree species, and forest homogeneity (Hessberg et al. 2005, North et al. 2009, Chiono et al. 2012). As a consequence, many forests in the western United States are experiencing higher-severity burns—in some cases, producing large patches of tree mortality that can severely hinder the reestablishment of conifer forests (Roccaforte et al. 2012, Collins and Roller 2013). Consequently, one of the primary focuses of contemporary forest management is the treatment of fuels and vegetation to reduce fire hazards, especially as climate continues to warm (Stephens et al. 2013).

There is increased recognition that forests adapted to low- to moderate-intensity fire regimes experienced some high-severity fire (Perry et al. 2011, Marlon et al. 2012). Patchy, high-severity fire provides opportunities for early-seral habitat development and the production of large pieces of deadwood resources that are important to many wildlife species (Fontaine and Kennedy 2012). As such, forest fuel treatments should not be used to eliminate all high-severity fire. Rather, treatments should allow for patterns of fire effects that approximate those occurring under more natural forest conditions. What little information we have on fire patterns under these conditions suggests that high-severity fire constitutes fairly low proportions of the overall burned area (5%–15%) in these forest types, which is generally aggregated in relatively small patches (smaller than 4 hectares [ha]), as is the case in the upper mixed-conifer forests in Yosemite National Park (Collins and Stephens 2010, Mallek et al. 2013).

Forest management involving habitat used by wildlife species at risk has been one of the principal challenges to US federal land managers for the last 25 years. In the Sierra Nevada, an ongoing debate is focused on several species that use old-growth forest, including the California spotted owl (CSO; *Strix occidentalis occidentalis*) and the Pacific fisher (*Martes pennanti pacifica*). Forest managers need information on appropriate levels of forest manipulations to create the desired balance between habitat conservation for wildlife populations and modifications of forests to improve their resilience to large high-severity fires that could prove more expensive and detrimental than the short-term effects of restoration treatments.

Fuel-reduction treatments reduce the potential impacts of wildfire by reducing the only aspect of the fire behavior

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Figure 1. Fuel treatments implemented in the Meadow Valley project area. (a) Pretreatment mixed-conifer forest. (b) Whole-tree harvester cutting small trees (thinning from below). (c) Small trees, tree tops, and limbs being chipped and shipped by truck to a bioenergy plant to produce electricity. (c) Posttreatment defensible fuel profile zone, taken from the same perspective as in panel (a). Photographs: Keith Perchemlides.

triangle (i.e., topography, weather, fuel) that can be modified by managers: the quantity and continuity of fuel. A number of techniques are employed to reduce fire hazards, and each technique has associated effects on forest structure (Agee and Skinner 2005). Mechanical treatments can reduce stand density, basal area, and ladder and canopy fuel. To reduce accumulated surface fuel and to offset the detritus added from harvest operations, prescribed fire is sometimes used following forest thinning to reduce fire hazards, but whole-tree harvesting (i.e., complete tree removal, with the materials chipped and trucked to a processing facility; figure 1) can also effectively keep much of the harvest detritus from being added to the forest floor. Broadcast burning alone is very effective in elevating canopy base height and in reducing surface fuel (Agee and Skinner 2005).

Recent research confirms the ability of fuel treatments to alter potential fire behavior (Fulé et al. 2012) and actual wildfire effects (Safford et al. 2012). Research has also determined that fuel-reduction treatments achieve their objectives with generally positive or neutral ecological effects (Stephens et al. 2012); however, almost all research on the effects of fuel treatments has been performed at the stand scale (10–25 ha). Given the large home ranges of many key wildlife species commonly at the crux of forest management issues in the western United States (e.g., the CSO, the northern spotted owl [*Strix occidentalis caurina*], the Pacific fisher), it is important to understand fuel-treatment impacts at larger spatial scales. This is particularly relevant because many fuel-treatment projects are being proposed—and, in a few instances, implemented—at landscape scales (15,000–40,000 ha; Ager et al. 2007, Collins et al. 2010).

Fuel treatments directly alter wildlife habitat by removing both aerial (trees) and ground (coarse wood, shrubs) cover. These altered conditions can affect both habitat suitability, which influences the number of individuals that an area can support, and habitat quality, which directly affects the fitness

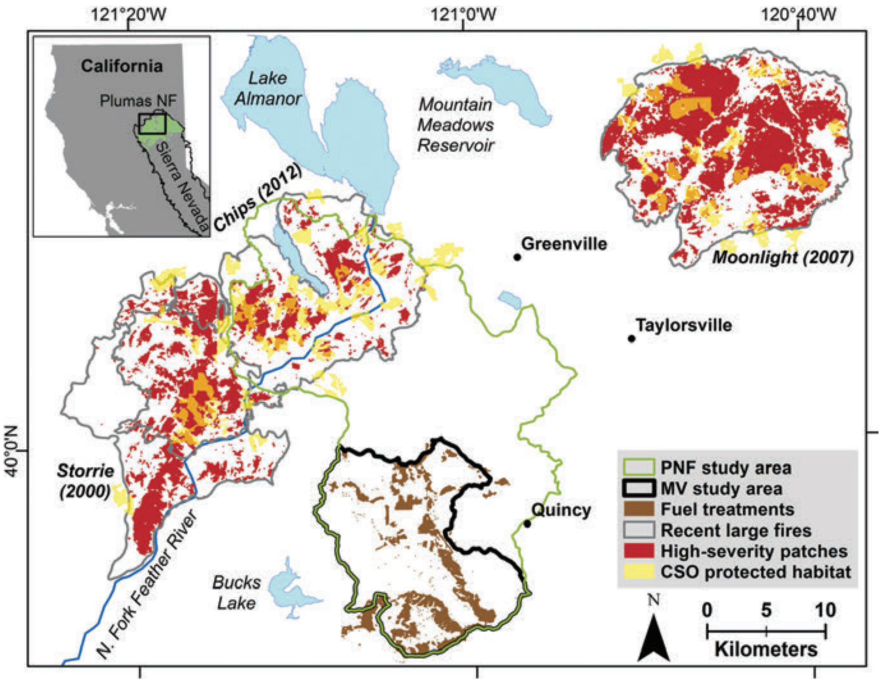

Figure 2. Meadow Valley study area with completed landscape fuel-treatment network. Recent large wildfires and the resulting patches of high-severity fire effects are also indicated. Three wildfires are shown: Storrie (2000), Moonlight (2007), and Chips (2012). These were selected on the basis of the following criteria: proximity to the study area (closer than 25 kilometers), vegetation type (conifer dominated), size (larger than 10,000 hectares), and age (since 2000). Abbreviations: CSO, California spotted owl; MV, Meadow Valley; N, north; NF, national forest; PNF, Plumas National Forest; W, west.

and productivity of individuals. Because more-suitable habitat for certain at-risk wildlife species is associated with greater aerial and ground cover, the effects of fuel treatments are generally perceived as negative. However, large patches of wildfire-caused tree mortality can also negatively affect both habitat suitability and quality (Tempel et al. in press). To the extent that fuel treatments reduce the potential for large patches of tree mortality in wildfire, there may also be an indirect benefit of fuel treatments to certain species' habitat. Finding a balance between these influences is a crucial management need.

Over the past decade, we have studied the ecological effects of one of the few completed landscape-level fuel-treatment networks in western US forests. Here, we distill the results of these efforts. We quantify change in vegetation structure and modeled fire behavior as a result of fuels treatments and assess treatment effects on the CSO, songbirds, and small mammals. Modeling studies have been published in which the trade-offs in these systems have been conceptually examined (Lee DC and Irwin 2005), but this is one of the first studies in which these questions have been empirically examined at landscape scales.

Study area and design

Our study area is located in the Meadow Valley area of the Plumas National Forest, situated in the northern Sierra Nevada, at 39 degrees (°) 56 minutes (′) north, 121°3′ west (figure 2). The climate is Mediterranean, with warm, dry summers and cool, wet winters, which is when most precipitation (1050 millimeters per year; Ansley and Battles 1998) occurs. The core study area is 19,236 ha, with elevations ranging from 850–2100 meters (m). The vegetation is primarily mixed-conifer forest, consisting of white fir (*Abies concolor*), Douglas-fir (*Pseudotsuga menziesii*), sugar pine (*Pinus lambertiana*), ponderosa pine (*Pinus ponderosa*), Jeffrey pine (*Pinus jeffreyi*), incense-cedar (*Calocedrus decurrens*), California black oak (*Quercus kelloggii*), and other less common hardwood species. White fir is the most abundant tree, although large (e.g., larger than 1 m in diameter) stumps of pines encountered frequently in the forest attest to a change in composition and structure in recent history. Red fir (*Abies magnifica*) is common at higher elevations, where it mixes with white fir. In addition, a number of species are found occasionally in or on the edge of the mixed-conifer forest, including western white pine (*Pinus monticola*) at higher elevations, lodgepole pine (*Pinus contorta* var. *murrayana*) in cold air pockets, and western juniper (*Juniperus occidentalis*) on xeric sites. California hazelnut (*Corylus cornuta*), dogwood (*Cornus* spp.), and willow (*Salix* spp.) are found in moister riparian areas. Montane chaparral and some meadows are interspersed in the landscape. Tree density varies as a result of recent fire- and timber-management history, elevation, slope, aspect, and edaphic conditions. Historical fire occurrence, which can be inferred from fire scars recorded in tree rings, suggests that the fire regime was predominantly frequent, low- to moderate-severity fires, at intervals ranging from 7–19 years, with the last widespread fires occurring 85–125 years ago (Moody et al. 2006).

Fire activity in the last 15–20 years has been notably higher in the northern Sierra Nevada than in the rest of the range (Collins 2014). Since 2000, there have been three megafires (covering more than 10,000 ha; Stephens et al. 2014) within 25 kilometers (km) of our study area, burning a total of 73,000 ha (figure 2). These fires burned predominantly in mixed-conifer forests, encompassing approximately 60 CSO protected activity centers (figure 2). Cumulatively, 34% of the area burned in these three fires suffered high-severity fire (more than 95% dominant tree mortality; figure 3a; Miller et al. 2009). More important than the total proportion of area severely burned is the distribution of high-severity patches over the burned area, because this can limit tree seed

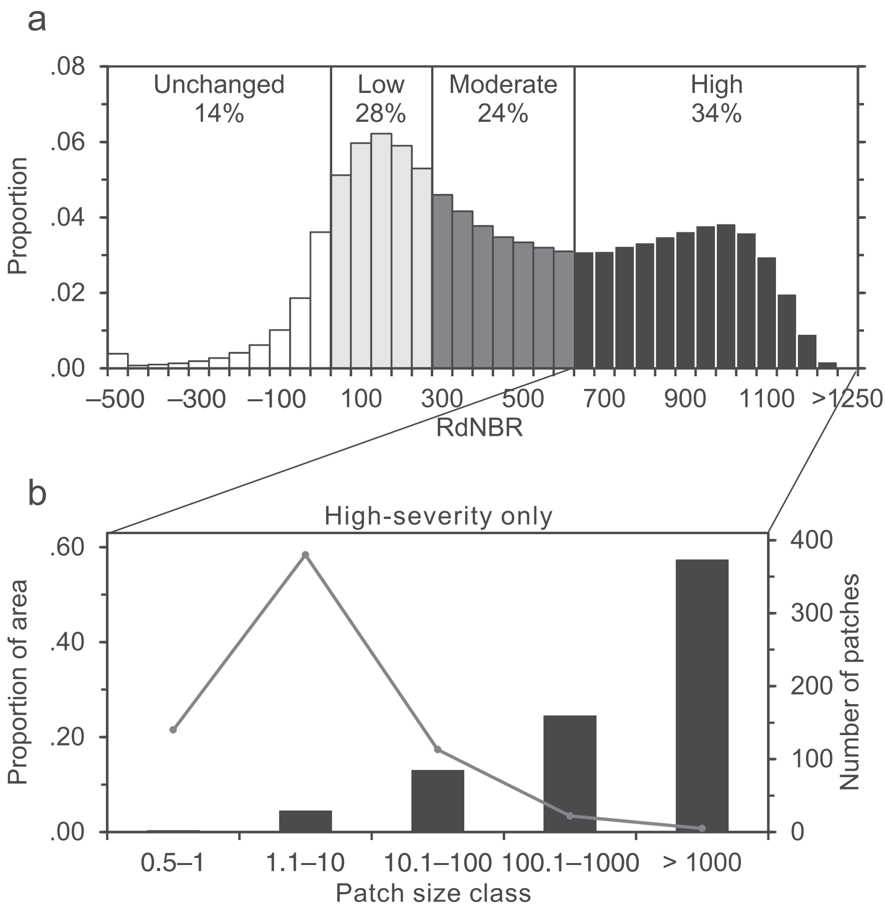

Figure 3. (a) Fire severity distribution for the three recent large fires in the Meadow Valley study area (see figure 2). The fire-severity estimates are based on the relative differenced normalized burn ratio (RdNBR; Miller and Thode 2007). (b) The proportion of total high-severity area (bars) and the number of patches (line) as a function of patch size class.

dispersal from wind and animals (Perry et al. 2011, Collins and Roller 2013). Large patches (defined here as larger than 1000 ha) accounted for a disproportionate amount of the total high-severity-fire area in the recent wildfires near the study area (figure 3b).

The projects that contributed to the fuel-treatment network are part of the larger Herger-Feinstein Quincy Library Group Pilot Project (USHR 1998). This project was directed by the US Congress to involve local communities in forest management. The project objectives included improving forest health, reducing uncharacteristic high-severity fire, conserving wildlife habitats, and stabilizing economic conditions in local communities. The projects in Meadow Valley encompassed a range of treatment types and intensities reflecting changes in regional management directions and differing land-management constraints across a complex landscape (Collins et al. 2010, Moghaddas et al. 2010). The primary fuel treatment used in Meadow Valley was defensible fuel profile zones (DFPZs), which are areas approximately 0.4–0.8 km wide in which surface, ladder, and crown fuel loads are reduced with a combination of moderate thinning from below (Moghaddas et al. 2010) and prescribed fire treatments (figure 1).

The DFPZs were excluded from portions of the landscape set aside as reserves and from designated CSO protected activity centers, which are 121-ha areas of high-suitability nesting habitat designated by forest biologists. In addition, the project predominantly excluded all riparian habitat conservation areas or stream buffers intended to protect riparian and aquatic resources (figure 4). The activities conducted in the DFPZs were chainsaw thinning and pile burning of trees up to 30 centimeters (cm) in diameter at breast height (dbh); mastication: primarily shrubs and small trees were shredded and chipped in place, with the material left on site; prescription burning: stands were burned under conditions of moderate relative humidity and fuel moisture; and a combination of mechanical thinning and prescription burning of trees up to 51 or 76 cm dbh, depending on whether the stands were in the wildland–urban interface, using a whole-tree harvest system (figure 1) to achieve a residual canopy cover of approximately 40%, and some were underburned (Moghaddas et al. 2010). In addition to the DFPZs, group-selection treatments were implemented as part of the project. The group-selection treatments included the removal of all conifers up to 76 cm dbh within an area of 0.8 ha, followed by residue piling and burning, then either natural regeneration or replanting to a density of 270 trees per ha with a mix of sugar pine, ponderosa pine, and Douglas-fir. These treatments collectively covered 3688 ha (3448 ha in the DFPZs, 240 ha in the group-selection treatment), or 19% of our study area, and were implemented between 2003 and 2008.

Forest structure and microclimate

Although they are designed to reduce fire hazards, forest treatments alter stand conditions directly by reducing tree density and canopy cover, and indirectly by altering microclimate conditions affecting the understory community. To assess these changes we measured stand structure, light, understory plant cover, micro-meteorological variables, soil moisture, and fuel moisture in replicated control, thinning, and group-selection treatments plots embedded within the landscape-level treatments (see Bigelow al. 2009, 2011, Bigelow and North 2012 for detailed methods).

The mean forest canopy cover was 69% (standard deviation [SD] = 7%) before treatment; after treatment it was 53%

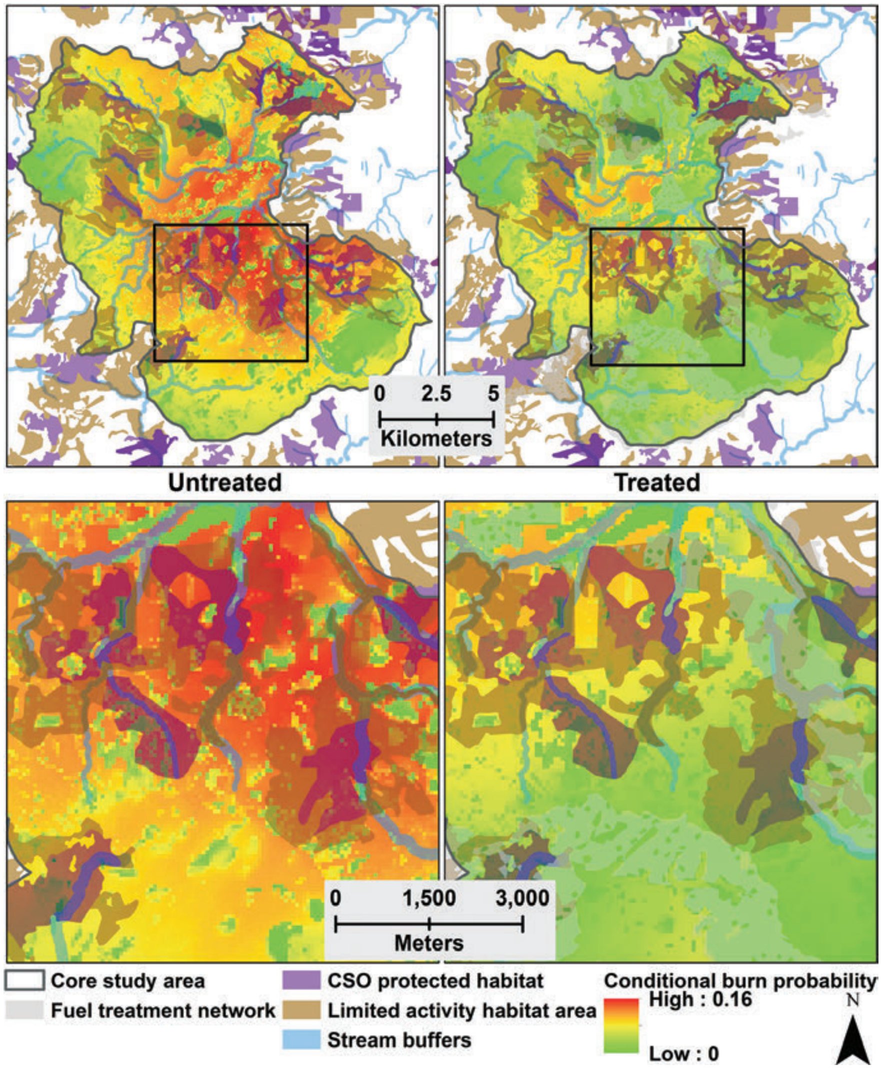

Figure 4. Hazardous fire potential across the Meadow Valley study area for the untreated and treated landscape conditions. This fire potential is based on the conditional burn probability of fire occurring with flame lengths greater than 2 meters, which is consistent with tree torching (see Collins et al. 2013 for specific details). Land designations that often limit or exclude active forest management (e.g., California spotted owl [CSO] protected habitat, stream buffers) are also shown to illustrate off-site effects of the landscape fuel-treatment network. The black square in the upper panels indicates the focal area shown in the bottom panels.

(SD = 7%) in thinned stands and 12% (SD = 6%) in the group-selection openings (Bigelow et al. 2011). These differences were reflected in growing-season understory light, which averaged 17% of full sun before treatment and increased to 26% in thinned stands and 67% in group-selection openings. Models of regenerating tree growth and light availability demonstrated that the height growth rates of shade-intolerant yellow pines (ponderosa and Jeffrey pines) and shade-tolerant white fir were equal at 41% of full sun. Light levels greater than this correlated exponentially with the height growth of the pines. The group-selection treatments provided ample light to recruit shade-intolerant species to the canopy, but only 8% of the sample locations in the thinning treatments had light levels exceeding the 41% crossover point, which suggests that these treatments would not substantially contribute to pine restoration across the landscape. An analysis of hemispherical photographs showed that the treatments decreased canopy closure following thinning. At the plot (1-ha) scale 3 years after treatment, cover of understory plant life-forms only changed under group selection ($p < .05$). Shade-tolerant conifers decreased, and graminoids, forbs, and broad-leaved trees (mainly California black oak and dogwood) increased (figure 5). There was no increase in exotic plant species cover with any of the treatments (Chiono 2012).

Changes in abiotic conditions followed differences in canopy cover for only some of the variables measured (Bigelow and North 2012). Soil moisture increased and duff moisture decreased in the group-selection treatments relative to the thinned and pretreatment conditions. Wind gust speeds (measured 2.5 m above ground) averaged 31% higher in the thinned stands than in the controls, but this was far less than the 128% increase in the group-selection openings. However, there was no difference in air temperature or relative humidity among the treatments, possibly because the increase in understory wind increased air mixing and eliminated any gradients in air temperature and humidity that might have resulted from increased irradiance.

Treatment increased within-stand variability for some vegetation and microclimate conditions but, in general, did not create the landscape-level heterogeneity characteristic of historic forest conditions in the Sierra Nevada (North et al. 2009). Mixed-conifer forests support the highest vertebrate diversity of California forests (Verner and Boss 1980), and studies suggest that this may result from habitat variability associated with the observed range of tree species diversity, canopy cover, microclimate, and deadwood conditions (Rambo and North 2009, Ma et al. 2010, White et al. 2013). This historic forest heterogeneity appears to reflect differences in fire intensity and site productivity associated with local and large-scale changes in slope, aspect, soil, and slope position (North et al. 2009, Lydersen and North 2012). On average, more mesic sites (e.g., drainage bottoms and north-facing slopes) historically supported greater stem density, canopy cover, and tree basal area, whereas drier and

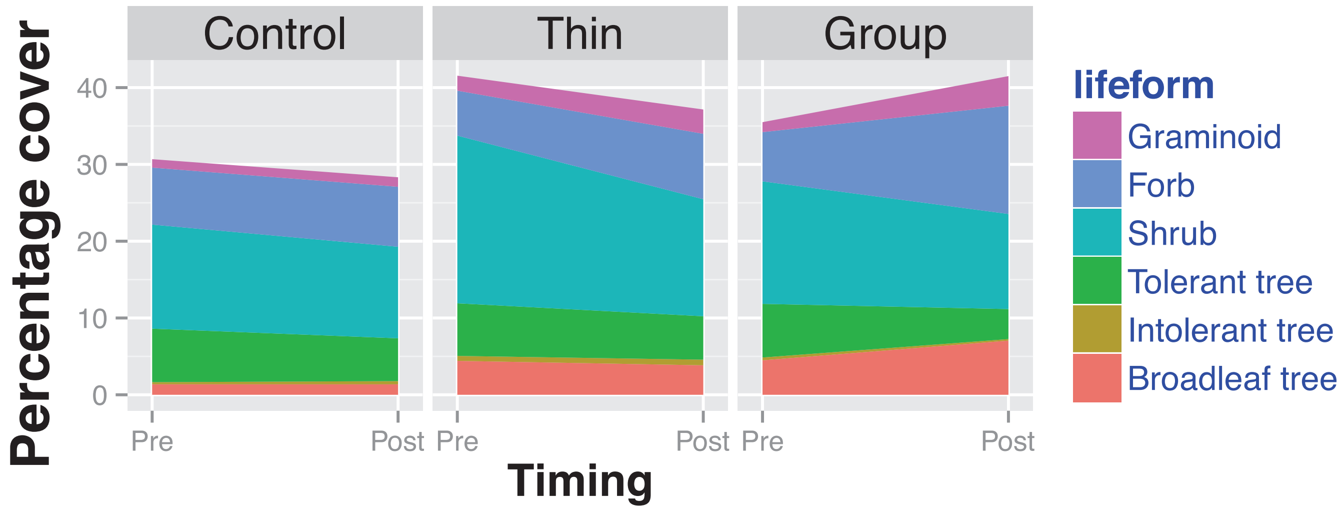

Figure 5. The percentage cover of plant life forms before (pre) *and 3 years after* (post) *fuel-reduction thinning and group-selection treatments (*n *= 300 subplots per treatment) that were implemented in 2007 in Meadow Valley. Changes in understory cover in thinned stands were not significant* ($p > .16$). *Graminoids, forbs, and broadleaf trees increased and shade-tolerant conifers decreased* ($p < .05$) *in group selection openings.*

steeper areas burned more frequently and intensely, creating more-open, pine-dominated forests (North et al. 2009). Although the Meadow Valley treatments did increase within-stand heterogeneity, they were not explicitly designed to vary with site topography or local productivity to produce this historic landscape variability.

Potential fire behavior

We employed a spatially explicit fire behavior model (Finney et al. 2007) to simulate fire spread across the Meadow Valley area. We simulated 10,000 individual fire events, with random ignition locations, and compared patterns of burn probability based on the number of times a particular area burned with the given ignition locations and simulated flame lengths for the study area prior to and following the implementation of landscape fuel treatments. Each fire event simulated burning for 240 minutes (one 4-hour burn period) under 97th percentile fuel moisture and wind conditions. These are the conditions associated with large-fire growth in this region (Collins et al. 2013). The burn period duration was selected such that the simulated fire sizes (for one burn period) approximated large-spread events observed (daily) in nearby recent wildfires (Collins et al. 2013). One of the primary assumptions with this approach is that, during these large-spread events (burn periods), fire suppression operations have limited impact, which is consistent with observed large-fire occurrence throughout the western United States (Finney et al. 2007). We summarized the burn probabilities across the Meadow Valley area into land allocations determined by the US Forest Service (USFS; Moghaddas et al. 2010).

The simulated fire behavior indicated that the landscape-scale network of DFPZs and prior fuel treatments were effective at reducing conditional burn probabilities across all land-allocation types, except the small area of off-base lands (figure 4; Moghaddas et al. 2010). Because burn probabilities are correlated directly and positively to fire size (Finney et al. 2007), it is clear that the pretreatment landscape was more conducive to large-fire growth than the posttreatment landscape was (Moghaddas et al. 2010, Collins et al. 2013). Although the influence of the treatments on the modeled burn probabilities of each land allocation varied, the untreated stands (e.g., those designated for protected CSO habitat, riparian and aquatic resources, and reserve lands) and the remaining private and unclassified lands all experienced reduced burn probabilities from the application of fuel treatments at the landscape scale (figure 4; Moghaddas et al. 2010). A similar reduced burn severity immediately adjacent to treated areas has been reported for actual fires across the western United States (Finney et al. 2005).

The substantial reduction in both the total area and the area burned at higher flame lengths under a posttreatment wildfire scenario was notable, given that only 19% of the study area had been treated (Moghaddas et al. 2010, Collins et al. 2013). Both the orientation of the treatments (approximately orthogonal to the predominant wind direction throughout the duration of the simulated fire), and the long, continuous shape of the DFPZs resulted in potential wildfires' intersecting fuel treatments in multiple places. In addition, the treatments were somewhat concentrated in the southwestern portion of the study area (figure 2), which is the dominant direction of strong winds during the fire season (Collins et al. 2013). In combination, these factors limited the ability of the simulated fire to both circumvent the treated areas and to regain spread and intensity after encountering the treatments. These results are important to managers, because similar installations of fuel and restoration treatments are needed in many Sierra Nevada

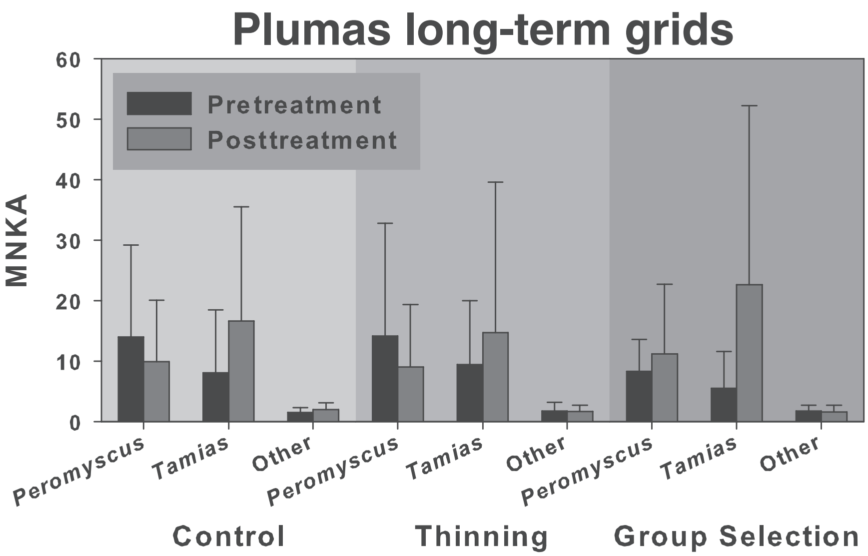

Figure 6. The mean minimum number of small animals known alive (MNKA), recorded before and after fuel treatments in the Plumas National Forest study area. For ease of presentation, we present three species groups (**Peromyscus boylii** ***and*** **Peromyscus maniculatus*****;*** **Tamias quadrimaculatus** ***and*** **Tamias senex*****; all other species; see Kelt et al. 2013 for details). The bars represent the means of the replicate sampling grids. The error bars represent the positive standard deviation.***

mixed-conifer forests, where the present treatment rates are very low (North et al. 2012).

Small mammals

The northern Sierra Nevada supports a diverse fauna of small mammals that play key ecological roles as consumers, seed and fungal dispersers, and prey for both terrestrial and aerial predators (Hallett et al. 2003, Kelt et al. 2013). We studied small mammals in the Meadow Valley study area and the greater Plumas National Forest study area (PNFSA; figure 2), with a particular focus on two species that are key prey of the CSO (Gutiérrez et al. 1995): the dusky-footed woodrat (*Neotoma fuscipes*) and the northern flying squirrel (*Glaucomys sabrinus*). Results on focal species efforts have been reported elsewhere (Innes et al. 2007, Smith et al. 2011), but one finding merits emphasis here. California black oak, the primary hardwood in mixed-conifer forests, is an important habitat element for both the woodrat and the flying squirrel. Woodrat density was positively correlated with black oak density (Innes et al. 2007), and both species strongly preferred black oaks for nest sites (Innes et al. 2008, Smith et al. 2011). California black oak may be important for other wildlife species as well (Zielinski et al. 2004), but its persistence in our study landscape is in doubt. California black oak is shade intolerant, and across our study area, there were few thriving seedlings and many mature trees in decline as adjacent conifers overtopped them. California black oak trees were present in only 133 of 602 plots placed randomly in the PNFSA and were in a codominant canopy position in less than 10% of the plots in which it was present (see supplement S1).

Our broader studies on the management needs of entire small mammal assemblages included two complementary efforts. We sampled small mammals annually for 8 years on replicate trapping grids in treated and untreated mixed-conifer forests dominated by white fir in order to evaluate the responses of the small mammal community to canopy thinning (Kelt et al. 2013). To determine whether the habitat associations of the mammals in these forests were similar to those of mammals in other forest types, we expanded our efforts to include stratified random sampling of the PNFSA that encompassed the Meadow Valley study area (figure 2).

Whereas canopy thinning in white-fir-dominated mixed-conifer forests caused some significant changes in forest structure, small mammal assemblages were similar before and after canopy thinning and group selection (Kelt et al. 2013), which suggests a minimal response in the short-term to these treatments (*contra* Suzuki and Hayes 2003, Gitzen et al. 2007, but see Carey and Wilson 2001). Although each treatment may have elicited somewhat different responses (figure 6), the variance across replicate plots eroded any such differences even in the face of the substantial variation in canopy cover. The lack of a short-term response may not be surprising in a system characterized by high interannual variation in weather and in a system dominated by generalist species; we look forward to resampling these sites after 10–15 years to assess potential longer-term responses. Because our manipulative experiment was focused on white-fir-dominated mixed-conifer forests, we pursued a more general assessment of mammalian responses to habitat and environmental variation across the entire PNFSA, capitalizing on a series of point-count transects established throughout the forest in a stratified (by forest type) random manner (see the "Songbirds" section below). We sampled eight randomly selected points on each of 74 transects to characterize how small mammals respond to broader variation in forest structure.

We assessed assemblage-wide responses to this variation with ordination (canonical correspondence and canonical correlation) and species-specific responses with multiple stepwise regression. All data were standardized (both rows and columns) by centering and normalizing, and the mammal data were log-transformed to prevent domination of the axes by common species. The results from all of the analyses were qualitatively identical to those of the Meadow Valley experimental grids, which indicates minimal responses of small mammal assemblages to variation in forest structure or composition. Although the spatial arrangement of the

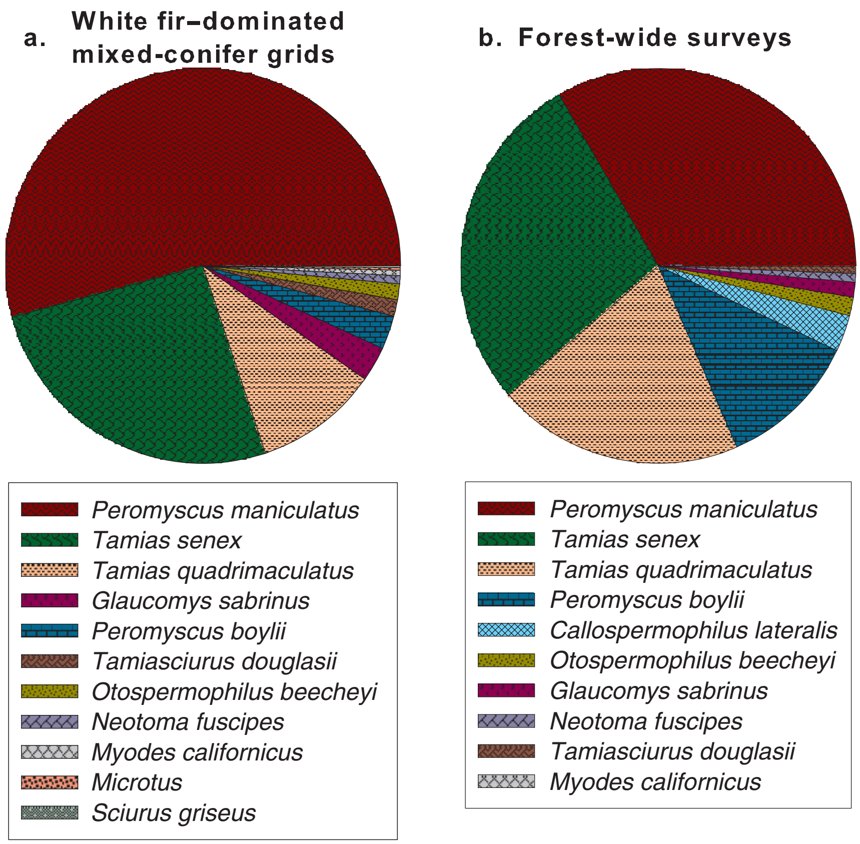

Figure 7. Small mammal composition at two spatial scales in the Plumas National Forest study area. At both scales, captures were dominated by three species. At the forest scale, only one other species was highly represented. All other species at both scales were only minor elements.

small mammal species in the ordination space was ecologically reasonable (e.g., woodrats and brush mice [*Peromyscus boylii*] associated with oaks, and chipmunks [*Tamias*] and Douglas squirrels [*Tamiasciurus douglasii*] associated with conifers and with a high basal area of trees and snags), ordination explained only a small proportion of variance in the distribution of small mammals. Similarly, regression failed to produce compelling associations for any species (or for community metrics such as species richness or diversity). The coefficients for both sets of analyses were universally low (Kelt et al. 2013).

In trapping efforts on the Meadow Valley experimental grids and in the larger PNFSA (figure 2), our captures were overwhelmingly dominated by 3–5 species (figure 7). Deer mice (*Peromyscus maniculatus*) dominated the captures at both spatial scales, comprising a full 55% of the captures on the Meadow Valley experimental grids and just over one-third of the captures in the PNFSA. Two species of chipmunk (*Tamias quadrimaculatus*, *Tamias senex*) represented an additional 40%–44%, and brush mice were an additional 8% in the PNFSA. Therefore, our samples were dominated by ecological generalists known to be tolerant of diverse habitats. What appears to be missing is a reasonable representation of species with more restricted niche requirements. Our sampling was not designed to sample shrews (*Sorex*), but California red-backed voles (*Myodes* [formerly *Clethrionomys*] *californicus*) may have been more common in this region in the 1940s and 1950s (Kelt et al. 2013) and should have been present in our study. This species forages on fungi, however, and requires large downed woody debris and a closed-canopy forest to allow sufficient moisture retention to promote fungal growth (Alexander and Verts 1992). In 177,216 trap nights of effort, we captured only 11 *Myodes* (all but one on Meadow Valley experimental grids). Other species that are mesic habitat specialists were not sampled (e.g., *Zapus trinotatus, Sorex palustris*).

It is not clear whether the taxonomically depauperate assemblage structure documented in our study represents a relatively recent reduction or is more historic for this region. No data on mammal assemblages exist prior to European settlement and the beginning of widespread changes to the Sierra Nevada forest ecosystems (Merchant 2012). However, one implication of this research is that, in spite of nearly a kilometer of vertical elevation relief and diverse forest types from ponderosa pine to red fir, the current forest conditions support a relatively homogeneous small mammal community dominated by ruderal species. It is unclear whether this reflects a legacy of fire exclusion and the resulting accumulation of fine woody debris or, perhaps, a response to a history of logging and fire suppression in this region. In contrast, other recent work in Yosemite (Roberts et al. 2008) confirms that small mammals respond strongly to variation in burn history. Taken together, these results support the fundamental ecological role of fire and broadscale forest heterogeneity in managing mixed-conifer forests in the Sierra Nevada (North et al. 2009).

Songbirds

To evaluate the effects of the Meadow Valley fuel-treatment network on songbirds, we compared avian community diversity before and after treatment. From 2004 to 2011, we surveyed the breeding community in and adjacent to Meadow Valley, using standardized point-count surveys with a 50-m radius (Ralph et al. 1995). Surveys were conducted at 51 stations where DFPZs were implemented (treated) and 201 stations where no treatments were implemented (untreated), proportional to the 19% of the study area treated. An additional 180 stations were surveyed in adjacent untreated PNFSA (figure 2) watersheds (the reference group). We used geographic information systems to establish locations

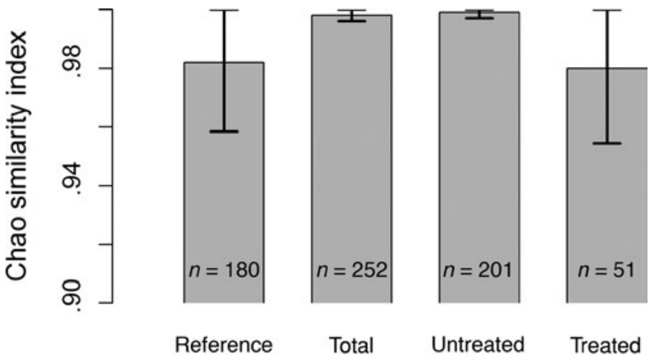

Figure 8. Chao similarity index for the avian community (60 species) before and after treatment at treated and untreated locations in the Meadow Valley study area and reference locations in the adjacent Plumas National Forest study area that also received no treatment. This metric ranges from 0–1, with 1 representing perfect similarity (all species and relative abundances shared among both samples). The error bars are 95% confidence intervals.

for the untreated and reference stations from a randomly selected origin (constrained by slopes lower than 35% and on USFS land) along a random compass bearing in a linear array of 4–12 points. The treated stations were placed within proposed DFPZ treatments across the breadth of treatment types and geography described above. All of the stations were a minimum of 250 m apart.

We surveyed all of the stations in both 2004 and 2005, prior to treatment, and for 2 years after all treatments were implemented (2010–2011). In each year, we surveyed every station twice during the peak of the breeding season (15 May–10 July), with a minimum of 10 days between visits. We limited our analyses to the 60 species breeding in upland habitats that were reliably recorded with point counts (Hutto et al. 1986). The results were summarized at the level of the three treatment groups described above (treated, untreated, reference) and for treated and untreated locations in Meadow Valley combined. For all of the analyses, we summed detections across four surveys (two visits per year over 2 years) for the pre- and posttreatment periods. We compared avian assemblages before and after the treatment with Chao–Jaccard's similarity index (Chao et al. 2005), calculated using EstimateS (version 9.1, University of Connecticut, Storrs). Chao–Jaccard similarity is sensitive to changes in species composition and abundance. Differences in avian diversity were evaluated using the exponent of the Shannon index (Nur et al. 1999). For both analyses, 95% confidence intervals were derived from estimated standard errors from 1000 bootstrap samples.

Our results indicate little change in the Meadow Valley avian communities in response to treatment. The communities were similar across the treated, untreated, and reference samples (figure 8). There was some evidence that the treated areas were less similar to each other than were the untreated areas, but this was not statistically significant ($p > .05$). Avian diversity (the Shannon index) was lowest for the treated sample prior to treatment but increased more in the posttreatment period, such that the Shannon index after treatment was equivalent in the treated and untreated samples (figure 9).

Evaluating the effects of fuel treatments with coarse metrics such as similarity and diversity can cause one to overlook large effects on select species (Hurteau et al. 2008). Numerous studies in seasonally dry fire-prone US forests have shown that fuel treatments can result in at least modest changes in the abundance of a broad range of avian species (Fontaine and Kennedy 2012). We recently reported that mechanical fuel-reduction treatments in the northern Sierra Nevada (including Meadow Valley) resulted in modest decreases in the abundance of a few closed-canopy associates and increases in some edge and open forest associates (Burnett et al. 2013). None of the 15 species evaluated in that study showed a significant decline following the construction of shaded fuel break DFPZ treatments—the primary treatment used in the Meadow Valley study area. With the moderate portion of the landscape treated, small differences in avian community similarity and diversity resulting from treatment, and the results from our previous evaluation of individual species response, we conclude that the effects of the Meadow Valley fuel-treatment network on the songbird community were minimal.

The fuel treatments implemented in Meadow Valley were typically less intense than those shown to result in large changes in avian communities (for a review, see Vanderwel et al. 2007). The treatments were applied to 19% of the landscape, and the prescriptions left relatively high canopy cover. Fire suppression and silvicultural practices over the last century have reduced forest heterogeneity and increased stand density (Scholl and Taylor 2010, Collins et al. 2011). In the Sierra Nevada, most fuel treatments changed the forest structure moderately from historic forest conditions (North et al. 2007). The Meadow Valley mechanical treatments primarily removed ladder fuels, which reduced crown fire potential but did not substantially alter the existing habitat features associated with songbirds, such as shrub cover or large overstory trees.

Our results should be considered in the context of the conditions that existed in the study area prior to the implementation of the landscape treatments. If an objective of these treatments was to maintain the existing avian assemblage and diversity, they appear to have been successful. However, a frequently stated objective for fuel reduction is to act as a surrogate for the natural fire regime (Stephens et al. 2012). Therefore, the maintenance of the pretreatment wildlife community may not always be the most desirable outcome in landscapes such as Meadow Valley and the larger PNFSA, where fire has been excluded for 85–125 years (Moody et al. 2006). Creating or enhancing

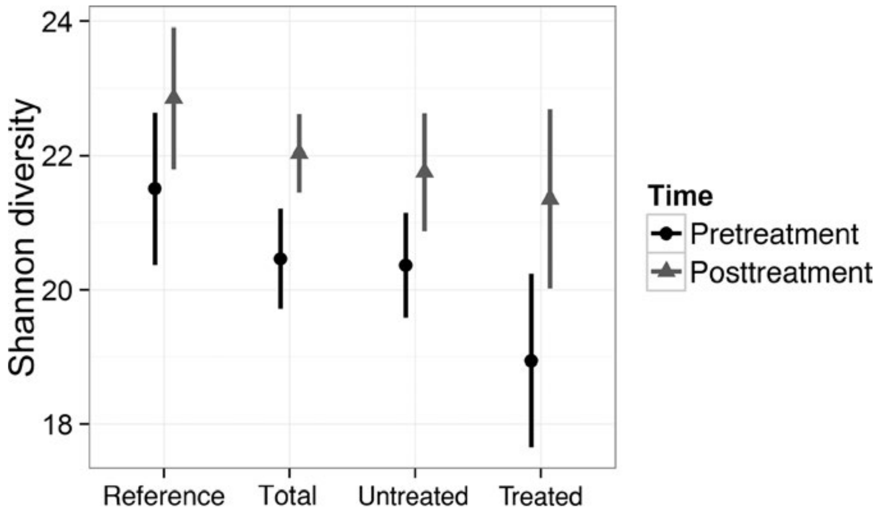

*Figure 9. Shannon diversity index of avian diversity before (*pretreatment*) and after (*posttreatment*) fuel treatments were implemented at treated (*n = *51) and untreated (*n = *201) locations and the first two combined (*Total; n = *252) in the Meadow Valley study area and in reference locations in the adjacent Plumas National Forest study area, which received no treatment (*n = *181). The error bars are 95% confidence intervals.*

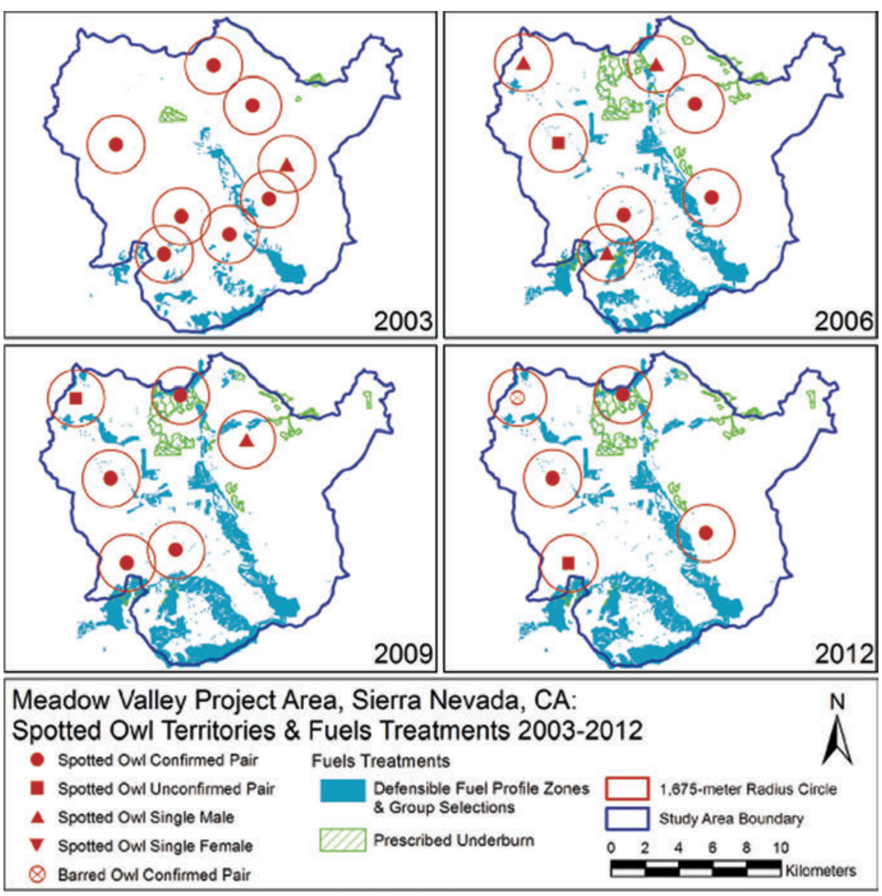

Figure 10. Distribution of territorial California spotted owl sites and landscape forest fuel treatments within the Meadow Valley study area from 2003 to 2012.

conditions for species associated with postdisturbance habitat, some of which have experienced recent declines, may be a prudent approach for achieving some biological diversity objectives (Fontaine and Kennedy 2012). If fuel-reduction treatments are to be a complementary tool to fire in achieving biological objectives, we suggest that they be designed to further increase landscape heterogeneity in fire-excluded forests.

California spotted owls

Modeling studies have projected that fuel treatments on a portion of the landscape (20%–35%) may have minimal effects on owl habitat and that the longer-term benefits of reduced wildfire risk may outweigh the short-term treatment effects on owl habitat (Ager et al. 2007, Roloff et al. 2012). However, no empirical data are available to assess the effects of landscape fuel treatments on the CSO and its habitat.

We used standardized surveys and color banding of individual owls to monitor the distribution, occupancy, survival, and reproduction of CSO sites annually across 1889 square kilometers between 2003 and 2012 in the Plumas and Lassen National Forests. Within this area, four areas were identified for implementation of landscape-scale fuel and restoration treatments. Our initial objectives were to establish baseline values for CSO distribution and abundance and to monitor the owl's response in the treated and untreated landscapes in posttreatment years. However, complete implementation of the fuel-treatment network only occurred on one (Meadow Valley; figure 10) of the four landscapes because of legal challenges to the proposed US Forest Service management strategy.

In the Meadow Valley study area, the number of territorial owl sites declined after treatment. Prior to and throughout the implementation of the treatment, the number of owl sites ranged from seven to nine. Between the final year of the DFPZ and group-selection installations (2008) and 2 years after treatment (2009–2010), the number of owl sites declined by one (six territorial sites), and by 3–4 years after treatment (2011–2012), the number of sites had declined to four—a decline of 43% from the pretreatment numbers (figure 11). These results mirror similar declines of the CSO in the larger Plumas-Lassen CSO study area over the past 20 years (Conner et al. 2013) but suggest a greater magnitude of decline within Meadow Valley (figure 11).

The CSO nests and roosts in dense, multilayered, mature forest patches, and the adult survival and territory occupancy of these owls is positively correlated to the amounts of mature forest in core areas around CSO sites (Dugger et al. 2011). For foraging, however, the CSO uses a broader range of vegetative conditions. Radio-telemetry conducted in Meadow Valley indicates that the CSO avoids foraging in DFPZs in the first 1–2 years after fuel treatments and that the owl's home range size was positively correlated with the amount of treatment within the home range (Gallagher 2010). Barred owls (*Strix varia*) began to colonize the Meadow Valley study area in 2012 and are likely to become a threat to the CSO and a confounding factor to be accounted for in assessments of forest management effects (Keane 2014).

Although inference must be tempered from a single study, the Meadow Valley area is the first large area to receive full the implementation of landscape-scale DFPZ and group-selection treatments in which CSOs were monitored annually both before and after treatment. CSOs are long-lived (up to 20 years) and exhibit high site fidelity as adults, although there is high annual variation in reproduction associated with weather and food (Gutierrez et al. 1995). Given these traits, individual CSOs may exhibit both short- and long-term responses to fuel treatments or wildfire, and understanding both is important to land-use managers. Our results documented a decline in CSO territories as a result of landscape fuel treatments, but the factors driving the decline remain unknown.

Conclusions

This study has shown that coordinated landscape-scale fuel treatments can substantially reduce the potential for hazardous fire across a large montane region, even when a moderate proportion of the area that could not be treated because of management constraints. In many cases, lands with designated management emphasis, such as wildlife habitat reserves and stream buffers, are distributed throughout the landscape. Creating fuel treatments that exclude these lands can result in a patchwork of treated areas heavily dissected by, for example, untreated stream buffers. Hazardous fire potential decreased in untreated areas, but that effect is not stable over time. Even if the existing network was maintained in a "treated" condition (i.e., periodic prescribed fire to keep surface and ladder fuels low) hazards will continue to increase in untreated areas because of stand development (Collins et al. 2013).

Our results indicate negative CSO responses to treatments, supported by the avoidance of DFPZs by foraging owls, larger owl home ranges associated with increasing amounts of treatment within the home ranges, and a 43% decline in the number of territorial CSO sites across the Meadow Valley study area within 3–4 years of the implementation of landscape treatments. In addition to changes in the number of owls, we also observed spatial redistribution of owl sites over time across the landscape (figure 10). The specific mechanisms driving these observations are unclear, but given the region-wide decline in the CSO population (Conner et al. 2013) and the increasing barred owl populations, it is difficult to disentangle fuel treatment effects from background or external pressures. Despite the challenges of working at landscape scales, studies such as this provide opportunities for addressing scale-dependent ecological phenomena, such as population-level species responses and responses to management strategies that cannot be addressed at smaller spatial scales.

To date, little discussion has been focused on what may constitute sustainable, viable CSO populations under various landscape conditions designed to address projected fire and climate scenarios. Furthermore, there is not a clear understanding of the balance between the potential short-term impacts from treatments and the longer-term benefits provided by introducing landscape heterogeneity (North et al. 2009), reducing potential for severe fire (Ager et al. 2007, Collins et al. 2013), increasing the potential for more desirable fire effects (North et al. 2012), and increasing resilience to climate change (Stephens et al. 2010). The Meadow Valley study is an important step in learning about the responses of wildlife species to fuel-reduction treatments.

Recent research in Yosemite National Park suggests that CSOs are not adversely affected by low- to moderate-severity fire (Roberts et al. 2011, Lee et al. 2013). Studies of the CSO both in Yosemite and in Sequoia and Kings Canyon National Parks have not shown population declines that have been found in several national forests in California. There are many differences between the two ownerships: National forest lands generally contain younger forests and lack the large tree structures associated with preferred owl habitat. With continued fire suppression, national forest lands continue to develop dense, small-tree stand conditions, reducing the habitat heterogeneity associated with a variety of small mammals that constitute the CSO's prey base. Because of these differences, it is difficult to determine whether more recent mechanical treatments or existing fire-suppressed conditions might be associated with declining CSO populations. Uncertainty also persists regarding the potential thresholds at which the amounts and patch sizes of high-severity fire reduce the postfire probabilities of CSO occupancy, survival, and reproduction. This is a significant information gap, given the trend for increasing amounts and patch sizes of high-severity fire in many Sierra Nevada forests (Miller et al. 2009). Unfortunately, only one CSO pair in Meadow Valley used an area that received prescribed burn treatments, but unlike those in some of the mechanically treated areas, these owls continued to occupy the burned area through the duration of the study and foraged within the burn-treatment areas (Gallagher 2010). The introduction of barred owls to Meadow Valley adds another important factor that may

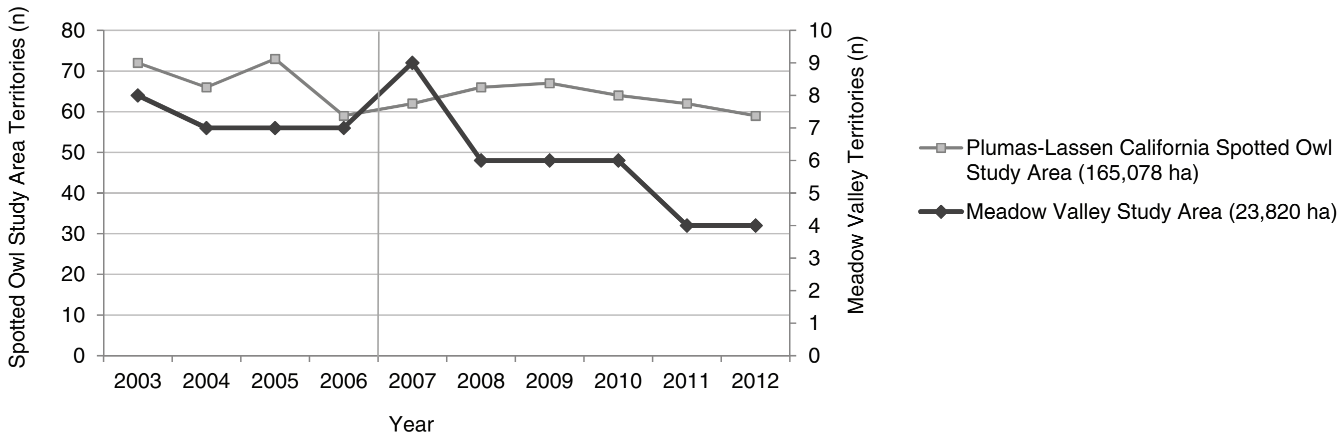

Figure 11. The annual number of territorial California spotted owl sites from 2003 to 2012 within the Meadow Valley study compared with the rest of the Plumas-Lassen study area (in the Plumas and Lassen National Forests). Vertical line represents completion of >80% of treatments.

reduce the population and viability of the CSO, possibly independent of forest structure.

Mechanical treatments can reduce fuels, but, in this study, they also left the largest trees and retained more than 40% canopy cover, two structural characteristics associated with CSO habitat use (Verner et al. 1992). However, although mechanical treatments retain these live features, they often remove snags for operator safety and fuel objectives; reduce tree density and canopy layering; reduce canopy cover to the minimum level (around 40%) considered to function as owl foraging habitat; and simplify the ground structure through a reduction of logs and small trees. Furthermore, DFPZ treatments are often uniformly implemented over large areas along roads, which results in extensive patches of simplified stand structure with regularly spaced trees. Another concern is that treatment size and placement are determined by land-use constraints (gentle slopes, access to roads) and opportunities to affect fire behavior. We have little information about how the location of treatments may affect CSOs' use of areas outside their core nesting locations. Several small mammals appeared to favor sites with steeper slopes (Kelt et al. 2013), possibly reflecting the spatial allocation of treatments in this landscape.

The importance of increasing heterogeneity within stands and across the landscape in mixed-conifer forests is well documented to meet restoration objectives (North et al. 2009, Stephens et al. 2010). Our ability to optimize heterogeneity at large scales may be more effectively achieved with prescribed and managed fires that are allowed to burn under moderate weather conditions. This type of burn often produces variable forest conditions that mimic historic patterns (Collins et al. 2011) to which this fauna, including the CSO, has adapted. Alternatively, mechanical treatments that produce the complex forest structure and composition that more closely mimic the patterns generated under a more active fire regime (North et al. 2009) may provide habitat conditions to support CSOs and a diverse fauna superior to those of the DFPZ and group-selection treatments implemented in Meadow Valley. Although mean stand conditions (e.g., canopy cover) have often been used to infer management impacts on preferred habitat (Tempel et al. in press), the historic heterogeneity of frequent-fire forests suggests we have yet to identify the optimal scales at which to create variable forest conditions.

We encourage further work to examine landscape-level treatments that are intended to emulate the influence of fire in creating spatial heterogeneity in vegetation and fuel conditions. A working hypothesis is that increased habitat heterogeneity, including the retention and development of currently limited but ecologically important forest conditions (areas of large, old trees) and more-open, patchy, early-seral stage conditions, would promote a diverse wildlife community while providing a more fire-resilient landscape. The results from the Meadow Valley study area illustrate the benefits and challenges of working at the landscape scale. Rigorous and controlled experiments are difficult because of the inherent variability across landscapes, sociopolitical constraints, and competing management objectives that can influence planned treatments. However, inferences from these studies can be strengthened by careful replication of management strategies across multiple landscapes.

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Supplemental material

The supplemental material is available online at *http://bioscience.oxfordjournals.org/lookup/suppl/doi:10.1093/biosci/biu137/-/DC1.*

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Frontiers in Ecology and the Environment

The forgotten stage of forest succession: early-successional ecosystems on forest sites

Mark E Swanson, Jerry F Franklin, Robert L Beschta, Charles M Crisafulli, Dominick A DellaSala, Richard L Hutto, David B Lindenmayer, and Frederick J Swanson

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The forgotten stage of forest succession: early-successional ecosystems on forest sites

Mark E Swanson[1*], Jerry F Franklin[2], Robert L Beschta[3], Charles M Crisafulli[4], Dominick A DellaSala[5], Richard L Hutto[6], David B Lindenmayer[7], and Frederick J Swanson[8]

Early-successional forest ecosystems that develop after stand-replacing or partial disturbances are diverse in species, processes, and structure. Post-disturbance ecosystems are also often rich in biological legacies, including surviving organisms and organically derived structures, such as woody debris. These legacies and post-disturbance plant communities provide resources that attract and sustain high species diversity, including numerous early-successional obligates, such as certain woodpeckers and arthropods. Early succession is the only period when tree canopies do not dominate the forest site, and so this stage can be characterized by high productivity of plant species (including herbs and shrubs), complex food webs, large nutrient fluxes, and high structural and spatial complexity. Different disturbances contrast markedly in terms of biological legacies, and this will influence the resultant physical and biological conditions, thus affecting successional pathways. Management activities, such as post-disturbance logging and dense tree planting, can reduce the richness within and the duration of early-successional ecosystems. Where maintenance of biodiversity is an objective, the importance and value of these natural early-successional ecosystems are underappreciated.

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Severe natural disturbances – such as wildfires, windstorms, and insect epidemics – are characteristic of many forest ecosystems and can produce a "stand-replacement" event, by killing all or most of the dominant trees therein (Figure 1). Typically, limited biomass is actually consumed or removed in such events, but many trees and other organisms experience mortality, leaving behind important biological legacies (structures inherited from the pre-disturbance ecosystem; Franklin *et al.* 2000), including standing dead trees and downed boles (tree trunks; Franklin *et al.* 2000). Such legacies provide diverse physical/biological properties and suitable microclimatic conditions for many species. Thereafter, species-diverse plant communities develop because substantial amounts of previously limited resources (light, moisture, and nutrients) become available. These emerging plant communities create additional habitat complexity and provide various energetic resources for terrestrial and aquatic organisms.

In a nutshell:

- Naturally occurring, early-successional ecosystems on forest sites have distinctive characteristics, including high species diversity, as well as complex food webs and ecosystem processes
- This high species diversity is made up of survivors, opportunists, and habitat specialists that require the distinctive conditions present there
- Organic structures, such as live and dead trees, create habitat for surviving and colonizing organisms on many types of recently disturbed sites
- Traditional forestry activities (eg clearcutting or post-disturbance logging) reduce the species richness and key ecological processes associated with early-successional ecosystems; other activities, such as tree planting, can limit the duration (eg by plantation establishment) of this important successional stage

The ecological importance of early-successional forest ecosystems (ESFEs) has received little attention, except as a transitional phase, before resumption of tree dominance. In forestry, this period is often called the "cohort re-establishment" or "stand initiation" stage, with attention obviously focused on tree regeneration and the re-establishment of closed forest canopies (Franklin *et al.* 2002). Ecological studies have focused primarily on plant-community development and the needs of selected animal (mostly game) species, and not on the diverse ecological roles of ESFEs.

Here, we highlight important features of ESFEs, including their role in sustaining ecosystem processes and biodiversity, so that they may be appropriately considered by resource managers and scientists, and included within management/research programs dedicated to maintaining these functions, particularly at larger spatio-temporal scales. Most published examples focus on sites in western North America, but ESFEs are important elsewhere (Angelstam 1998; DeGraaf *et al.* 2003). We also discuss how traditional forestry practices, such as clearcutting, tree planting, and post-disturbance logging, can affect early-successional communities.

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Figure 1. *Stand-replacement disturbance events in forests create large areas free of tree dominance and rich in physical and biological resources, including legacies of the pre-disturbance ecosystem.*

(Franklin *et al.* 2002). Consequently, the duration of ESFEs varies inversely with rapidity of tree regeneration and growth, which, in turn, depend on such variables as tree propagule availability, conditions affecting seedling or sprout establishment, and site productivity. ESFE longevity after natural disturbances is therefore highly variable.

Development of a closed forest canopy may require a century or more in areas with limited seed sources, harsh environmental conditions, severe shrub competition (in some instances), or combinations thereof (Hemstrom and Franklin 1982). For example, tree canopy closure after wildfire in the Douglas fir region of western North America often requires several decades (Poage *et al.* 2009), but can occur much more rapidly when canopy seedbanks are abundant (eg Larson and Franklin 2005). Closed forest canopies may develop quickly in forests dominated by trees with strong sprouting ability (eg many angiosperms) or when windstorms "release" understories of shade-tolerant tree seedling banks by removing all or most of the overstory (Foster *et al.* 1997).

■ Early-successional ecosystems on forest sites

Initial conditions after stand-replacing forest disturbances vary generically, depending on the type of disturbance; this includes the types of physical and biological legacies available. For example, aboveground vegetation may be limited immediately after the disturbance, as in the case of severe wildfires or volcanic eruptions. Conversely, intact understory communities may persist where forests have been blown down by severe windstorms. Spatial heterogeneity in conditions is characteristic, given that disturbances vary greatly in the amount of damage they cause (Turner *et al.* 1998). For instance, severe wildfires frequently include substantial areas of unburned as well as low to medium levels of mortality, creating variability in shade, litterfall, soil moisture, seed distribution, and other factors.

We define ESFEs as those ecosystems that occupy potentially forested sites in time and space between a stand-replacement disturbance and re-establishment of a closed forest canopy. These ecosystems undergo compositional and structural changes (succession) during their occupancy of a site. Changes begin immediately post-disturbance, as a result of the activities of surviving organisms (eg plants, animals, and fungi), including plant growth and seed production. Developmental processes are enriched by colonization of flora and fauna from outside the disturbed area. Successional change is often characterized by progressive dominance of annual and perennial herbs, shrubs, and trees, although all of these species are typically represented throughout the entire sequence of forest stand development (or sere; Halpern 1988).

The ESFE developmental stage ends with re-establishment of tree cover that is sufficiently dense to suppress and often eliminate many smaller shade-intolerant plants

■ Attributes of early-successional ecosystems

After severe disturbances, forest sites are characterized by open, non-tree-dominated environments, but have high levels of structural complexity and spatial heterogeneity and retain legacy materials.

Environmental conditions

Removal of the overstory forest canopy during disturbances dramatically alters the site's microclimate, including light regimes. These changes lead to increased exposure to sunlight, more extreme temperatures (ground and air), higher wind velocities, and lower levels of relative humidity and moisture in litter and surface soil. Shifts in these environmental metrics favor some species, while creating suboptimal or intolerable conditions for others. For example, post-disturbance plant community composition, cover, and physiognomy are altered as shade-tolerant understory herbs are largely displaced by shade-intolerant and drought-tolerant species. New substrates deposited by floods or volcanic eruptions may lack nutrients, provide additional water-holding capacity, or have high albedo, all of which favor shifts in plant communities.

Survivors

Organisms (in a variety of forms) that survive severe disturbances are extremely important for repopulating and

restoring ecosystem functions in the post-disturbance landscape. Even in severely disturbed areas, organisms may survive as individuals (mature or immature) or as reproductive structures (eg spores, seeds, rootstocks, and eggs), which become in situ propagule sources. For example, after the 1980 volcanic eruption of Mount St Helens (Washington State), most pre-eruption flora and many fauna (especially aquatic and burrowing terrestrial species) survived within the blast zone through several different mechanisms (Dale *et al*. 2005).

Surviving organisms are also often vital for the prompt re-establishment of important ecosystem functions, such as conservation of nutrients and stabilization of substrates. For instance, the important role of resprouting vegetation in curbing massive losses of nitrogen was demonstrated by experimentally clearcutting and applying herbicides in a watershed at Hubbard Brook Experimental Forest (Bormann and Likens 1979).

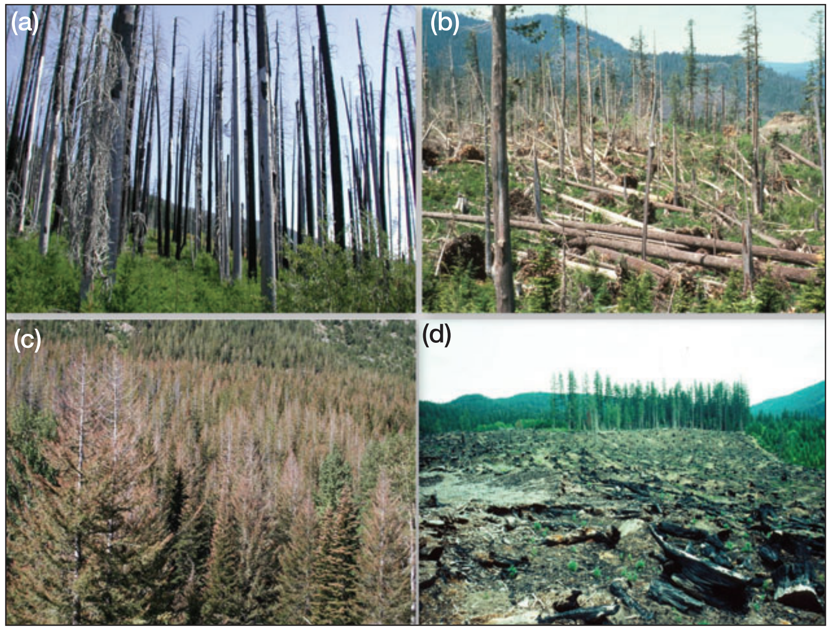

Figure 2. *Different types of disturbances produce different types of biological legacies, including living organisms and structures: (a) standing dead trees (snags) are dominant structural legacies after severe wildfires; (b) downed tree trunks and nearly intact understory communities are characteristic legacies after major windstorms; (c) standing dead trees are also dominant structural legacies after heavy insect infestations; and (d) clearcuts typically eliminate most aboveground structural legacies. Values for each metric are shown in Table 1 and are described in detail in the text.*

Structural complexity

The structural complexity of ESFEs depends initially on legacies, the general nature of which varies with the type of disturbance (Table 1; Figure 2); for example, snags and shrubs originating from belowground perennating (ie resprouting) parts or seeds are dominant legacies after wildfires, whereas downed boles and largely intact understories are typical post-disturbance characteristics of windstorms.

Woody legacies, such as snags and downed boles, play numerous roles in structuring and facilitating the development of the recovering ecosystem – providing habitat for survivors and colonists, moderating the physical environment, enriching aquatic systems in the disturbed area (Jones and Daniels 2008), and providing long-term sources of energy and nutrients (Harmon *et al*. 1986). Although subject to decomposition, these legacies can persist for many decades and sometimes even centuries.

Table 1. Different types of intense disturbances generate different types of biological legacies

	Disturbance				
Biological legacies	*Wildfire*	*Wind*	*Insect*	*Volcano*	*Clearcut*
Live trees	Infrequent	Variable	Variable (depends on stand composition)	Infrequent – confined to margins	Infrequent or absent
Snags	Abundant	Variable	Abundant	Abundant (spatially variable)	Infrequent or absent
Downed woody debris	Variable, but typically abundant	Abundant	Variable, but eventually abundant	Abundant (spatially variable)	Infrequent
Undisturbed understory	Infrequent	Abundant	Abundant	Infrequent – confined to disturbance margins	Infrequent
Spatial heterogeneity of recovery	High	Variable	High	High	Variable – usually low
Time in early-successional condition	Variable	Variable	Long	Variable – usually long	Variable – usually short

Figure 3. *Plant communities with well-developed shrub and perennial herb species are characteristic of early-successional communities on forest sites and provide diverse food resources. Twenty-five years after the Mount St Helens eruption in 1980, this community, which was within the blast zone, includes well-developed shrubs (eg* Sorbus *and* Vaccinium *spp), trees, and perennial herbs (eg* Epilobium angustifolium*).*

Structural complexity is further enhanced by the establishment and development of a variety of plant species, which often include perennial herbs and shrubs characteristic of open environments, as well as individual trees (Figure 3). The diversity of plant morphologies (maximum height, crown width, etc) increases structural richness, so that this associated flora contributes to both horizontal and vertical heterogeneity.

Spatial heterogeneity

Spatial heterogeneity is evident in early-successional ecosystems and has multiple causes: (1) natural variability in the geophysical template (topography and lithology) of the affected landscape; (2) variability in conditions in the pre-disturbance forest ecosystem; (3) variability in the intensity of the disturbance event; and (4) variability in rates and patterns of subsequent developmental processes in the ESFE. The first two sources relate to existing geophysical and biological patterns within the disturbed area. Land formations and patterns of geomorphic processes are certainly key geophysical elements (Swanson *et al.* 1988). The presence of surface water, such as streams and ponds, can be particularly influential in facilitating survival and re-establishment of biota.

Natural disturbances create heterogeneous environments at multiple spatial scales (Heinselmann 1973), because disturbances do not cause damage uniformly. Disturbances such as wildfires and windstorms are variable in intensity (eg "spotting", or initiation of new flame fronts by wind-thrown firebrands, during fire events). Alternatively, geographic variation in environmental conditions and topography (Swanson *et al.* 1988) influences the intensity of the disturbance and results in heterogeneity at multiple scales. Variability in the structure and composition of the pre-disturbance forest also creates spatial and temporal variability (Wardell-Johnson and Horowitz 1996). Some of these patterns may be transient, such as residual snowbanks protecting tree regeneration after the aforementioned Mount St Helens eruption (Dale *et al.* 2005).

Post-disturbance developmental processes also lead to spatial heterogeneity. For example, varying distances to sources of tree seed result in different rates and densities of tree re-establishment (Turner *et al.* 1998). Structural legacies can greatly influence the rates at which wind- or waterborne organic (including propagules) and inorganic materials are deposited. Finally, animal activity can strongly influence patterns of revegetation, as illustrated by the multiple effects that gophers (*Thomomys* spp) can have on post-disturbance landscapes (Crisafulli *et al.* 2005b) or the way ungulate browsing may impede tree regeneration (Hessl and Graumlich 2002).

■ Biological diversity

ESFEs in temperate forest seres show great diversity in the abundance of plant and animal species (Fontaine *et al.* 2009). Species composition may consist of a mix of forest survivors, opportunists, or ruderals (plants that grow on disturbed or poor-quality lands), and habitat specialists that co-exist in the resource-rich ESFE environment (Figure 3). Most forest understory flora can survive disturbances as established plants, perennating rootstocks, or seeds. In one study, in western North America, over 95% of understory species survived the combined disturbance of logging and burning of an old-growth Douglas-fir–western hemlock stand (Halpern 1988). Some important early-successional species (eg *Rubus* spp [blackberry; raspberry], *Ribes* spp [gooseberry], and *Ceanothus* spp [buckbrush]) may persist as long-lived seedbanks.

Opportunistic herbaceous species are often conspicuous dominants early in the development of ESFEs (Figure 4). Many of these weedy species (particularly annuals) decline quickly, although other opportunists will persist as part of the plant community until overtopped by slower growing shrubs or trees. Consequently, diverse plant communities of herbs, shrubs, and young trees emerge in ESFEs; this, combined with the structural legacies from the pre-disturbance ecosystem, often results in high levels of structural richness (Figure 3).

Many animals, including habitat specialists and species typically absent from the eventual tree-dominated com-

munities, thrive under the conditions found in ESFEs. For some species, this is the only successional stage that can provide suitable foraging or nesting habitat. As an example, many butterflies and moths (Lepidoptera) found in forested regions depend on the high diversity and quality of plant forage in ESFEs (eg Miller and Hammond 2007), whereas jewel beetles (Coleoptera: Buprestideae) depend on abundant coarse woody debris. Also, a number of ground-dwelling beetle species occur as habitat specialists in early-successional communities (Heyborne *et al.* 2003).

Figure 4. *Early-successional communities are often dominated by annual herbaceous species for the first few years after disturbance; these are quickly displaced by perennial herbaceous species and shrubs.*

Many vertebrates also respond positively to ESFEs, which may provide the only suitable habitat at a regional scale for some species. Ectothermic animals, such as reptiles (eg Rittenhouse *et al.* 2007), generally respond favorably to sunnier and drier conditions, colonizing early-successional habitat or increasing in abundance if present as survivors. Many amphibians also thrive in ESFEs, provided resources such as water bodies and key structures (eg logs) are available. The diversity and abundance of amphibians in the area affected by the 1980 Mount St Helens eruption is illustrative (Crisafulli *et al.* 2005a); eleven of 15 amphibian species survived the event, and some (eg western toad, *Bufo boreas*) have since had exceptional breeding success.

The broad array of birds using the abundant and varied food sources (eg fruits, nectar, herbivorous insects) and nesting habitat in ESFEs includes many raptors and neotropical migrants, often making bird diversity highest during the ESFE stage of succession (Klaus *et al.* in press). Some species are habitat specialists that directly utilize the legacy of recently killed trees; for instance, black-backed woodpeckers (*Picoides arcticus*) are almost completely restricted to early post-fire conditions (Hutto 2008). Mountain bluebirds (*Sialia currucoides*) and several other woodpecker species also favor structurally rich, early-successional habitats (Figure 5). Observed population declines of many avian species in eastern North America – which, in some cases, have proceeded to a point of conservation concern – are linked to conversion of early-successional habitat to closed forest (Litvaitis 1993).

Small mammal communities in ESFEs typically show high levels of diversity as well, including some obvious habitat specialists. The eastern chestnut mouse (*Pseudomys gracilicaudatus*), for example, inhabits early-successional environments in coastal eastern Australia for 2–5 years after a wildfire, and then declines dramatically until these environments are burned again (Fox 1990). Populations of mesopredators (medium-sized predators, such as raccoons [*Procyon lotor*] and fox species) benefit from the abundance of small vertebrate prey items characteristic of ESFEs. Likewise, some species of large mammals are well known to favor ESFEs (Nyberg and Janz 1990). Utilizing the diverse and luxuriant forage characteristically present in these ecosystems, ungulates, such as members of the Cervidae, in turn serve to benefit large predators (eg wolves [*Canis lupus*]) as well as scavengers, making ESFEs important elements within those species' typically extensive home ranges. Omnivores, such as bears (*Ursus* spp), also rely on the diversity of food sources often present in ESFEs.

Food web diversity

ESFEs are exceptional in the diversity and complexity of food webs they support. Simply stated, a diverse plant community produces many food sources. Food resources for herbivores (grasses, shrubs, forbs) – as well as nectar, seeds, and shrub-borne fruit (eg produced by *Rubus* and *Vaccinium* spp [huckleberry]) – can reach high levels before site dominance by trees. In the temperate Northern Hemisphere, biologically important berry production is maximized in slowly reforesting ESFEs. Resource production in early-successional patches may even augment the richness of adjacent undisturbed forests, as in the case of fluxes of key prey species (Sakai and Noon 1997).

Aquatic biologists have, perhaps, best appreciated the greater complexity of food chains in early-successional versus closed forest environments (Bisson *et al.* 2003). In established forest stands, trees strongly dominate the physical and biological conditions in nearby small streams by controlling light and temperature, stabilizing channels, providing woody debris, and, importantly, offering allochthonous inputs (organic matter originating outside the aquatic ecosystem) – the primary energy and nutrient source for such ecosystems (Vannote *et al.* 1980).

Stand-replacement disturbances remove forest constraints on conditions and processes, and shift streams to an early-

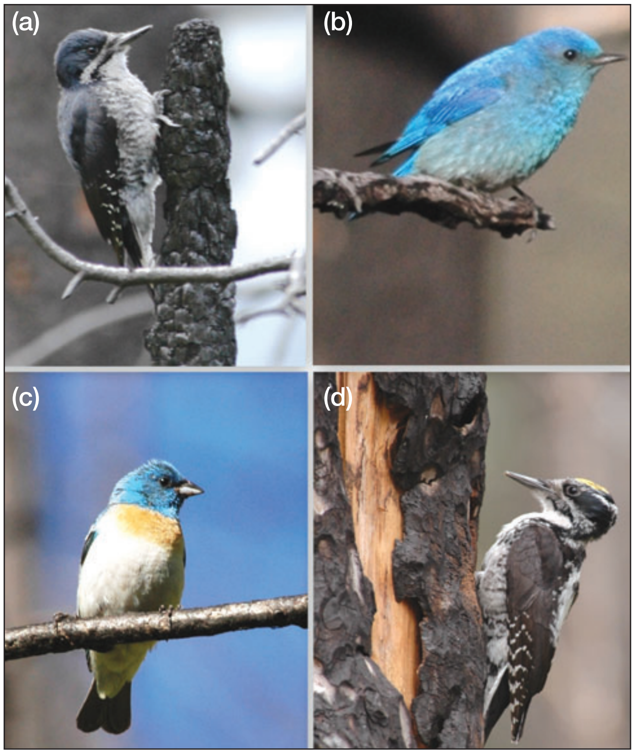

Figure 5. *Bird diversity is typically high in early-successional communities on forest sites and includes many habitat specialists: (a) black-backed woodpeckers (*Picoides arcticus*) are almost entirely restricted to early post-fire habitat; (b) mountain bluebirds (*Sialia currucoides*) favor early-successional ecosystems; (c) lazuli buntings (*Passerina amoena*) and (d) three-toed woodpeckers (*Picoides tridactylus*) have similar requirements.*

successional context (Minshall 2003; Figure 6). This greatly diversifies the types and timing of allochthonous inputs, as well as increases primary productivity. Allochthonous inputs are shifted from primarily tree-derived litter (coniferous-based in many systems) to material from a range of flowering herbs, shrubs, and trees, as well as from conifers. Consequently, litter inputs are highly variable in quality (eg decomposability) and delivery time, as compared with litterfall contributed primarily by evergreen conifer species. Also, inputs to post-disturbance streams often include material with a high nitrogen content, such as litter from the early-successional genera *Alnus* and *Ceanothus* (Hibbs *et al.* 1994).

Greater algal production may increase the diversity and abundance of aquatic invertebrate populations, which, in turn, become prey for fish and other organisms. However, increases in sediment production associated with disturbances can negate some benefits to aquatic processes and organisms (Gregory *et al.* 1987).

Processes in ESFEs

Ecosystem processes in ESFEs can be more diverse than those in closed forest systems, where the primary productivity of trees is dominant and organic matter is processed primarily through detrital food webs. Development of more diverse, and perhaps more "balanced", trophic pathways is possible when a disturbance opens a previously closed forest canopy. The contrast is probably greatest in forests dominated by a single tree type, such as evergreen conifers, as opposed to more diverse forests, such as mixed evergreen associations.

Recharging nutrient pools

ESFEs provide major opportunities for recharge of nutrient pools, such as additions to the nitrogen pool by leguminous (eg *Lupinus*) and some non-leguminous early-successional (eg *Alnus* and *Ceanothus*) plant species. These genera are commonly absent from late-successional forests, but are well represented in ESFEs. Nitrogenous additions from these sources are particularly important where the disturbance – eg a wildfire – has volatilized a substantial amount of the existing nitrogen pool.

Mineralization rates of organic material are typically accelerated (sometimes profoundly) after disturbances, as a result of warmer growing season temperatures. Diversified litter inputs in ESFEs, including a greater proportion of easily decomposed litter from herbs and deciduous shrubs, also result in more rapid mineralization. Finally, successional changes in the fungal and microbial communities can also hasten decomposition processes. As noted, these changes will be most profound in forest ecosystems dominated by a single species, including evergreen conifers or hard-leaved, evergreen hardwoods (such as the ash-type eucalypt forests of southeastern Australia).

In aquatic ecosystems that experience fire in adjacent forests, greater post-disturbance light and nutrient availability enhance primary productivity within the water body, causing shifts in food webs from the level of primary producers up through high-level consumers, such as fish (Spencer *et al.* 2003).

Modifying hydrologic and geomorphic regimes

Hydrologic regimes associated with ESFEs contrast greatly with those characterizing closed forest cover. For example, transpiration and interception are dramatically reduced and recover only gradually as forest canopies redevelop. Increases in normally low summer flows and annual water yields may occur immediately after a disturbance, as compared with levels in the dense young forests that may subsequently develop (Jones and Post 2004). The opposite may be true in systems where condensation of cloud or fog on tree crowns is an important component of the hydrologic cycle. ESFEs may also contribute to increased discharge peak runoff flows in hydrologic events of smaller magnitude (Harr 1986), but appear to have little effect on the magnitude of peak flows during large runoff events (Grant *et al.* 2008). From an ecological perspective, this may have a positive outcome, however, because floods restructure and rejuvenate many riparian communities (Gregory *et al.* 1991).

Rates and patterns of geomorphic processes, such as erosion and nutrient leaching losses, are also different between ESFEs and later successional stages. Tree death results in a loss of root strength that is critical for stabilizing soils and deeper rock layers on mountain slopes (Perry *et al.* 2008). Erosion and landslides may occur at higher rates in ESFEs, contributing to the variability of sediment budgets in watersheds (Reeves *et al.* 1995) and creating long-lasting substrates for ruderals. While enhancing erosion processes, ESFEs also provide materials and processes that counteract this effect, such as woody debris, which retain sediments and organic materials, and surviving vegetation, which stabilizes slopes and nutrient stores (eg Bormann and Likens 1979).


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Figure 6. *Streams within early-successional forest ecosystems contrast with forest-dominated reaches in many ecosystem attributes, including physical parameters (temperature and insolation), structure, plant and animal composition, and ecosystem processes, such as primary productivity.*

Land management implications

Incorporating ESFE attributes into forest policy and management is highly desirable, given the numerous advantages provided by these ecosystems. Many species and ecological processes are strongly favored by conditions that develop after stand-replacement disturbances. Rapid, artificially accelerated "recovery" of disturbed forest areas (eg via dense planting) to closed forest conditions has serious implications for many species. Clearly the term "recovery" has a different meaning for such early-successional specialists or obligates.

To fulfill their full ecological potential, ESFEs require their full complement of biological legacies (eg dead trees and logs) and sufficient time for early-successional vegetation to mature. Where land managers are interested in conservation of the biota and maintenance of ecological processes associated with such communities, forest policy and practices need to support the maintenance of structurally rich ESFEs in managed landscapes. Natural disturbance events will provide major opportunities for these ecosystems, and managers can build on those opportunities by avoiding actions that (1) eliminate biological legacies, (2) shorten the duration of the ESFEs, and (3) interfere with stand-development processes. Such activities include intensive post-disturbance logging, aggressive reforestation, and elimination of native plants with herbicides.

In particular, post-disturbance logging removes key structural legacies, and damages recolonizing vegetation, soils, and aquatic elements of disturbed areas (Foster and Orwig 2006; Lindenmayer *et al.* 2008). Where socioeconomic considerations necessitate post-disturbance logging, variable retention harvesting (retention of snags, logs, live trees, and other structures through harvest) can maintain structural complexity in logged areas (Eklund *et al.* 2009).

Prompt, dense reforestation can have negative consequences for biodiversity and processes associated with ESFEs, by dramatically shortening their duration. Such efforts reduce spatial and compositional variability characteristic of natural tree-regeneration processes, promote structural uniformity, and initiate intense competitive processes that eliminate elements of biodiversity that might otherwise persist. Artificial reforestation can also reduce genetic diversity by favoring dominance by fewer tree species/genotypes, and may make the system more prone to subsequent, high-severity disturbances (Thompson *et al.* 2007). The elimination of shrubs and broad-leaved trees through herbicide application can alter synergistic relationships, such as the belowground mycorrhizal processes provided by certain shrub species (eg *Arctostaphylos* spp).

Naturally regenerated ESFEs are likely to be better adapted to the present-day climate and may be more adaptable to future climate change. The diverse genotypes in naturally regenerated ESFEs are likely to provide greater resilience to environmental stresses than nursery-grown, planted trees of the same species. Given that climate change is also resulting in altered behavior of pests and pathogens (Dale *et al.* 2001), encouraging greater tree species diversity may also increase ecosystem resilience.

Clearcutting has been proposed as a technique to create ESFEs, but this can provide only highly abridged and simplified ESFE conditions. First, traditional clearcuts leave few biological legacies (eg Lindenmayer and McCarthy 2002), limiting habitat and biodiversity potential. Second, clearcuts are often quickly and densely reforested, and often involve the use of herbicides to limit competition with desired tree species. Clearcuts can provide some early-successional functionality (eg serving as nurseries or post-breeding habitat for many bird species in the southern US; Faaborg 2002), but this service is often truncated by prompt reforestation.

Management plans should provide for the maintenance of areas of naturally developing ESFEs as part of a diverse landscape. This should be in reasonable proportion to *historical* occurrences of different successional stages, as based on region-specific historical ecology. Major disturbance events provide managers with opportunities to incorporate a greater diversity of species and processes in forest landscapes and to enhance landscape heterogeneity. Some aspects of ESFEs can be incorporated into areas managed for production forestry as well, such as through variable retention harvest methods, the incorporation of natural tree regeneration, and extending the duration of herb/shrub communities in some portions of a stand by deliberately maintaining low tree stocking levels.

Finally, we suggest that adjustments in language are needed. Ecologists and managers often refer to "recovery" when discussing post-disturbance ecosystems, inferring that early seral conditions are undesirable and need to be restored to closed canopy conditions as quickly as possible. Emphasizing recovery as the management goal fails to acknowledge the essential ecological roles played by early-successional ecosystems on forest sites. It should also be considered that climate change and other factors may not permit "recovery" to pre-disturbance conditions.

■ Conclusions

Twentieth-century forest management objectives were centered on wood production and, later, on conservation and development of late-successional forests. Rapid regeneration of dense timber stands was frequently seen as a way to address both of these divergent objectives. Recognizing the ecological value of early-successional ecosystems on forest sites extends the ecological concerns associated with old growth to another "rich" period in a forest sere. This represents an important development in the evolution of holistic management of forest ecosystems, whereby large landscapes are managed for diverse seral stages.

ESFEs provide a distinctive mix of physical, chemical, and biological conditions, are diverse in species and processes, and are poorly represented and undervalued in traditional forest management. Forest policy and practice must give serious attention to sustaining substantial areas of ESFEs and their biological legacies. Similarly, scientists need to initiate research on the structure, composition, and function of ESFEs in different regions and under different disturbance regimes, as well as on the historical extent of these systems, to serve as a reference for conservation planning.

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Comparing selected fire regime condition class (FRCC) and LANDFIRE vegetation model results with tree-ring data

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Abstract. Fire Regime Condition Class (FRCC) has been developed as a nationally consistent interagency method in the US to assess degree of departure between historical and current fire regimes and vegetation structural conditions across differing vegetation types. Historical and existing vegetation map data also are being developed for the nationwide LANDFIRE project to aid in FRCC assessments. Here, we compare selected FRCC and LANDFIRE vegetation characteristics derived from simulation modeling with similar characteristics reconstructed from tree-ring data collected from 11 forested sites in Utah. Reconstructed reference conditions based on trees present in 1880 compared with reference conditions modeled by the Vegetation Dynamics Development Tool for individual Biophysical Settings (BpS) used in FRCC and LANDFIRE assessments showed significance relationships for ponderosa pine, aspen, and mixed-conifer BpS but not for spruce–fir, piñon–juniper, or lodgepole pine BpS. LANDFIRE map data were found to be ~58% accurate for BpS and ~60% accurate for existing vegetation types. Results suggest that limited sampling of age-to-size relationships by different species may be needed to help refine reference condition definitions used in FRCC assessments, and that more empirical data are needed to better parameterize FRCC vegetation models in especially low-frequency fire types.

Additional keywords: reference conditions, successional classes, Vegetation Dynamics Development Tool (VDDT).

Introduction

Altered fire regimes and associated changes in vegetation structure, composition, and fuels pose risks to biodiversity, sustainable ecosystems, and economic and community interests across the United States (USDA/USDI 2000). However, the magnitude of these risks varies between ecosystems as a result of differences in their fire and vegetation histories, successional, compositional, and structural dynamics, and the influence of invasive species (Morgan *et al*. 2001; Schoennagel *et al*. 2004). Fire exclusion over the 20th century has not affected all ecosystems uniformly, and accurate characterization of historical fire regimes and recent vegetation changes is critical to inform management decisions about the need for fuel treatments or ecological restoration across differing plant communities.

Use of historical fire regimes and vegetation conditions to inform fire and fuel management decisions in the US has been refined into the Fire Regime Condition Class (FRCC) concept (Hann and Bunnell 2001; Schmidt *et al*. 2002; Hann and Strohm 2003; Hann *et al*. 2003; Shlisky and Hann 2003). FRCC is an index that compares current with historical fire regimes and vegetation composition and structure to assess degree of departure on a scale from one (least departed) to three (most departed). FRCC is based on an assumption that historical processes and patterns (those present before widespread Euro-American settlement in the mid- to late-1800s) represent longer-term sustainable ecosystem conditions, and that greater departure in current conditions represents a greater risk for uncharacteristic fire behavior and associated ecosystem impacts. Initial coarse-level (1-km^2 resolution) FRCC maps described the degree of departure at a national scale (Schmidt *et al*. 2002). After this initial effort, a set of standard guidebook methods was developed to assess FRCC at landscape to stand scales for local management and planning needs (at time of writing, FRCC Guidebook v1.3; Hann *et al*. 2004). FRCC maps of 30-m^2 resolution are also being developed as part the LANDFIRE project, an effort to provide consistent vegetation, fuels, and fire regime data for the entire US (Rollins and Frame 2006; www.landfire.gov, accessed 19 October 2007). FRCC is now a key variable for defining wildfire risk to ecosystems as a result of its explicit incorporation into the Healthy Forests Restoration Act of 2003 (HFRA 2003). FRCC represents a significant advance in the integration of fire and forest histories and landscape and vegetation ecology to provide an ecologically based method for setting fire-management priorities and objectives across the US (Shlisky and Hann 2003).

Definition of departure indices in FRCC assessments begins with simulation modeling of historical vegetation composition and structure using the Vegetation Dynamics Development Tool (VDDT; Beukema and Kurz 2003). VDDT is used to develop non-spatially defined reference conditions within Biophysical Settings (BpS; formerly referred to as Potential Natural Vegetation Groups (PNVG); Küchler 1964; NRCS 2003). For LANDFIRE, BpS are derived from Nature Serve's ecological

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classification system (Comers *et al*. 2003) and are not directly comparable with those used in FRCC assessments. However, both systems use BpS in a similar manner to represent the vegetation communities that would likely exist under given environmental conditions (climate, soils, and landscape physiography) and historical disturbance regimes. BpS in LANDFIRE are assigned to specific locations in their nationwide mapping efforts, whereas BpS in FRCC assessments are non-spatial and assigned based on individual user needs for specific projects or management requirements. Reference conditions are the proportions of vegetation successional stages (community structure and composition) as affected by varying fire frequencies, severities, and successional pathways within each BpS (Hann *et al*. 2004).

FRCC and LANDFIRE vegetation models (also known as Vegetation Dynamics Models) were defined during regional professional workshops conducted between 2002 and 2009 (2005–09 for LANDFIRE). VDDT model inputs for individual BpS are based on historical fire regime characteristics (frequency and severity) and vegetation data derived from published and unpublished studies and expert opinion developed both at the regional workshops and through subsequent peer reviews (Hann *et al*. 2004). The amount and quality of available historical data for each BpS vary, which can affect the quality and accuracy of the resulting modeled reference conditions. In an FRCC assessment, a field evaluation is conducted of existing vegetation structure, which, in forests, is based on cover type, density of tree stands, tree size, and current successional status. Successional status is determined by visually estimating stand composition, tree density, and average tree age, the latter of which is based on tree diameters. Proportions of current successional classes in a project or management area are estimated during the field assessment and then compared with the proportions of reference conditions derived from VDDT model output. The FRCC departure index (1 to 3) is assigned based at least partially on differences in proportions of successional classes present in the current forest relative to modeled reference conditions in the historical forest.

There is a need to test the process of development of reference conditions by comparing VDDT model output with known fire and vegetation histories. This comparison is critical for assessing consistency and accuracy in the modeling process. Here, we compare VDDT-modeled reference conditions with tree-ring-based reconstructions of reference conditions from 11 forested sites in Utah and eastern Nevada (tree-ring data reported in Heyerdahl *et al*. 2005, and Brown *et al*. 2008*a*). The tree-ring reconstructions span transects aligned along elevation gradients that include multiple forest types. We ask the following questions with this comparison: (1) do FRCC methods of evaluating stand structure based on diameter estimates accurately represent ages of forest vegetation and is there variation based on species and site? (2) Do FRCC and LANDFIRE BpS models adequately capture the range of variation in proportions of reference conditions reconstructed by the tree-ring data? (3) Do LANDFIRE mapped data layers for BpS and Existing Vegetation Types (EVT) match the tree-ring plot data? (4) Can further empirical fire history and tree recruitment data be used to strengthen FRCC evaluation and reference condition modeling outputs? We consider this study to be only an initial test of FRCC and LANDFIRE vegetation modeling methods, but one that may provide an example for future testing needs.

Methods

Study area

Tree-ring sites used for this study extend from the Colorado Plateau of southern Utah, west to the Wah Wah Mountains in the eastern Great Basin of western Utah, and north to the Uinta and Bear River Mountains in northern Utah (Fig. 1, Table 1; Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). The region is a complex of valleys, mesas, canyons, plateaus, and mountains that range in elevation from ~900 to >3600 m. Forest types vary generally across elevation gradients. Piñon (*Pinus edulis* (PIED; four-letter codes are used in tables) and *P. monophylla* (PIMO)) and juniper (*Juniperus scopulorum* (JUSC) and *J. osteosperma* (JUOS)) savannas and woodlands occur at the lowest forest margins above desert shrublands or grasslands. Ponderosa pine (*Pinus ponderosa* (PIPO)) forests occur in montane zones in pure and mixed stands. Douglas-fir (*Pseudotsuga menziesii* (PSME)) often occurs in association with ponderosa pine on north-facing aspects and in relatively mesic sites. Mixed-conifer forests occur at intermediate elevations and include combinations of ponderosa pine, Douglas-fir, piñons, junipers, and firs (*Abies lasiocarpa* (ABLA) or *A. concolor* (ABCO)). Mixed-conifer forests also often occur in association with aspen (*Populus tremuloides* (POTR)). Aspen forms large (>100 ha) pure stands throughout the upper montane and lower subalpine zones across the study area except in the Great Basin. Lodgepole pine (*Pinus contorta* (PICO)) often forms pure stands at mid-elevations (1900 to 2800 m) or occurs in the mixed-conifer zone in northern Utah. Subalpine forests dominated by Engelmann spruce (*Picea engelmannii* (PIEN)) and firs occur at upper elevations (2350 to 3500 m). At the highest forested elevations (generally above 3000 m), pure Engelmann spruce forests occur in mesic sites whereas bristlecone pine (*Pinus longaeva* (PILO)) or limber pine (*P. flexilis* (PIFL)) are typically found in dry or rocky sites.

There was, in general, a gradient in fire frequency across the elevational gradient before fire exclusion that began at all sites in the late 1800s (Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). Fire occurrence was highest in the middle of the elevation range in ponderosa pine and drier mixed-conifer sites. Fire frequency progressively declined both above and below this middle-elevation zone. At upper elevations, generally moist conditions led to high fuel biomass, both living and dead, in many stands, but fewer years in which fuels were dry enough to ignite and spread. At lower elevations in the piñon–juniper woodlands, fuels were often dry enough to burn because of hotter and dryer fire seasons, but because of lower productivity, there were in general less continuous both aerial and surface fuels and fires were not able to spread. In the middle zone, both fuel amounts and moistures were just right (what has come to informally be called the 'Goldilocks effect'), and able to burn often in wide-spreading fires.

Utah forests underwent a period of intensive grazing and land use beginning in the 1850s as a result of Euro-American settlement. Intensive grazing removed understorey species and began alteration of longer-term historical forest dynamics. Logging also affected forest structure in many areas. The tree-ring study found that cessation of historical patterns of fires began in

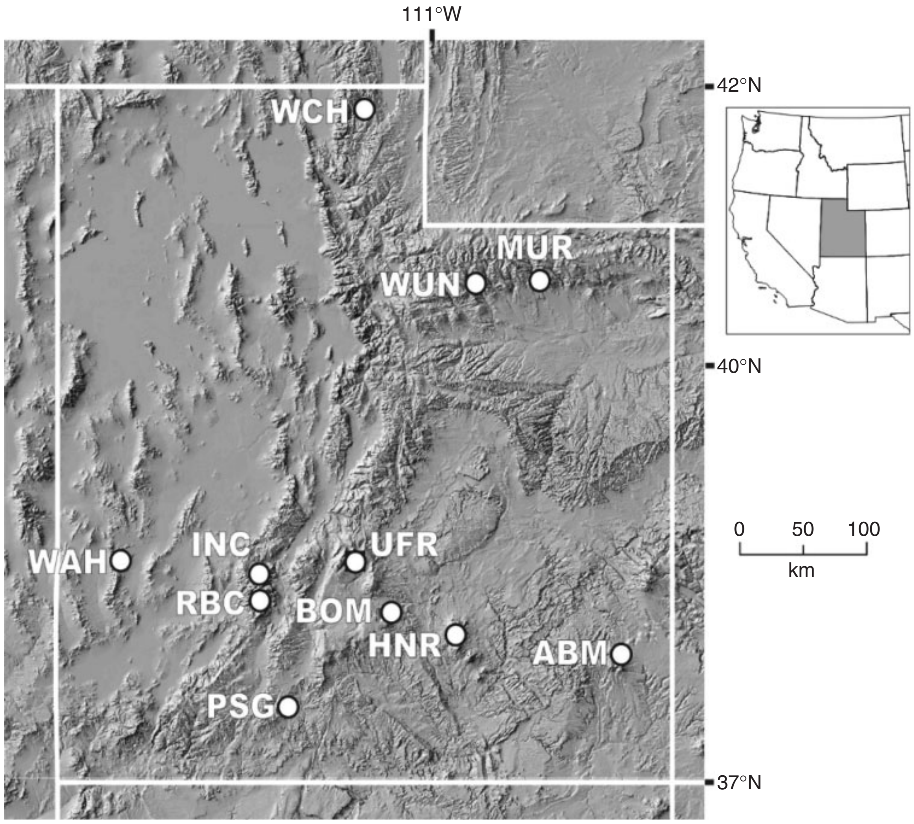

Fig. 1. Locations of tree-ring sites. Three-letter codes correspond to those in Table 1.

Table 1. Tree-ring sites used in the present study arranged from north to south
FRCC (Fire Regime Condition Class) and LANDFIRE BpS (biophysical settings) forest types are listed in Table 4

Site	Minimum elevation (m)	Maximum elevation (m)	Average precipitation (cm)	FRCC and LANDFIRE BpS
Wasatch Mountains (WCH)	2255	2588	100	SPFI, SPDF, CHPI, 10510, 10520, 10500, 10550
Western Uinta (WUN)	2207	3133	60	PPIN, SPDF, SPFI, CHPI, 10540, 10510, 10520, 10500, 10550
Middle Uinta River (MUR)	2308	3250	70	PPIN, SPDF, SPFI, CHPI, 10540, 10510, 10520, 10500, 10550
Wah Wah Mountains (WAH)	2195	2686	40	JUPI, PPIN, SPDF, 10160, 10540, 10500
Upper Fremont River (UFR)	2800	3039	80	SPDF, SPFI, 10510, 10520, 10500
Indian Creek (INC)	2364	2518	65	PPIN, SPDF, 10540, 10500
Beaver Creek (RBC)	2358	3077	90	PPIN, SPDF, SPFI, 10540, 10510, 10520, 10500
Boulder Mountain (BOM)	2405	3377	80	JUPI, PPIN, SPDF, SPFI, 10160, 10540, 10510, 10520, 10500
Henry Mountains (HNR)	2407	3138	60	JUPI, PPIN, SPDF, 10160, 10540, 10500
Abajo Mountains (ABM)	2557	3231	85	JUPI, PPIN, SPDF, SPFI, 10160, 10510, 10520, 10500
Paunsaugunt Plateau (PSG)	2309	2736	45	JUPI, PPIN, SPDF, SPFI, 10160, 10540, 10510, 10520, 10500

the 1860s to 1890s depending on location (Brown *et al*. 2008*a*), similar to patterns seen in forests throughout the western US. Initial reduction in fire frequency was likely the result of grazing that removed grass and herbaceous fuels, followed later by direct fire suppression in the 20th to 21st centuries.

Tree-ring data

The tree-ring study used a systematic sampling design to characterize stand and age structure and fire regimes across forest gradients in each site (Table 1; Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). Similar methods have been used in multiple studies

around the western US and are described in more detail in Heyerdahl *et al*. (2005, 2006), Brown and Wu (2005), Brown (2006), Brown *et al*. (2008*a*, 2008*b*), and Brown and Schoettle (2008). A 500-m grid was established at each site and plots sampled at grid points. Plot centers were located in the field using hand-held global positioning system (GPS) units. An *n*-tree density-adapted sampling method (Jonsson *et al*. 1992) was used to collect data from the nearest ~30 remnant (logs, snags, or stumps) or living trees >20 cm diameter at breast height (DBH) to each plot center. Maximum plot radius was set at 40 m (~0.5 ha) and most plots were ~<0.2 ha in size. For each plot tree, species was recorded and an increment core (on living trees) or cross-section (from logs, snags, and stumps) was collected from ~10 cm height above ground. Sampled cores had to be no more than a field-estimated 10 years from pith to minimize pith offset when assessing pith date. Diameter at sample height (DSH) and DBH were measured on living trees, and DSH was measured or estimated for remnant trees missing bark, sapwood, or heartwood. Distance from plot center and azimuth were measured on all trees for reconstruction of tree basal areas, density, and spatial patterning. To reconstruct surface fire history, cross-sections were cut from any fire-scarred trees found within plots. Additional fire-scarred trees also were sampled within ~80 m of each grid point and between grid points when discovered. GPS coordinates and species of fire-scarred trees outside of plots were recorded.

Standard dendrochronological methods were used to crossdate all samples using locally developed master chronologies (Heyerdahl *et al*. 2005). Pith dates were estimated on cores that did not intersect pith based on the curvature of the innermost rings sampled. The tree recruitment date is considered to be the date of tree pith at 10-cm height. No corrections were made for time to grow from germination to 10 cm sample heights because of the widely varying species and environmental conditions at the sites that were collected for the study. Once crossdating of ring series was completed on all samples, dates for any fire scars seen within the ring series were assigned. Any trees that were not able to be dated were not used in subsequent analyses.

FRCC and LANDFIRE vegetation models

VDDT modeling estimates the relative proportions of non-spatially defined reference conditions that would have occurred under a historical fire regime and an equilibrium (current) climate regime within each BpS (Beukema and Kurz 2003). VDDT input includes average fire frequencies, severities, and other disturbances defined as probabilistic events, and vegetation structural stage development pathways, including changes in species composition and density through a successional sequence. VDDT runs are commonly made for 500 years to allow vegetation conditions to equilibrate over time. VDDT output is proportions of vegetation successional classes – the reference conditions – across a non-spatially referenced landscape at the end of the 500-year model run. Reference conditions for most forest types are summarized into five seral stages that approximate overall developmental characteristics of community age and structure: early-replacement, mid-open, mid-closed, late-open, and late-closed. Each developmental stage represents a successional class defined by average tree age, species composition, structural characteristics, and response to disturbances. LANDFIRE and FRCC assessments use VDDT in a similar manner, but in LANDFIRE, reference condition proportions are then coupled with the spatial model LANDSUM (Keane *et al*. 2002) to map resulting vegetation conditions for each BpS across actual landscapes at a 30-m^2 spatial resolution.

FRCC and LANDFIRE developed their own BpS models using two different vegetation classification systems (Küchler 1964 *v.* Comers *et al*. 2003). Both systems attempt to describe the same historical vegetation using VDDT; however, their models use different probabilities for disturbance, and have somewhat different species distributions and geographic extents (often based on expert opinion; see http://frcc.gov, accessed 19 October 2007; www.landfire.gov for details).

Comparing tree-ring with FRCC and LANDFIRE data

We performed three tests to compare the tree-ring data with FRCC and LANDFIRE vegetation models. First, we compiled age and DBH data derived from the tree-ring study to assess whether FRCC methods of visual estimates of tree diameters accurately represent the age of forest vegetation for defining mid- and late-development classes of reference conditions. FRCC guidebook methods define >23 cm DBH as a visual indicator of a mature tree when conducting field assessments. For this analysis, we assumed that plots with trees averaging ≤23 cm DBH would be considered to be in a mid-development reference condition, and >23 cm would be in late-development. We conducted least-squares linear regressions to estimate fitness of tree age to DBH by species and site. As many of the regression models did not meet the statistical requirements of homoscedasticity, normality, and constant variance in model residuals, a logarithmic transformation was applied to the tree ages before regression. Models that had significant *P* values ($P < 0.05$) were considered to be representative of species growth estimates. We also conducted an analysis of variance (ANOVA) of age and diameter by species and site to both determine the strength of these relationships and how they varied by species and location across the region. All statistical analyses were conducted using the *Statistica* software (StatSoft Inc. 2008). The tree-ring study sampled a total of ~10 000 remnant and living trees; however, we only used data from the living trees for this part of our assessment. Dead trees (stumps, snags, and logs) often were missing bark, sapwood, or portions of the heartwood that reduced confidence in diameter estimates. The DBH-to-age analysis therefore consisted of 5173 living trees from 13 species from the 11 sites.

Our second test was whether VDDT modeled reference conditions captured the range of variation in reference conditions reconstructed by the tree-ring data as of a date of 1880. Dates of initial Euro-American settlement varied across the study area but all sites showed some Euro-American impact by 1880, including cessation of spreading fires in almost all of the sites (Brown *et al*. 2008*a*). As current vegetation may not be representative of past vegetation type, only species present in 1880 and their corresponding ages were used to assign BpS and reference condition to each of a total of 273 plots that were sampled from the 11 sites (Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). Both living and remnant trees were used to estimate the 1880 plot compositions. FRCC and LANDFIRE use key species to

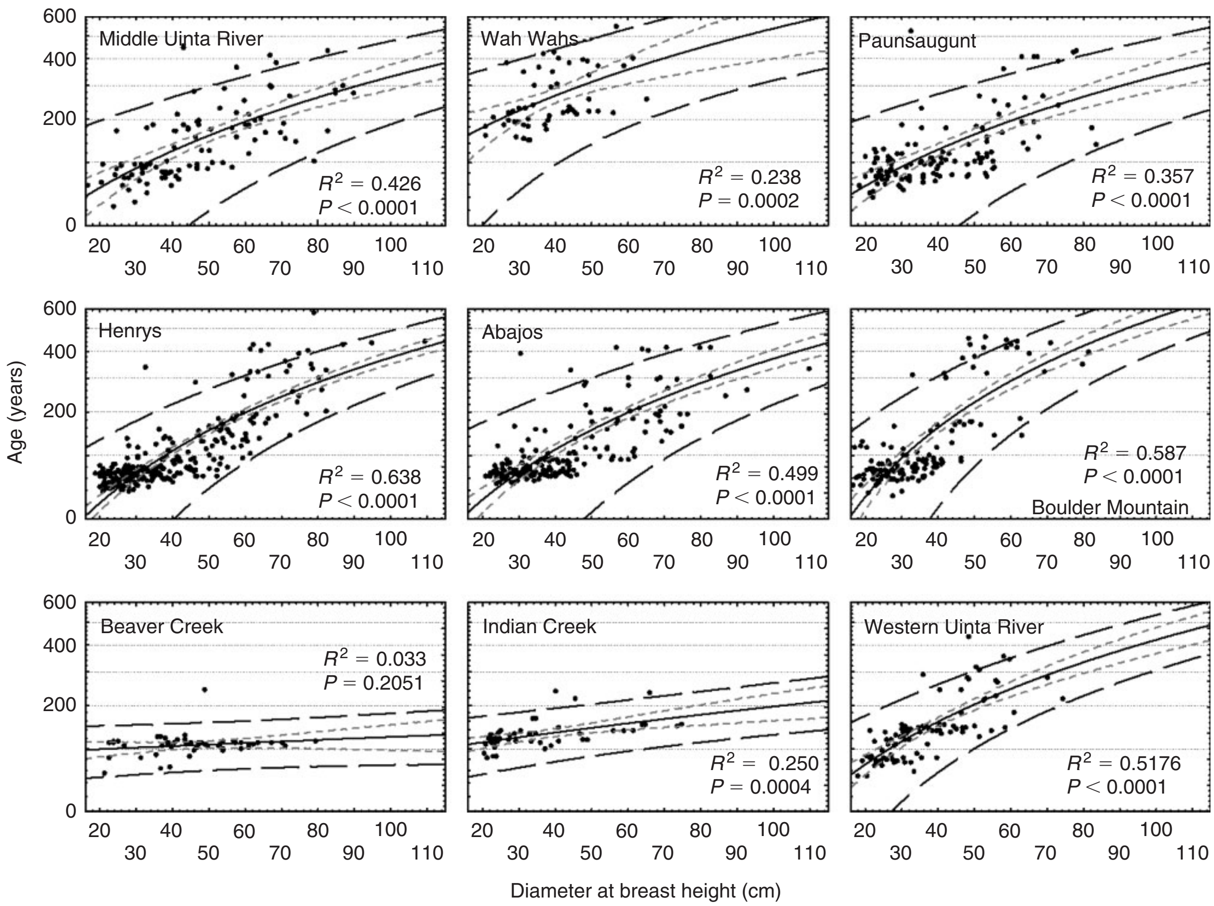

Fig. 2. Diameter at breast height (DBH) and log(age) regressions for ponderosa pine trees by site, with linear fits (solid lines), 95% confidence intervals (gray dashed lines), and 95% prediction intervals (black dashed lines). Overall R^2 for ponderosa pine trees across all sites was 0.44.

define vegetation characteristics when conducting an assessment and we used these species as the basis for assigning BpS and reference condition to each plot.

Historical age class and species composition in 1880 for each plot were compared with FRCC and LANDFIRE reference conditions for selected BpS. FRCC and LANDFIRE BpS descriptions are available on their respective project websites (www.frcc.gov; www.landfire.gov). We did not evaluate the typical five-stage VDDT models because of difficulties in using the tree-ring data to accurately recreate smaller size classes in historical stand densities as a result of probable tree mortality and decay since pre-settlement periods (e.g. Brown and Cook 2006; Brown *et al*. 2008*b*). However, we assume that we are able to define with some confidence mid- and late-development stands based on crossdated ages of trees present in each plot in 1880. The mean age of a 23-cm-DBH live tree varied by species, and we used the tree-ring results to estimate the upper 95% confidence interval for predicted tree size to consider whether a stand was late developmental stage in 1880. We grouped data from open and closed stands together based on age and composition for comparison with succession classes from VDDT output. If any trees in a plot were older than their predicted age-to-size confidence interval, the plot was considered to be in late-development in 1880. If there were no older trees during the historical period, then the plot was considered to have been in mid-development. If there were no trees during the historical period, the plot was considered to have no data and not used in this analysis. Once plots were categorized by BpS and reference condition, they were compared with FRCC and LANDFIRE BpS model proportions of mid- and late-development vegetation based on VDDT output. We used a Chi-square test to determine if the observed tree-ring reference condition proportions were significantly different than the expected based on the VDDT output.

Finally, we compared tree-ring plot data with LANDFIRE BpS and EVT map layers produced by the LANDFIRE project. LANDFIRE data are spatially mapped, which provided a unique opportunity to evaluate vegetation models at a high spatial resolution through comparison with the mapped tree-ring data. Plots were first located through their GPS coordinates relative to LANDFIRE map data. The BpS assignments we made for each plot in 1880 were then compared with LANDFIRE BpS map data. We also compiled the living tree composition in each plot and compared that with the LANDFIRE EVT map data. If key

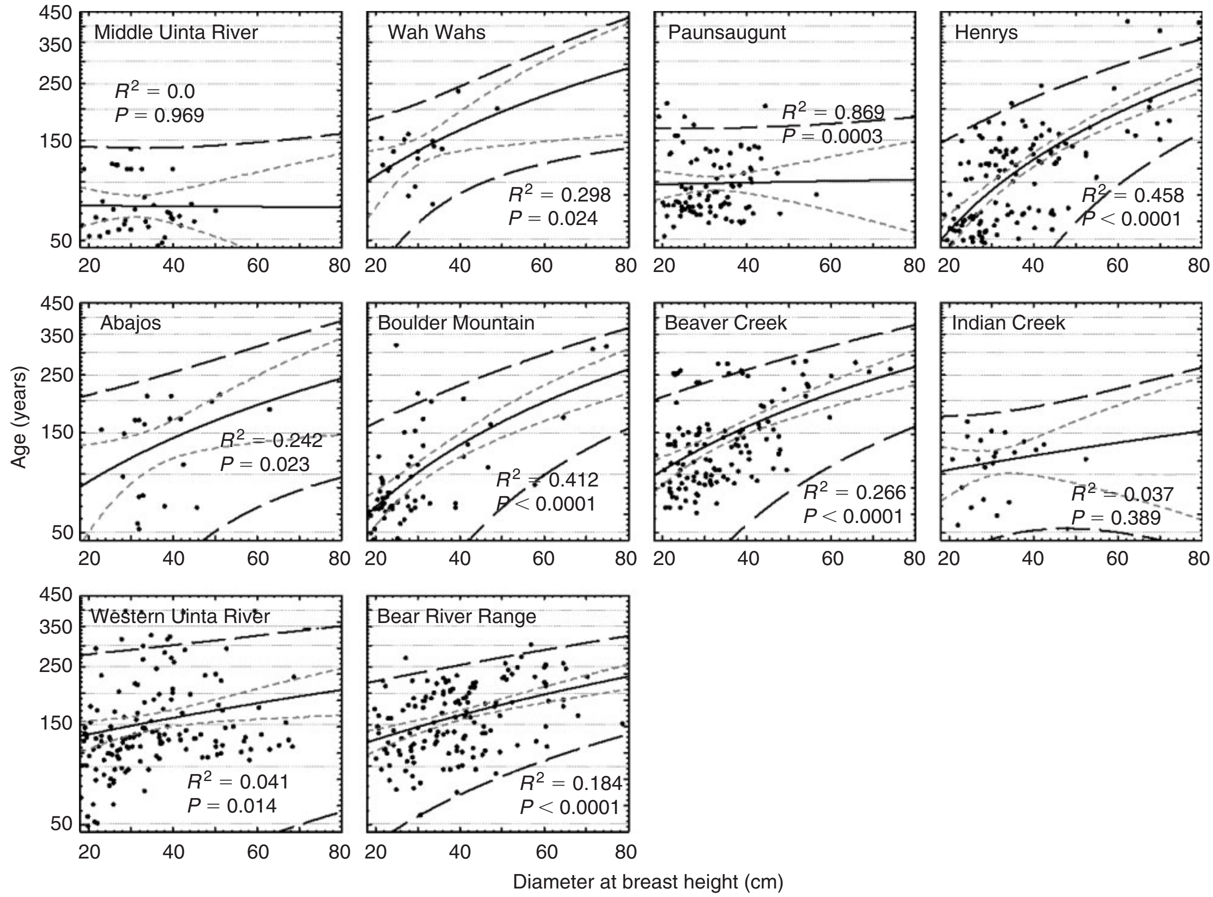

Fig. 3. Diameter at breast height (DBH) and log(age) regressions for Douglas-fir trees by site, with linear fits (solid lines), 95% confidence intervals (gray dashed lines), and 95% prediction intervals (black dashed lines). Overall R^2 for Douglas-fir trees across all sites was 0.21.

species were present in the tree-ring data in comparison with the mapped BpS or EVT, then the grid point was considered to have been accurately mapped in LANDFIRE.

Results

Age–diameter relationships

DBH and tree ages exhibited generally broad relationships, both within species and among sites (Figs 2–4; Tables 2, 3). Ponderosa pine was the only species where age and size were strongly correlated using data from all sites ($R^2 = 0.438$, $P < 0.001$) and were strongly correlated over most of the individual sites (Table 2). There were outliers for most species by DBH and age; however, their deviance did not significantly change the results. Median tree age was predicted for trees at 23 cm using an inverse prediction with 95% confidence interval (Table 3). ANOVA results indicate that species associated with infrequent fire regimes (piñon–juniper, spruce–fir, and bristlecone pine; Heyerdahl *et al*. 2005) were found to have greater average ages than frequent fire species (especially ponderosa pine and Douglas-fir; Fig. 5). Variance of diameters relative to ages for species that contained a large sample n, such as Douglas-fir (PSME), ponderosa pine (PIPO), and Engelmann spruce (PIEN) was small. There was greater variance found in species that had fewer sampled trees and plots, such as bristlecone pine (PILO), Rocky Mountain juniper (JUSC), one-seed juniper (JUOS), limber pine (PIFL), and single leaf piñon (PIMO), but this result is likely an artifact of the smaller number of trees used in each regression. ANOVA indicated that DBH and age estimates for all sites were similar with the exception of WAH (Fig. 5). This may be explained by the large presence of fire-infrequent and older species (bristlecone pine, Rocky Mountain juniper, and one-seed juniper) that were sampled in that site.

FRCC and LANDFIRE BpS models

Median ages of trees >23 cm DBH were used to define the proportions of mid- and late-development reference conditions for trees present in plots in 1880 (Table 3). Reference condition proportions reconstructed from the tree-ring data compared favorably with FRCC BpS models for ponderosa pine (PPIN5), mixed-conifer (SPDF), and lodgepole (CHPI) but not for piñon–juniper (JUPI1, JUPI2), south-western mixed-conifer (MCAN) and spruce–fir (SPFI2, SPFI7; Table 4, Fig. 6). Reference condition proportions reconstructed from the tree-ring data

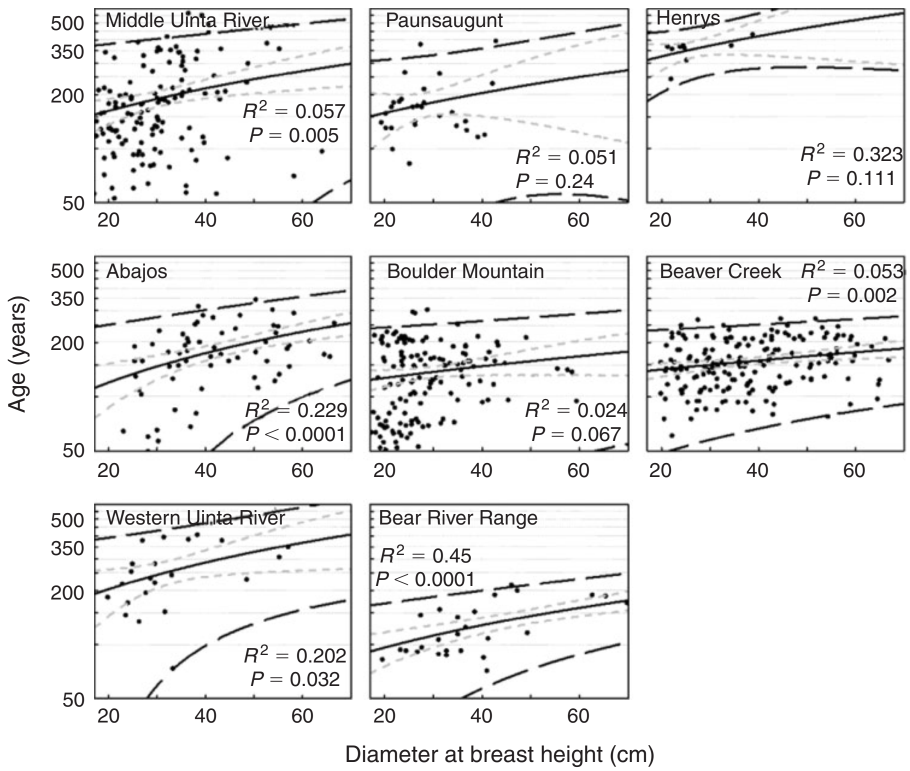

Fig. 4. Diameter at breast height (DBH) and log(age) regressions for Engelmann spruce trees by site, with linear fits (solid lines), 95% confidence intervals (gray dashed lines), and 95% prediction intervals (black dashed lines). Overall R^2 for Engelmann spruce trees across all sites was 0.06.

Table 2. Observed two-sided *P* values for DBH–age regressions for all species at all sites

Bold values represent locations where *P* values are significant at the 95% confidence interval (<0.05) based on sample size (>10 trees)

Species	Site									
	WCH	RBC	ABM	BOM	HNR	PSG	INC	WUN	MUR	WAH
PIPO		0.021	**<0.0001**	**<0.0001**	**<0.0001**	**<0.0001**	**0.0004**	**<0.0001**	**<0.0001**	**0.0002**
PSME	**<0.0001**	**<0.0001**	**0.0234**	**<0.0001**	**<0.0001**	0.88	0.39	**0.01**	0.969	0.024
PIEN	**<0.0001**	**<0.0001**	**<0.0001**	0.066	0.111	0.241		0.03	0.005	
ABLA	**<0.0001**	0.19	**<0.0001**		0.147			0.37		
POTR	**0.01**	0.01	**<0.0001**	0.63	0.107		0.40	0.81	**0.020**	
ABCO		0.22				**<0.0001**	0.22		0.069	0.002
PICO	**<0.0001**				**<0.0001**	**0.0007**				
PIFL	0.28				**<0.0001**	0.090	0.28			
PIED				**<0.0001**	**<0.0001**	0.025				
PIMO										**<0.0001**
JUSC				0.152		**0.111**				0.903
JUOS				**0.0003**		0.677			0.797	**0.0002**
PILO										0.574

compared favorably with LANDFIRE BpS models for Rocky Mountain dry–mesic montane mixed-conifer (10510), aspen and aspen–mixed-conifer low- and high-elevation forests (10110, 10611, 10612), but not for piñon–juniper (10160), ponderosa pine (10540), Rocky Mountain mesic montane mixed-conifer (10520), Rocky Mountain subalpine dry–mesic spruce–fir forest and woodland (10550), and Rocky Mountain lodgepole pine (10500; Table 4, Fig. 6). The JUPI1 BpS model (Table 4) was the most different from the tree-ring data, although the JUPI2 model had a similar trend of a larger proportion of late-successional

stands in comparison with the tree-ring data (Fig. 6). Spruce–fir and lodgepole pine data both showed low correspondence with VDDT model results, including opposite trends of more older than younger stands in the tree-ring data in contrast to the VDDT modeled reference conditions (Fig. 6).

Table 3. Expected median ages of trees >23 cm DBH (diameter at breast height) by species, with lower and upper 95% confidence intervals derived from tree-ring data

NS, age–DBH regression not significant

Species	Age (years) at 23 cm	R^2	*P* value
PIPO	40.9 ± 3.2	0.438	<0.0001
JUOS	114.9 ± 41.9	0.438	<0.0001
PIED	135.3 ± 21.9	0.28	<0.0001
PIFL	66 ± 11.4	0.271	<0.0001
PIMO	176.3 ± 29.8	0.231	<0.0001
PSME	42.9 ± 6	0.213	<0.0001
PICO	54.3 ± 12.6	0.112	<0.0001
POTR	104 ± 9.1	0.095	<0.0001
PIEN	24.7 ± 14.7	0.055	<0.0001
JUSC	NS	0.05	0.0961
ABCO	14.8 ± 14.4	0.023	<0.0001
PILO	NS	0.012	0.6295
ABLA	50.2 ± 10.2	0.01	<0.0001

LANDFIRE map data

LANDFIRE map layers were found to be overall ~58% accurate for BpS and 60% accurate for EVT when compared with the tree-ring data for each plot (Table 5). LANDFIRE maps were 38% accurate for both BpS and EVT, 28% accurate for at least one type (17% EVT accurate and BpS inaccurate, with 11% BpS accurate and EVT inaccurate), and 34% inaccurate. Mixed-conifer and spruce–fir types had the highest accuracies by BpS for LANDFIRE with accuracies ranging from 64 to 82% for BpS and 67 to 79% for EVT. Piñon–juniper was the least accurately mapped BpS and EVT with 13 and 37% accuracy respectively.

Discussion

FRCC and LANDFIRE BpS models

Current stand conditions are determined through visual estimates of stand structure, including tree diameters, in FRCC assessments (Hann *et al*. 2004). FRCC assessments are designed to be a relatively rapid method of characterizing current vegetation and fire regime departures from historical conditions. The expense of collecting field data, such as canopy closure, canopy base height, tree density, stand age structure, and fire and stand histories, make field sampling impractical for FRCC assessments. However, based on the limited findings of this study, it appears that FRCC methods may result in inaccurate measures of plant community departure based on visually estimated

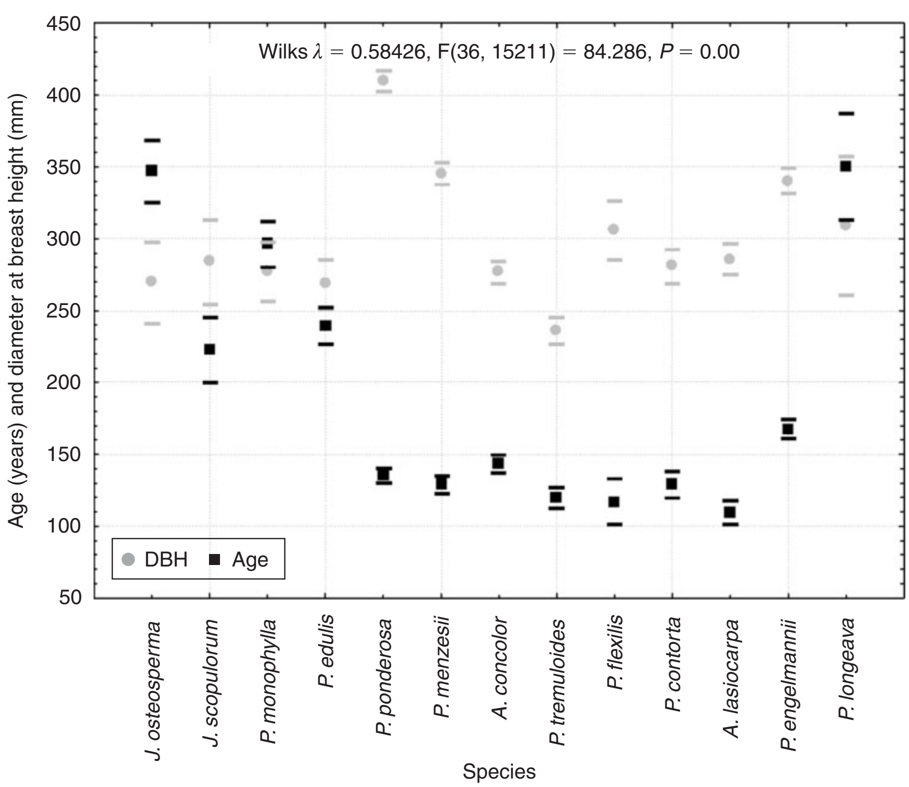

Fig. 5. ANOVA of age and diameter at breast height (DBH) by species and site. Horizontal bars represent 95% confidence intervals.

Table 4. Observed proportions of mid- and late-development reference conditions reconstructed from tree-ring data in plots collected in Utah, compared with FRCC (Fire Regime Condition Class) BpS (biophysical settings) model output for mid- and late-development reference conditions from Hann *et al*. (2004) and LANDFIRE

MFI, mean fire interval (years); *n*, number of plots in the observed data in mid- or late-seral stages. Chi-square fit is for the observed plots *v.* BpS models with 1 degree of freedom, $P = 0.05$ significance for types <3.84. BpS that meet the range of variability in the observed data are highlighted in bold

BpS description *v.* observed data	FRCC code	LANDFIRE code	Mid (%)	Late (%)	MFI	*n*		Chi-square	*P* value
						Mid	Late		
Observed PIED, PIMO, JUSC, JUPI			4	96		1	24		
Piñon–juniper infrequent fire	JUPI2		30	70	435			12.99	0.0003
Piñon–juniper frequent fire	JUPI1		50	50	31			21.16	<0.0001
Colorado Plateau piñon–juniper woodland		10160	55	45	128			26.273	<0.0001
Observed PIPO			26	74		13	37		
Colorado plateau ponderosa	**PPIN5**		25	75	6			**0.027**	0.87
Southern Rocky Mountain ponderosa pine woodland		10540	44	56	15			6.575	0.01
Observed PSME, ABCO, PIPO, PIEN			48	52		35	38		
South-western mixed-conifer	MCAN		35	65	10			5.377	0.02
Rocky Mountain dry–mesic montane mixed-conifer forest and woodland		**10510**	40	60	10			**1.92**	0.166
RM mesic montane mixed-conifer forest and woodland		10520	75	25	33			28.498	<0.0001
Spruce–fir–Douglas-fir[A]	**SPDF**		58	42	19			**0.658**	0.417
Observed PICO			36	64		4	7		
Lodgepole pine	CHPI		65	35	125			3.965	0.046
Rocky Mountain lodgepole pine forest		10500	100	0	124			4.455	0.035
Observed PIEN, ABLA, ABCO			26	74		16	58		
RM subalpine dry–mesic spruce–fir forest and woodland		10550	65	35	212			61.206	<0.0001
Lower subalpine forest	SPFI7		80	20	91			157.622	<0.0001
Upper subalpine forest	SPFI2		70	30	143			82.474	<0.0001
Observed POTR			90	10		33	4		
Deciduous woodland–oak or aspen	DWOA		55	45	100			17.474	<0.0001
Intermountain basins aspen–mixed-conifer forest – low		**10611**	85	15	10			**0.509**	0.475
Intermountain basins aspen–mixed-conifer forest – high		**10612**	95	5	32			**2.63**	0.105
Rocky Mountain aspen forest and woodland		**10110**	80	20	27			**2.146**	0.142
PILO			0	100		0	3		

[A] Includes observed POTR plots.

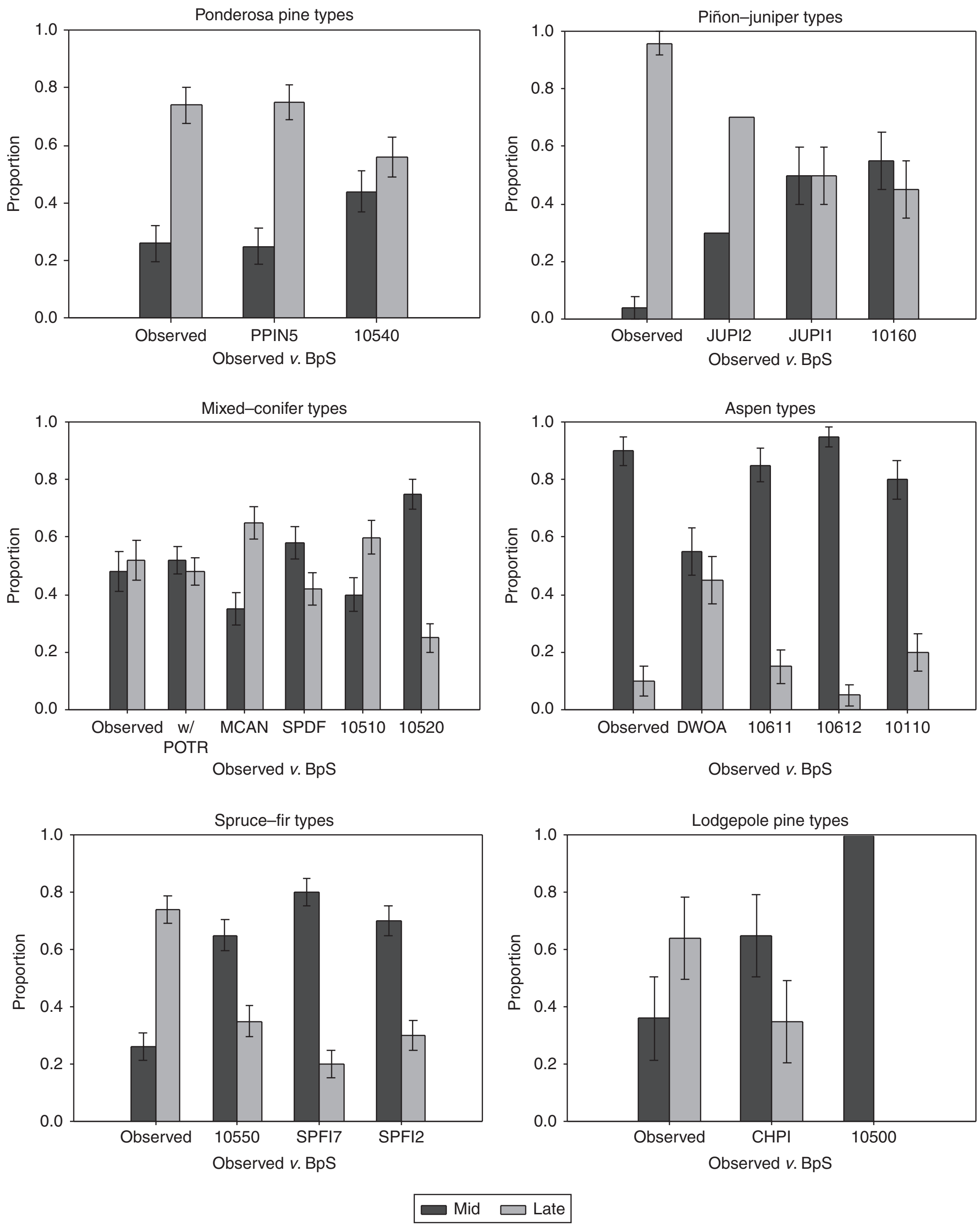

Fig. 6. Proportion of plots observed in the tree-ring data compared to FRCC (Fire Regime Condition Class) and LANDFIRE modeled reference condition proportions. Error bars were generated by calculating the 95% confidence interval from sample variance and standard error of observed points. Tree-ring results are on the left (e.g. observed), FRCC and LANDFIRE models are listed by their four-letter abbreviations on the right (e.g. PPIN5, 10540, etc.).

age–diameter relationships for determining reference condition proportions. Variations in age–size relationships both within species and among sites (Figs 2–5) may limit the ability to accurately gauge departure from estimated historical composition based on VDDT model results. Generally poor relationships between size and age may result in misassignment of current reference condition proportions based only on visual estimates, which may in turn lead to misassignment of the FRCC index.

Better correspondence between the tree-ring data and some BpS models indicates that VDDT models more accurately reflect historical forest structure in frequent-fire forest types such as ponderosa pine, mixed-conifer and aspen, than in

Table 5. LANDFIRE accuracy by BpS (biophysical settings) and EVT (existing vegetation type)

Code is the LANDFIRE map code for BpS or EVT type, *n* is number of plots tested, and % is percentage that were accurately mapped based on tree-ring data at plot scale

	Code	*n*	%
BpS			
Rocky Mountain aspen forest and woodland	10110	31	32
Colorado Plateau piñon–juniper woodland	10160	29	14
Rocky Mountain lodgepole pine forest	10500	7	43
Rocky Mountain dry–mesic montane mixed-conifer forest and woodland	10510	6	33
Rocky Mountain mesic montane mixed-conifer forest and woodland	10520	11	64
Southern Rocky Mountain ponderosa pine woodland	10540	19	53
Rocky Mountain subalpine dry–mesic spruce–fir forest and woodland	10550	82	66
Intermountain basins aspen–mixed-conifer forest – low elevation	10611	22	82
Intermountain basins aspen–mixed-conifer forest – high elevation	10612	31	77
Intermountain basins mountain mahogany woodland and shrubland	10620	6	50
EVT			
Rocky Mountain aspen forest and woodland	2011	26	50
Colorado Plateau piñon–juniper woodland and shrubland	2016	43	37
Rocky Mountain lodgepole pine forest	2050	19	63
Rocky Mountain montane mesic mixed-conifer forest and woodland	2052	9	78
Southern Rocky Mountain ponderosa pine woodland	2054	24	46
Rocky Mountain subalpine dry–mesic spruce–fir forest and woodland	2055	53	79
Intermountain basins aspen–mixed-conifer forest and woodland	2061	64	67
Abies concolor forest alliance	2208	14	71

infrequent-fire types such as spruce–fir and piñon–juniper (Fig. 6). BpS reference condition models were determined by managers and scientists familiar with the local ecology of each region during regional workshops. BpS types that are considered to be representative of each region were identified and described based on available historical and ecological data. Some BpS types, such as ponderosa pine and dry mixed-conifer forests, have extensive fire and forest history data with which to parameterize VDDT model runs. Other BpS types are less well studied and their fire and vegetation histories less certain, especially across the range of environmental and community variation within and between regions. The better correspondence between modeled and reconstructed reference conditions in frequent-fire-type models (ponderosa pine and mixed-conifer forest types; Fig. 6) is likely related to the greater amount of fire and forest history research that has been conducted in these forest types. Conversely, fire-infrequent types (spruce, and piñon–juniper woodland types; Fig. 6) have had less fire history research conducted, with the result that their fire regimes and successional patterns are less well documented for input to VDDT modeling. Furthermore, infrequent-fire types generally have fewer observations of historical fires and forest successional changes available for adequate characterization of fire regime parameters for VDDT modeling (e.g. Brown *et al*. 2008*a*).

Another factor that undoubtedly results in varying model and data results is that individual-site fire histories often have experienced contingent historical events that lead to differences from a ‘typical’ or average fire regime of a particular forest type. Stochastic modeling in FRCC and LANDFIRE generalizes vegetation and its fire regimes into generic types and does not take into account site-specific variability or, more importantly, the history of climate variations or other disturbances that may have affected changes in community structure through time. Variations in site histories undoubtedly contribute to variations in ratios of actual from modeled reference conditions. For example, spruce–fir and lodgepole pine FRCC and LANDFIRE BpS models predict more mid- than late-development stands, but the Utah tree-ring data found the opposite (Fig. 6). This may be due to longer fire intervals in this region than in other areas, leading to generally older stands across landscapes. Many spruce trees found in the tree-ring study were >300 years old at the time of sampling and probably resulted from fires that occurred in the late 1600s, most commonly in 1685 (Heyerdahl *et al*. 2005). However, the current presence of older rather than younger stands does not mean that these forests are outside their historical ranges of variability in either their fire regime or forest structure, but rather that they have not had extensive fires in the intervening period that would have resulted in a larger proportion of mid-successional stands as suggested should be present based on VDDT model results. Without taking into account this history of the forest landscapes, the VDDT models suggest that there is current departure in the landscape proportions of reference conditions in Utah spruce–fir and lodgepole pine forests.

Taking into account differences in fire histories, the trend of model results toward older or younger successional classes in each BpS may be more important to consider in FRCC assessments rather than the absolute proportions of stand structures. This may provide a more realistic perspective for assessing whether a particular BpS should be considered as inside or outside of its historical range of variation. For example, the tree-ring fire data for piñon–juniper (P-J) woodlands show the majority of stands are currently in late-development structural stages (Fig. 6). The FRCC BpS model JUPI2 (Table 4) also predicts more late-development trees than younger, but underpredicts what was found in the tree-ring data. The sensitivity of the VDDT model to fire frequency is critical to the setting of reference conditions. The model inaccuracy may be due to the model’s fire

return interval, currently predicted to be ~450 years. If the interval is increased (~1000 years), the model begins to more closely reflect the tree-ring results. A recent assessment of (P-J) ecosystems in the western US concluded that fire was only a minor disturbance in many less productive stands because of lack of both surface and crown fuels with which to carry fire (Romme *et al*. 2009). We believe that many of the Utah stands sampled probably fell into this category of fire regime historically, which means that if the longer intervals had been used in VDDT modeling, the reference conditions would likely be closer to what was found in the tree-ring data. The error may also be due to the definition of a mid-development stand in terms of the age; the mean ages of sampled piñon and juniper were among the highest in the tree-ring study. The mid-definition could be changed for P-J to an older age class by species to define the mid- from late-successional classes in the reference conditions.

Good correspondence between the tree-ring data and models for ponderosa pine (PPIN5), aspen (10110), and mixed-conifer (SPDF, 10510; Fig. 6) suggests that the reference conditions for these BpS were accurately modeled by VDDT parameters, at least in the Utah study sites. However, results of this study suggest that inaccuracy in piñon–juniper and subalpine types makes any decision based on a VDDT output possibly subject to error. For BpS types in which disturbance may not be the major or only factor in tree recruitment, VDDT models may need further evaluation. Additional empirical disturbance and forest history sampling in piñon–juniper, spruce–fir, and lodgepole pine types should increase the available information about these systems to use in VDDT modeling. However, because of generally longer fire intervals in these forests, any departure from historical to present conditions may be less than in frequent-fire BpS such as ponderosa pine and mixed-conifer forests. A possible result of inaccurate estimations of departure and wrong FRCC classification may be the application of incorrect management actions that could lead to even further departure from historical conditions (see also Romme *et al*. 2009).

The only accurate way to establish the age of a stand is to physically sample the trees for ages. We suggest based on the results of our comparison that at least some limited age sampling is needed for FRCC assessments. This sampling probably should include removing cores from the field and crossdating by trained dendrochronologists to most accurately characterize age and successional status of stands. Additional field-sampled fire history and stand establishment data, especially in the less-well-studied ecosystems, should further increase the accuracy of VDDT models through better dynamic estimations of age structures and relationships with fire regimes. However, we also realize that this type of sampling is expensive and – perhaps more critically to the efficient use of FRCC in forest management decisions – more time-consuming than FRCC visual assessment methods as currently practiced. Nevertheless, we suggest that some sort of compromise solution could be found that would provide both the most accurate as well as timely data possible for FRCC assessment needs.

LANDFIRE maps

Zhu *et al*. (2006) used a cross-validation technique to determine that existing vegetation data layer accuracies are between 60 and 89% in LANDFIRE maps. Our study's comparison of LANDFIRE and tree-ring data falls on the lower end of the estimate of Zhu *et al*. (2006) (Table 4). When broken down by BpS and EVT, some types are more accurately represented in LANDFIRE data than others. EVT mapping in LANDFIRE is most accurate for the mixed-conifer and spruce–fir types. These forests generally have the densest and most continuous canopies, and may have been easiest to identify through remote sensing methods because of their continuous canopies and more distinctive NDVI reflectance in Landsat spectral bands (Zhu *et al*. 2006). Conversely, sparser canopy cover may have led to lower accuracy in other types such as piñon–juniper, which is similar to what Zhu *et al*. (2006) found. It should be noted, however, that piñon–juniper plots sampled for the tree-ring study were generally found in ecotonal areas near lower ends of study sites, and may not be wholly representative of piñon–juniper BpS as defined in the LANDFIRE mapping effort.

Conclusion

Historical forest conditions reconstructed from tree-ring data provide opportunities for comparison with FRCC and LANDFIRE modeled vegetation data across multiple forest types. The tree-ring reconstructions we examined suggest that reference conditions are better modeled in frequent-fire forest types but not necessarily in infrequent-fire forest types, at least in Utah forests. Additional studies in fire-infrequent forest types should increase understanding of historical stand compositions, fire histories, and other disturbances with which to better parameterize VDDT reference condition models. The greatest amount of fire history research has been conducted in ponderosa pine and mixed-conifer forests, which likely contributed to the better correspondence between tree-ring data and VDDT model results that we found in this study. We consider this study as only a first step in comparison of empirical vegetation data with vegetation models used in both FRCC assessments and the nationwide LANDFIRE mapping effort. Tree-ring data provide an opportunity to compare site-specific vegetation patterns and fire regime variations that are often not easily accounted for in modeling efforts. Revised methods for assessing FRCC may need to take into greater account both tree ages and stand histories to more accurately compare with model results. We also suggest that ranges of reference conditions be incorporated into the BpS classifications to better take into account fire and forest histories rather than trying to establish average conditions that must be met for a FRCC index to be assigned.

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Land Use Planning and Wildfire: Development Policies Influence Future Probability of Housing Loss

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Abstract

Increasing numbers of homes are being destroyed by wildfire in the wildland-urban interface. With projections of climate change and housing growth potentially exacerbating the threat of wildfire to homes and property, effective fire-risk reduction alternatives are needed as part of a comprehensive fire management plan. Land use planning represents a shift in traditional thinking from trying to eliminate wildfires, or even increasing resilience to them, toward avoiding exposure to them through the informed placement of new residential structures. For land use planning to be effective, it needs to be based on solid understanding of where and how to locate and arrange new homes. We simulated three scenarios of future residential development and projected landscape-level wildfire risk to residential structures in a rapidly urbanizing, fire-prone region in southern California. We based all future development on an econometric subdivision model, but we varied the emphasis of subdivision decision-making based on three broad and common growth types: infill, expansion, and leapfrog. Simulation results showed that decision-making based on these growth types, when applied locally for subdivision of individual parcels, produced substantial landscape-level differences in pattern, location, and extent of development. These differences in development, in turn, affected the area and proportion of structures at risk from burning in wildfires. Scenarios with lower housing density and larger numbers of small, isolated clusters of development, i.e., resulting from leapfrog development, were generally predicted to have the highest predicted fire risk to the largest proportion of structures in the study area, and infill development was predicted to have the lowest risk. These results suggest that land use planning should be considered an important component to fire risk management and that consistently applied policies based on residential pattern may provide substantial benefits for future risk reduction.

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Introduction

The recognition that homes are vulnerable to wildfire in the wildland-urban interface (WUI) has been established for decades [e.g., 1,2]; but with a recent surge in structures burning, this issue is now receiving widespread attention in policy, the media, and the scientific literature. Single fire events, like those in Greece, Australia, southern California, and Colorado have resulted in scores of lost lives, thousands of structures burned, and billions of dollars in expenditures [3–6]. With the potential for increasingly severe fire conditions under climate change [7] and projections of continued housing development [8], it is becoming clear that more effective fire-risk reduction solutions are needed. "Fire risk" here refers to the probability of a structure burning in a wildfire within a given time period.

Traditional fire-risk reduction focuses heavily on fire suppression and manipulation of wildland vegetation to reduce hazardous fuels [9]. Enormous resources are invested in vegetation management [10], but as increasing numbers of homes burn down despite this massive investment, the "business-as-usual" approach to fire management is undergoing reevaluation. One issue is that fuel treatments may not be located in the most strategic positions, i.e., in the wildland-urban interface [11]. Yet, even if treatments surrounded all communities, scattered development patterns are difficult for firefighters to reach [12–14], and fuel treatments do little to protect homes without firefighter access [15–16]. Fuel treatments may also be ineffective against embers or flaming materials that blow ahead of the fire front [17].

One alternative to traditional fire management that is receiving widespread attention is to prepare communities through the use of fire-safe building materials or creating defensible space around structures [17–18]. These actions represent an important shift in emphasis from trying to prevent wildfires in fire-prone areas to better anticipating fires that are ultimately inevitable. Nevertheless, the cost of building and retrofitting homes to be fire-safe can be prohibitive, and these actions do not guarantee immunity from fire [19].

Land use planning is an alternative that represents a further shift in thinking, beyond the preparation of communities to withstand an inevitable fire, to preventing new residential structures from being exposed to fire in the first place. The reason homes are vulnerable to fires at the wildland-urban interface is a function of its very definition: "where homes meet or intermingle with wildland vegetation" [20]. In other words, the location and

pattern of homes influence their fire risk, and past land-use decision-making has allowed homes to be constructed in highly flammable areas [21]. Land use planning for fire safety is beginning to receive some attention in the literature [22–23], and there is growing recognition of the potential benefits of directing development outside of the most hazardous locations [8,19,24].

Despite recent attention in the literature, land use planning for wildfire has yet to gain traction in practice, particularly in the United States. However, fire history has been used to help define land zoning for fire planning in Italy [22], and bushfire hazard maps are integrated into planning policy in Victoria, Australia [25]. Although some inertia inevitably arises from complications with existing policy and plans, a primary impediment to the design and implementation of fire-smart land use planning is lack of guidance about specific locations, patterns of development, or appropriate methodology to direct the placement of new development. Without a solid knowledge base to draw from, planners will be misinformed about which planning decisions may result in the greatest overall reduction of residential landscape risk. Even worse, poor science could result in placement of homes in areas that actually have high fire hazard.

Research on how planning decisions contributed to structures burning in the past provides some guidance about what actions may work in the future. Analysis of hundreds of homes that burned in southern California the last decade showed that housing arrangement and location strongly influence fire risk, particularly through housing density and spacing, location along the perimeter of development, slope, and fire history [26]. Although high-density structure-to-structure loss can occur [27–28], structures in areas with low- to intermediate- housing density were most likely to burn, potentially due to intermingling with wildland vegetation or difficulty of firefighter access. Fire frequency also tends to be highest at low to intermediate housing density, at least in regions where humans are the primary cause of ignitions [29–30].

These results suggest, for example, that placing new residential development within the boundaries of existing high-density developments or in areas of low relief may reduce fire risk. However, it is difficult to know whether broad-scale planning policies would actually result in the intended housing arrangement and pattern at the landscape scale, and whether those patterns would result in lower fire risk. Our objective here was to simulate three scenarios of future residential development, and to project wildfire risk, in a rapidly urbanizing and fire-prone region where we have studied past structure loss [25]. We based all future development on an econometric subdivision model, but we varied the emphasis of subdivision decision-making based on three broad and common growth types.

Although cities vary in extent, fragmentation, and residential density [31–32], urban form typically adheres to a set of common patterns [33–34], and we based our development scenarios on the three primary means by which residential development typically occurs: infill, expansion, or leapfrog [34]. Infill is characterized by development of vacant land surrounded by existing development, typically in built-up areas where public facilities already exist. [35–36], and should result in higher structure density rather than increased urban extent. Expansion growth occurs along the edge of existing development, extends the size of the urban patch to which it is adjacent, and may have variable influence on structure density. Leapfrog growth occurs when development occurs beyond existing urban areas such that the new structure is surrounded by undeveloped land. This type of growth would expand the urban extent and initially result in lower structure density; but these areas may eventually become centers of growth from which infill or expansion can occur. We asked:

1) Do residential development policies reflecting broad growth types affect the resulting pattern and footprint of development across the landscape?
2) Do differences in extent, location, and pattern of residential development translate into differences in wildfire risk, based on the current configuration of structures?
3) Which development process, infill, expansion, or leapfrog, results in the lowest projected fire risk across the landscape?

Methods

Study Area

The study area included all land within the South Coast Ecoregion of San Diego County, California, US, encompassing an area of 8312 km^2. The region is topographically diverse with high levels of biodiversity, and urban development has been the primary cause of natural habitat loss and species extinction [37]. Owing to the Mediterranean climate, with mild, wet winters and long summer droughts, the native shrublands dominating the landscape are extremely fire-prone. San Diego County was the site of major wildfire losses in 2003 and 2007 [38], although large wildfire events have occurred in the county since record-keeping began, and are expected to continue, as fire frequency has steadily increased in recent decades [29,39]. The county is home to more than three million residents, and approximately one million more people are expected by 2030 [40]. Although most residential development has been concentrated along the coast, expansion of housing is expected in the eastern, unincorporated part of the county.

Econometric Subdivision Model

A host of alternative modeling approaches exist to simulate future land use scenarios [41], including a cellular automaton model that we previously applied to the study area [42]. We chose to use an econometric modelling approach for this study because we wanted to capture fine-scale, structure-level patterns and processes that are correlated with housing loss to wildfire [26]; and econometric models may perform better at the scale of individual parcels [43].

Although we based the three development scenarios on generalized planning policies, we also wanted to ensure that the residential projections were realistic and adhered to current planning regulations. The objective of the econometric modeling was to estimate the likelihood that residential parcels will subdivide in the future. Therefore, we used a probit model to estimate the transition probability of each parcel based on a range of potential explanatory variables typically associated with parcel subdivision and housing development [44–45].

To develop the model of subdivision probability, we acquired GIS data of the county's parcel boundaries in years 2005 and 2009 from the San Diego Association of Governments (SANDAG). The dependent variable was equal to 1 if a parcel subdivided between 2005 and 2009, and zero otherwise. Using these data layers we first determined which parcels were legally able to subdivide given current land use regulations. Minimum lot size restrictions are typically considered the most import restriction for determining future land use. We deemed a parcel eligible for subdivision if the current lot size was greater than twice the minimum legal size given the land class. To determine which parcels subdivided between 2005 and 2009, we queried parcel IDs where the total

area was reduced by at least the minimum lot size between the two time periods. Finally, we were able to generate a suite of variables that determine the likelihood of a parcel developing in the future (Table S1).

We overlaid the parcel boundaries over a range of GIS layers representing our explanatory variables. These data are available to download at (http://www.sandag.org/index.asp?subclassid = 100&fuseaction = home.subclasshome). Our explanatory variables included: parcel size, parcel size squared, six dummy variables which capture non-linear effects of parcel size, distance to the coast, distance to the coast squared; distance to city center and its square, current zoning, slope, land use, roads, if the parcel is in a protected area, if the parcel is in a development area, if the parcel is in the redevelopment area (Table 1).

Spatial Model of Future Development under Planning Alternatives

The outcome of the land use change econometric model is the subdivision probability for each parcel for a five-year time step. Based on these probabilities, we developed a GIS spatial simulation model of future land use under three distinct planning scenarios: infill (development in open or low density parcels within already developed areas), expansion (development on the fringe of developed areas), and leapfrog (development in open areas). The model runs in four 5-year time steps from 2010 to 2030, and generates the spatial locations of new housing units in the county.

Although development decisions could feasibly depend on fire risk, we did not model that here. There is no evidence that fire has influenced past regional planning decisions, so it was not used as an explanatory variable in the econometric model. Although we could have evaluated the potential for future development decisions to be based in part on fire risk, this would have required simulation of feedbacks between fires and probability of development. Because our objective in this study was to isolate the effects of the three distinct growth types, we modeled fire risk only as a function of development pattern and not vice versa.

We constructed a complete spatial database of existing residential structures in the study area [26]. These structures and their corresponding parcel boundaries served as the initial conditions for all three scenarios of the spatial simulation model. The current and projected future GIS layers of structures were also subsequently used in the fire risk model (see below). The

Table 1. Variables and results from the probit regression model of parcel subdivision in San Diego County.

Subdivided (1 = yes,0 = no)	Coefficient	Std. Err.	z	P>|z|	[95% Conf. Interval]	
Acres of lot	0.0026342	0.00075	3.51	0	0.001164	0.004105
Acres of lot ^{2}	−3.02E-06	1.29E-06	−2.34	0.019	−5.55E-06	−4.93E-07
Distance to ocean	−7.42E-06	1.33E-06	−5.59	0	−0.00001	−4.82E-06
Distance to ocean ^{2}	2.33E-11	8.28E-12	2.82	0.005	7.11E-12	3.96E-11
Distance to major road	2.17E-07	2.74E-06	0.08	0.937	−5.16E-06	5.59E-06
Distance to major road ^{2}	−1.94E-11	1.70E-11	−1.14	0.252	−5.27E-11	1.38E-11
Distance to nearest city center	−0.0000115	1.70E-06	−6.76	0	−1.5E-05	−8.16E-06
Distance to nearest city center ^{2}	2.89E-11	9.70E-12	2.98	0.003	9.91E-12	4.79E-11
Slope between 0–5%	0.6211289	0.211761	2.93	0.003	0.206085	1.036173
Slope between 5–10%	0.3911427	0.210684	1.86	0.063	−0.02179	0.804076
Slope between 10–25%	0.0716669	0.212725	0.34	0.736	−0.34527	0.4886
Rural Residential	−0.3563149	0.071512	−4.98	0	−0.49648	−0.21615
Single Family	0.1361149	0.068678	1.98	0.047	0.001509	0.270721
Multi-Family	−0.2505093	0.151486	−1.65	0.098	−0.54742	0.046397
Road	0.015329	0.086094	0.18	0.859	−0.15341	0.184069
Open Space	−0.7440933	0.099145	−7.51	0	−0.93841	−0.54977
Orchard/Vineyard	−0.5813305	0.097867	−5.94	0	−0.77315	−0.38951
Agriculture	−0.9785208	0.132734	−7.37	0	−1.23867	−0.71837
Vacant Land	−0.5222501	0.074586	−7	0	−0.66844	−0.37606
Zoned protected	0.253769	0.076881	3.3	0.001	0.103086	0.404452
Area marked for redevelopment	−0.2680261	0.14069	−1.91	0.057	−0.54377	0.007722
Area marked for development	0.5780101	0.064103	9.02	0	0.452371	0.703649
Parcel between 10–20 acres	−0.3379532	0.065899	−5.13	0	−0.46711	−0.20879
Parcel between 5–10 acres	−0.6119036	0.067012	−9.13	0	−0.74325	−0.48056
Parcel between 2–5 acres	−1.16297	0.07062	−16.47	0	−1.30138	−1.02456
Parcel between 1–2 acres	−1.563956	0.090286	−17.32	0	−1.74091	−1.387
Parcel between.5–1 acres	−1.999939	0.099893	−20.02	0	−2.19573	−1.80415
Parcel between.25–.5 acres	−2.178273	0.117101	−18.6	0	−2.40779	−1.94876
Constant	−1.397931	0.227467	−6.15	0	−1.84376	−0.9521

Sample size 113 001, LR Chi^2 1535.23, pro>chi 0, pseudo R^2 0.22. Further description of the variables is provided in Table S1.
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dataset of existing housing includes locations of 687,869 structures, of which 4% were located within the perimeter of one of 40 fires that burned since 2001. During these fires, 4315 structures were completely destroyed, and another 935 were damaged.

For future development scenarios, we wanted to allocate an equal number of new structures to the landscape. This was to ensure that any predicted difference in fire risk was a function of the arrangement and location of structures, not the total number of structures. Nevertheless, differences in the total number of structures were simulated with each of the 5-year time steps. We determined the number of housing units to add during the simulations based on projections made by San Diego County [46]. Using factors such as development proposals, general plan densities, and information from jurisdictions, the county estimated that between 331,378 units and 486,336 units could be supported within the developable residential land by 2030. Because the eastern, desert portion of the county was not included in our study area, we used a conservative approach and simulated the addition of 331,378 new dwelling units. We divided this number by four to define the number of new dwelling units to add at each time step, assuming a linear growth rate.

One output of the econometric model was the prediction of the maximum number of new dwelling units that could be added to each parcel. However, dwelling units may consist of apartments as well as single family homes. The mix of single and multifamily units in the region has remained relatively constant over time, and the overall trend has been a mix of roughly 1/3 multifamily and 2/3 single family units. Because the fire risk model is based on points representing structure locations across the landscape, regardless of the number of dwelling units per structure, we needed to generate a conversion factor from dwelling units to structures. We therefore defined a minimum lot size of 0.25 acre on which no more than a single structure could be built, regardless of the number of dwelling units in it (i.e., a single family home or apartment complex). Then, once a parcel was selected for development by the model (see details below), we divided its total area by the maximum number of dwelling units to be added, according to the econometric model. If the result was larger than 0.25, we subdivided parcels according to the result. If not, we quantified how many 0.25 acre parcels fit into the original parcel, and generated the new parcel boundaries accordingly.

Using the initial map of parcels (year 2010), we classified each parcel that was defined as eligible for development (in the previous stage) as suitable for one of the three planning scenarios described above, according to the number of developed parcels in its immediate neighborhood (i.e., those parcels that share a boundary with the focal parcel). We defined 'developed parcels' as ones that had more than one house per 20 acres (8.09 ha). Therefore, according to these density thresholds, we allowed some parcels with nonzero housing density to be considered as 'undeveloped' because these large, rural parcels might contain a single or a handful of houses but they exist within a large open area. In other words, the overall land cover of these parcels was effectively undeveloped, and we therefore assumed that development in adjacent parcels would be akin to development in open areas.

We defined infill parcels as those that were completely surrounded by developed parcels. Expansion parcels had at least one neighboring parcel that was undeveloped; and leapfrog parcels were those with no developed parcels in their immediate surroundings. We reclassified the type of each available parcel in the same manner after each time step, to account for changing dynamics in the development map of the county.

We conducted three simulations, one for each development scenario (infill, expansion, and leapfrog). In each simulation, all parcels were eligible to subdivide, regardless of their class. Therefore, to build a simulation for a specific scenario, we increased the development probability of parcels of the selected scenario by 20%, to favor their development compared to the other types of parcels, without prohibiting development in the other parcel types. This approach was necessary because the projected number of dwelling units was much larger than it would be possible to fit in infill and leapfrog class parcels solely. For example, as the spatial coverage of developed parcel expands, there is less contiguous area that is undevelopable and suitable for leapfrog development. Therefore, the scenarios are not exclusive, but rather a mixture of the three development types. Yet, in each scenario, there is one main type of development, and smaller amounts of development events of the other two types.

Due to the immense computational demand of the simulations, we adopted a deterministic, rather than a stochastic approach to decide on which parcels were subdivided. After enhancing the transition probability according to the corresponding scenario, we ranked and then sorted all parcels according to their probability of subdivision. We then sequentially selected parcels, while simultaneously tallying the number of dwelling units in them, until the development target in that time step (one fourth of the total number of dwelling units to be added: 82,795) was reached. Once the development target was reached, we moved to the next time step. After each time step, the remaining parcels that were still eligible for development were re-classified to development types according to the new spatial configuration of the landscape.

Once a parcel was selected for subdivision, and the number of new parcels to develop in it was calculated (as detailed above), an equal-area spatial splitting model was employed to split the parent parcel to the predefined number of equal-area child parcels. We developed a simple splitting model which is based on iterative splitting of larger parcels into two smaller parcels using a straight line splitting boundary. Once the parcel was fully split into the needed number of sub-parcels, we allocated a new structure inside each new parcel by generating a point at its centroid (center of gravity). The point datasets of all structure locations per time step per scenario were passed over to the fire risk model, which is described below.

Fire Risk Modeling and Analysis

To project the distribution of fire risk under alternative scenarios, we used MaxEnt [47–48], a map-based modeling software used primarily for species distribution modeling [48], but we have used it successfully for ignition modeling [50] and for projecting current fire risk in the study area [26]. For this study, we slightly modified the model from Syphard et al. [26]. The dependent variable was the location of structures destroyed by fire between 2001 and 2010. Although inclusion of damaged structures in the data set does not significantly affect results [26], we only included completely destroyed structures to avoid the introduction of any uncertainty.

The MaxEnt software uses a machine-learning algorithm that iteratively evaluates contrasts among values of predictor values at locations where structures burned versus values distributed across the entire study area. The model assumes that the best approximation of an unknown distribution (i.e., structure destruction) is the one with maximum entropy. The output is an exponential function that assigns a probability to every cell of a map. Thus, the resulting continuous maps of fire risk represented the probability of a structure being destroyed by fire. In these output maps, areas of predicted high fire risk that did not have structures on them represented environmental conditions similar to those in which structures have actually burned.

We based the explanatory variables on those that were significantly related to burned structures in Syphard et al. [26], including maps depicting housing arrangement and pattern, housing location, and biophysical factors. Housing pattern variables reflected individual structure locations as well as the arrangement of structures within housing clusters. We calculated housing clusters, defined as groups of structures located within a maximum of 100 m from each other, by creating 100 m buffers around all structures and dissolving the overlapping boundaries [51].

Because burned structures were significantly related to small housing clusters [26], we calculated the area of every cluster as an attribute, and then created raster grids based on that attribute. Low-to intermediate housing density and distance to the edge of the cluster were also significant explanatory variables relative to housing pattern and location [26], so we also created raster grids for those. GIS buffer measures at 1-km have been found to explain approximately 90% of the variation in rural residential density [52], so we developed density grids using simple density interpolation based on a 1-km search radius, with area determined through square map units. To create grids representing distance to the edge of clusters, we first collapsed the cluster polygons into vector polyline files, and then created grids of interpolated Euclidean Distance to the edge within each cluster.

Because the MaxEnt model randomly selects background samples in the map to compare with locations of destroyed structures, we used a mask to restrict sampling to the developed environment within cluster boundaries; the distance to the edge of the cluster would represent a different relationship inside a cluster boundary versus outside in the wildland. We also modified the grids to ensure that any random sample located within the 100m buffer zone would receive a value of 100m; thus, all points within the buffer were considered "the edge of the development".

After creating the grids representing housing pattern and arrangement of the current configuration of structures, we applied the same algorithms to the maps of simulated future structure locations. We thus generated grids representing future housing pattern and arrangement under alternative development scenarios. The other explanatory variables, including fire history, slope, fuel type, southwest aspect, and distance to coast [26] remained constant through time for current and future scenarios. Although historic fire frequency and fuel type typically change through time, we did not simulate their dynamics here because we wanted to isolate the effect of planning decisions on housing pattern and arrangement while holding everything else constant.

We conditioned the MaxEnt model on present distributions of housing using ten thousand random background points and destroyed structures located no closer than 500-m to minimize any effect of spatial autocorrelation. We used 80% (260 records) of these data for model training, and 20% [66 records) for testing. We repeated the process using cross-validation with five replicates and used the average of these five models for analyses. For smoother functions of the explanatory variables, we used hinge features, linear, and quadratic with an increase in regularization of beta set at 2.5, based on Elith et al. [48]. The smoother response curves minimize over fitting of the model. We conducted jackknife tests of explanatory variable importance.

We first developed the model using mapped explanatory variables derived from the current configuration of structures. To project fire risk under the different time steps of the alternative development scenarios, projected the model conditioned upon current conditions onto maps representing future conditions by substituting the grids representing future housing pattern and arrangement. This is similar to how potential future distributions of species are projected under climate change scenarios [49].

To quantify differences among current and future alternative scenarios, we calculated metrics representing housing density, pattern, and footprint to determine the extent to which the planning policies produced differences in housing pattern and location. We compared the modeled structure fire risk of the scenarios by overlaying all maps of structure locations with their respective mapped output grids from the MaxEnt models and calculating probability of burning for every structure point. We also calculated total area of risk by selecting three threshold criteria [53]. These criteria, at 0.05, 0.25, and 0.5 represented three different degrees of risk, and we calculated the proportion of structures that were located in risk areas for every time step in all scenarios.

Results

The probit econometric model, run on 113 001 observations, showed that larger parcels were most likely to subdivide, although the relationship between parcel size and subdivision probability was non-linear (Table 1). Parcels closer to existing roads, the ocean, those with lower slopes, and those designated as fit for development were all most likely to develop. Parcels designated in redevelopment areas were less likely to develop. Overall, the model had a pseudo r –squared of 0.22.

The land use simulation model, based on a combination of the econometric subdivision model and three different growth policies, resulted in substantial differences in the extent and pattern of housing of the three scenarios. The total area of housing development, or the housing footprint, was largest for simulations where leapfrog growth dominated, followed by expansion-type development, and then infill (Figure 1a). The differences in the housing footprint became larger among the scenarios over time, but the largest difference was between infill and the other two development types. As the housing footprint expanded in the three scenarios, the corresponding housing density declined, so that leapfrog growth resulted in the lowest housing density per 1-km, followed by expansion and then infill (Figure 2b). Despite the near inverse of this relationship, there was generally a larger separation among scenarios with regard to housing density. With larger housing footprints and lower housing density, the number of separate housing clusters increased while their size decreased (Figure 2c).

In the first two time steps of the model (2015 and 2020), the simulated development pattern closely followed the desired pattern in the scenario, although some of the growth in the infill scenario ended up becoming expansion or leapfrog (Table 2). In the last two time steps (2025 and 2030), there were not enough infill parcels left, and thus, the majority of growth in these simulations became expansion, followed by infill, and then leapfrog. In the last time step, there were not enough isolated parcels in the leapfrog scenario and thus, the majority of development became expansion. Thus in general, as more development occurred in the simulations by the year 2030, the majority took the form of expansion.

The area under the curve (AUC) of receiver operating characteristic (ROC) plots, indicating the ability of the MaxEnt model to discriminate between burned and unburned structures, averaged across five cross-validated replicate runs was 0.91. The AUC represents the probability that, for a randomly selected set of observations, the model prediction was higher for a burned structure than for an unburned structure [49].The two most important variables in the model according to the internal jackknife tests in MaxEnt [47] were related to housing pattern:

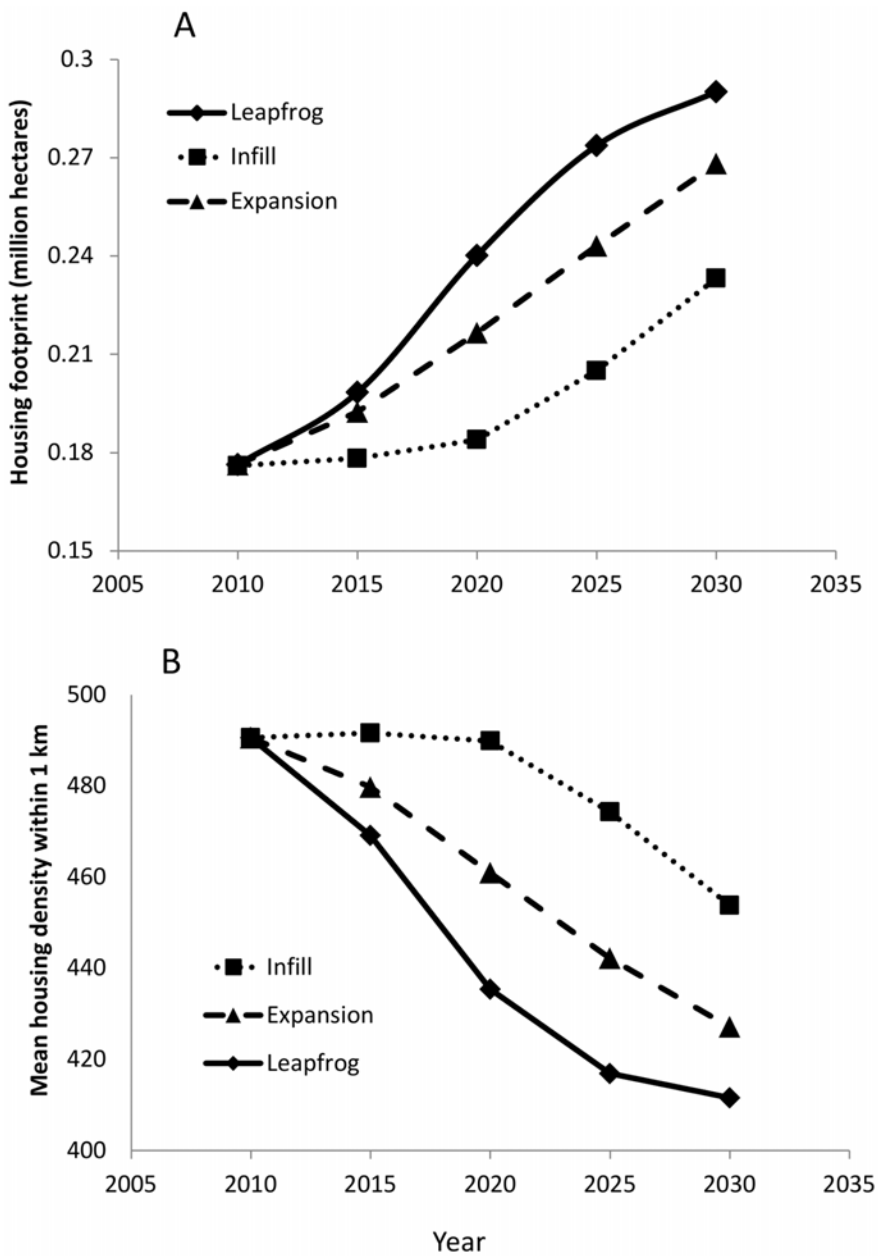

Figure 1. Trends of development extent and pattern for three planning policy simulations from 2010–2030, including A) total housing footprint representing the area of land within all housing clusters, and B) mean housing density averaged across all housing clusters.
doi:10.1371/journal.pone.0071708.g001

low to intermediate housing density and small cluster size and housing density (Figure 3). The distance to the edge of housing cluster was a less important contribution.

Maps showing the probability of a structure being destroyed in a wildfire, displayed as a gradient from low to high risk, show broad agreement relative to the general areas of the landscape that are riskiest, with correlation coefficients ranging from 0.85–0.91 (Figure 4). Nevertheless, subtle differences are apparent in the three development-scenario maps by year 2030, with the highest-risk areas in the expansion scenario located farther east than infill, and the highest-risk areas in leapfrog occupying a wider extent than either of the other two scenarios.

Differences among current housing and the three development scenarios are clearly illustrated through the mean landscape risk, or total probability of all structures burning (Figure 5). All three development scenarios were predicted to experience an increase in mean landscape risk over the duration of the simulations, except for infill at year 2015. The highest landscape risk to structures was predicted for the leapfrog scenario, followed by expansion, and then infill. The increase in risk over time is more gradual for the infill scenario than the other two scenarios.

The ranking of scenarios varied according to the proportion of structures located within different levels of risk defined through binary thresholding (Figure 6). When the continuous risk maps were thresholded at the lowest number of 0.05, a large proportion

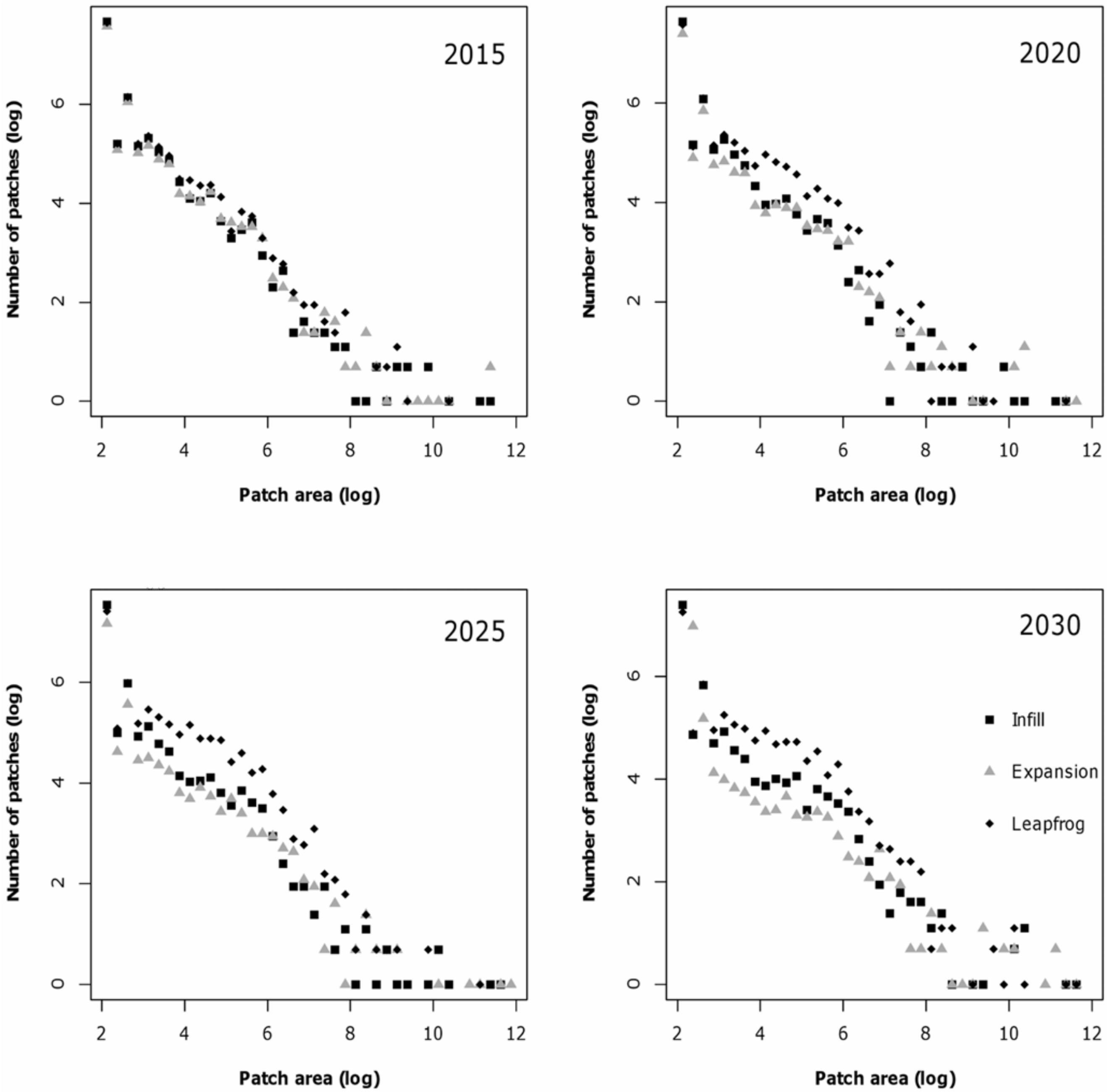

Figure 2. Trends in number of patches and patch area for three planning policy simulations from 2010–2030. Numbers were log-transformed for better visual representation of the scenarios.
doi:10.1371/journal.pone.0071708.g002

of structures in all scenarios fell within areas defined as risky according to this criterion. At this threshold, the proportion of structures in high-risk areas increased linearly for the expansion and leapfrog development scenarios while the proportion of infill homes increased more gradually. When risk was defined more conservatively at 0.25, temporal trends for the leapfrog and infill scenarios were similar to the 0.05 threshold. However, the proportion of structures at risk in the expansion scenario initially increased to 2020, but this proportion leveled off and declined by 2030. When the threshold was highest at 0.50, a very low proportion of structures in any scenario were located in areas at risk. But in these high-risk areas, the expansion scenario switched places with infill to have the lowest proportion of structures at risk in all time steps. Leapfrog had the largest proportion of homes at risk. This proportion of homes located in areas at risk with a threshold at 0.5 declined over time for all three scenarios.

Discussion

Our simulations of residential development showed that planning policies based on different growth types, applied locally for subdivision of individual parcels, will likely produce substantial and cumulative landscape-level differences in pattern, location, and extent of development. These differences in development pattern, in turn, will likely affect the area and proportion of

Table 2. Pattern of simulated development under infill, expansion, and leapfrog growth policies.

		Actual development		
Development scenario	**year**	***Infill***	***Expansion***	***Leapfrog***
Infill	2015	9450	18	6
	2020	11787	153	29
	2025	236	624	144
	2030	325	890	179
Expansion	2015	0	772	0
	2020	0	1243	2
	2025	0	1871	1
	2030	0	2662	0
Leapfrog	2015	0	10	408
	2020	0	5	1132
	2025	1	83	3563
	2030	34	917	0

The numbers in the table denote the numbers of patches of a given development type.
doi:10.1371/journal.pone.0071708.t002

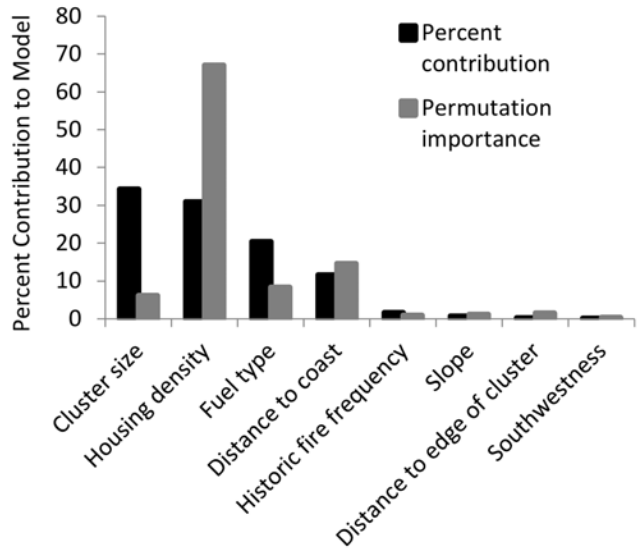

Figure 3. The importance of explanatory variables averaged across five cross-validated replications in the MaxEnt fire risk model. Percent contribution is determined as a function of the information gain from each environmental variable throughout the MaxEnt model iterations. Permutation importance reflects the drop in model accuracy that results from random permutations of each environmental variable, normalized to percentages.
doi:10.1371/journal.pone.0071708.g003

structures at risk from burning in wildfires. In particular, the scenarios with lower housing density and larger numbers of small, isolated clusters of development, i.e., leapfrog followed by expansion and infill, were generally predicted to have the highest predicted fire risk to the largest proportion of structures in the study area. Nevertheless, rankings of scenarios were affected by the definition of risk.

Theoretically, it makes sense that leapfrog development produced fragmented development with larger numbers of small patches, lower housing density, and a larger housing footprint; and that infill resulted in the opposite, with expansion in the middle. By definition, leapfrog development requires open space around all sides of the newly developed parcel, whereas infill requires development on all sides, and expansion requires development on one side and open space on another. Implementing these planning policies on real landscapes, however, can be complex if there are more houses to build than there are parcels that meet the definitions of the three planning rules, and thus not all development conforms strictly to the policy [54]. In our simulations, parcels meeting the definition of each growth type had a higher probability of subdividing; yet, as we were simulating a real landscape, many newly developed parcels did not meet the scenario criteria. That the three scenarios nevertheless produced substantial differences in landscape-level development patterns shows that decision-making at the individual level can lead to meaningful broad-scale effects.

The objective of the econometric model was to provide a baseline probability to predict which parcels were most likely to subdivide; thus, the econometric model itself provides no explanation of how a given policy affects likelihood of subdivision, although it does indicate the correlation between the policy and the outcome. In our setting, which areas are protected, marked for redevelopment, or marked for development may be endogenous to the land owner decision to subdivide. In the case of these variables especially, our results should not be interpreted as causal predictors. Likewise, we use data only from 2005–2009 to predict changes to 2030. If major changes in the land market take place over this time horizon our model will not be able to take this into account.

Although some differences in predicted fire risk among the three scenarios likely stemmed from location of new structures relative to variables such as distance to coast, fuel type, or slope, the most important variables in the fire risk model were housing density and cluster size, with most structure loss historically occurring in areas with low housing density and in small, isolated housing clusters. Thus, leapfrog development was generally the riskiest scenario and infill the least risky. The most surprising result was the variation in predicted risk for the expansion scenario over time and at different thresholds. While leapfrog and infill showed similar trajectories across thresholds, expansion went from being the highest-risk scenario at the low threshold to being the lowest-risk scenario at the highest threshold. Because the threshold is merely a way to group structures into a binary classification, this means that, while the average risk calculated across all homes shows expansion to rank in the middle of infill and leapfrog throughout the simulation (Figure 5), the other two scenarios have a relatively larger proportion of homes that are modeled to be at a very high risk (i.e., 0.25 or 0.5), particularly by the end of the simulations. Because the total number of structures with a risk greater than 0.25 or 0.5 is relatively low in all scenarios, this difference in distribution of homes at the highest risk is not reflected in the mean. Another reason for the shift in rank of expansion over time is that, as more development occupied the landscape, there were fewer parcels remaining to accomplish infill or leapfrog type growth in the other scenarios. Thus, by the end of the simulations in year 2030, the majority of growth in all scenarios was expansion, and there was some convergence between scenarios. Finally, the change in risk of expansion growth over time may reflect that, despite the relatively low importance of distance to edge of cluster as an explanatory variable, expansion growth is characterized as having an initially fragmented landscape pattern that eventually merges into large patches with low edge.

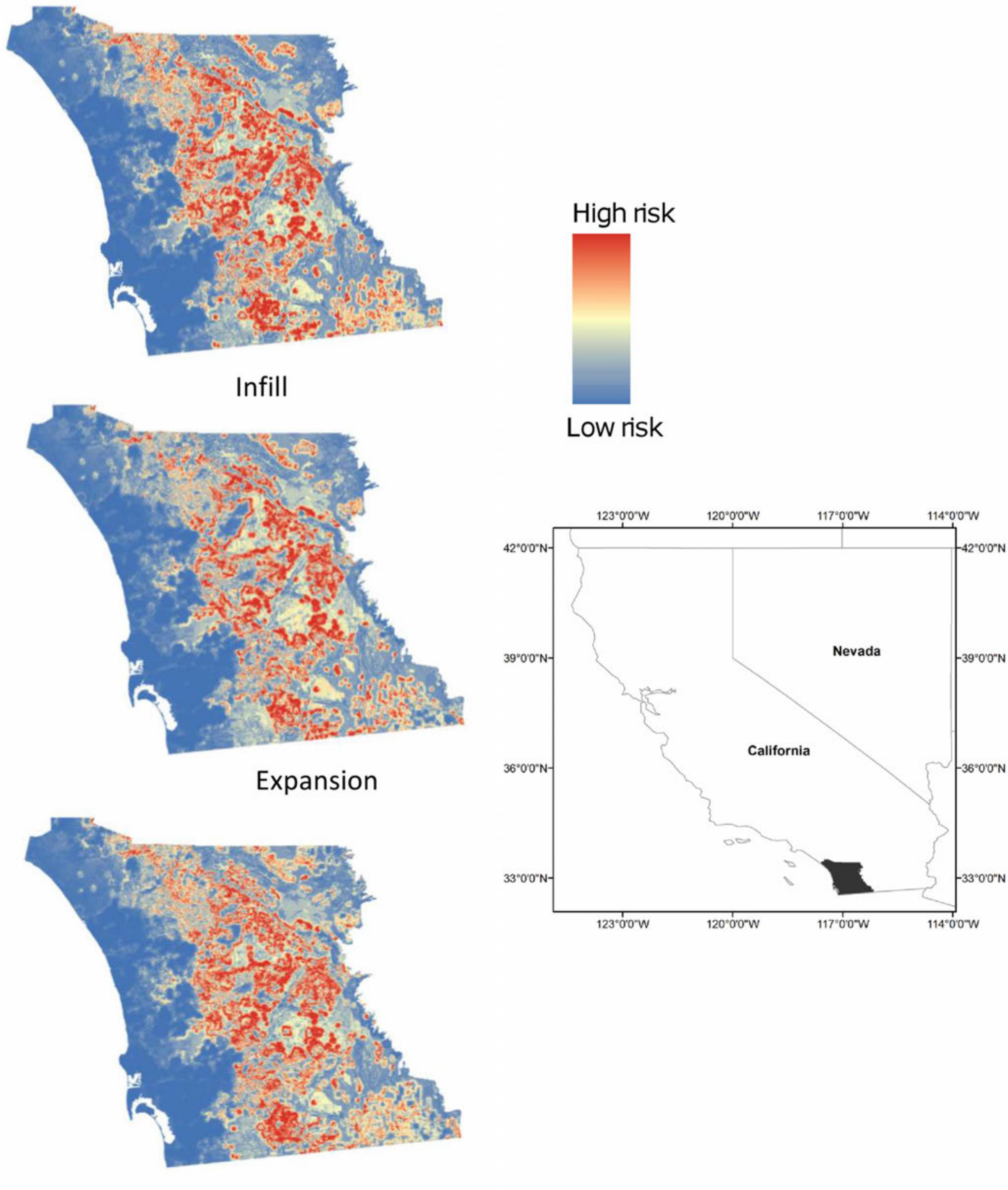

Figure 4. Maps of the study area showing projected wildfire risk at year 2030 for simulations of residential development under policies emphasizing infill, expansion, or leapfrog growth.
doi:10.1371/journal.pone.0071708.g004

Although leapfrog development clearly ranked highest in terms of fire risk, the interpretation of which planning policy is best may depend on fire management objectives and resources, as well as other considerations such as biodiversity or ecological impacts.

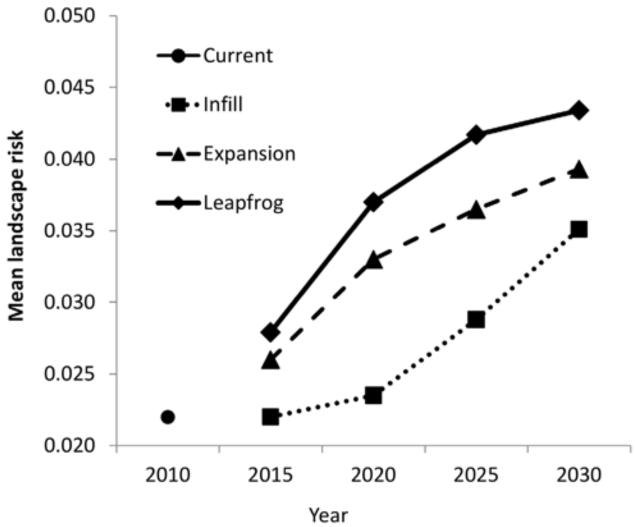

Figure 5. Projected landscape fire risk, reflecting the probability of burning in a wildfire averaged across all residential structures on the current landscape and in three development scenarios of infill, expansion, and leapfrog for year 2030.
doi:10.1371/journal.pone.0071708.g005

The spatial pattern of development affects multiple ecological functions and services [55], with potentially varying conservation implications; both leapfrog and expansion development consumed more land than infill, which would likely lead to more ecological degradation [56]; nevertheless, higher-density clustered development may be dominated by more invasive species [57]. Trade-offs between fire protection and conservation are common, but techniques are available for identifying mutually beneficial solutions [58].

Different perceptions of the fire risk results could also potentially translate into different planning priorities for management. For example, if the priority is to plan for the lowest overall risk to structures, then the mean landscape risk clearly delineates the rankings of options, with infill being the winner. However, if the objective is to reduce the number of structures at the highest risk threshold, i.e., $>=0.5$, then expansion is the best option, at least by 2030. An important consideration for fire management is the total area that needs to be protected, as well as the length of wildland-urban interface [8,13]. Therefore, despite the lower number of structures at the highest risk thresholds, expansion creates more edge than infill and may translate into greater challenges for firefighter protection.

Although we did not create separate scenarios for high or low growth, the results at different time steps can be substituted to envision the potential outcome of developing more or fewer houses. In the short term, the total fire risk is projected to increase proportionately as more land is developed. However, given the inverse relationship between housing density and fire risk, it is possible that this trend could reverse if housing growth eventually resulted in expansive high-density development.

Land use planning is one of a range of options available for reducing fire risk, and the best outcome will likely be achieved through a combination of strategies that include homeowner actions, improvements in fire-safe building codes, and advanced fire suppression tactics. Although we isolated the effect of land use planning policy in the three development scenarios, the fire risk model nevertheless showed that the pattern and location of structures in this study area were the most important out of a suite of factors influencing structure loss. We used a correlative approach that did not incorporate mechanisms or feedbacks, but our models clearly illustrated differences in the cumulative effects of individual planning decisions. The relationship between spatial pattern of development and fire risk is likely related to the intermixing of development and wildland vegetation [29,59]; thus, these results likely apply to a wide range of fire-prone ecosystems with large proportions of human-caused ignitions. Nevertheless, because fire risk is highly variable over space and time, and due to a range of human and biophysical variables [60], we recommend planners develop their own models for the best understanding of where the most fire-prone areas are in their region [19].

With projections of substantial global change in climate and human development, we recommend that land use planning should be considered as an important component to fire risk management, potentially to become as successful as the prevention of building on flood plains [61]. History has shown us that preventing fires is impossible in areas where large wildfires are a natural ecological process [4,9]. As Roger Kennedy put it, "the

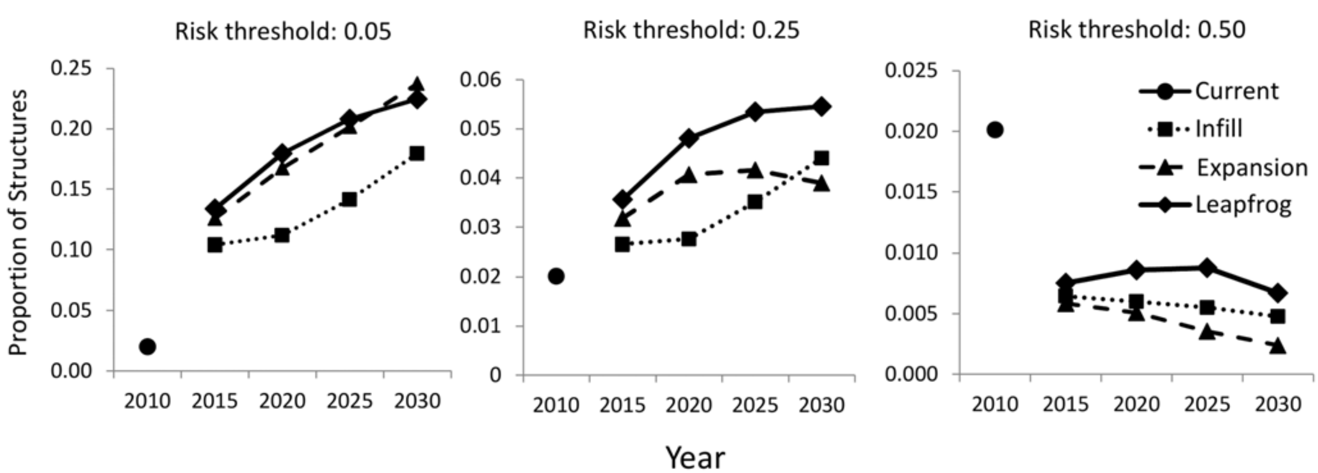

Figure 6. Proportion of residential structures that are located in areas of high fire risk defined using thresholds from the fire risk model of 0.05, 0.25, and 0.5 for current structures and for structures simulated under infill, expansion, and leapfrog growth policies.
doi:10.1371/journal.pone.0071708.g006

problem isn't fires; the problem is people in the wrong places [62]."

Supporting Information

Table S1 Definitions and summary statistics for variables used in the probit model.
(DOCX)

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Author Contributions

Conceived and designed the experiments: ADS JEK ABM VB. Performed the experiments: ADS ABM VB. Analyzed the data: ADS ABM VB. Wrote the paper: ADS JEK ABM VB.

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The role of defensible space for residential structure protection during wildfires

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Abstract. With the potential for worsening fire conditions, discussion is escalating over how to best reduce effects on urban communities. A widely supported strategy is the creation of defensible space immediately surrounding homes and other structures. Although state and local governments publish specific guidelines and requirements, there is little empirical evidence to suggest how much vegetation modification is needed to provide significant benefits. We analysed the role of defensible space by mapping and measuring a suite of variables on modern pre-fire aerial photography for 1000 destroyed and 1000 surviving structures for all fires where homes burned from 2001 to 2010 in San Diego County, CA, USA. Structures were more likely to survive a fire with defensible space immediately adjacent to them. The most effective treatment distance varied between 5 and 20 m (16–58 ft) from the structure, but distances larger than 30 m (100 ft) did not provide additional protection, even for structures located on steep slopes. The most effective actions were reducing woody cover up to 40% immediately adjacent to structures and ensuring that vegetation does not overhang or touch the structure. Multiple-regression models showed landscape-scale factors, including low housing density and distances to major roads, were more important in explaining structure destruction. The best long-term solution will involve a suite of prevention measures that include defensible space as well as building design approach, community education and proactive land use planning that limits exposure to fire.

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Introduction

Across the globe and over recent decades, homes have been destroyed in wildfires at an unprecedented rate. In the last decade, large wildfires across Australia, southern Europe, Russia, the US and Canada have resulted in tens of thousands of properties destroyed, in addition to lost lives and enormous social, economic and ecological effects (Filmon 2004; Boschetti *et al.* 2008; Keeley *et al.* 2009; Blanchi *et al.* 2010; Vasquez 2011). The potential for climate change to worsen fire conditions (Hessl 2011), and the projection of continued housing growth in fire-prone wildlands (Gude *et al.* 2008) suggest that many more communities will face the threat of catastrophic wildfire in the future.

Concern over increasing fire threat has escalated discussion over how to best prepare for wildfires and reduce their effects. Although ideas such as greater focus on fire hazard in land use planning, using fire-resistant building materials and reducing human-caused ignitions (e.g. Cary *et al.* 2009; Quarles *et al.* 2010; Syphard *et al.* 2012) are gaining traction, the traditional strategy of fuels management continues to receive the most attention. Fuels management in the form of prescribed fires or mechanical treatments has historically occurred in remote, wildland locations (Schoennagel *et al.* 2009), but recent studies suggest that treatments located closer to homes and communities may provide greater protection (Witter and Taylor 2005; Stockmann *et al.* 2010; Gibbons *et al.* 2012). In fact, one of the most commonly recommended strategies in terms of fuels and fire protection is to create defensible space immediately around structures (Cohen 2000; Winter *et al.* 2009). Defensible space is an area around a structure where vegetation has been modified, or 'cleared,' to increase the chance of the structure surviving a wildfire. The idea is to mitigate home loss by minimising direct contact with fire, reducing radiative heating, lowering the probability of ignitions from embers and providing a safer place for fire fighters to defend a structure against fire (Gill and Stephens 2009; Cheney *et al.* 2001). Many jurisdictions provide specific guidelines and practices for creating defensible space, including minimum distances that are required among trees and shrubs as well as minimum total distances from the structure. These distances may be enforced through local ordinances or state-wide laws. In California, for example, a state law in 2005 increased the required total distance from 9 m (30 ft) to 30 m (100 ft).

Despite these specific guidelines on how to create defensible space, there is little scientific evidence to support the amount and location of vegetation modification that is actually effective

at providing significant benefits. Most spacing guidelines and laws are based on 'expert opinion' or recommendations from older publications that lack scientific reference or rationale (e.g. Maire 1979; Smith and Adams 1991; Gilmer 1994). However, one study has provided scientific support for, and forms the basis of, most guidelines, policy and laws requiring a minimum of 30 m (100 ft) of defensible space (Cohen 1999, 2000). The modelling and experimental research in that study showed that flames from forest fires located 10–40 m (33–131 ft) away would not scorch or ignite a wooden home; and case studies showed 90% of homes with non-flammable roofs and vegetation clearance of 10–20 m (33–66 ft) could survive wildfires (Cohen 2000). However, the models and experimental research in that study focussed on crown fires in spruce or jack pine forests, and the primary material of home construction was wood. Therefore, it is unknown how well this guideline applies to regions dominated by other forest types, grasslands, or nonforested woody shrublands and in regions where wooden houses are not the norm.

Some older case studies showed that most homes with non-flammable roofs and 10–18 m (33–ft) of defensible space survived the 1961 Bel Air fire in California (Howard *et al.* 1973); most homes with non-flammable roofs and more than 10 m (33 ft) of defensible space also survived the 1990 Painted Cave fire (Foote and Gilless 1996). Also, several fire-behaviour modelling studies have been conducted in chaparral shrublands. One study showed that reducing vegetative cover to 50% at 9–30 m (30–ft) from structures effectively reduced fireline intensity and flame lengths, and that removal of 80% cover would result in unintended consequences such as exotic grass invasion, loss of habitat and increase in highly flammable flashy fuels (A. Fege and D. Pumphrey, unpubl. data). Another showed that separation distances adequate to protect firefighters varied according to fuel model and that wind speeds greater than 23 km h^{-1} negated the effect of slope, and wind speed above 48 km h^{-1} negated any protective effect of defensible space (F. Bilz, E. McCormick and R. Unkovich, unpubl. data, 2009). Results obtained through modelling equations of thermal radiation also found safety distances to vary as a function of fuel type, type of fire, home construction material and protective garments worn by firefighters (Zárate *et al.* 2008).

Although there is no empirical evidence to support the need for more than 30 m (100 ft) of defensible space, there has been a concerted effort in some areas to increase this distance, particularly on steep slopes. In California, a senate bill was introduced in 2008 (SB 1618) to encourage property owners to clear 91 m (300 ft) through the reduction of environmental regulations and permitting needed at that distance. Although this bill was defeated in committee, many local ordinances do require homeowners to clear 91 m (300 ft) or more, and there are reports that some people are unable to get fire insurance without 91 m (300 ft) of defensible space (F. Sproul, pers. comm.). In contrast, homeowner acceptance of and compliance with defensible space policies can be challenging (Winter *et al.* 2009; Absher and Vaske 2011), and in many cases homeowners do not create any defensible space.

It is critically important to develop empirical research that quantifies the amount, location and distance of defensible space that provides significant fire protection benefits so that guidelines and policies are developed with scientific support. Data that are directly applicable to southern California are especially important, as this region experiences the highest annual rate of wildfire-destroyed homes in the US. Not having sufficient defensible space is obviously undesirable because of the hazard to homeowners. However, there are clear trade-offs involved when vegetation reduction is excessive, as it results in the loss of native habitats, potential for increased erosion and invasive species establishment, and it potentially even increases fire risk because of the high flammability of weedy grasslands (Spittler 1995; Keeley *et al.* 2005; Syphard *et al.* 2006).

It is also important to understand the role of defensible space in residential structure protection relative to other factors that explain why some homes are destroyed in fires and some are not. Recent research shows that landscape-scale factors, such as housing arrangement and location, as well as biophysical variables characterising properties and neighbourhoods such as slope and fuel type, were important in explaining which homes burned in two southern California study areas (Syphard *et al.* 2012; 2013). Understanding the relative importance of different variables at different scales may help to identify which combinations of factors are most critical to consider for fire safety.

Our objective was to provide an empirical analysis of the role of defensible space in protecting structures during wildfires in southern California shrublands. Using recent pre-fire aerial photography, we mapped and measured a suite of variables describing defensible space for burned and unburned structures within the perimeters of major fires from 2001 to 2010 in San Diego County to ask the following questions:

1. How much defensible space is needed to provide significant protection to homes during wildfires, and is it beneficial to have more than the legally required 30 m (100 ft)?
2. Does the amount of defensible space needed for protection depend on slope inclination?
3. What is the role of defensible space relative to other factors that influence structure loss, such as terrain, fuel type and housing density?

Methods

Study area

The properties and structures analysed were located in San Diego County, California, USA (Fig. 1) – a topographically diverse region with a Mediterranean climate characterised by cool, wet winters and long summer droughts. Fire typically is a direct threat to structures adjacent to wildland areas. Native shrublands in southern California are extremely flammable during the late summer and fall (autumn) and when ignited, burn in high-intensity, stand-replacing crown fires. Although 500 homes on average have been lost annually since the mid-1900s (Calfire 2000), that rate has doubled since 2000. Most of these homes have burned during extreme fire weather conditions that accompany the autumn Santa Ana winds. The wildland–urban interface here includes more than 5 million homes, covering more than 28 000 km^2 (Hammer *et al.* 2007).

Property data

The data for properties to analyse came from a complete spatial database of existing residential structures and their

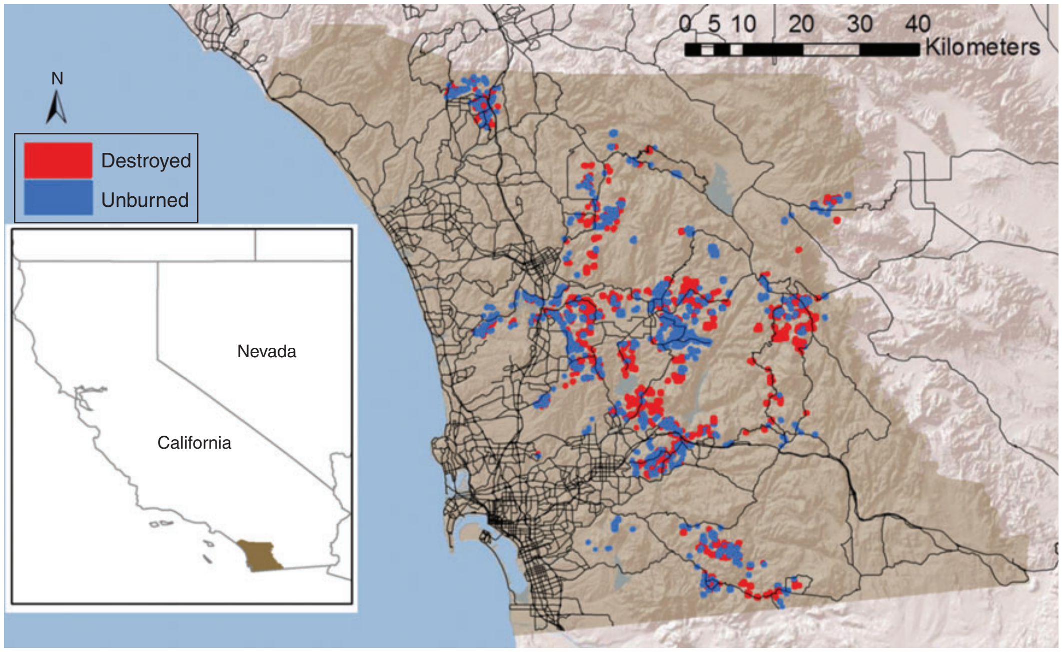

Fig. 1. Location of destroyed and unburned structures within the South Coast ecoregion of San Diego County, California, USA.

corresponding property boundaries developed for San Diego County (Syphard *et al.* 2012). This dataset included 687 869 structures, of which 4315 were completely destroyed by one of 40 major fires that occurred from 2001 to 2010. Our goal was to compare homes that were exposed to wildfire and survived with those that were exposed and destroyed. To determine exposure to fire, we only considered structures located both within a GIS layer of fire perimeters and within areas mapped as having burned at a minimum of low severity through thematic Monitoring Trends in Burn Severity produced by the USA Geological Survey and USDA Forest Service. From these data, we used a random sample algorithm in GIS software to select 1000 destroyed and 1000 unburned homes that were not adjacent to each other, to minimise any potential for spatial autocorrelation. Our final property dataset included structures that burned across eight different fires. More than 97% of these structures burned in Santa Ana wind-driven fire events (Fig. 1).

Calculating defensible space and additional explanatory variables

To estimate defensible space, we developed and explored a suite of variables relative to the distance and amount of defensible space surrounding structures, as well as the proximity of woody vegetation to the structure (Table 1). We measured these variables based on interpretation of Google Earth aerial imagery. We based our measurements on the most recent imagery before the date of the fire. In almost all cases, imagery was available for less than 1 year before the fire.

Our definition of defensible space followed the guidelines published by the California Department of Forestry and Fire Protection (Calfire 2006). 'Clearance' included all areas that were not covered by woody vegetation, including paved areas or grass. Although Google Earth prevents the identification of understorey vegetation, woody trees and shrubs were easily distinguished from grass, and our objective was to measure horizontal distances as required by Calfire rather than assess the relative flammability of different vegetation types. Trees or shrubs were allowed to be within the defensible space zone as long as they were separated by the minimum horizontal required distance, which was 3 m (10 ft) from the edge of one tree canopy to the edge of the next (Fig. 2). Although greater distances between trees or shrubs are recommended on steeper slopes, we followed the same guidelines for all properties. For all structures, we started the distance measurements by drawing lines from the centre of the four orthogonal sides of the structure that ended when they intersected anything that no longer met the requirements in the guidelines. A fair number of structures are not four sided; thus, the start of the centre point was placed at a location that approximated the farthest extent of the structure along each of four orthogonal sides.

We developed two sets of measurements of the distance of defensible space based on what is feasible for homeowners within their properties *v.* the total effective distance of defensible space. We made these two measurements because homeowners are only required to create defensible space within their own property, and this would reflect the effect of individual homeowner compliance. Therefore, even if cleared vegetation extended beyond the property line, the first set of distance measurements ended at the property boundary. The second set of measurements ignored the property boundaries and accounted for the total potential effect of treatment. For all measurements, we recorded the cover types (e.g. structure >3 m (10 ft) long, property boundary, or vegetation type) at which the distance measurements stopped (Table 1). Because property

Table 1. Defensible space variables measured for every structure

Urban veg, landscaping vegetation that was not in compliance with regulations within urban matrix; wildland veg, wildland vegetation that was not in compliance with regulations; orchard, shrub to tree-sized vegetation in rows; urban to wildland, landscaping vegetation that leads into wildland vegetation; structure, any building longer than 3 m (10 ft)

Variable	Definition
Distance defensible space within property	Measure of clearance from side of structure to property boundary calculated for four orthogonal directions from structure and averaged
Total distance defensible space	Measure of clearance from side of structure to end of clearance calculated for four orthogonal directions from structure and averaged
Cover type at end of defensible space	Type of cover encountered at end of measurement (urban veg, wildland veg, orchard, urban to wildland, structure)
Percentage clearance	Percentage of clearance calculated across the entire property
Neighbours' vegetation	Binary indicator of whether neighbours' uncleared vegetation was located within 30 m (100 ft) of the main structure
Vegetation touching structure	Number of sides on which woody vegetation touches main structure (1–4) Structure with more than 4 sides were viewed as a box and given a number between 1 and 4
Vegetation overhanging roof	Was vegetation overhanging the roof? (yes or no)

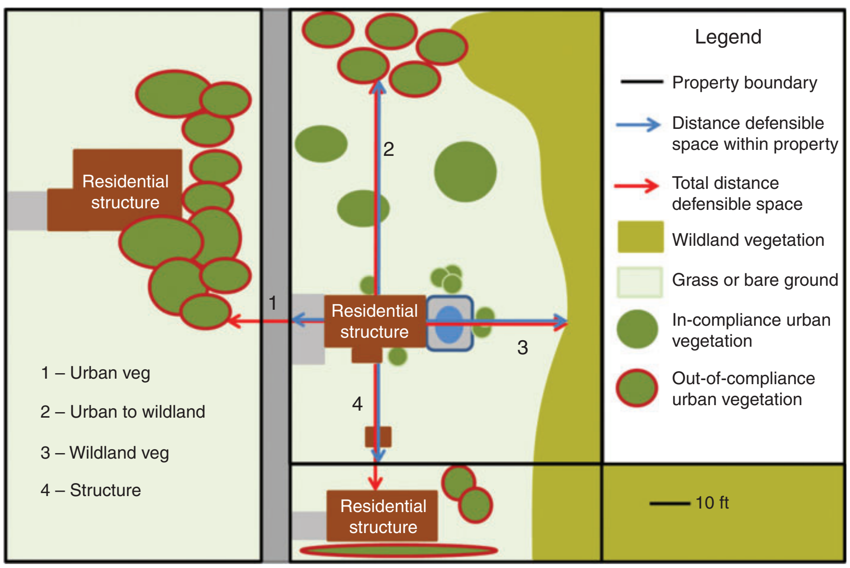

Fig. 2. Illustration of defensible space measurements. See Table 1 for full definition of terms.

owners usually can only clear vegetation on their own land, it is possible that the effectiveness of defensible space partly depends upon the actions of neighbouring homeowners. Therefore, we also recorded whether or not any neighbours' un-cleared vegetation was located within 30 m (100 ft) of the structure.

To assess the total amount of woody vegetation that can safely remain on a property and still receive significant benefits of defensible space, we calculated the total percentage of cleared land, woody vegetation and structure area across every property. This was accomplished by overlaying a grid on each property and determining the proportion of squares falling into each class. Preliminary results showed these three measurements to be highly correlated, so we only retained percentage clearance for further analysis. To evaluate the relative effect of woody vegetation directly adjacent to structures, we also calculated the number of sides of the structure with vegetation touching and recorded whether any trees were overhanging structures' roofs.

In addition to defensible space measurements, we evaluated other factors known to influence the likelihood of housing loss to fire in the region (Syphard *et al.* 2012, 2013). Using the same data as in Syphard *et al.* (2012, 2013), we extracted spatial information from continuous grids of explanatory variables for the locations of all structures in our analysis. Variables included interpolated housing density based on a 1-km search radius; percentage slope derived from a 30-m digital elevation model (DEM); Euclidean distance to nearest major and minor road and fuel type, which was based on a simple classification of US Forest Service data (Syphard *et al.* 2012), including urban, grass, shrubland and forest & woodland.

Analysis

We performed several analyses to determine whether relative differences in home protection are provided by different distances and amounts of defensible space, particularly beyond the legally required 30 m (100 ft), and to identify the effective treatment distance for homes on low and steep slopes.

Categorical analysis

For the first analysis, we divided our data into several groups to identify potential differences among specific categories of defensible space distance around structures located on shallow and steep slopes. We first sorted the full dataset of 2000 structures by slope and then split the data in the middle to create groups of homes with shallow slope and steep slope. We divided the data in half to keep the number of structures even within both groups and to avoid specifying an arbitrary number to define what constitutes shallow or steep slope. The two equal-sized subsets of data ranged from 0 to 9%, with a mean of 8% for shallow slope, and from 9 to 40%, with a mean of 27% for steep slope. Within these data subsets, we next created groups reflecting different mean distances of defensible space around structures. We also performed separate analyses based on whether defensible space measurements were calculated within the property boundary or whether measurements accounted for the total distance of defensible space.

Within all groups, we calculated the proportion of homes that were destroyed by wildfire. We performed Pearson's Chi-square tests of independence to determine whether or not the proportion of destroyed structures within groups was significantly different (Agresti 2007). We based one test on four equal-interval groups within the legally required distance of 30 m (100 ft): 0–7 m (0–25 ft), 8–15 m (26–50 ft), 16–23 m (51–75 ft) and 24–30 m (76–100 ft). A second test was based on three groups (24–30 m (75–100 ft), 31–90 m (101–300 ft) and >90 m (>300 ft) or >60 m (>200 ft)) to evaluate whether groups with mean defensible space distances >30 m (>100 ft) were significantly different from groups with <30 m (<100 ft). When defensible space distances were only measured to the property boundary, few structures had mean defensible space >90 m (>300 ft). Therefore, we used a cut-off of 60 m (200 ft) to increase the sample size in the Chi-square analysis. In addition to the Chi-square analysis, we calculated the relative risk among every successive pair of categories (Sheskin 2004). The relative risk was calculated as the ratio of proportions of burned homes within two groups of homes that had different defensible space distances.

Effective treatment analysis

In addition to comparing the relative effect of defensible space among different groups of mean distances, as described above, we also considered that the protective effect of defensible space for structures exposed to wildfire is conceptually similar to the effect of medication in producing a therapeutic response in people who are sick. In addition to pharmacological applications, treatment–response relationships have been used for radiation, herbicide, drought tolerance and ecotoxicological studies (e.g. Streibig *et al.* 1993; Cedergreen *et al.* 2005; Knezevic *et al.* 2007; Kursar *et al.* 2009). The effect produced by a drug or treatment typically varies according to the concentration or amount, often up to a point at which further increase provides no additional response. The effective treatment (ET50), therefore, is a specific concentration or exposure that produces a therapeutic response or desired effect. Here we considered the treatment to be the distance or amount of defensible space.

Using the software package DRC in R (Knezevic *et al.* 2007; Ritz and Streibig 2013), we evaluated the treatment–response relationship of defensible space in survival of structures during wildfire. To calculate the effective treatment, we fit a log-logistic model with logistic regression because we had a binary dependent variable (burned or unburned). We specified a 2-parameter model where the lower limit was fixed at 0 and the upper limit was fixed at 1. We again performed separate analyses for data subsets reflecting shallow and steep slope, as well as from measurements of defensible space taken within, or regardless of, property boundaries. We also performed analyses to find the effective treatment of percentage clearance of trees and shrubs within the property.

Multiple regression analysis

To evaluate the role of defensible space relative to other variables, we developed multiple generalised linear regression models (GLMs) (Venables and Ripley 1994). We again had a binary dependent variable (burned versus unburned), so we specified a logit link and binomial response. Although the proportion of 0s and 1s in the response may be important to consider for true prediction (King and Zeng 2001; Syphard *et al.* 2008), our objective here was solely to evaluate variable importance. We developed multiple regression models for all possible combinations of the predictor variables and used the corrected Akaike's Information Criterion (AICc) to rank models and select the best ones for each region using package MuMIn in R (R Development Core Team 2012; Burnham and Anderson 2002). We recorded all top-ranked models that had an AICc value within 2 of that of the model with lowest AICc to identify all models with empirical support. To assess variable importance, we calculated the sum of Akaike weights for all models that contained each variable. On a scale of 0–1, this metric represents the weight of evidence that models containing the variable in question are the best model (Burnham and Anderson 2002). The distance of defensible space measured within property boundaries was highly correlated with the distance of defensible space measured beyond property boundaries ($r = 0.82$), so we developed two separate analyses – one using variables measured only within the property boundary and the other using variables that accounted for defensible space outside of the property boundary as well as the potential effect of neighbours having uncleared vegetation within 30 m (100 ft) of the structure. A test to avoid multicollinearity showed all other variables within each multiple regression analysis to be uncorrelated ($r < 0.5$).

Surrounding matrix

To assess whether the proportion of destroyed structures varied according to their surrounding matrix, we summarised the most common cover type at the end of defensible space measurements (descriptions in Table 1) for all structures. These summaries

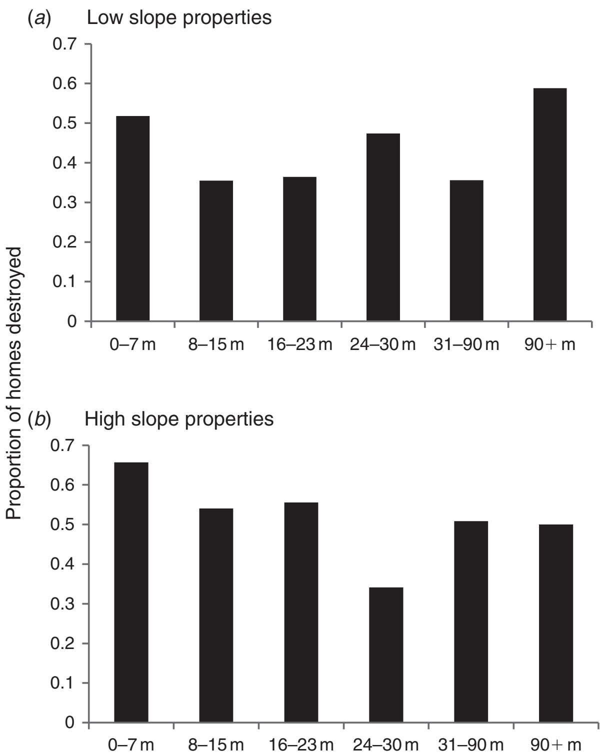

Fig. 3. Proportion of destroyed homes grouped by distances of defensible space based upon total distance of clearance within property boundary, for structures on (*a*) shallow slopes (mean 8%) and (*b*) steep slopes (mean 27%).

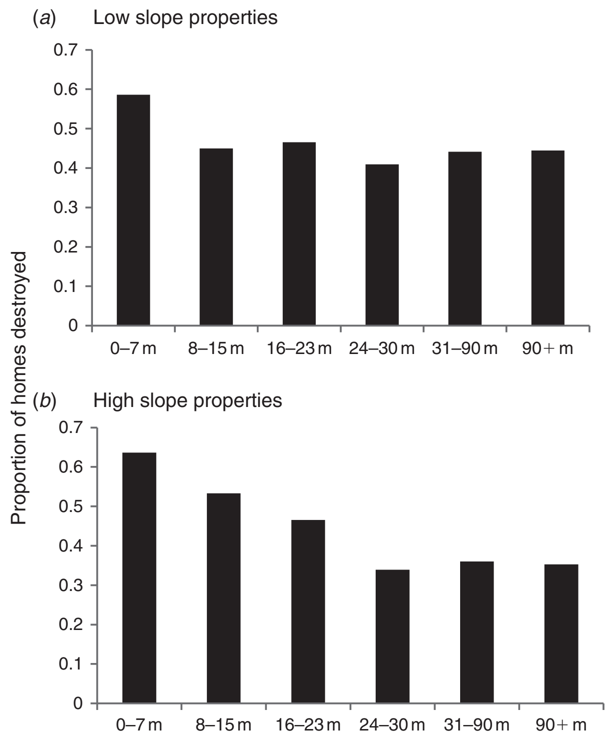

Fig. 4. Proportion of destroyed homes grouped by distances of defensible space based upon total distance of clearance regardless of property boundary, for structures on (*a*) shallow slopes (mean 8%) and (*b*) steep slopes (mean 27%).

were based on the majority surrounding cover type from the four orthogonal sides of the structure. We also noted cases in which there was a tie (e.g. two sides were urban vegetation and two sides were structures).

Results

Categorical analysis

When the distance of defensible space was measured both 'only within property boundaries' (Fig. 3) and 'regardless of property boundaries' (Fig. 4), the Chi-square test showed a significant difference ($P < 0.001$) in the proportion of destroyed structures among the four equal-interval groups of distance ranging from 0 to 30 m (0–100 ft). This relationship was consistent on both shallow-slope and steep-slope properties, although the relative risk analysis showed considerable variation among classes (Table 2) There was a steadily decreasing proportion of destroyed structures at greater distances of defensible space up to 30 m (100 ft) on the steep-slope structures with defensible space measured regardless of property boundaries (Fig. 4*b*). Otherwise, the biggest difference in proportion of destroyed structures occurred between 0 and 7 m (0–25 ft) and 8–15 m (26–50 ft) (Figs 3*a–b*, 4*a*).

When the distance of defensible space was measured in intervals from 24 m (75 ft) and beyond, the Chi-square test showed no significant difference among groups ($P = 0.96$ for shallow-slope properties and $P = 0.74$ for steep-slope properties) (Figs 3, 4), although again, the relative risk analysis showed considerable variation (Table 2).There was a slight increase in the proportion of homes destroyed at longer distance intervals when the defensible space was measured only to the property boundaries (Fig. 3*a–b*). This slight increase is less apparent when distances were measured regardless of boundaries (Fig. 4*a–b*).

The relative risk calculations showed that the ratio of proportions was generally more variable among successive pairs when the distances were measured within property boundaries (Table 2). For these calculations, the risk of a structure being destroyed was significantly lower when the defensible space distance was 8–15 m (25–50 ft) compared to 0–7 m (0–25 ft) on both shallow- and steep-slope properties. On the steep-slope properties, there was an additional reduction of risk when comparing 24–30 m (75–100 ft) to 16–23 m (50–75 ft). However, the risk of a home being destroyed was slightly significantly higher when there was 31–90 m (101–225 ft) compared to 16–23 m (50–75 ft). For distances that were measured regardless of property boundary (total clearance), the only significant differences in risk of burning were a reduction in risk for 8–15 m (25–50 ft) compared to 0–7 m (0–25 ft).

Table 2. Number of burned and unburned structures within defensible space distance categories (m), their relative risk and significance
A relative risk of 1 indicates no difference; <1 means the chance of a structure burning is less than the other group; >1 means the chance is higher than the other group. The relative risk is calculated for pairs that include the existing row and the row above. Confidence intervals are in parentheses

	Distance within property				Total distance			
	Burned	Unburned	Relative risk	*P*	Burned	Unburned	Relative risk	*P*
Shallow slope								
0–7	200	186			162	114		
8–15	109	198	0.69 (0.12)	<0.001	108	132	0.77	0.002
16–23	51	89	1.03 (0.30)	0.850	78	90	1.03	0.770
24–30	36	40	1.30 (0.39)	0.110	50	70	0.90	0.430
31–90	28	47	0.79 (0.24)	0.220	79	99	1.06	0.640
60 or 90+	10	6	1.67 (0.63)	0.040	8	9	1.01	0.830
Steep slope								
0–7	245	128			224	128		
8–15	174	148	0.82 (0.10)	0.001	158	139	0.84	0.008
16–23	85	68	1.03 (0.16)	0.750	73	83	0.87	0.210
24–30	29	56	0.61 (0.17)	0.004	26	50	0.73	0.080
31–	29	28	1.49 (0.48)	0.050	39	68	1.06	0.760
60 or 90+	5	5	0.98 (0.47)	0.950	4	8	0.91	0.830

Table 3. Effective treatment results reflecting the distance (in metres, with feet in parentheses) and percentage clearance within properties that provided significant improvement in structure survival during wildfires
The property mean is the average distance of defensible space or percentage clearance that was calculated on the properties before the wildfires and provides a means to compare the effective treatment result to the actual amount on the properties

	All parcels effective treatment ($n = 2000$)	Parcel mean	Shallow slope (mean 8%) effective treatment ($n = 1000$)	Parcel mean	Steep slope (mean 27%) effective treatment ($n = 1000$)	Parcel mean
Defensible space within parcel	10 (33)	13 (44)	4 (13)	14 (45)	25 (82)	11 (35)
Total distance defensible space	10 (32)	19 (63)	5 (16)	20 (67)	20 (65)	18 (58)
Mean percentage clearance on property	36	48	31	51	37	35

Effective treatment analysis

Analysis of the treatment–response relationships among defensible space and structures that survived wildfire showed that, when all structures are considered together, the mean actual defensible space that existed around structures before the fires was longer than the calculated effective treatment (Table 3). Regardless of whether the defensible space was measured within or beyond property boundaries, the estimated effective treatment of defensible space was nearly the same at 10 m (32–33 ft).

The effective treatment distance was much shorter for structures on shallow slopes (4–5 m (13–16 ft)) than for structures on steep slopes (20–25 m (65–82 ft)), but in all cases was <30 m (<100 ft). Although longer distances of defensible space were calculated as effective on steeper slopes, these structures actually had shorter mean distances of defensible space around their properties than structures on low slopes (Table 3).

The calculated effective treatment of the mean percentage clearance on properties was 36% for all properties, 31% for structures on shallow slopes and 37% for structures on steep slopes (Table 3). In total, the properties all had higher actual percentage clearance on their property than was calculated to be effective. However, this mainly reflects the shallow-slope properties, as those structures on steep slopes had less clearance than the effective treatment.

Multiple regression analysis

When defensible space was measured only to the property boundaries, it was not included in the best model, according to the all-subsets multiple regression analysis (Table 4). However, it was included in the best model when factoring in the distance of defensible space measured beyond property boundaries (Table 5). In both multiple regression analyses, low housing density and shorter distances to major roads were ranked as the most important variables according to their Akaike weights. Slope and surrounding fuel type were also in both of the best models as well as other measures of defensible space, including the percentage clearance on property and whether vegetation was overhanging the structure's roof. The number of sides in which vegetation was touching the structure was included in the best model when defensible space was only measured to the property boundary. The total explained deviance for the multiple regression models was low (12–13%) for both analyses.

Table 4. Results of multiple regression models of destroyed homes using all possible variable combinations and corrected Akaike's Information Criterion (AICc)

Includes variables measured within property boundary only. Top-ranked models include all those ($n = 12$) with AICc within 2 of the model with the lowest AICc. Relative variable importance is the sum of 'Akaike weights' over all models including the explanatory variable

Variable in order of importance	Relative variable importance	Model-averaged coefficient	Number inclusions in top-ranked models
Housing density	1	−0.003	12
Distance to major road	1	−0.0005	12
Percentage clearance	1	−0.02	12
Slope	1	0.03	12
Vegetation overhang roof	1	0.5	12
Fuel type	0.67	Factor	9
Vegetation touch structure	0.49	0.07	6
Distance defensible space within property	0.45	−0.0002	5
South-westness	0.36	−0.0007	3
Distance to minor road	0.28	−0.0002	1
D^2 of top-ranked model			0.123

Table 5. Results of multiple regression models of destroyed homes using all possible variable combinations and corrected Akaike's Information Criterion (AICc)

Includes variables measured beyond property boundary. Top-ranked models include all those ($n = 6$) with AICc within 2 of the model with the lowest AICc. Relative variable importance is the sum of 'Akaike weights' over all models including the explanatory variable

Variable in order of importance	Relative variable importance	Model-averaged coefficient	Number inclusions in top-ranked models
Housing density	1	−0.003	6
Distance to major road	1	−0.0005	6
Total distance defensible space	1	−0.004	6
Percentage clearance	1	−0.01	6
Vegetation overhang roof	0.99	0.4	6
Slope	0.99	0.03	6
Fuel type	0.86	Factor	4
South-westness	0.42	−0.0009	2
Distance to minor road	0.36	−0.0009	2
Neighbours' vegetation	0.27	0.08	1
Vegetation touch structure	0.27	0.18	1
D^2 of top-ranked model			0.125

Surrounding matrix

The cover type that most frequently surrounded the structures at the end of the defensible space measurements was urban vegetation, followed by urban vegetation leading into wildland vegetation, and wildland vegetation (Fig. 5). Many structures were equally surrounded by different cover types. There were no significant differences in the proportion of structures destroyed depending on the surrounding cover type. However, a disproportionately large proportion of structures burned (28 *v.* 9% unburned) when they were surrounded by urban vegetation that extended straight into wildland vegetation.

Discussion

For homes that burned in southern Californian urban areas adjacent to non-forested ecosystems, most burned in high-intensity Santa Ana wind-driven wildfires and defensible space increased the likelihood of structure survival during wildfire. The most effective treatment distance varied between 5 and 20 m (16–58 ft), depending on slope and how the defensible space was measured, but distances longer than 30 m (100 ft) provided no significant additional benefit. Structures on steeper slopes benefited from more defensible space than structures on shallow slopes, but the effective treatment was still less than 30 m (100 ft). The steepest overall decline in destroyed structures occurred when mean defensible space increased from 0–7 m (0–25 ft) to 8–15 m (26–50 ft). That, along with the multiple regression results showing the significance of vegetation touching or overhanging the structure, suggests it is most critical to modify vegetation immediately adjacent to the house, and to move outward from there. Similarly, vegetation overhanging the structure was also strongly correlated with structure loss in Australia (Leonard *et al.* 2009).

In terms of fuel modification, the multiple regression models also showed that the percentage of clearance was just as, or more important than, the linear distance of defensible space.

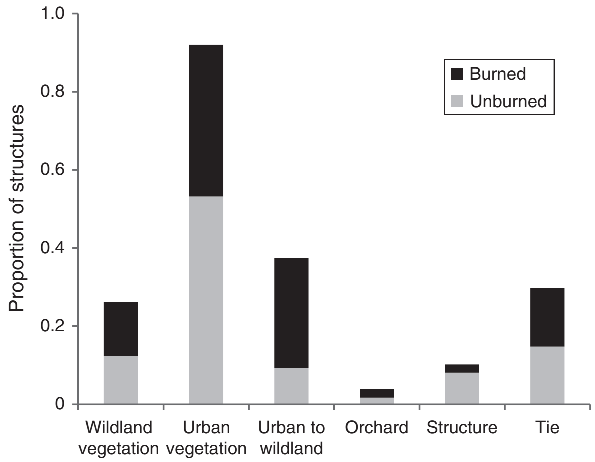

Fig. 5. Proportion of destroyed and unburned structures based on the primary surrounding cover type at the end of defensible space measurements. There were no significant differences in the proportion of burned and unburned structures within cover types ($P = 0.14$). Cover types are defined in Table 1.

However, as with defensible space, percentage clearance did not need to be draconian to be effective. Even on steep slopes, the effective percentage clearance needed on the property was <40%, with no significant advantage beyond that. Although these steep-slope structures benefited more from clearance, they tended to have less clearance than the effective amount, which may be why slope was such an important variable in the multiple regression models. Shallow-slope structures, in contrast, had more clearance on average than was calculated to be effective, suggesting these property owners do not need to modify their behaviours as much relative to people living on steep slopes.

Although the term 'clearance' is often used interchangeably with defensible space, this term is incorrect when misinterpreted to mean clearing all vegetation, and our results underline this difference. The idea behind defensible space is to reduce the continuity of fuels through maintenance of certain distances among trees and shrubs. Although we could not identify the vertical profile of fuels through Google Earth imagery, the fact that at least 60% of the horizontal woody vegetative cover can remain on the property with significant protective effects demonstrates the importance of distinguishing defensible space from complete vegetation removal. Thus, we suggest the term 'clearance' be replaced with 'fuel treatment' as a better way of communicating fire hazard reduction needs to home owners.

The percentage cover of woody shrubs and trees was not evenly distributed across properties, and we did not collect data describing how the cover was distributed. Considering the importance of defensible space and vegetation modification immediately adjacent to the structure, it should follow that actions to reduce cover should also be focussed in close proximity to the structure. The hazard of vegetation near the structure has apparently been recognised for some time (Foote *et al.* 1991; Ramsey and McArthur 1994), but it is not stressed enough, and rarely falls within the scope of defensible space guidelines or ordinances.

In addition to the importance of vegetation overhanging or touching the structure, it is important to understand that ornamental vegetation may be just as, if not more, dangerous than native vegetation in southern California. Although the results showed no significant differences in the cover types in the surrounding matrix, there was a disproportionately large number of structures destroyed (28% burned *v.* 9% unburned) when ornamental vegetation on the property led directly into the wildland. Ornamental vegetation may produce highly flammable litter (Ganteaume *et al.* 2013) or may be particularly dangerous after a drought when it is dry, or has not been maintained, and species of conifer, juniper, cypress, eucalypt, *Acacia* and palm have been present in the properties of many structures that have been destroyed (Franklin 1996). Nevertheless, ornamental vegetation is allowed to be included as defensible space in many codes and ordinances (Haines *et al.* 2008).

One reason that longer defensible space distances did not significantly increase structure protection may be that most homes are not destroyed by the direct ignition of the fire front but rather due to ember-ignited spot fires, sometimes from fire brands carried as far as several km away. Although embers decay with distance, the difference between 30 and 90 m (100 and 300 ft) may be small relative to the distance embers travel under the severe wind conditions that were present at the time of the fires. The ignitability of whatever the embers land on, particularly adjacent to the house, is therefore most critical for propagating the fire within the property or igniting the home (Cohen 1999; Maranghides and Mell 2009).

Aside from roofing or home construction materials and vegetation immediately adjacent to structures (Quarles *et al.* 2010; Keeley *et al.* 2013), the flammability of the vegetation in the property may also play a role. Large, cleared swaths of land are likely occupied at least in part by exotic annual grasses that are highly ignitable for much of the year. Conversion of woody shrubs with higher moisture content into low-fuel-volume grasslands could potentially increase fire risk in some situations by increasing the ignitability of the fuel; and if the vegetation between a structure and a fire is not readily combustible, it could protect the structure by absorbing heat flux and filtering fire brands (Wilson and Ferguson 1986).

The slight increase in proportion of structures destroyed with longer distances of defensible space within parcel boundaries was surprising. However, that increase was not significant in the Chi-square analysis, although there were some significant differences in the pairwise relative risk analysis. Nevertheless, the largest significant effect of defensible space was between the categories of 0–7 m (0–25 ft) to 8–15 m (26–50 ft), and it may be that differences in categories beyond these distances are not highly meaningful or reflect an artefact of the definition of distance categories. These relationships at longer distances are likely also weak compared to the effect of other variables operating at a landscape scale. Although the categorical analysis allowed us to answer questions relative to legal requirements and specific distances, the effective treatment analysis was important for identifying thresholds in the continuous variable.

The multiple regression models showed that landscape factors such as low housing density and longer distances to major roads were more important than distance of defensible space for explaining structure destruction, and the importance of

these variables is consistent with previous studies (Syphard *et al.* 2012, 2013), despite the smaller spatial extent studied here. Whereas this study used an unburned control group exposed to the same fires as the destroyed structures, previous studies accounted for structures across entire landscapes. The likelihood of a fire destroying a home is actually a result of two major components: the first is the likelihood that there will be a fire, and the second is the likelihood that a structure will burn in that fire. In this study, we only focussed on structure loss given the presence of a fire, and the total explained variation for the multiple regression models was quite low at ~12%. However, when the entire landscape was accounted for in the total likelihood of structure destruction, the explained variation of housing density alone was >30% (Syphard *et al.* 2012). One reason for the relationship between low housing density and structure destruction is that structures are embedded within a matrix of wildland fuel that leads to greater overall exposure, which is consistent with Australian research that showed a linear decrease of structure loss with increased distance to forest (Chen and McAneney 2004). That research, however, only focussed on distance to wildland boundaries and did not quantify variability in defensible space or ornamental vegetation immediately surrounding structures. Thus, fire safety is important to consider at multiple scales and for multiple variables, which will ultimately require the cooperation of multiple stakeholders.

Conclusions

Structure loss to wildfire is clearly a complicated function of many biophysical, human and spatial factors (Keeley *et al.* 2009; Syphard *et al.* 2012). For such a large sample size, we were unable to account for home construction materials, but this is also well understood to be a major factor, with older homes and wooden roofs being most vulnerable (Franklin 1996; Cohen 1999, 2000). In terms of actionable measures to reduce fire risk, this study shows a clear role for defensible space up to 30 m (100 ft). Although the effective distances were on average much shorter than 30 m (100 ft), we recognise that additional distance may be necessary to provide sufficient protection to firefighters, which we did not address in this study (Cheney *et al.* 2001). In contrast, the data in this study do not support defensible space beyond 30 m (100 ft), even for structures on steep slopes. In addition to the fact that longer distances did not contribute significant additional benefit, excessive vegetation clearance presents a clear detriment to natural habitat and ecological resources. Results here suggest the best actions a homeowner can take are to reduce percentage cover up to 40% immediately adjacent to the structure and to ensure that vegetation does not overhang or touch the structure.

In addition to defensible space, this study also underlines the potential importance of land use planning to develop communities that are fire safe in the long term, in particular through their reduction to exposure to wildfire in the first place. Localised subdivision decisions emphasising infill-type development patterns may significantly reduce fire risk in the future, in addition to minimising habitat loss and fragmentation (Syphard *et al.* 2013). This study was conducted in southern California, which has some of the worst fire weather in the world and many properties surrounded by large, flammable exotic trees. Therefore, recommendations here should apply to other non-forested ecosystems as well as many forested regions.

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Importance of Roadless Areas in Biodiversity Conservation in Forested Ecosystems: Case Study of the Klamath-Siskiyou Ecoregion of the United States

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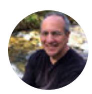
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Importance of Roadless Areas in Biodiversity Conservation in Forested Ecosystems: Case Study of the Klamath-Siskiyou Ecoregion of the United States

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Abstract: *Although many roadless areas on federal lands have been mapped in the United States since the 1970s, there has been little specific research on how and to what degree roadless areas contribute to biodiversity conservation. We examined the ecological attributes of mapped roadless areas for the Klamath-Siskiyou ecoregion of northwestern California and southwestern Oregon (U.S.A.). Attributes examined include special elements (such as natural heritage, serpentine geology, late-seral forests, Port Orford cedar* [Chamaecyparis lawsoniana]*), and key watersheds; elevation and habitat representation; and overall landscape connectivity. We compared designated wilderness to roadless areas, giving special attention to the relative importance of small roadless areas (405–2024 ha). We mapped nearly 500 roadless areas of ≥405 ha. Roadless areas occupied more than twice the land area of wilderness (approximately 27% of the entire ecoregion) and contained approximately 36% of the known occurrences of heritage elements, 37% of the mapped serpentine habitats, 36% of the remaining late-seral forests, 60% of Port Orford cedar strongholds, and 42% of key watersheds for aquatic biodiversity. In addition, roadless areas were composed of significant amounts of low- and mid-elevation sites and a substantial number of the 214 mapped physical-biological habitat types with strong complementarity with designated wilderness. Fragmentation analyses showed that roadless areas contributed to regional connectivity in important ways. Also, small roadless areas were an important component of the roadless-areas conservation assessment. For the Klamath-Siskiyou ecoregion, roadless areas and designated wilderness provide an important foundation upon which to develop a comprehensive regional conservation strategy.*

Importancia de Áreas sin Caminos para la Conservación de la Biodiversidad en Ecosistemas Forestales: Estudio de Caso de la Ecoregión Klamath-Siskiyou, E.U.A.

Resumen: *Aunque se ha mapeado una gran cantidad de superficie sin caminos en tierras federales de los E.U.A. desde la década de 1970, se ha investigado poco sobre como y hasta qué grado las áreas sin caminos contribuyen a la conservación de la biodiversidad. Examinamos los atributos ecológicos de áreas sin caminos en la ecoregión de Klamath-Siskiyou en el noroeste de California y suroeste de Oregon (E.U.A.). Los atributos examinados incluyeron elementos especiales (patrimonio natural, geología de serpentina, bosques serales recientes, cedro [*Chamaecyparis lawsoniana*] y cuencas clave), elevación y representación de hábitat y la conectividad general del paisaje. Comparamos áreas designadas para vida silvestre con áreas sin caminos, poniendo especial atención a la importancia relativa de áreas pequeñas sin caminos (405-2024 ha). Hicimos mapas de cerca de 500 áreas sin caminos ≥ (405 ha). Las áreas sin caminos ocuparon más del doble de la superficie de las áreas de vida silvestre (aproximadamente el 27% de toda la ecoregión). Las áreas sin caminos y contenían aproximadamente el 36% de elementos ancestrales conocidos, el 37% de los habitats serpentinos mapeados, el 36% de los bosques serales recientes, 60% de la población de cedro y 42% de las cuencas*

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clave para la biodiversidad acuática. Adicionalmente, las áreas sin caminos estaban compuestas de cantidades significativas de sitios bajos y de elevación media y cantidades sustanciales de 214 tipos de hábitats fisicobiológicos con una fuerte complementariedad con áreas designadas para vida silvestre. Los análisis de fragmentación mostraron que las áreas sin caminos contribuyeron a la conectividad regional de manera importante. También, las áreas sin caminos pequeñas fueron un componente importante de la evaluación de conservación de las áreas sin caminos. Para la ecoregión Klamath-Siskiyou, las áreas sin caminos y las designadas para vida silvestre son un fundamento importante sobre el que se puede desarrollar una estrategia integral de conservación regional.

Introduction

Natural habitat destruction and fragmentation are the leading causes of the decline and loss of species worldwide and have been a topic of considerable research and review (Harris 1984; Wilcove et al. 1986; Usher 1987; Saunders et al. 1991; Wilson 1992). Fragmentation of forested landscapes has been under particular scrutiny and has been shown to contribute to population declines in many species (Matthiae & Stearns 1981; Harris 1984). The process of deforestation and fragmentation can be complex, having both spatial and temporal components (Zipperer 1993). Roads allow access to pristine areas and fragment native ecosystems into smaller and smaller patches of various sizes and shapes (Dickman 1987; Atkinson & Cairns 1992). Nearly all native ecosystems are destined to resemble ever smaller and more isolated habitat islands as humans continue to encroach on remaining natural habitats (Wilcox 1980).

According to the National Research Council (1997), there are approximately 4 million miles of roadway in the United States. That covers about 1% of the conterminous United States, but the negative ecological effects of the "road-effect zone" are often much greater—18–22% (Forman 2000). Roads constructed to gain access to resources on public lands have been substantial, and in some cases extremely heavy, over the last 50 years. It is difficult to determine the number of all roads currently on public lands, but agency estimates exist. The U.S. Forest Service (USFS) maintains approximately 440,000 miles of roads, nearly 10 times the total length of the interstate highway system.

Roads and the maintenance of roads affect natural terrestrial and aquatic environments in many ways. Increased erosion, air, and water pollution, spread of invasive exotics, road mortality and avoidance, and habitat fragmentation all accompany roads (reviewed by Andrews 1990; Spellerberg 1998; Jones et al. 2000; Trombulak & Frissell 2000). Roads directly fragment natural ecosystems (Reed et al. 1996), but—more importantly—they also provide access to areas, which leads to subsequent human disturbances from activities such as logging, mining, grazing, agriculture, and urban development. These disturbances result in substantial declines in native species and an overall degradation of ecosystem integrity. Roads, deforestation, and fragmentation are intimately related.

On USFS land, two lengthy roadless-area reviews and evaluations (RARE I and RARE II) were performed in the 1970s. The objective of the reviews was to inventory the roadless areas of the national forest system to determine which ones should be considered for wilderness designation as a result of the Wilderness Act of 1964 (Crowell & Cutler 1983). This substantial mapping activity set a minimum size of 2024 ha for designation as "wilderness;" smaller areas would be ineligible for such designation. During the 96th Congress, over half of the 6 million ha of roadless areas recommended by RARE II were declared wilderness. In the early 1980s, with the Reagan administration and new leadership in the 97th Congress, all actions on roadless areas were suspended (Crowell & Cutler 1983). Since that time, roadless areas have been mapped by a wide range of conservationists using a variety of techniques. As large unroaded lands disappear, a minimum size of 405 ha is now being examined for wilderness designation. Geographic information systems (GIS) have been employed extensively in recent years to address this issue. The nongovernmental conservation community has widely promoted roadless areas as important conservation lands (Foreman & Wolke 1992; Noss & Cooperrider 1994), but few studies have evaluated their actual ecological benefits.

On 13 October 1999, President Clinton directed the USFS to provide strong and lasting protection for roadless areas in the national forest system. The President's directive initiated a rule-making process on roadless areas by the USFS that lead to over 517,000 comments from the public (USFS 2000). On 10 May 2000, the USFS released a proposed rule and draft environmental impact statement for which the public provided over 1.1 million additional responses (USFS 2000). The rule-making process for roadless areas marked the biggest public commenting process ever undertaken by the USFS. Given the significance of the impending roadless-area policy, we sought to evaluate the ecological attributes of existing roadless areas in a forest ecoregion, the Klamath-Siskiyou. The ecological attributes we examined included five special elements of conservation concern, two representation evaluations, and regional landscape connectivity.

Methods

Study Area

The Klamath-Siskiyou ecoregion of northwestern California and southwestern Oregon has long been recognized for its outstanding biodiversity (Whittaker 1960; Kruckeberg 1984; DellaSala et al. 1999). The World Conservation Union (IUCN) considers the Klamath-Siskiyou an area of global botanical significance (Wagner 1997), and the World Wildlife Fund chose it as a global 200 ecoregion, meaning that it is of high biodiversity value and under considerable threat (Ricketts et al. 1999). The ecoregion, as we define it, covers over 43,000 km^2, of which approximately 63% is in public ownership (83% of this by the USFS). Nearly 13% of the ecoregion is considered strictly protected, primarily through a number of relatively large, scattered wilderness areas (Fig. 1).

Roadless Area Mapping and Evaluation

As part of a regional conservation assessment (Strittholt et al. 1999), we assembled road data and mapped roadless areas (≥405 ha). We used 1:100,000 road data for the entire ecoregion and 1:24,000 for the public-lands portion. Using the 1:100,000 scaled data, we calculated road density with a 5 × 5 km moving-window operation on density results from a 1 × 1 km fixed grid, which provided a context for the subsequent mapping and evaluation of roadless areas. We mapped roadless areas on existing public lands using a newly developed technique that combines raster and vector GIS operations on 1:24,000–scaled roads data (Strittholt et al. 1999). We included all public lands in mapping and assessing ecological values regardless of agency jurisdiction. In both road density and roadless-area mapping, we excluded trails from the analysis. We used Arc/Info GIS to conduct the computer-mapping analyses.

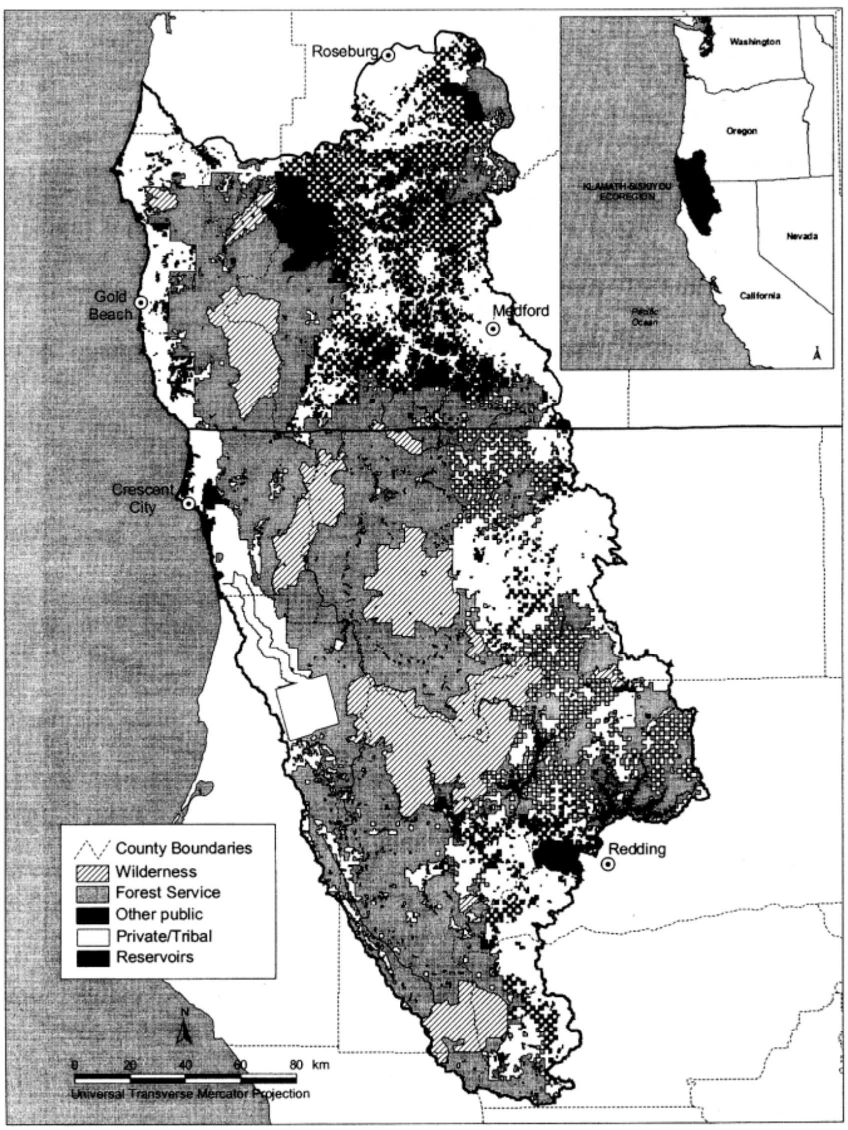

Figure 1. Klamath-Siskiyou primary ownership and existing wilderness areas.

Using all available GIS data layers, we focused on evaluating the ecological or conservation attributes of the remaining roadless areas in the Klamath-Siskiyou ecoregion, and examined large (>2024 ha) and small (405–2024 ha) roadless areas separately. We also subjected the designated wilderness areas to the same data queries and summarized the conservation benefits of adding the roadless areas to the current wilderness system for the ecoregion.

Special Elements

We evaluated the mapped roadless areas in terms of five special-element criteria in the region, including the presence or absence of natural-heritage elements, serpentine geology (of special importance to this region), late-seral forest, high abundances of Port Orford cedar (*Chamaecyparis lawsoniana*) with few or no signs of infestation by a lethal root-rot fungus, and watersheds of aquatic biodiversity importance (or key watersheds) as identified by Forest Ecosystem Management Assessment Team (1993).

Occurrences of natural-heritage elements are known point locations for plant and animal species of special conservation interest, including rare and endangered species. Computer databases are collected and maintained on these species by natural-heritage programs organized by individual U.S. states. For this study, heritage data were obtained from the Oregon Natural Heritage Program and California Natural Diversity Database, and all records since 1970 were included in the analysis. We examined 8793 heritage-element occurrence records, which fell within the Klamath-Siskiyou ecoregion boundary. We treated each record (or point location) equally; rarer species were not weighted. We subdivided all point locations into six categories, primarily by taxonomic group, and overlaid them against data layers for mapped roadless areas and designated wilderness areas.

The Klamath-Siskiyou ecoregion is noted for its abundance of serpentine bedrock geology (13%, or 575,550 ha of the ecoregion). Serpentine is a metamorphic rock upon which ultramafic soils are built. Ultramafic soils are unique in many of their physical and chemical properties: they are low in exchangeable calcium and high in magnesium, and they tend to be deficient in many soil nutrients. Many of these soils contain high levels of heavy metals, such as nickel, chromium, and cobalt, that impede normal plant growth and development (Coleman & Kruckeberg 1999). For these reasons, serpentine geology is one of several known factors that have contributed to species endemism in the Klamath-Siskiyou (DellaSala et al. 1999; Strittholt et al. 1999). Protection of areas of serpentine geology therefore becomes an important regional conservation objective. From existing U.S. Geological Survey state geology maps (1:500,000 map scale) for Oregon and California, we digitized serpentine geology. Using overlay operations, we then calculated the area of serpentine geology in designated wilderness and mapped roadless areas.

Based on interpretation of mid-1990s satellite images, approximately 22% of the Klamath-Siskiyou ecoregion contained late-seral forest (928,356 ha), and 80% of it was on public land (Strittholt et al. 1999). Late-seral forest, which once dominated much of the Pacific Northwest, has been in significant decline since the end of World War II. Many species and natural processes depend upon older forests, and they are of special conservation concern in the Klamath-Siskiyou. We calculated the area of late-seral forest in designated wilderness and mapped roadless areas. Spatial resolution of the forest data was 30 × 30 m and based on two different datasets, one for the Oregon portion of the ecoregion (Cohen et al. 1995) and the other for the California portion (Legacy, unpublished data). Late seral was defined as any forest type more than 100 years of age. Late-seral condition is not equally important among the various forest types found in the Klamath-Siskiyou. For example, some globally imperiled forest types are found in the ecoregion, including white fir (*Abies concolor*), Port Orford cedar, Brewer spruce (*Picea breweriana*), and huckleberry oak (*Quercus vaccinifolia*), and these are of particular concern (DellaSala et al. 1999), but we did not have the data necessary to evaluate this criterion in greater depth. In addition, only a subset of this area could be regarded as old-growth forest, but, again, such detailed information on forest age was not available for the ecoregion.

Port Orford cedar is an important southwestern Oregon–northwest California endemic tree species (Lang 1999) that grows primarily in riparian areas, where it provides channel stabilization, shade for waterways, and microhabitat for numerous aquatic species (Jimerson & Creasy 1997). In an area where migratory species—most notably salmon—make up the bulk of the region's fish fauna, streamside integrity is of paramount importance, and Port Orford cedar is one of the dominant riparian tree species throughout the western sections of the Klamath-Siskiyou ecoregion. These cedars are at risk because of their value in Asian markets (there is no major domestic market for Port Orford cedar) and, more important, from an imported root-rot fungus (*Phytophthora lateralis*) (Lang 1999). This fungus is water-borne and is usually associated with the building and use of roads for logging, mining, and recreation (Jimerson & Creasy 1997). Spores are easily picked up from infected sites and transported to uninfected ones on the tires of vehicles. Infestation usually results in mortality (Zobel et al. 1985). Because of their limited distribution and current threats from management and the exotic root-rot fungus, Port Orford cedar plant communities were classified as G2 (or globally imperiled) by The Nature Conservancy (Grossman et al. 1994).

We mapped Port Orford cedar distribution and root-rot fungus infestation by sixth-field subwatershed, and we evaluated both cedar density and infestation rate against designated wilderness and mapped roadless areas. Watersheds (or catchments) include all lands enclosed by a continuous hydrologic surface-drainage divide and lying upslope from a specified point on a stream (Maxwell et al. 1995). Watersheds are hierarchical and can be modeled from digital elevation models (Maidment & Djokic 2000) or delineated from hydrology and contour maps. The U.S. Geological Survey developed its own hierarchical watershed-delineation system based on a hydrologic unit code-naming convention. In this system, every two digits of a hydrologic unit code correspond to each level in the hydrologic unit system: A 12-digit code identifies six nested subwatersheds, the sixth pair of digits representing the finest subwatershed.

Of the 1135 sixth-field subwatersheds mapped for the Klamath-Sisikyou ecoregion, 215 contained cedar data, all on USFS public lands. We assigned ordinal scores (1-5, with 5 representing the highest cedar density and lowest level of infestation) to cedar density and degree of infestation separately, added them together to obtain a composite score, and assigned the score to each subwatershed (Strittholt et al. 1999). Of the 903,559 ha (n = 215 subwatersheds) containing cedar and infestation data, 19% (179,430 ha, n = 40 subwatersheds) scored low, meaning they had a combination of low cedar density and high infestation. Approximately 22% of the subwatersheds (199,389 ha, n = 48) scored high (i.e., high cedar density and low infestation). We classified the remaining subwatersheds (58% or 524,740 ha, n = 127) as medium (i.e., moderate cedar density and moderate infestation). We then overlaid the designated wilderness and mapped roadless areas on these composite-score categories and summarized the results.

Of the 1135 sixth-field subwatershed basins mapped for the Klamath-Siskiyou ecoregion (Strittholt et al. 1999), 333 were previously identified as key watersheds by fisheries biologists of the Forest Ecosystem Management Assessment Team (1993). Key watersheds are those believed to be of special importance for aquatic biodiversity, with particular emphasis on the salmonid species and stocks at risk throughout the Pacific Northwest. We determined the number and area of key watersheds in designated wilderness and mapped roadless areas.

Representation

We measured elevation bands and natural habitat types to evaluate the contribution roadless areas make to ecosystem representation. Elevations in the Klamath-Siskiyou ecoregion range from sea level to 2700 m. To simplify the assessment, we pooled the elevation values from the U.S. Geological Survey digital elevation models (DEMs) into 305-m intervals, which resulted in nine elevation bands (or classes). We then calculated the area of each elevation class occupied by designated wilderness and roadless areas. We further simplified the results into classes of low (sea level-915 m), medium (915-1525 m), and high (>1525 m) elevation.

We also examined representation by merging physical and biological habitat types. We combined 19 physical habitat types (Vance-Borland 1999) with 26 natural vegetation types, resulting in 214 distinct natural habitat classes (Strittholt et al. 1999). We calculated degree of representation for each of the 214 natural habitat types according to designated wilderness and mapped roadless areas, and we assigned each to one of four possible representation categories: (1) >50% represented, (2) 25-50% represented, (3) <25% represented, and (4) none represented.

Regional Landscape Connectivity

We examined landscape connectivity of the ecoregion with Arc/Info and FRAGSTATS, a versatile fragmentation software. As with the other analyses, we considered the designated wilderness, and mapped large and small roadless areas separately. FRAGSTATS calculates a large number of fragmentation metrics (or indices) on three distinct levels, patch, class, and landscape (McGarigal & Marks 1995). We focused the assessment on six class-level metrics, including (1) class area, (2) core percent of landscape, (3) mean core area, (4) total core area index, (5) mean nearest-neighbor distance, and (6) edge density (see Appendix for definitions).

We applied the fragmentation metrics in two ways. First, we ran FRAGSTATS on a merged raster file containing designated wilderness and the two roadless-area size categories, treating each as a distinct landscape class. Second, we ran FRAGSTATS on a series of map layers to ascertain the level of connectivity each provided. One map contained only designated wilderness, a second contained designated wilderness and large roadless areas (>2024 ha), and a third contained designated wilderness and all roadless areas (≥405 ha). In each case, we assigned all areas of interest as the same class. By analyzing these three cases, we tested the level of landscape connectivity for the current condition (existing protected areas only), a landscape with the existing protected areas plus all large roadless areas, and existing protected areas plus all large and small roadless areas. We put the edge buffer distance at 90 m for all trials, based on the mean edge-effect distance determined for changes in the physical environment at the edges of Pacific Northwest forests (Chen et al. 1992).

Results

Roads and Roadless Areas

The total road length of all surface types for the Klamath-Siskiyou mapped at 1:100,000 scale was 44,522 km. The road data available at 1:24,000 scale (approximately 75% of the ecoregion), added 42% more road length than the data for coarser roads over the same area. Road-density results calculated from the coarser dataset showed sizeable, somewhat isolated regions of low road density. Designated wilderness was at the center of these larger areas.

Designated wilderness in the ecoregion covered 533,700 ha in several large, scattered patches. We mapped nearly 500 roadless areas of ≥405 ha on public lands of the Klamath-Siskiyou ecoregion area, which totaled 1,186,422 ha, approximately 27% of the ecoregion and more than twice the area of designated wilderness (Fig. 2). We mapped 131 large roadless areas covering 871,815 ha and 367 smaller roadless areas covering 314,607 ha, or 26% of the total roadless area. The USFS lands accounted for most (92%) of the roadless area mapped, followed by Bureau of Land Management (7.6%) and other public lands (e.g., National Park Service, 0.4%).

Special Elements

Roadless areas contained nearly four times more heritage elements than designated wilderness areas; the largest gains occurred in the plant and vertebrate categories (Table 1). In general, roadless areas captured approximately 36% of all known heritage elements. When added to those within designated wilderness, the total number increased to 3816 records, or 43% of all known records (Table 1). Small roadless areas accounted for 931 element records, or 29% of the total roadless-area records, adding substantially more plant, vertebrate, and invertebrate occurrences.

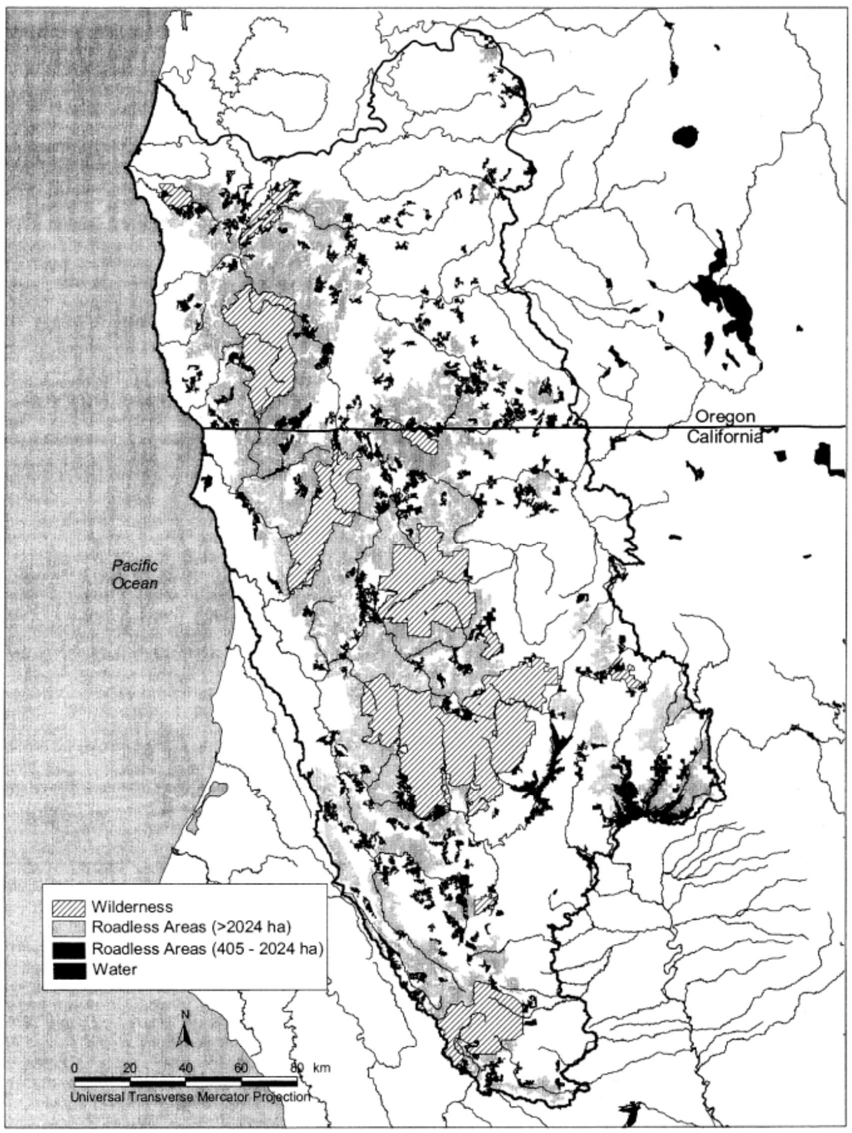

Figure 2. Large (>2024 ha) and small (405–2024 ha) roadless areas within the Klamath-Siskiyou ecoregion.

Table 1. Frequency of heritage-element occurrence records for existing wilderness and mapped roadless areas within the Klamath-Siskiyou ecoregion.

Element category	*No. of records*	*No. in wilderness*	*No. in roadless areas (≥405 ha)*	*No. in roadless areas (405–2,024 ha)*	*Wilderness (%)*	*Roadless areas (%)*	*Combined (%)*
Plants	3837	389	1306	341	10.1	34.0	44.1
Vertebrates	4652	212	1749	562	4.6	37.6	41.2
Invertebrates	132	2	80	26	1.5	60.6	62.1
Community	8	0	6	2	0.0	75.0	75.0
Aquatic	6	0	0	0	0.0	0.0	0.0
Special feature	158	36	36	0	22.8	22.8	45.6
Total	8793	639	3177	931	7.3	36.1	43.4

We found 209,051 ha (36%) of the existing serpentine geology in the Klamath-Siskiyou ecoregion in roadless areas. The contribution from the smaller roadless-area class was 47,090 ha, 22% of the roadless-area total. Designated wilderness areas captured 100,170 ha (17%) of serpentine geology in the ecoregion. Adding all remaining roadless areas to designated wilderness would more than triple the area of serpentine currently protected, bringing the ecoregion protection total to approximately 54%.

Of the 928,356 ha of late-seral forest mapped for the Klamath-Siskiyou, 337,180 ha (36%) were in roadless areas. Small roadless areas accounted for 93,508 ha, or 28% of the roadless-area total. In comparison, designated wilderness areas contained only 16% (149,386 ha) of the remaining late-seral forest. We found no difference in the average density of late-seral forest between wilderness and roadless areas: both contained approximately 28% late-seral forest. The smaller roadless-area class showed a slightly higher percentage of late-seral cover as a group (30%). Of course, individual wilderness areas and roadless areas showed higher and lower percentages of late-seral forest cover. Adding roadless areas into protection brought the total area of late-seral forest protected in the ecoregion to 52% (486,566 ha).

In the Port Orford cedar assessment, roadless areas contained far more subwatershed areas classified as having either low or medium cedar composite scores than did designated wilderness areas (Table 2). These areas require special management, such as road closure and removal, to stop the spread of the root-rot fungus. The area identified as having a high cedar composite score—high cedar density and low fungus infestation—was included largely in designated wilderness or roadless areas and is the remaining stronghold on the public-land portion of the region for protecting cedar and the many ecosystem services this species provides.

Key watersheds covered 1,157,812 ha, or 26% of the Klamath-Siskiyou ecoregion. Over 42% of the key watershed area (491,954 ha) was also roadless. The contribution from smaller roadless areas was 89,506 ha, or 18% of the roadless area total. Of the 333 key watersheds, 190 (57%) were 80% contained (54 were completely contained) within wilderness and roadless areas. Only 13 (4%) contained no roadless area. Key watershed in designated wilderness covered 303,054 ha (26%). The combined key watershed total for both wilderness and roadless areas was 795,008 ha, 68% of all key watershed area.

Representation

Wilderness and roadless areas showed a marked difference in elevation representation (Fig. 3). When compared to wilderness, roadless areas captured much more

Table 2. Scored results of Port Orford cedar density and infestation by root-rot fungus, organized by subwatershed (n = 215) compared with wilderness and roadless areas within the Klamath-Siskiyou ecoregion.

*Cedar composite score categories**	*Total area (ha)*	*Wilderness (ha)*	*Roadless (ha)*	*Wilderness (%)*	*Roadless (%)*	*Combined (%)*
Low (n = 40)	179,430	8,948	79,948	5.0	44.5	49.5
Medium (n = 127)	524,740	88,166	228,352	16.8	43.5	60.3
High (n = 48)	199,389	60,927	113,933	30.5	57.1	87.6
Total (n = 215)	903,559	158,041	422,233	17.5	46.7	64.2

**For details on cedar composite scoring method resulting in categories of low (low cedar density and high infestaion), medium, and high (high cedar density and low infestation), see Strittholt et al. (1999).*

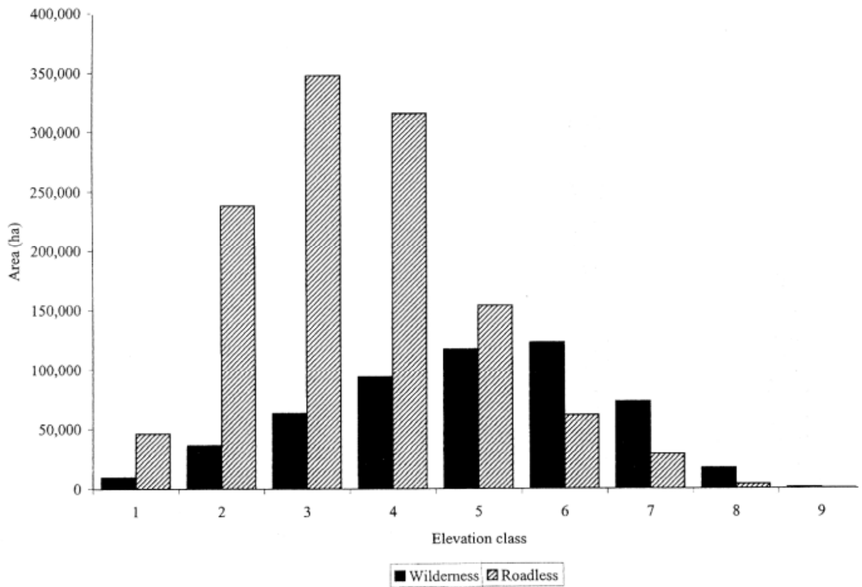

Figure 3. Representation of elevation classes by wilderness and mapped roadless areas for the Klamath-Siskiyou ecoregion. Elevation classes are reported in approximately 305-m intervals from mean sea level to 305 m (elevation class 1).

of the low- and medium-elevation classes (<1525 m) (Table 3). The small roadless areas made up about the same proportion (approximately 26%) of each elevation class, providing additional representation of low and medium elevations. Roadless areas did proportionally better than designated wilderness at representing low and medium elevations. Most of the existing high-elevation sites in the Klamath-Siskiyou were in designated wilderness.

Of the 214 combined physical-biological habitat types described and mapped for the Klamath-Siskiyou, roadless areas contained a wider array of habitat types than designated wilderness. Roadless areas also contained many different habitat types than designated wilderness, with 96 new types (45%) represented at the ≥25% level (Fig. 4). When combined, designated wilderness and roadless areas complimented each other well, with 64% (138/214) of the classes represented at the ≥25% representation level. Although not visible in Fig. 4, smaller roadless areas made important contributions to 148 different habitat types, including 24 not found in any designated wilderness or large roadless areas. In addition, 54 habitat types were highly concentrated in smaller road-

Table 3. Representation of elevation classes by wilderness and mapped roadless areas for the Klamath-Siskiyou ecoregion.

Elevation class[a]	*Total area (ha)*	*Wilderness (ha)*	*Roadless (ha)*	*Combined (ha)*	*Wilderness (%)*	*Roadless (%)*	*Combined (%)*[b]	*Roadless contribution (%)*
Low								
1	351,765	9,500	46,182	55,682	2.7	13.1	15.8	82.9
2	934,857	36,000	238,407	274,407	3.8	25.5	29.3	86.9
3	1,145,382	62,900	347,664	410,564	5.5	30.3	35.8	84.7
subtotal	2,432,004	108,400	632,253	740,653	4.4	26.0	30.4	85.4
Medium								
4	900,862	93,800	315,876	409,676	10.4	35.1	45.5	77.1
5	523,358	117,100	153,840	270,940	22.4	29.4	51.8	56.8
subtotal	1,424,220	210,900	469,716	680,616	14.8	33.0	47.8	69.0
High								
6	283,128	122,900	61,476	184,376	43.4	21.7	65.1	33.3
7	124,405	72,900	28,989	101,889	58.6	23.3	81.9	28.4
8	21,449	17,500	3,949	21,449	81.6	18.4	100.0	18.4
9	1,716	1,100	400	1,500	64.1	23.3	87.4	26.7
subtotal	430,698	214,400	94,814	309,214	49.8	22.0	71.8	30.7
Total	4,286,922	533,700	1,196,783	1,730,483				

[a]*Elevation classes are reported in approximately 305-m intervals from mean sea level, to 305 m (elevation class 1), to the highest elevation band of 2440 m (elevation class 9).*
[b]*Wilderness plus roadless totals.*

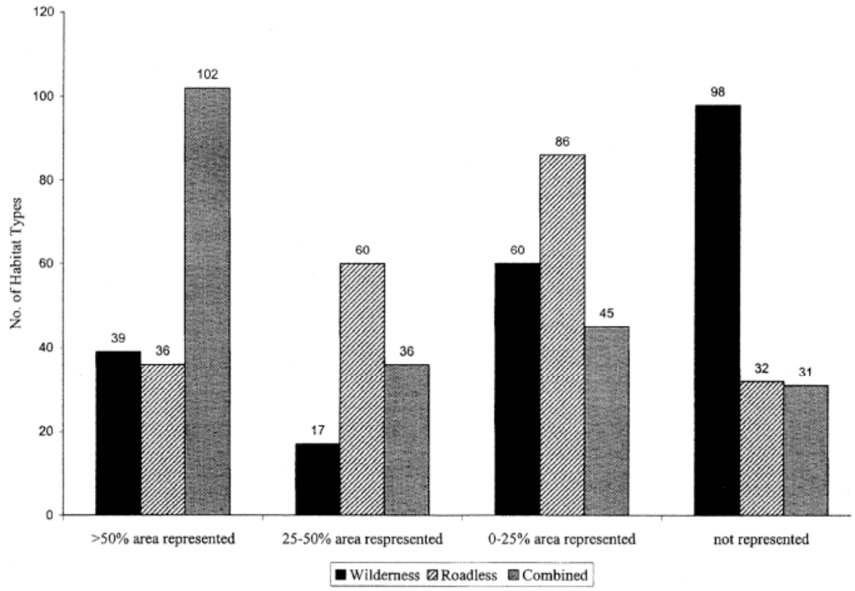

Figure 4. Physical-biological habitat representation (n = 214) for wilderness and mapped roadless areas for the Klamath-Siskiyou ecoregion.

less areas. More than 50% of the total area of these habitat types was located in this roadless-area size class alone.

Regional Landscape Connectivity

Treating each component of a single landscape—wilderness, larger roadless areas, and smaller roadless areas—as a separate class produces a number of interesting but not unexpected results (Table 4). Designated wilderness areas in the Klamath-Siskiyou are large and scattered, centering on the major high-elevation portions of the region. Based on this description, the wilderness areas showed some expected fragmentation results: large values for total core area, mean core area, and mean nearest-neighbor distance. Edge density was also predictably low. Compared to wilderness areas, the larger roadless areas showed a substantial decrease in the mean core-area measurement (26,736 ha) and in the mean nearest-neighbor distance (5356 m). Core percent of landscape and edge density were higher in larger roadless areas than in any other class. The higher edge density signifies that many of the patches, while large, are relatively convoluted. Total core area index was lower than for wilderness, but still high.

Mean core area for the smaller roadless areas was predictably low, but this group still showed a relatively high total core-area index value. Based on a lower edge-density value, smaller roadless areas were less convoluted in shape than the larger size class. This class accounted for nearly 15% of all core area among the three classes examined.

The fragmentation results for each of the three landscape scenarios—wilderness only, wilderness plus large roadless areas, and wilderness plus all roadless areas of ≥405 ha—showed predictable increases in class area and edge density as more land parcels were given theoretical protection status (Table 5). Because only one class type was used for each landscape scenario, the values for core percentage of landscape and total core-area index were identical. Going from the least-protected to the most-protected scenarios, declines were observed for core percentage of landscape; total core-area index, mean core area, and mean nearest neighbor, and increases were observed in class area and edge density. The substantial declines in mean nearest-neighbor values

Table 4. Fragmentation results for the Klamath-Siskiyou ecoregion calculated for a single landscape containing wilderness, small roadless areas (405–2024 ha), and large roadless areas (>2024 ha) treated as separate classes.

Fragmentation index	*Wilderness*	*Roadless areas (>2024 ha)*	*Roadless areas (405-2024 ha)*
Class area (ha)	529,619	872,013	315,297
Core percentage of landscape	30	45	15
Total core area index (%)	97	88	80
Mean core area (ha)	32,178	5,442	665
Mean nearest-neighbor distance (m)	6,459	1,103	864
Edge density (m/ha)	3.88	9.13	5.42

Table 5. Fragmentation results for the Klamath-Siskiyou ecoregion, calculated for three separate landscapes: wilderness only, wilderness + large roadless areas, and wilderness + all roadless areas.

Fragmentation index	*Wilderness only*	*Wilderness + roadless areas (>2024 ha)*	*Widlerness + roadless areas (≥405 ha)*
Class area (ha)	529,619	1,401,632	1,716,929
Core percentage of landscape	97	92	90
Total core area index (%)	97	92	90
Mean core area (ha)	32,178	12,087	3,554
Mean nearest-neighbor distance (m)	6,459	1,370	225
Edge density (m/ha)	3.88	11.50	14.46

provide the best evidence out of these indices that adding roadless areas to the current distribution of protected areas would provide improved regional landscape connectivity. Including the smaller roadless areas added significantly to regional landscape connectivity.

Discussion

The extensive literature on the importance of intact natural habitats makes a compelling case for the potential role of roadless areas as refugia for native biodiversity and as areas crucial to forest integrity and function. Equally impressive is the mounting body of evidence showing the ecological costs of roads. Although some species may benefit from the physical changes and fragmentation caused by roads, their overall and cumulative effect in natural forest landscapes is negative, often seriously so. The suggestion that research on the effects of roads on natural ecosystems is inconclusive (e.g., Heinz Center 1999) is largely unsupported by the literature. Thus, the exclusion of roads from fragmentation assessments presents an incomplete picture of the effects of one of the most predominate anthropogenically induced changes to forested ecosystems in North America. For a variety of reasons, some of which are ecological, roadless areas are once again on the political agenda.

On numerous scientific grounds, our analyses strongly support protecting roadless areas in the Klamath-Siskiyou ecoregion. Roadless areas contained many known locations of species of concern, including rare and endangered species, many more than could be explained by additional land area alone, and the contribution by small roadless areas was noteworthy. That is not to say that roadless areas contained the majority of the known locations of rare species or even all of the known biological hotspots in the ecoregion. For example, there are a few well-known biological hotspots largely within the roaded public and private lands of the Klamath-Siskiyou (Noss et al. 1999; Strittholt et al. 1999). If one conservation goal is to protect most or all of the known rare species, roadless areas contribute only partially to this objective. Additional conservation measures and approaches are needed to achieve this goal in the Klamath-Siskiyou. In addition, our assumption is that rare species located in designated wilderness and roadless areas would be better protected than if they existed under other management practices. This assumption would not always be substantiated by the natural history of a particular species and the individual site conditions under which it lives; a species-specific needs assessment is required to make an accurate determination. We also suspect that many more rare species are present in roadless areas than are presently known because of the remoteness of roadless areas, a lack of organized biological surveys, and a current sampling bias toward areas with road access. The actual importance of roadless areas with regard to this criterion may actually be higher than the current data analysis indicates.

The amount of late-seral forest in roadless areas is considerable, but the percentage is about that found in wilderness areas (both are around 28% late seral). What is important is that these forests are part of relatively large, intact blocks of habitat representing important ecological values. Larger patches of forest that can support a wider range of species, including those requiring large home ranges, are more secure from human-induced effects and are large enough to allow natural processes such as fire to operate without human interference. As with heritage elements, although roadless areas offer some protection to late-seral forests, not all important late-seral stands are included in roadless areas; This includes particular stands of late-seral forest within the Klamath-Siskiyou.

The contribution roadless areas made to Port Orford cedar conservation was particularly significant. The roadless area and designated wilderness components of the best-condition category were high; when combined, they accounted for nearly 88% of the best remaining areas of uninfected cedar. Because root-rot fungus is easily spread by the tires of vehicles moving on roads from infected watersheds to uninfected ones, maintaining large roadless areas is one important strategy for stopping the spread of the disease and therefore protecting this important tree species. At this level of analysis, smaller roadless areas do not seem to be as important for cedar as larger ones; but as in the case of key watersheds, the position of these roadless areas may be as important as the total area they protect.

Our representation analyses suggest that an important role of roadless areas is adding low and medium elevations and many combined physical-biological habitat classes to the reserve network. Other researchers have shown that many of the existing protected areas (especially wilderness areas) are concentrated on higher elevations (Harris 1984; Scott et al. 1993; Strittholt & Frost 1997), and the assumption could be made that most of the remaining roadless areas are much the same. Based on our assessment, this was not the case in the Klamath-Siskiyou. Roadless areas contributed significantly to low and medium elevations. Lower elevations contained most of the region's biological richness (DellaSala et al. 1999), and roadless areas were well represented at these elevations. If full representation of existing habitat types is a regional conservation goal, roadless areas contribute significantly to this objective, but conservation actions on roaded public and private lands need to be taken as well (Strittholt et al. 1999).

The way wilderness and roadless areas complimented one another in representing the various physical-biological classes defined for the Klamath-Siskiyou ecoregion is an important finding for regional conservation. As one would expect from the elevation results, wilderness areas contained large portions of the physical-biological habitat types found at the higher elevations. Wilderness areas are concentrated on most of the forested and nonforested ecosystems at high elevations, including most of the red fir (*Abies magnifica*) and white fir (*A. concolor*) forests and much of the higher Jeffrey pine (*Pinus jeffreyi*), ponderosa pine (*P. ponderosa*), montane-hardwood conifer, and Klamath mixed-conifer forest types. Roadless areas added additional area to some of these habitat types, and—more important—picked up different physical zones for the same plant community types as well as totally new habitat types, including forests of Douglas-fir, montane hardwood, and Sierra mixed conifers growing under various physical zones defined and mapped for the region (Strittholt et al. 1999; Vance-Borland 1999). Smaller roadless areas were particularly important in contributing to 54 different habitat types and were exclusively responsible for capturing 24 habitat types not encountered at all in wilderness or the larger roadless areas.

Our fragmentation results demonstrated the importance of roadless areas to regional connectivity. The larger roadless areas contributed the most, but protecting the smaller roadless areas would contribute to the protection of the overall landscape connectivity in this ecoregion, and it would do so while maintaining a high level of core interior habitat. In the Klamath-Siskiyou, wilderness and roadless-area patches are positioned in such a way that, when they are protected, once-isolated habitat islands become linked throughout the heart of the ecoregion. In the future, the issue of inter-regional linkage will have to be addressed, and roadless areas could provide the nuclei around which future corridors targeting individual species or ecosystem processes could be designed.

Small roadless areas in the Klamath-Siskiyou did not equally address all conservation issues examined, but they address many of them significantly, especially heritage elements, late-seral forests, elevation representation, habitat-type representation, and overall landscape connectivity. Based on the results, small roadless areas warrant inclusion in future assessment and planning of roadless areas. If small roadless areas are important in the Klamath-Siskiyou, where a high proportion of larger roadless areas exist, then they are likely to be even more important in regions where the majority of roadless areas are small, such as the southern Appalachians.

We considered the importance of roadless areas as a whole to the conservation of regional biodiversity. The next logical step in the roadless-areas assessment for the Klamath-Siskiyou and elsewhere will be to evaluate each roadless area individually to measure its relative ecological attributes. To some extent, this has already been done for the Klamath-Siskiyou (Strittholt et al. 1999), and an expansion of this assessment is currently underway that will include the national forests of western Washington, Oregon, and northern California. The better the ecological attributes of each roadless area are quantified, the better regional conservation can be planned. That is not to say that the ecological criteria we examined (or ecological criteria in general) are the only values roadless areas possess, and ecological reasons are not the only criteria for evaluation. Other values humans place on wild places, such as recreational, aesthetic, educational, economic, and scientific values, are all important reasons for establishing greater environmental protection of roadless areas.

The opportunity to protect some of the largest remaining forest habitat in the United States may be a fleeting one. Time is of the essence. Based on agency inventories in 1979, the USFS had about 32 million ha (42%) of roadless areas. Since the 1979 RARE II process, the estimated rate of loss of roadless areas has been about 400,000 ha each year nationally (Foreman & Wolke 1992). Under the new roadless-area rule making (USFS 2000), an average of 95 km of new roads each year are planned for roadless country through 2004.

Providing protection of large natural areas is an indispensable component of an overall conservation strategy (Noss & Cooperrider 1994), but a protected-area network cannot be compoed primarily of poor-quality conservation lands (Hall 1988). Protected-area networks must be carefully designed to maintain regional biodiversity, ecosystem processes, and overall ecosystem integrity. The existing roadless areas on federal lands in the United States are the remaining pieces of the natural landscape that once covered the nation. Most biologically productive lands are already developed, and opportunities are limited to design effective protected-area networks without the necessity for substantial restoration. Roadless areas provide the remaining building blocks toward this end, so it is important to understand the contribution these areas make to an overall conservation strategy. Without this information, there is little hope for effective conservation or informed land-use planning.

The roadless areas in the Klamath-Siskiyou, if protected, would provide a wide range of ecological attributes important in this ecoregion, which is why roadless areas became the cornerstone of a recent reserve design for the ecoregion (Noss et al. 1999; Strittholt et al. 1999). Our results underscore the significance of roadless areas for regional conservation, but roadless areas alone cannot provide all the ecological elements needed to maintain regional biodiversity. Thus, it will be important to manage roadless areas responsibly and restore them where necessary (DellaSala et al. 1999; Strittholt et al. 1999). We do not know if roadless areas in other regions will yield similar results, particularly since, according to the USFS (2000), 60% of roadless areas on national forests are at elevations above 1524 m. But we believe that our methodologies may be applied elsewhere, particularly as the inventory of small roadless areas takes on greater significance in federal-lands policy.

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Appendix

Descriptions of the six fragmentation metrics used in the Klamath-Siskiyou landscape connectivity analysis.*

Metric	*Description*
Class area (ha)	sum of the areas of all patches of the corresponding patch type (range: ≥0, without limit)
Core percentage of landscape	sum of the core areas of each patch of the corresponding patch type, divided by total landscape area (range: ≥0, ≤ 100)
Mean core area (ha)	sum of the core areas of each patch of the corresponding patch type, divided by the number of patches of the same type (range: ≥0, without limit)
Total core-area index (%)	sum of the core areas of each patch of the corresponding patch type, divided by the sum of the areas of each patch of the same type, multiplied by 100 (range: ≥ 0, ≤ 100)
Mean nearest-neighbor distance (m)	sum of the distance to the nearest neighboring type of the same type, based on nearest edge-to-edge distance for each patch of the corresponding patch type, divided by the number of patches of the same type (range: ≥0, without limit)
Edge density (m/ha)	sum of the lengths of all edge segments involving the corresponding patch type, divided by the total landscape area (range: ≥0, without limit)

* *Adapted from McGarigal and Marks (1995).*

Frontiers in Ecology and the Environment

The forgotten stage of forest succession: early-successional ecosystems on forest sites

Mark E Swanson, Jerry F Franklin, Robert L Beschta, Charles M Crisafulli, Dominick A DellaSala, Richard L Hutto, David B Lindenmayer, and Frederick J Swanson

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The forgotten stage of forest succession: early-successional ecosystems on forest sites

Mark E Swanson[1*], Jerry F Franklin[2], Robert L Beschta[3], Charles M Crisafulli[4], Dominick A DellaSala[5], Richard L Hutto[6], David B Lindenmayer[7], and Frederick J Swanson[8]

Early-successional forest ecosystems that develop after stand-replacing or partial disturbances are diverse in species, processes, and structure. Post-disturbance ecosystems are also often rich in biological legacies, including surviving organisms and organically derived structures, such as woody debris. These legacies and post-disturbance plant communities provide resources that attract and sustain high species diversity, including numerous early-successional obligates, such as certain woodpeckers and arthropods. Early succession is the only period when tree canopies do not dominate the forest site, and so this stage can be characterized by high productivity of plant species (including herbs and shrubs), complex food webs, large nutrient fluxes, and high structural and spatial complexity. Different disturbances contrast markedly in terms of biological legacies, and this will influence the resultant physical and biological conditions, thus affecting successional pathways. Management activities, such as post-disturbance logging and dense tree planting, can reduce the richness within and the duration of early-successional ecosystems. Where maintenance of biodiversity is an objective, the importance and value of these natural early-successional ecosystems are underappreciated.

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Severe natural disturbances – such as wildfires, windstorms, and insect epidemics – are characteristic of many forest ecosystems and can produce a "stand-replacement" event, by killing all or most of the dominant trees therein (Figure 1). Typically, limited biomass is actually consumed or removed in such events, but many trees and other organisms experience mortality, leaving behind important biological legacies (structures inherited from the pre-disturbance ecosystem; Franklin *et al.* 2000), including standing dead trees and downed boles (tree trunks; Franklin *et al.* 2000). Such legacies provide diverse physical/biological properties and suitable microclimatic conditions for many species. Thereafter, species-diverse plant communities develop because substantial amounts of previously limited resources (light, moisture, and nutrients) become available. These emerging plant communities create additional habitat complexity and provide various energetic resources for terrestrial and aquatic organisms.

In a nutshell:

- Naturally occurring, early-successional ecosystems on forest sites have distinctive characteristics, including high species diversity, as well as complex food webs and ecosystem processes
- This high species diversity is made up of survivors, opportunists, and habitat specialists that require the distinctive conditions present there
- Organic structures, such as live and dead trees, create habitat for surviving and colonizing organisms on many types of recently disturbed sites
- Traditional forestry activities (eg clearcutting or post-disturbance logging) reduce the species richness and key ecological processes associated with early-successional ecosystems; other activities, such as tree planting, can limit the duration (eg by plantation establishment) of this important successional stage

The ecological importance of early-successional forest ecosystems (ESFEs) has received little attention, except as a transitional phase, before resumption of tree dominance. In forestry, this period is often called the "cohort re-establishment" or "stand initiation" stage, with attention obviously focused on tree regeneration and the re-establishment of closed forest canopies (Franklin *et al.* 2002). Ecological studies have focused primarily on plant-community development and the needs of selected animal (mostly game) species, and not on the diverse ecological roles of ESFEs.

Here, we highlight important features of ESFEs, including their role in sustaining ecosystem processes and biodiversity, so that they may be appropriately considered by resource managers and scientists, and included within management/research programs dedicated to maintaining these functions, particularly at larger spatio-temporal scales. Most published examples focus on sites in western North America, but ESFEs are important elsewhere (Angelstam 1998; DeGraaf *et al.* 2003). We also discuss how traditional forestry practices, such as clearcutting, tree planting, and post-disturbance logging, can affect early-successional communities.

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Figure 1. *Stand-replacement disturbance events in forests create large areas free of tree dominance and rich in physical and biological resources, including legacies of the pre-disturbance ecosystem.*

■ Early-successional ecosystems on forest sites

Initial conditions after stand-replacing forest disturbances vary generically, depending on the type of disturbance; this includes the types of physical and biological legacies available. For example, aboveground vegetation may be limited immediately after the disturbance, as in the case of severe wildfires or volcanic eruptions. Conversely, intact understory communities may persist where forests have been blown down by severe windstorms. Spatial heterogeneity in conditions is characteristic, given that disturbances vary greatly in the amount of damage they cause (Turner *et al.* 1998). For instance, severe wildfires frequently include substantial areas of unburned as well as low to medium levels of mortality, creating variability in shade, litterfall, soil moisture, seed distribution, and other factors.

We define ESFEs as those ecosystems that occupy potentially forested sites in time and space between a stand-replacement disturbance and re-establishment of a closed forest canopy. These ecosystems undergo compositional and structural changes (succession) during their occupancy of a site. Changes begin immediately post-disturbance, as a result of the activities of surviving organisms (eg plants, animals, and fungi), including plant growth and seed production. Developmental processes are enriched by colonization of flora and fauna from outside the disturbed area. Successional change is often characterized by progressive dominance of annual and perennial herbs, shrubs, and trees, although all of these species are typically represented throughout the entire sequence of forest stand development (or sere; Halpern 1988).

The ESFE developmental stage ends with re-establishment of tree cover that is sufficiently dense to suppress and often eliminate many smaller shade-intolerant plants (Franklin *et al.* 2002). Consequently, the duration of ESFEs varies inversely with rapidity of tree regeneration and growth, which, in turn, depend on such variables as tree propagule availability, conditions affecting seedling or sprout establishment, and site productivity. ESFE longevity after natural disturbances is therefore highly variable.

Development of a closed forest canopy may require a century or more in areas with limited seed sources, harsh environmental conditions, severe shrub competition (in some instances), or combinations thereof (Hemstrom and Franklin 1982). For example, tree canopy closure after wildfire in the Douglas fir region of western North America often requires several decades (Poage *et al.* 2009), but can occur much more rapidly when canopy seedbanks are abundant (eg Larson and Franklin 2005). Closed forest canopies may develop quickly in forests dominated by trees with strong sprouting ability (eg many angiosperms) or when windstorms "release" understories of shade-tolerant tree seedling banks by removing all or most of the overstory (Foster *et al.* 1997).

■ Attributes of early-successional ecosystems

After severe disturbances, forest sites are characterized by open, non-tree-dominated environments, but have high levels of structural complexity and spatial heterogeneity and retain legacy materials.

Environmental conditions

Removal of the overstory forest canopy during disturbances dramatically alters the site's microclimate, including light regimes. These changes lead to increased exposure to sunlight, more extreme temperatures (ground and air), higher wind velocities, and lower levels of relative humidity and moisture in litter and surface soil. Shifts in these environmental metrics favor some species, while creating suboptimal or intolerable conditions for others. For example, post-disturbance plant community composition, cover, and physiognomy are altered as shade-tolerant understory herbs are largely displaced by shade-intolerant and drought-tolerant species. New substrates deposited by floods or volcanic eruptions may lack nutrients, provide additional water-holding capacity, or have high albedo, all of which favor shifts in plant communities.

Survivors

Organisms (in a variety of forms) that survive severe disturbances are extremely important for repopulating and

restoring ecosystem functions in the post-disturbance landscape. Even in severely disturbed areas, organisms may survive as individuals (mature or immature) or as reproductive structures (eg spores, seeds, rootstocks, and eggs), which become in situ propagule sources. For example, after the 1980 volcanic eruption of Mount St Helens (Washington State), most pre-eruption flora and many fauna (especially aquatic and burrowing terrestrial species) survived within the blast zone through several different mechanisms (Dale *et al*. 2005).

Surviving organisms are also often vital for the prompt re-establishment of important ecosystem functions, such as conservation of nutrients and stabilization of substrates. For instance, the important role of resprouting vegetation in curbing massive losses of nitrogen was demonstrated by experimentally clearcutting and applying herbicides in a watershed at Hubbard Brook Experimental Forest (Bormann and Likens 1979).

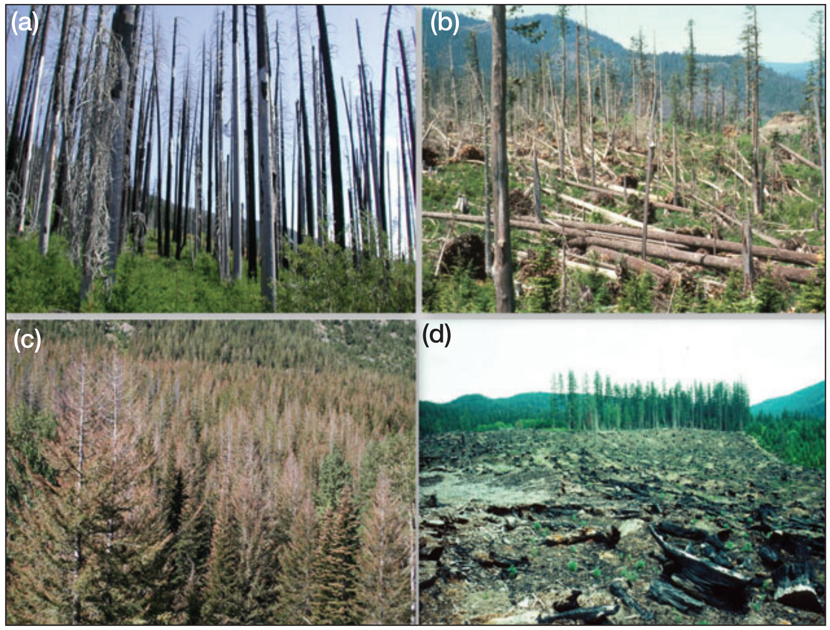

Figure 2. *Different types of disturbances produce different types of biological legacies, including living organisms and structures: (a) standing dead trees (snags) are dominant structural legacies after severe wildfires; (b) downed tree trunks and nearly intact understory communities are characteristic legacies after major windstorms; (c) standing dead trees are also dominant structural legacies after heavy insect infestations; and (d) clearcuts typically eliminate most aboveground structural legacies. Values for each metric are shown in Table 1 and are described in detail in the text.*

Structural complexity

The structural complexity of ESFEs depends initially on legacies, the general nature of which varies with the type of disturbance (Table 1; Figure 2); for example, snags and shrubs originating from belowground perennating (ie resprouting) parts or seeds are dominant legacies after wildfires, whereas downed boles and largely intact understories are typical post-disturbance characteristics of windstorms.

Woody legacies, such as snags and downed boles, play numerous roles in structuring and facilitating the development of the recovering ecosystem – providing habitat for survivors and colonists, moderating the physical environment, enriching aquatic systems in the disturbed area (Jones and Daniels 2008), and providing long-term sources of energy and nutrients (Harmon *et al*. 1986). Although subject to decomposition, these legacies can persist for many decades and sometimes even centuries.

Table 1. Different types of intense disturbances generate different types of biological legacies

	Disturbance				
Biological legacies	*Wildfire*	*Wind*	*Insect*	*Volcano*	*Clearcut*
Live trees	Infrequent	Variable	Variable (depends on stand composition)	Infrequent – confined to margins	Infrequent or absent
Snags	Abundant	Variable	Abundant	Abundant (spatially variable)	Infrequent or absent
Downed woody debris	Variable, but typically abundant	Abundant	Variable, but eventually abundant	Abundant (spatially variable)	Infrequent
Undisturbed understory	Infrequent	Abundant	Abundant	Infrequent – confined to disturbance margins	Infrequent
Spatial heterogeneity of recovery	High	Variable	High	High	Variable – usually low
Time in early-successional condition	Variable	Variable	Long	Variable – usually long	Variable – usually short

Figure 3. *Plant communities with well-developed shrub and perennial herb species are characteristic of early-successional communities on forest sites and provide diverse food resources. Twenty-five years after the Mount St Helens eruption in 1980, this community, which was within the blast zone, includes well-developed shrubs (eg* Sorbus *and* Vaccinium *spp), trees, and perennial herbs (eg* Epilobium angustifolium*).*

Structural complexity is further enhanced by the establishment and development of a variety of plant species, which often include perennial herbs and shrubs characteristic of open environments, as well as individual trees (Figure 3). The diversity of plant morphologies (maximum height, crown width, etc) increases structural richness, so that this associated flora contributes to both horizontal and vertical heterogeneity.

Spatial heterogeneity

Spatial heterogeneity is evident in early-successional ecosystems and has multiple causes: (1) natural variability in the geophysical template (topography and lithology) of the affected landscape; (2) variability in conditions in the pre-disturbance forest ecosystem; (3) variability in the intensity of the disturbance event; and (4) variability in rates and patterns of subsequent developmental processes in the ESFE. The first two sources relate to existing geophysical and biological patterns within the disturbed area. Land formations and patterns of geomorphic processes are certainly key geophysical elements (Swanson *et al.* 1988). The presence of surface water, such as streams and ponds, can be particularly influential in facilitating survival and re-establishment of biota.

Natural disturbances create heterogeneous environments at multiple spatial scales (Heinselmann 1973), because disturbances do not cause damage uniformly. Disturbances such as wildfires and windstorms are variable in intensity (eg "spotting", or initiation of new flame fronts by wind-thrown firebrands, during fire events). Alternatively, geographic variation in environmental conditions and topography (Swanson *et al.* 1988) influences the intensity of the disturbance and results in heterogeneity at multiple scales. Variability in the structure and composition of the pre-disturbance forest also creates spatial and temporal variability (Wardell-Johnson and Horowitz 1996). Some of these patterns may be transient, such as residual snowbanks protecting tree regeneration after the aforementioned Mount St Helens eruption (Dale *et al.* 2005).

Post-disturbance developmental processes also lead to spatial heterogeneity. For example, varying distances to sources of tree seed result in different rates and densities of tree re-establishment (Turner *et al.* 1998). Structural legacies can greatly influence the rates at which wind- or waterborne organic (including propagules) and inorganic materials are deposited. Finally, animal activity can strongly influence patterns of revegetation, as illustrated by the multiple effects that gophers (*Thomomys* spp) can have on post-disturbance landscapes (Crisafulli *et al.* 2005b) or the way ungulate browsing may impede tree regeneration (Hessl and Graumlich 2002).

■ Biological diversity

ESFEs in temperate forest seres show great diversity in the abundance of plant and animal species (Fontaine *et al.* 2009). Species composition may consist of a mix of forest survivors, opportunists, or ruderals (plants that grow on disturbed or poor-quality lands), and habitat specialists that co-exist in the resource-rich ESFE environment (Figure 3). Most forest understory flora can survive disturbances as established plants, perennating rootstocks, or seeds. In one study, in western North America, over 95% of understory species survived the combined disturbance of logging and burning of an old-growth Douglas-fir–western hemlock stand (Halpern 1988). Some important early-successional species (eg *Rubus* spp [blackberry; raspberry], *Ribes* spp [gooseberry], and *Ceanothus* spp [buckbrush]) may persist as long-lived seedbanks.

Opportunistic herbaceous species are often conspicuous dominants early in the development of ESFEs (Figure 4). Many of these weedy species (particularly annuals) decline quickly, although other opportunists will persist as part of the plant community until overtopped by slower growing shrubs or trees. Consequently, diverse plant communities of herbs, shrubs, and young trees emerge in ESFEs; this, combined with the structural legacies from the pre-disturbance ecosystem, often results in high levels of structural richness (Figure 3).

Many animals, including habitat specialists and species typically absent from the eventual tree-dominated com-

munities, thrive under the conditions found in ESFEs. For some species, this is the only successional stage that can provide suitable foraging or nesting habitat. As an example, many butterflies and moths (Lepidoptera) found in forested regions depend on the high diversity and quality of plant forage in ESFEs (eg Miller and Hammond 2007), whereas jewel beetles (Coleoptera: Buprestideae) depend on abundant coarse woody debris. Also, a number of ground-dwelling beetle species occur as habitat specialists in early-successional communities (Heyborne *et al.* 2003).

Figure 4. *Early-successional communities are often dominated by annual herbaceous species for the first few years after disturbance; these are quickly displaced by perennial herbaceous species and shrubs.*

Many vertebrates also respond positively to ESFEs, which may provide the only suitable habitat at a regional scale for some species. Ectothermic animals, such as reptiles (eg Rittenhouse *et al.* 2007), generally respond favorably to sunnier and drier conditions, colonizing early-successional habitat or increasing in abundance if present as survivors. Many amphibians also thrive in ESFEs, provided resources such as water bodies and key structures (eg logs) are available. The diversity and abundance of amphibians in the area affected by the 1980 Mount St Helens eruption is illustrative (Crisafulli *et al.* 2005a); eleven of 15 amphibian species survived the event, and some (eg western toad, *Bufo boreas*) have since had exceptional breeding success.

The broad array of birds using the abundant and varied food sources (eg fruits, nectar, herbivorous insects) and nesting habitat in ESFEs includes many raptors and neotropical migrants, often making bird diversity highest during the ESFE stage of succession (Klaus *et al.* in press). Some species are habitat specialists that directly utilize the legacy of recently killed trees; for instance, black-backed woodpeckers (*Picoides arcticus*) are almost completely restricted to early post-fire conditions (Hutto 2008). Mountain bluebirds (*Sialia currucoides*) and several other woodpecker species also favor structurally rich, early-successional habitats (Figure 5). Observed population declines of many avian species in eastern North America – which, in some cases, have proceeded to a point of conservation concern – are linked to conversion of early-successional habitat to closed forest (Litvaitis 1993).

Small mammal communities in ESFEs typically show high levels of diversity as well, including some obvious habitat specialists. The eastern chestnut mouse (*Pseudomys gracilicaudatus*), for example, inhabits early-successional environments in coastal eastern Australia for 2–5 years after a wildfire, and then declines dramatically until these environments are burned again (Fox 1990). Populations of mesopredators (medium-sized predators, such as raccoons [*Procyon lotor*] and fox species) benefit from the abundance of small vertebrate prey items characteristic of ESFEs. Likewise, some species of large mammals are well known to favor ESFEs (Nyberg and Janz 1990). Utilizing the diverse and luxuriant forage characteristically present in these ecosystems, ungulates, such as members of the Cervidae, in turn serve to benefit large predators (eg wolves [*Canis lupus*]) as well as scavengers, making ESFEs important elements within those species' typically extensive home ranges. Omnivores, such as bears (*Ursus* spp), also rely on the diversity of food sources often present in ESFEs.

Food web diversity

ESFEs are exceptional in the diversity and complexity of food webs they support. Simply stated, a diverse plant community produces many food sources. Food resources for herbivores (grasses, shrubs, forbs) – as well as nectar, seeds, and shrub-borne fruit (eg produced by *Rubus* and *Vaccinium* spp [huckleberry]) – can reach high levels before site dominance by trees. In the temperate Northern Hemisphere, biologically important berry production is maximized in slowly reforesting ESFEs. Resource production in early-successional patches may even augment the richness of adjacent undisturbed forests, as in the case of fluxes of key prey species (Sakai and Noon 1997).

Aquatic biologists have, perhaps, best appreciated the greater complexity of food chains in early-successional versus closed forest environments (Bisson *et al.* 2003). In established forest stands, trees strongly dominate the physical and biological conditions in nearby small streams by controlling light and temperature, stabilizing channels, providing woody debris, and, importantly, offering allochthonous inputs (organic matter originating outside the aquatic ecosystem) – the primary energy and nutrient source for such ecosystems (Vannote *et al.* 1980).

Stand-replacement disturbances remove forest constraints on conditions and processes, and shift streams to an early-

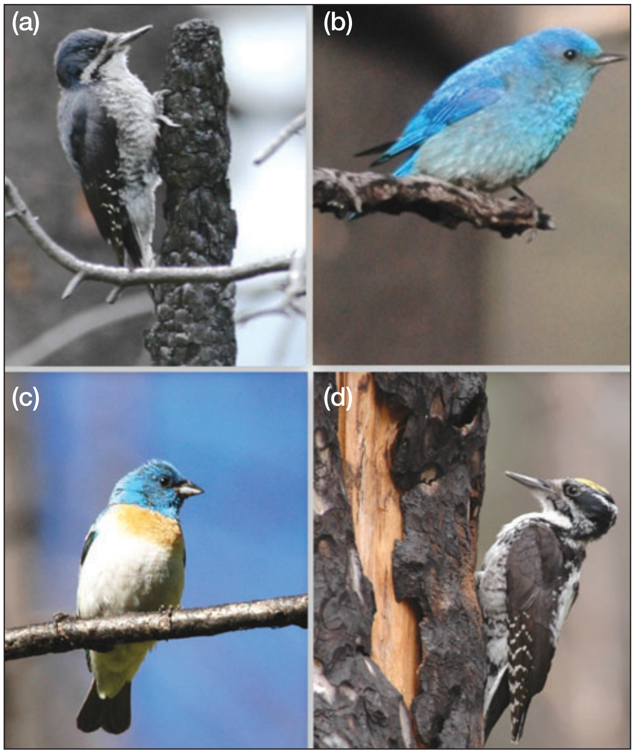

Figure 5. *Bird diversity is typically high in early-successional communities on forest sites and includes many habitat specialists: (a) black-backed woodpeckers (*Picoides arcticus*) are almost entirely restricted to early post-fire habitat; (b) mountain bluebirds (*Sialia currucoides*) favor early-successional ecosystems; (c) lazuli buntings (*Passerina amoena*) and (d) three-toed woodpeckers (*Picoides tridactylus*) have similar requirements.*

successional context (Minshall 2003; Figure 6). This greatly diversifies the types and timing of allochthonous inputs, as well as increases primary productivity. Allochthonous inputs are shifted from primarily tree-derived litter (coniferous-based in many systems) to material from a range of flowering herbs, shrubs, and trees, as well as from conifers. Consequently, litter inputs are highly variable in quality (eg decomposability) and delivery time, as compared with litterfall contributed primarily by evergreen conifer species. Also, inputs to post-disturbance streams often include material with a high nitrogen content, such as litter from the early-successional genera *Alnus* and *Ceanothus* (Hibbs *et al.* 1994).

Greater algal production may increase the diversity and abundance of aquatic invertebrate populations, which, in turn, become prey for fish and other organisms. However, increases in sediment production associated with disturbances can negate some benefits to aquatic processes and organisms (Gregory *et al.* 1987).

■ Processes in ESFEs

Ecosystem processes in ESFEs can be more diverse than those in closed forest systems, where the primary productivity of trees is dominant and organic matter is processed primarily through detrital food webs. Development of more diverse, and perhaps more "balanced", trophic pathways is possible when a disturbance opens a previously closed forest canopy. The contrast is probably greatest in forests dominated by a single tree type, such as evergreen conifers, as opposed to more diverse forests, such as mixed evergreen associations.

Recharging nutrient pools

ESFEs provide major opportunities for recharge of nutrient pools, such as additions to the nitrogen pool by leguminous (eg *Lupinus*) and some non-leguminous early-successional (eg *Alnus* and *Ceanothus*) plant species. These genera are commonly absent from late-successional forests, but are well represented in ESFEs. Nitrogenous additions from these sources are particularly important where the disturbance – eg a wildfire – has volatilized a substantial amount of the existing nitrogen pool.

Mineralization rates of organic material are typically accelerated (sometimes profoundly) after disturbances, as a result of warmer growing season temperatures. Diversified litter inputs in ESFEs, including a greater proportion of easily decomposed litter from herbs and deciduous shrubs, also result in more rapid mineralization. Finally, successional changes in the fungal and microbial communities can also hasten decomposition processes. As noted, these changes will be most profound in forest ecosystems dominated by a single species, including evergreen conifers or hard-leaved, evergreen hardwoods (such as the ash-type eucalypt forests of southeastern Australia).

In aquatic ecosystems that experience fire in adjacent forests, greater post-disturbance light and nutrient availability enhance primary productivity within the water body, causing shifts in food webs from the level of primary producers up through high-level consumers, such as fish (Spencer *et al.* 2003).

Modifying hydrologic and geomorphic regimes

Hydrologic regimes associated with ESFEs contrast greatly with those characterizing closed forest cover. For example, transpiration and interception are dramatically reduced and recover only gradually as forest canopies redevelop. Increases in normally low summer flows and annual water yields may occur immediately after a disturbance, as compared with levels in the dense young forests that may subsequently develop (Jones and Post 2004). The opposite may be true in systems where condensation of cloud or fog on tree crowns is an important component of the hydrologic cycle. ESFEs may also contribute to increased discharge peak runoff flows in hydrologic events of smaller magnitude (Harr 1986), but appear to have little effect on the magnitude of peak flows during large runoff events (Grant *et al.* 2008). From an ecological perspective, this may have a positive outcome, however, because floods restructure and rejuvenate many riparian communities (Gregory *et al.* 1991).

Rates and patterns of geomorphic processes, such as erosion and nutrient leaching losses, are also different between ESFEs and later successional stages. Tree death results in a loss of root strength that is critical for stabilizing soils and deeper rock layers on mountain slopes (Perry *et al.* 2008). Erosion and landslides may occur at higher rates in ESFEs, contributing to the variability of sediment budgets in watersheds (Reeves *et al.* 1995) and creating long-lasting substrates for ruderals. While enhancing erosion processes, ESFEs also provide materials and processes that counteract this effect, such as woody debris, which retain sediments and organic materials, and surviving vegetation, which stabilizes slopes and nutrient stores (eg Bormann and Likens 1979).

Figure 6. *Streams within early-successional forest ecosystems contrast with forest-dominated reaches in many ecosystem attributes, including physical parameters (temperature and insolation), structure, plant and animal composition, and ecosystem processes, such as primary productivity.*

■ Land management implications

Incorporating ESFE attributes into forest policy and management is highly desirable, given the numerous advantages provided by these ecosystems. Many species and ecological processes are strongly favored by conditions that develop after stand-replacement disturbances. Rapid, artificially accelerated "recovery" of disturbed forest areas (eg via dense planting) to closed forest conditions has serious implications for many species. Clearly the term "recovery" has a different meaning for such early-successional specialists or obligates.

To fulfill their full ecological potential, ESFEs require their full complement of biological legacies (eg dead trees and logs) and sufficient time for early-successional vegetation to mature. Where land managers are interested in conservation of the biota and maintenance of ecological processes associated with such communities, forest policy and practices need to support the maintenance of structurally rich ESFEs in managed landscapes. Natural disturbance events will provide major opportunities for these ecosystems, and managers can build on those opportunities by avoiding actions that (1) eliminate biological legacies, (2) shorten the duration of the ESFEs, and (3) interfere with stand-development processes. Such activities include intensive post-disturbance logging, aggressive reforestation, and elimination of native plants with herbicides.

In particular, post-disturbance logging removes key structural legacies, and damages recolonizing vegetation, soils, and aquatic elements of disturbed areas (Foster and Orwig 2006; Lindenmayer *et al.* 2008). Where socioeconomic considerations necessitate post-disturbance logging, variable retention harvesting (retention of snags, logs, live trees, and other structures through harvest) can maintain structural complexity in logged areas (Eklund *et al.* 2009).

Prompt, dense reforestation can have negative consequences for biodiversity and processes associated with ESFEs, by dramatically shortening their duration. Such efforts reduce spatial and compositional variability characteristic of natural tree-regeneration processes, promote structural uniformity, and initiate intense competitive processes that eliminate elements of biodiversity that might otherwise persist. Artificial reforestation can also reduce genetic diversity by favoring dominance by fewer tree species/genotypes, and may make the system more prone to subsequent, high-severity disturbances (Thompson *et al.* 2007). The elimination of shrubs and broad-leaved trees through herbicide application can alter synergistic relationships, such as the belowground mycorrhizal processes provided by certain shrub species (eg *Arctostaphylos* spp).

Naturally regenerated ESFEs are likely to be better adapted to the present-day climate and may be more adaptable to future climate change. The diverse genotypes in naturally regenerated ESFEs are likely to provide greater resilience to environmental stresses than nursery-grown, planted trees of the same species. Given that climate change is also resulting in altered behavior of pests and pathogens (Dale *et al.* 2001), encouraging greater tree species diversity may also increase ecosystem resilience.

Clearcutting has been proposed as a technique to create ESFEs, but this can provide only highly abridged and simplified ESFE conditions. First, traditional clearcuts leave few biological legacies (eg Lindenmayer and McCarthy 2002), limiting habitat and biodiversity potential. Second, clearcuts are often quickly and densely reforested, and often involve the use of herbicides to limit competition with desired tree species. Clearcuts can provide some early-successional functionality (eg serving as nurseries or post-breeding habitat for many bird species in the southern US; Faaborg 2002), but this service is often truncated by prompt reforestation.

Management plans should provide for the maintenance of areas of naturally developing ESFEs as part of a diverse landscape. This should be in reasonable proportion to *historical* occurrences of different successional stages, as based on region-specific historical ecology. Major disturbance events provide managers with opportunities to incorporate a greater diversity of species and processes in forest landscapes and to enhance landscape heterogeneity. Some aspects of ESFEs can be incorporated into areas managed for production forestry as well, such as through variable retention harvest methods, the incorporation of natural tree regeneration, and extending the duration of herb/shrub communities in some portions of a stand by deliberately maintaining low tree stocking levels.

Finally, we suggest that adjustments in language are needed. Ecologists and managers often refer to "recovery" when discussing post-disturbance ecosystems, inferring that early seral conditions are undesirable and need to be restored to closed canopy conditions as quickly as possible. Emphasizing recovery as the management goal fails to acknowledge the essential ecological roles played by early-successional ecosystems on forest sites. It should also be considered that climate change and other factors may not permit "recovery" to pre-disturbance conditions.

Conclusions

Twentieth-century forest management objectives were centered on wood production and, later, on conservation and development of late-successional forests. Rapid regeneration of dense timber stands was frequently seen as a way to address both of these divergent objectives. Recognizing the ecological value of early-successional ecosystems on forest sites extends the ecological concerns associated with old growth to another "rich" period in a forest sere. This represents an important development in the evolution of holistic management of forest ecosystems, whereby large landscapes are managed for diverse seral stages.

ESFEs provide a distinctive mix of physical, chemical, and biological conditions, are diverse in species and processes, and are poorly represented and undervalued in traditional forest management. Forest policy and practice must give serious attention to sustaining substantial areas of ESFEs and their biological legacies. Similarly, scientists need to initiate research on the structure, composition, and function of ESFEs in different regions and under different disturbance regimes, as well as on the historical extent of these systems, to serve as a reference for conservation planning.

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Comparing selected fire regime condition class (FRCC) and LANDFIRE vegetation model results with tree-ring data

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Abstract. Fire Regime Condition Class (FRCC) has been developed as a nationally consistent interagency method in the US to assess degree of departure between historical and current fire regimes and vegetation structural conditions across differing vegetation types. Historical and existing vegetation map data also are being developed for the nationwide LANDFIRE project to aid in FRCC assessments. Here, we compare selected FRCC and LANDFIRE vegetation characteristics derived from simulation modeling with similar characteristics reconstructed from tree-ring data collected from 11 forested sites in Utah. Reconstructed reference conditions based on trees present in 1880 compared with reference conditions modeled by the Vegetation Dynamics Development Tool for individual Biophysical Settings (BpS) used in FRCC and LANDFIRE assessments showed significance relationships for ponderosa pine, aspen, and mixed-conifer BpS but not for spruce–fir, piñon–juniper, or lodgepole pine BpS. LANDFIRE map data were found to be ~58% accurate for BpS and ~60% accurate for existing vegetation types. Results suggest that limited sampling of age-to-size relationships by different species may be needed to help refine reference condition definitions used in FRCC assessments, and that more empirical data are needed to better parameterize FRCC vegetation models in especially low-frequency fire types.

Additional keywords: reference conditions, successional classes, Vegetation Dynamics Development Tool (VDDT).

Introduction

Altered fire regimes and associated changes in vegetation structure, composition, and fuels pose risks to biodiversity, sustainable ecosystems, and economic and community interests across the United States (USDA/USDI 2000). However, the magnitude of these risks varies between ecosystems as a result of differences in their fire and vegetation histories, successional, compositional, and structural dynamics, and the influence of invasive species (Morgan *et al*. 2001; Schoennagel *et al*. 2004). Fire exclusion over the 20th century has not affected all ecosystems uniformly, and accurate characterization of historical fire regimes and recent vegetation changes is critical to inform management decisions about the need for fuel treatments or ecological restoration across differing plant communities.

Use of historical fire regimes and vegetation conditions to inform fire and fuel management decisions in the US has been refined into the Fire Regime Condition Class (FRCC) concept (Hann and Bunnell 2001; Schmidt *et al*. 2002; Hann and Strohm 2003; Hann *et al*. 2003; Shlisky and Hann 2003). FRCC is an index that compares current with historical fire regimes and vegetation composition and structure to assess degree of departure on a scale from one (least departed) to three (most departed). FRCC is based on an assumption that historical processes and patterns (those present before widespread Euro-American settlement in the mid- to late-1800s) represent longer-term sustainable ecosystem conditions, and that greater departure in current conditions represents a greater risk for uncharacteristic fire behavior and associated ecosystem impacts. Initial coarse-level (1-km^2 resolution) FRCC maps described the degree of departure at a national scale (Schmidt *et al*. 2002). After this initial effort, a set of standard guidebook methods was developed to assess FRCC at landscape to stand scales for local management and planning needs (at time of writing, FRCC Guidebook v1.3; Hann *et al*. 2004). FRCC maps of 30-m^2 resolution are also being developed as part the LANDFIRE project, an effort to provide consistent vegetation, fuels, and fire regime data for the entire US (Rollins and Frame 2006; www.landfire.gov, accessed 19 October 2007). FRCC is now a key variable for defining wildfire risk to ecosystems as a result of its explicit incorporation into the Healthy Forests Restoration Act of 2003 (HFRA 2003). FRCC represents a significant advance in the integration of fire and forest histories and landscape and vegetation ecology to provide an ecologically based method for setting fire-management priorities and objectives across the US (Shlisky and Hann 2003).

Definition of departure indices in FRCC assessments begins with simulation modeling of historical vegetation composition and structure using the Vegetation Dynamics Development Tool (VDDT; Beukema and Kurz 2003). VDDT is used to develop non-spatially defined reference conditions within Biophysical Settings (BpS; formerly referred to as Potential Natural Vegetation Groups (PNVG); Küchler 1964; NRCS 2003). For LANDFIRE, BpS are derived from Nature Serve's ecological

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classification system (Comers *et al*. 2003) and are not directly comparable with those used in FRCC assessments. However, both systems use BpS in a similar manner to represent the vegetation communities that would likely exist under given environmental conditions (climate, soils, and landscape physiography) and historical disturbance regimes. BpS in LANDFIRE are assigned to specific locations in their nationwide mapping efforts, whereas BpS in FRCC assessments are non-spatial and assigned based on individual user needs for specific projects or management requirements. Reference conditions are the proportions of vegetation successional stages (community structure and composition) as affected by varying fire frequencies, severities, and successional pathways within each BpS (Hann *et al*. 2004).

FRCC and LANDFIRE vegetation models (also known as Vegetation Dynamics Models) were defined during regional professional workshops conducted between 2002 and 2009 (2005–09 for LANDFIRE). VDDT model inputs for individual BpS are based on historical fire regime characteristics (frequency and severity) and vegetation data derived from published and unpublished studies and expert opinion developed both at the regional workshops and through subsequent peer reviews (Hann *et al*. 2004). The amount and quality of available historical data for each BpS vary, which can affect the quality and accuracy of the resulting modeled reference conditions. In an FRCC assessment, a field evaluation is conducted of existing vegetation structure, which, in forests, is based on cover type, density of tree stands, tree size, and current successional status. Successional status is determined by visually estimating stand composition, tree density, and average tree age, the latter of which is based on tree diameters. Proportions of current successional classes in a project or management area are estimated during the field assessment and then compared with the proportions of reference conditions derived from VDDT model output. The FRCC departure index (1 to 3) is assigned based at least partially on differences in proportions of successional classes present in the current forest relative to modeled reference conditions in the historical forest.

There is a need to test the process of development of reference conditions by comparing VDDT model output with known fire and vegetation histories. This comparison is critical for assessing consistency and accuracy in the modeling process. Here, we compare VDDT-modeled reference conditions with tree-ring-based reconstructions of reference conditions from 11 forested sites in Utah and eastern Nevada (tree-ring data reported in Heyerdahl *et al*. 2005, and Brown *et al*. 2008*a*). The tree-ring reconstructions span transects aligned along elevation gradients that include multiple forest types. We ask the following questions with this comparison: (1) do FRCC methods of evaluating stand structure based on diameter estimates accurately represent ages of forest vegetation and is there variation based on species and site? (2) Do FRCC and LANDFIRE BpS models adequately capture the range of variation in proportions of reference conditions reconstructed by the tree-ring data? (3) Do LANDFIRE mapped data layers for BpS and Existing Vegetation Types (EVT) match the tree-ring plot data? (4) Can further empirical fire history and tree recruitment data be used to strengthen FRCC evaluation and reference condition modeling outputs? We consider this study to be only an initial test of FRCC and LANDFIRE vegetation modeling methods, but one that may provide an example for future testing needs.

Methods

Study area

Tree-ring sites used for this study extend from the Colorado Plateau of southern Utah, west to the Wah Wah Mountains in the eastern Great Basin of western Utah, and north to the Uinta and Bear River Mountains in northern Utah (Fig. 1, Table 1; Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). The region is a complex of valleys, mesas, canyons, plateaus, and mountains that range in elevation from ~900 to >3600 m. Forest types vary generally across elevation gradients. Piñon (*Pinus edulis* (PIED; four-letter codes are used in tables) and *P. monophylla* (PIMO)) and juniper (*Juniperus scopulorum* (JUSC) and *J. osteosperma* (JUOS)) savannas and woodlands occur at the lowest forest margins above desert shrublands or grasslands. Ponderosa pine (*Pinus ponderosa* (PIPO)) forests occur in montane zones in pure and mixed stands. Douglas-fir (*Pseudotsuga menziesii* (PSME)) often occurs in association with ponderosa pine on north-facing aspects and in relatively mesic sites. Mixed-conifer forests occur at intermediate elevations and include combinations of ponderosa pine, Douglas-fir, piñons, junipers, and firs (*Abies lasiocarpa* (ABLA) or *A. concolor* (ABCO)). Mixed-conifer forests also often occur in association with aspen (*Populus tremuloides* (POTR)). Aspen forms large (>100 ha) pure stands throughout the upper montane and lower subalpine zones across the study area except in the Great Basin. Lodgepole pine (*Pinus contorta* (PICO)) often forms pure stands at mid-elevations (1900 to 2800 m) or occurs in the mixed-conifer zone in northern Utah. Subalpine forests dominated by Engelmann spruce (*Picea engelmannii* (PIEN)) and firs occur at upper elevations (2350 to 3500 m). At the highest forested elevations (generally above 3000 m), pure Engelmann spruce forests occur in mesic sites whereas bristlecone pine (*Pinus longaeva* (PILO)) or limber pine (*P. flexilis* (PIFL)) are typically found in dry or rocky sites.

There was, in general, a gradient in fire frequency across the elevational gradient before fire exclusion that began at all sites in the late 1800s (Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). Fire occurrence was highest in the middle of the elevation range in ponderosa pine and drier mixed-conifer sites. Fire frequency progressively declined both above and below this middle-elevation zone. At upper elevations, generally moist conditions led to high fuel biomass, both living and dead, in many stands, but fewer years in which fuels were dry enough to ignite and spread. At lower elevations in the piñon–juniper woodlands, fuels were often dry enough to burn because of hotter and dryer fire seasons, but because of lower productivity, there were in general less continuous both aerial and surface fuels and fires were not able to spread. In the middle zone, both fuel amounts and moistures were just right (what has come to informally be called the 'Goldilocks effect'), and able to burn often in wide-spreading fires.

Utah forests underwent a period of intensive grazing and land use beginning in the 1850s as a result of Euro-American settlement. Intensive grazing removed understorey species and began alteration of longer-term historical forest dynamics. Logging also affected forest structure in many areas. The tree-ring study found that cessation of historical patterns of fires began in

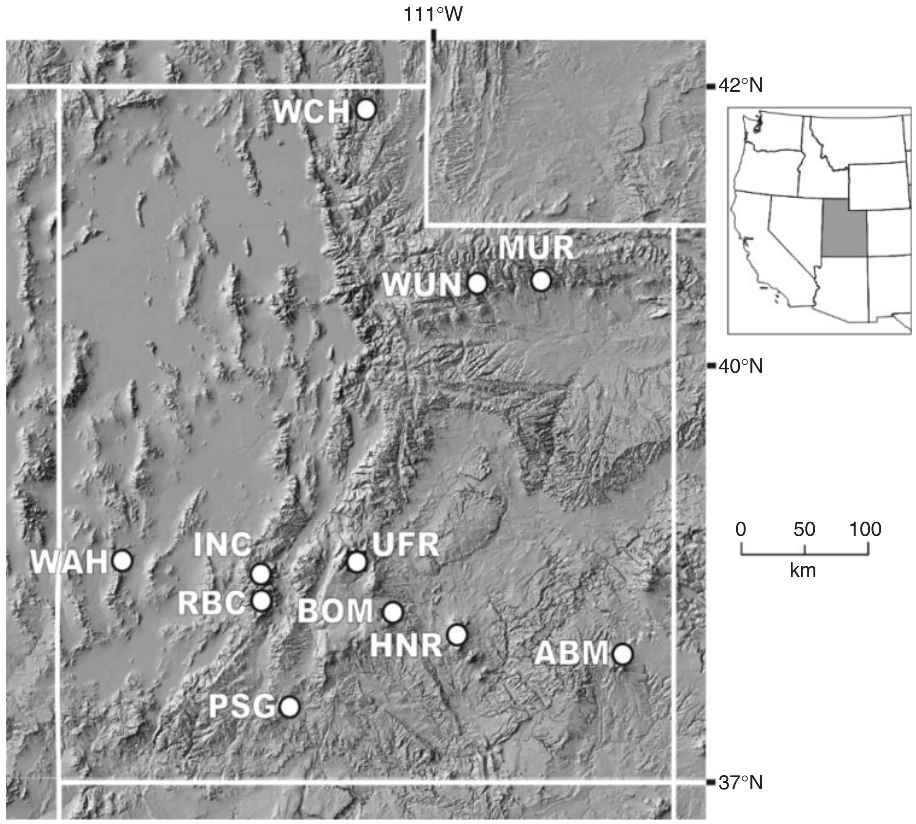

Fig. 1. Locations of tree-ring sites. Three-letter codes correspond to those in Table 1.

Table 1. Tree-ring sites used in the present study arranged from north to south
FRCC (Fire Regime Condition Class) and LANDFIRE BpS (biophysical settings) forest types are listed in Table 4

Site	Minimum elevation (m)	Maximum elevation (m)	Average precipitation (cm)	FRCC and LANDFIRE BpS
Wasatch Mountains (WCH)	2255	2588	100	SPFI, SPDF, CHPI, 10510, 10520, 10500, 10550
Western Uinta (WUN)	2207	3133	60	PPIN, SPDF, SPFI, CHPI, 10540, 10510, 10520, 10500, 10550
Middle Uinta River (MUR)	2308	3250	70	PPIN, SPDF, SPFI, CHPI, 10540, 10510, 10520, 10500, 10550
Wah Wah Mountains (WAH)	2195	2686	40	JUPI, PPIN, SPDF, 10160, 10540, 10500
Upper Fremont River (UFR)	2800	3039	80	SPDF, SPFI, 10510, 10520, 10500
Indian Creek (INC)	2364	2518	65	PPIN, SPDF, 10540, 10500
Beaver Creek (RBC)	2358	3077	90	PPIN, SPDF, SPFI, 10540, 10510, 10520, 10500
Boulder Mountain (BOM)	2405	3377	80	JUPI, PPIN, SPDF, SPFI, 10160, 10540, 10510, 10520, 10500
Henry Mountains (HNR)	2407	3138	60	JUPI, PPIN, SPDF, 10160, 10540, 10500
Abajo Mountains (ABM)	2557	3231	85	JUPI, PPIN, SPDF, SPFI, 10160, 10510, 10520, 10500
Paunsaugunt Plateau (PSG)	2309	2736	45	JUPI, PPIN, SPDF, SPFI, 10160, 10540, 10510, 10520, 10500

the 1860s to 1890s depending on location (Brown *et al*. 2008*a*), similar to patterns seen in forests throughout the western US. Initial reduction in fire frequency was likely the result of grazing that removed grass and herbaceous fuels, followed later by direct fire suppression in the 20th to 21st centuries.

Tree-ring data

The tree-ring study used a systematic sampling design to characterize stand and age structure and fire regimes across forest gradients in each site (Table 1; Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). Similar methods have been used in multiple studies

around the western US and are described in more detail in Heyerdahl *et al*. (2005, 2006), Brown and Wu (2005), Brown (2006), Brown *et al*. (2008*a*, 2008*b*), and Brown and Schoettle (2008). A 500-m grid was established at each site and plots sampled at grid points. Plot centers were located in the field using hand-held global positioning system (GPS) units. An *n*-tree density-adapted sampling method (Jonsson *et al*. 1992) was used to collect data from the nearest ~30 remnant (logs, snags, or stumps) or living trees >20 cm diameter at breast height (DBH) to each plot center. Maximum plot radius was set at 40 m (~0.5 ha) and most plots were ~<0.2 ha in size. For each plot tree, species was recorded and an increment core (on living trees) or cross-section (from logs, snags, and stumps) was collected from ~10 cm height above ground. Sampled cores had to be no more than a field-estimated 10 years from pith to minimize pith offset when assessing pith date. Diameter at sample height (DSH) and DBH were measured on living trees, and DSH was measured or estimated for remnant trees missing bark, sapwood, or heartwood. Distance from plot center and azimuth were measured on all trees for reconstruction of tree basal areas, density, and spatial patterning. To reconstruct surface fire history, cross-sections were cut from any fire-scarred trees found within plots. Additional fire-scarred trees also were sampled within ~80 m of each grid point and between grid points when discovered. GPS coordinates and species of fire-scarred trees outside of plots were recorded.

Standard dendrochronological methods were used to crossdate all samples using locally developed master chronologies (Heyerdahl *et al*. 2005). Pith dates were estimated on cores that did not intersect pith based on the curvature of the innermost rings sampled. The tree recruitment date is considered to be the date of tree pith at 10-cm height. No corrections were made for time to grow from germination to 10 cm sample heights because of the widely varying species and environmental conditions at the sites that were collected for the study. Once crossdating of ring series was completed on all samples, dates for any fire scars seen within the ring series were assigned. Any trees that were not able to be dated were not used in subsequent analyses.

FRCC and LANDFIRE vegetation models

VDDT modeling estimates the relative proportions of non-spatially defined reference conditions that would have occurred under a historical fire regime and an equilibrium (current) climate regime within each BpS (Beukema and Kurz 2003). VDDT input includes average fire frequencies, severities, and other disturbances defined as probabilistic events, and vegetation structural stage development pathways, including changes in species composition and density through a successional sequence. VDDT runs are commonly made for 500 years to allow vegetation conditions to equilibrate over time. VDDT output is proportions of vegetation successional classes – the reference conditions – across a non-spatially referenced landscape at the end of the 500-year model run. Reference conditions for most forest types are summarized into five seral stages that approximate overall developmental characteristics of community age and structure: early-replacement, mid-open, mid-closed, late-open, and late-closed. Each developmental stage represents a successional class defined by average tree age, species composition, structural characteristics, and response to disturbances. LANDFIRE and FRCC assessments use VDDT in a similar manner, but in LANDFIRE, reference condition proportions are then coupled with the spatial model LANDSUM (Keane *et al*. 2002) to map resulting vegetation conditions for each BpS across actual landscapes at a 30-m^2 spatial resolution.

FRCC and LANDFIRE developed their own BpS models using two different vegetation classification systems (Küchler 1964 *v.* Comers *et al*. 2003). Both systems attempt to describe the same historical vegetation using VDDT; however, their models use different probabilities for disturbance, and have somewhat different species distributions and geographic extents (often based on expert opinion; see http://frcc.gov, accessed 19 October 2007; www.landfire.gov for details).

Comparing tree-ring with FRCC and LANDFIRE data

We performed three tests to compare the tree-ring data with FRCC and LANDFIRE vegetation models. First, we compiled age and DBH data derived from the tree-ring study to assess whether FRCC methods of visual estimates of tree diameters accurately represent the age of forest vegetation for defining mid- and late-development classes of reference conditions. FRCC guidebook methods define >23 cm DBH as a visual indicator of a mature tree when conducting field assessments. For this analysis, we assumed that plots with trees averaging ≤23 cm DBH would be considered to be in a mid-development reference condition, and >23 cm would be in late-development. We conducted least-squares linear regressions to estimate fitness of tree age to DBH by species and site. As many of the regression models did not meet the statistical requirements of homoscedasticity, normality, and constant variance in model residuals, a logarithmic transformation was applied to the tree ages before regression. Models that had significant P values ($P < 0.05$) were considered to be representative of species growth estimates. We also conducted an analysis of variance (ANOVA) of age and diameter by species and site to both determine the strength of these relationships and how they varied by species and location across the region. All statistical analyses were conducted using the *Statistica* software (StatSoft Inc. 2008). The tree-ring study sampled a total of ~10 000 remnant and living trees; however, we only used data from the living trees for this part of our assessment. Dead trees (stumps, snags, and logs) often were missing bark, sapwood, or portions of the heartwood that reduced confidence in diameter estimates. The DBH-to-age analysis therefore consisted of 5173 living trees from 13 species from the 11 sites.

Our second test was whether VDDT modeled reference conditions captured the range of variation in reference conditions reconstructed by the tree-ring data as of a date of 1880. Dates of initial Euro-American settlement varied across the study area but all sites showed some Euro-American impact by 1880, including cessation of spreading fires in almost all of the sites (Brown *et al*. 2008*a*). As current vegetation may not be representative of past vegetation type, only species present in 1880 and their corresponding ages were used to assign BpS and reference condition to each of a total of 273 plots that were sampled from the 11 sites (Heyerdahl *et al*. 2005; Brown *et al*. 2008*a*). Both living and remnant trees were used to estimate the 1880 plot compositions. FRCC and LANDFIRE use key species to

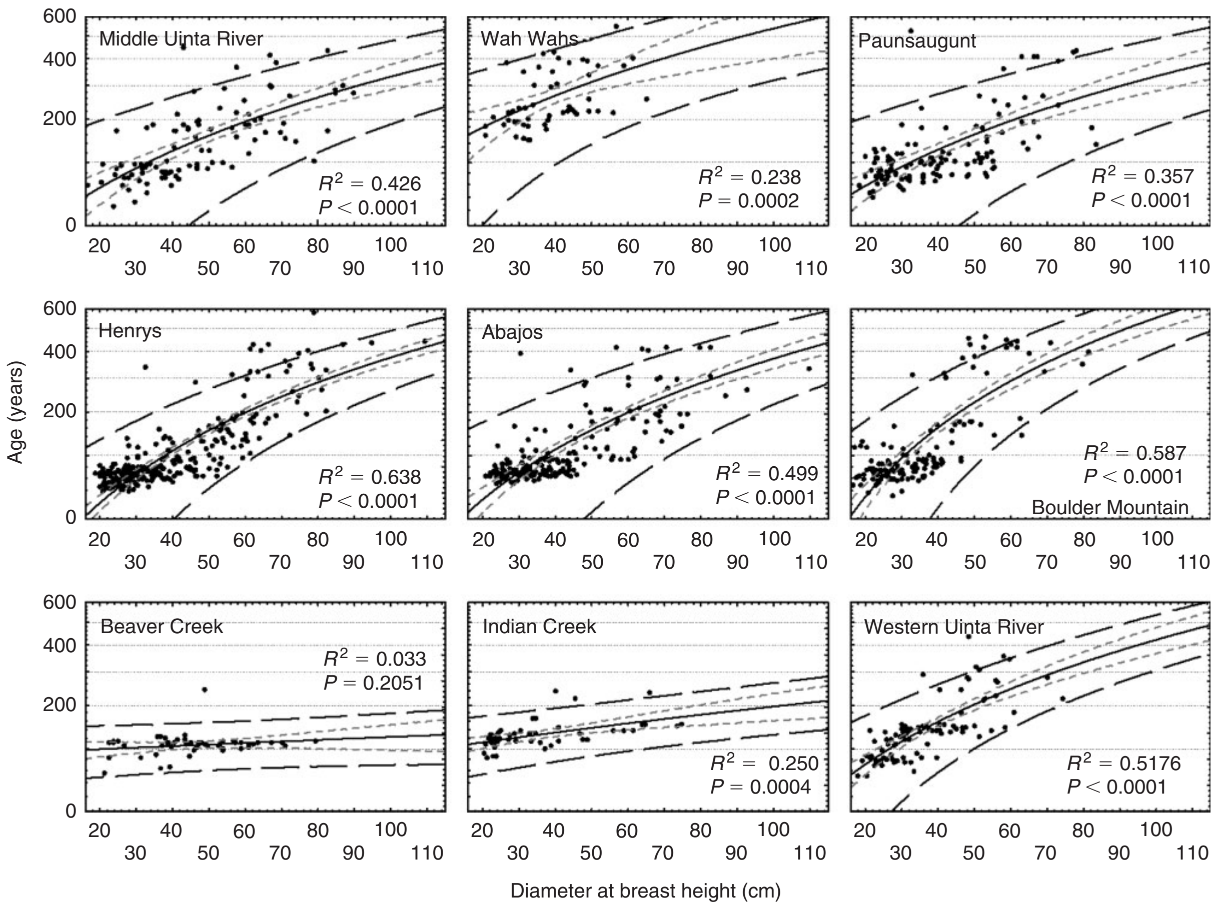

Fig. 2. Diameter at breast height (DBH) and log(age) regressions for ponderosa pine trees by site, with linear fits (solid lines), 95% confidence intervals (gray dashed lines), and 95% prediction intervals (black dashed lines). Overall R^2 for ponderosa pine trees across all sites was 0.44.

define vegetation characteristics when conducting an assessment and we used these species as the basis for assigning BpS and reference condition to each plot.

Historical age class and species composition in 1880 for each plot were compared with FRCC and LANDFIRE reference conditions for selected BpS. FRCC and LANDFIRE BpS descriptions are available on their respective project websites (www.frcc.gov; www.landfire.gov). We did not evaluate the typical five-stage VDDT models because of difficulties in using the tree-ring data to accurately recreate smaller size classes in historical stand densities as a result of probable tree mortality and decay since pre-settlement periods (e.g. Brown and Cook 2006; Brown *et al*. 2008*b*). However, we assume that we are able to define with some confidence mid- and late-development stands based on crossdated ages of trees present in each plot in 1880. The mean age of a 23-cm-DBH live tree varied by species, and we used the tree-ring results to estimate the upper 95% confidence interval for predicted tree size to consider whether a stand was late developmental stage in 1880. We grouped data from open and closed stands together based on age and composition for comparison with succession classes from VDDT output. If any trees in a plot were older than their predicted age-to-size confidence interval, the plot was considered to be in late-development in 1880. If there were no older trees during the historical period, then the plot was considered to have been in mid-development. If there were no trees during the historical period, the plot was considered to have no data and not used in this analysis. Once plots were categorized by BpS and reference condition, they were compared with FRCC and LANDFIRE BpS model proportions of mid- and late-development vegetation based on VDDT output. We used a Chi-square test to determine if the observed tree-ring reference condition proportions were significantly different than the expected based on the VDDT output.

Finally, we compared tree-ring plot data with LANDFIRE BpS and EVT map layers produced by the LANDFIRE project. LANDFIRE data are spatially mapped, which provided a unique opportunity to evaluate vegetation models at a high spatial resolution through comparison with the mapped tree-ring data. Plots were first located through their GPS coordinates relative to LANDFIRE map data. The BpS assignments we made for each plot in 1880 were then compared with LANDFIRE BpS map data. We also compiled the living tree composition in each plot and compared that with the LANDFIRE EVT map data. If key

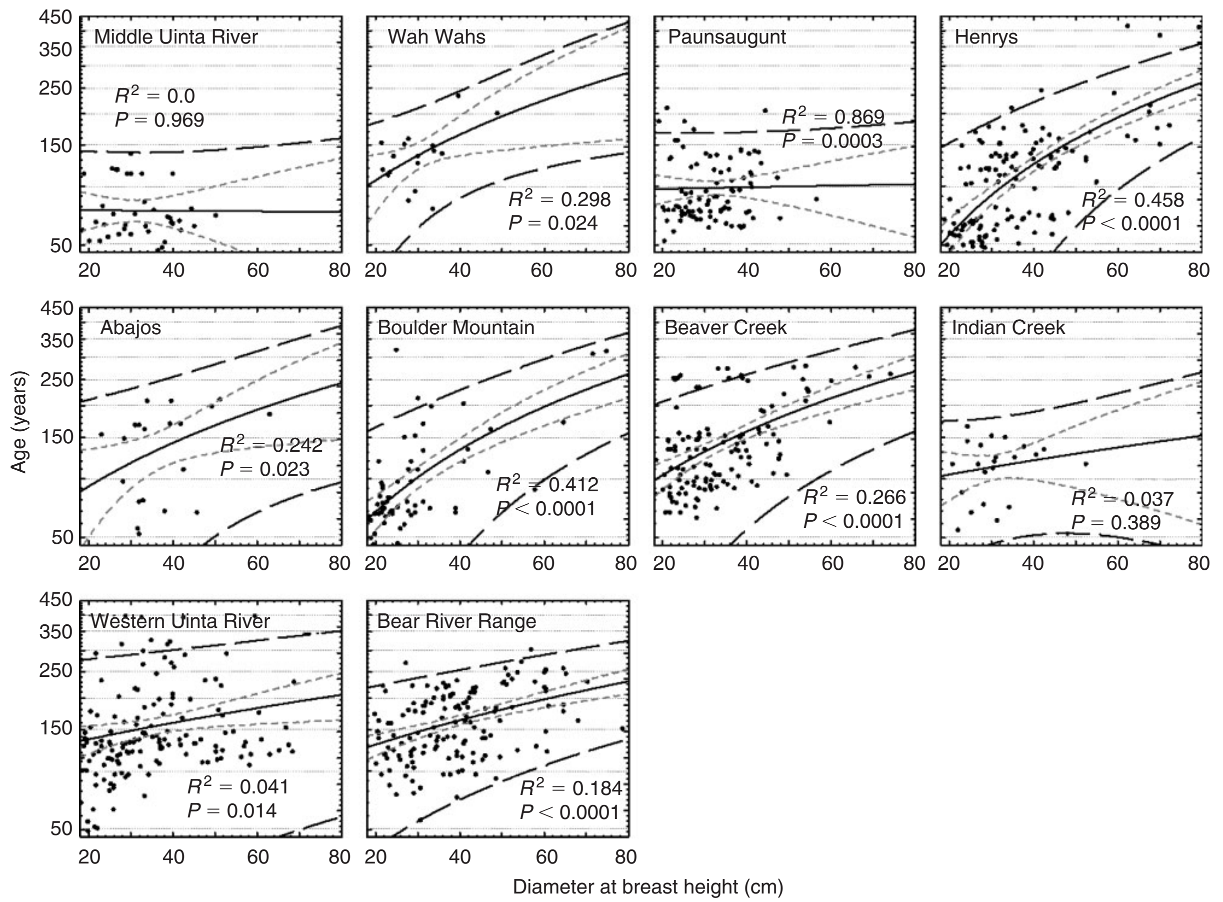

Fig. 3. Diameter at breast height (DBH) and log(age) regressions for Douglas-fir trees by site, with linear fits (solid lines), 95% confidence intervals (gray dashed lines), and 95% prediction intervals (black dashed lines). Overall R^2 for Douglas-fir trees across all sites was 0.21.

species were present in the tree-ring data in comparison with the mapped BpS or EVT, then the grid point was considered to have been accurately mapped in LANDFIRE.

Results

Age–diameter relationships

DBH and tree ages exhibited generally broad relationships, both within species and among sites (Figs 2–4; Tables 2, 3). Ponderosa pine was the only species where age and size were strongly correlated using data from all sites ($R^2 = 0.438$, $P < 0.001$) and were strongly correlated over most of the individual sites (Table 2). There were outliers for most species by DBH and age; however, their deviance did not significantly change the results. Median tree age was predicted for trees at 23 cm using an inverse prediction with 95% confidence interval (Table 3). ANOVA results indicate that species associated with infrequent fire regimes (piñon–juniper, spruce–fir, and bristlecone pine; Heyerdahl *et al.* 2005) were found to have greater average ages than frequent fire species (especially ponderosa pine and Douglas-fir; Fig. 5). Variance of diameters relative to ages for species that contained a large sample *n*, such as Douglas-fir (PSME), ponderosa pine (PIPO), and Engelmann spruce (PIEN) was small. There was greater variance found in species that had fewer sampled trees and plots, such as bristlecone pine (PILO), Rocky Mountain juniper (JUSC), one-seed juniper (JUOS), limber pine (PIFL), and single leaf piñon (PIMO), but this result is likely an artifact of the smaller number of trees used in each regression. ANOVA indicated that DBH and age estimates for all sites were similar with the exception of WAH (Fig. 5). This may be explained by the large presence of fire-infrequent and older species (bristlecone pine, Rocky Mountain juniper, and one-seed juniper) that were sampled in that site.

FRCC and LANDFIRE BpS models

Median ages of trees >23 cm DBH were used to define the proportions of mid- and late-development reference conditions for trees present in plots in 1880 (Table 3). Reference condition proportions reconstructed from the tree-ring data compared favorably with FRCC BpS models for ponderosa pine (PPIN5), mixed-conifer (SPDF), and lodgepole (CHPI) but not for piñon–juniper (JUPI1, JUPI2), south-western mixed-conifer (MCAN) and spruce–fir (SPFI2, SPFI7; Table 4, Fig. 6). Reference condition proportions reconstructed from the tree-ring data

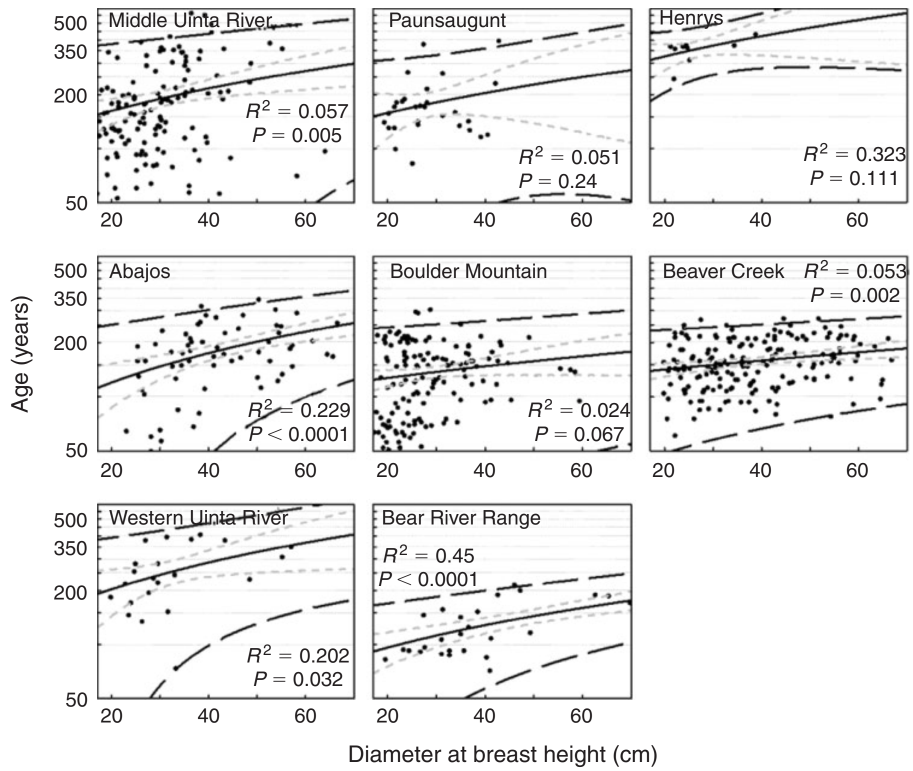

Fig. 4. Diameter at breast height (DBH) and log(age) regressions for Engelmann spruce trees by site, with linear fits (solid lines), 95% confidence intervals (gray dashed lines), and 95% prediction intervals (black dashed lines). Overall R^2 for Engelmann spruce trees across all sites was 0.06.

Table 2. Observed two-sided *P* values for DBH–age regressions for all species at all sites

Bold values represent locations where *P* values are significant at the 95% confidence interval (<0.05) based on sample size (>10 trees)

Species	Site									
	WCH	RBC	ABM	BOM	HNR	PSG	INC	WUN	MUR	WAH
PIPO		0.021	**<0.0001**	**<0.0001**	**<0.0001**	**<0.0001**	**0.0004**	**<0.0001**	**<0.0001**	**0.0002**
PSME	**<0.0001**	**<0.0001**	**0.0234**	**<0.0001**	**<0.0001**	0.88	0.39	**0.01**	0.969	0.024
PIEN	**<0.0001**	**<0.0001**	**<0.0001**	0.066	0.111	0.241		0.03	0.005	
ABLA	**<0.0001**	0.19	**<0.0001**		0.147			0.37		
POTR	**0.01**	0.01	**<0.0001**	0.63	0.107		0.40	0.81	**0.020**	
ABCO		0.22				**<0.0001**	0.22		0.069	0.002
PICO	**<0.0001**				**<0.0001**	**0.0007**				
PIFL	0.28				**<0.0001**	0.090	0.28			
PIED				**<0.0001**	**<0.0001**	0.025				
PIMO										**<0.0001**
JUSC				0.152		**0.111**				0.903
JUOS				**0.0003**		0.677			0.797	**0.0002**
PILO										0.574

compared favorably with LANDFIRE BpS models for Rocky Mountain dry–mesic montane mixed-conifer (10510), aspen and aspen–mixed-conifer low- and high-elevation forests (10110, 10611, 10612), but not for piñon–juniper (10160), ponderosa pine (10540), Rocky Mountain mesic montane mixed-conifer (10520), Rocky Mountain subalpine dry–mesic spruce–fir forest and woodland (10550), and Rocky Mountain lodgepole pine (10500; Table 4, Fig. 6). The JUPI1 BpS model (Table 4) was the most different from the tree-ring data, although the JUPI2 model had a similar trend of a larger proportion of late-successional

stands in comparison with the tree-ring data (Fig. 6). Spruce–fir and lodgepole pine data both showed low correspondence with VDDT model results, including opposite trends of more older than younger stands in the tree-ring data in contrast to the VDDT modeled reference conditions (Fig. 6).

Table 3. Expected median ages of trees >23 cm DBH (diameter at breast height) by species, with lower and upper 95% confidence intervals derived from tree-ring data

NS, age–DBH regression not significant

Species	Age (years) at 23 cm	R^2	P value
PIPO	40.9 ± 3.2	0.438	<0.0001
JUOS	114.9 ± 41.9	0.438	<0.0001
PIED	135.3 ± 21.9	0.28	<0.0001
PIFL	66 ± 11.4	0.271	<0.0001
PIMO	176.3 ± 29.8	0.231	<0.0001
PSME	42.9 ± 6	0.213	<0.0001
PICO	54.3 ± 12.6	0.112	<0.0001
POTR	104 ± 9.1	0.095	<0.0001
PIEN	24.7 ± 14.7	0.055	<0.0001
JUSC	NS	0.05	0.0961
ABCO	14.8 ± 14.4	0.023	<0.0001
PILO	NS	0.012	0.6295
ABLA	50.2 ± 10.2	0.01	<0.0001

LANDFIRE map data

LANDFIRE map layers were found to be overall ~58% accurate for BpS and 60% accurate for EVT when compared with the tree-ring data for each plot (Table 5). LANDFIRE maps were 38% accurate for both BpS and EVT, 28% accurate for at least one type (17% EVT accurate and BpS inaccurate, with 11% BpS accurate and EVT inaccurate), and 34% inaccurate. Mixed-conifer and spruce–fir types had the highest accuracies by BpS for LANDFIRE with accuracies ranging from 64 to 82% for BpS and 67 to 79% for EVT. Piñon–juniper was the least accurately mapped BpS and EVT with 13 and 37% accuracy respectively.

Discussion

FRCC and LANDFIRE BpS models

Current stand conditions are determined through visual estimates of stand structure, including tree diameters, in FRCC assessments (Hann *et al*. 2004). FRCC assessments are designed to be a relatively rapid method of characterizing current vegetation and fire regime departures from historical conditions. The expense of collecting field data, such as canopy closure, canopy base height, tree density, stand age structure, and fire and stand histories, make field sampling impractical for FRCC assessments. However, based on the limited findings of this study, it appears that FRCC methods may result in inaccurate measures of plant community departure based on visually estimated

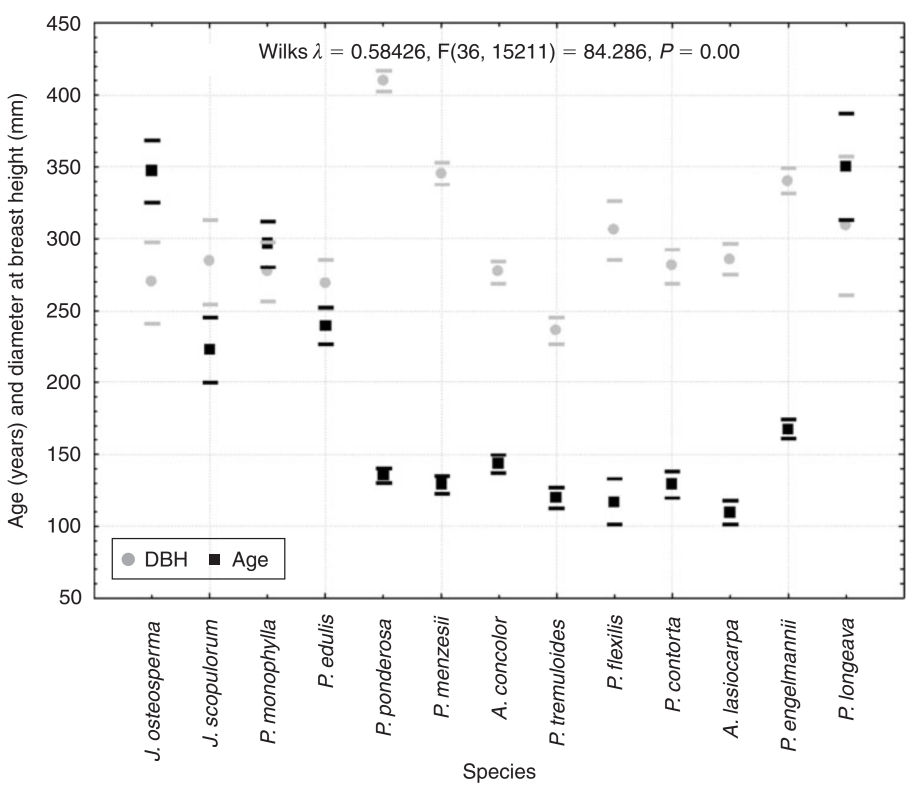

Fig. 5. ANOVA of age and diameter at breast height (DBH) by species and site. Horizontal bars represent 95% confidence intervals.

Table 4. Observed proportions of mid- and late-development reference conditions reconstructed from tree-ring data in plots collected in Utah, compared with FRCC (Fire Regime Condition Class) BpS (biophysical settings) model output for mid- and late-development reference conditions from Hann *et al*. (2004) and LANDFIRE

MFI, mean fire interval (years); *n*, number of plots in the observed data in mid- or late-seral stages. Chi-square fit is for the observed plots *v.* BpS models with 1 degree of freedom, $P = 0.05$ significance for types <3.84. BpS that meet the range of variability in the observed data are highlighted in bold

BpS description *v.* observed data	FRCC code	LANDFIRE code	Mid (%)	Late (%)	MFI	*n*		Chi-square	*P* value
						Mid	Late		
Observed PIED, PIMO, JUSC, JUPI			4	96		1	24		
Piñon–juniper infrequent fire	JUPI2		30	70	435			12.99	0.0003
Piñon–juniper frequent fire	JUPI1		50	50	31			21.16	<0.0001
Colorado Plateau piñon–juniper woodland		10160	55	45	128			26.273	<0.0001
Observed PIPO			26	74		13	37		
Colorado plateau ponderosa	**PPIN5**		25	75	6			**0.027**	0.87
Southern Rocky Mountain ponderosa pine woodland		10540	44	56	15			6.575	0.01
Observed PSME, ABCO, PIPO, PIEN			48	52		35	38		
South-western mixed-conifer	MCAN		35	65	10			5.377	0.02
Rocky Mountain dry–mesic montane mixed-conifer forest and woodland		**10510**	40	60	10			**1.92**	0.166
RM mesic montane mixed-conifer forest and woodland		10520	75	25	33			28.498	<0.0001
Spruce–fir–Douglas-fir[A]	**SPDF**		58	42	19			**0.658**	0.417
Observed PICO			36	64		4	7		
Lodgepole pine	CHPI		65	35	125			3.965	0.046
Rocky Mountain lodgepole pine forest		10500	100	0	124			4.455	0.035
Observed PIEN, ABLA, ABCO			26	74		16	58		
RM subalpine dry–mesic spruce–fir forest and woodland		10550	65	35	212			61.206	<0.0001
Lower subalpine forest	SPFI7		80	20	91			157.622	<0.0001
Upper subalpine forest	SPFI2		70	30	143			82.474	<0.0001
Observed POTR			90	10		33	4		
Deciduous woodland–oak or aspen	DWOA		55	45	100			17.474	<0.0001
Intermountain basins aspen–mixed-conifer forest – low		**10611**	85	15	10			**0.509**	0.475
Intermountain basins aspen–mixed-conifer forest – high		**10612**	95	5	32			**2.63**	0.105
Rocky Mountain aspen forest and woodland		**10110**	80	20	27			**2.146**	0.142
PILO			0	100		0	3		

[A]Includes observed POTR plots.

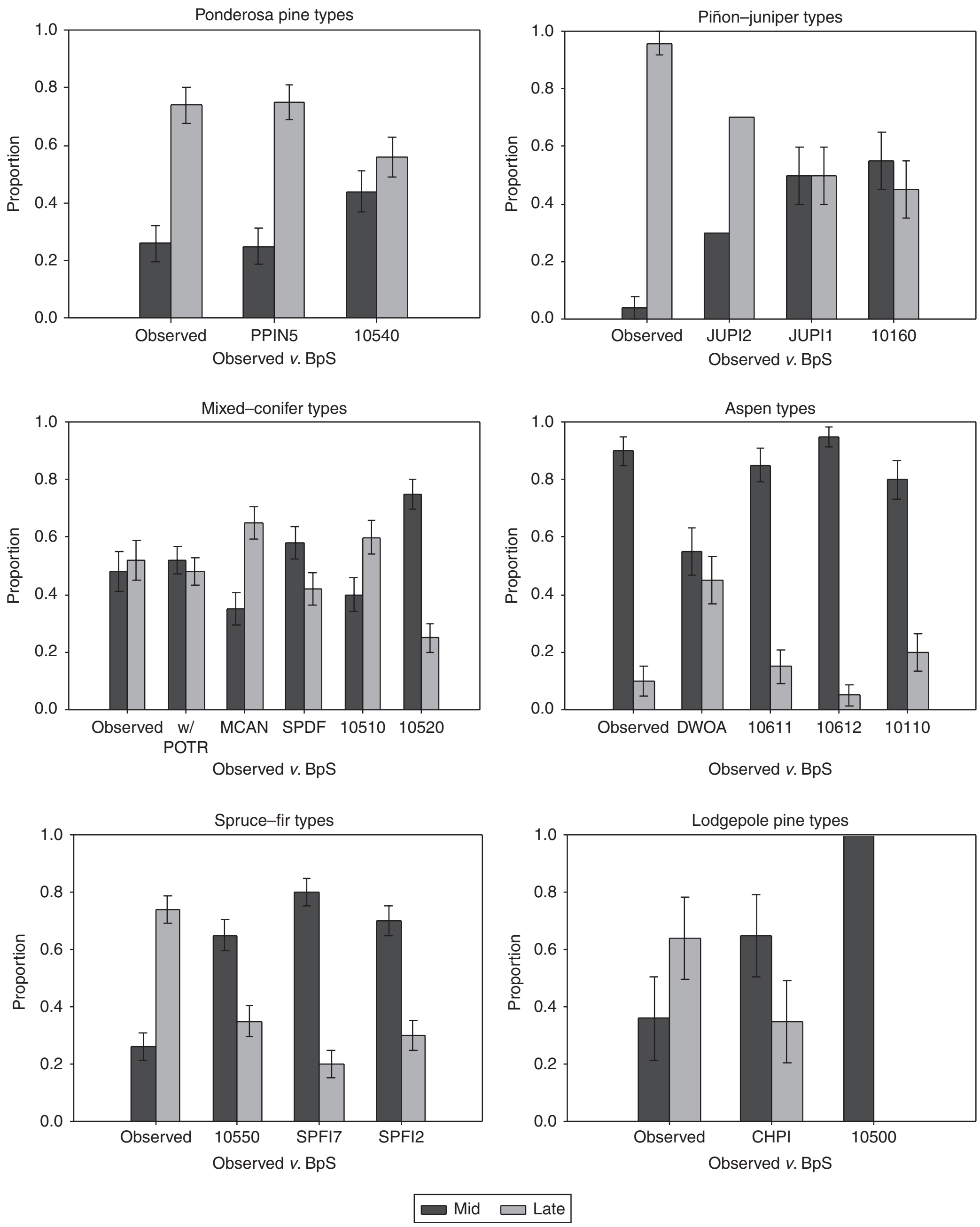

Fig. 6. Proportion of plots observed in the tree-ring data compared to FRCC (Fire Regime Condition Class) and LANDFIRE modeled reference condition proportions. Error bars were generated by calculating the 95% confidence interval from sample variance and standard error of observed points. Tree-ring results are on the left (e.g. observed), FRCC and LANDFIRE models are listed by their four-letter abbreviations on the right (e.g. PPIN5, 10540, etc.).

age–diameter relationships for determining reference condition proportions. Variations in age–size relationships both within species and among sites (Figs 2–5) may limit the ability to accurately gauge departure from estimated historical composition based on VDDT model results. Generally poor relationships between size and age may result in misassignment of current reference condition proportions based only on visual estimates, which may in turn lead to misassignment of the FRCC index.

Better correspondence between the tree-ring data and some BpS models indicates that VDDT models more accurately reflect historical forest structure in frequent-fire forest types such as ponderosa pine, mixed-conifer and aspen, than in

Table 5. LANDFIRE accuracy by BpS (biophysical settings) and EVT (existing vegetation type)
Code is the LANDFIRE map code for BpS or EVT type, *n* is number of plots tested, and % is percentage that were accurately mapped based on tree-ring data at plot scale

	Code	*n*	%
BpS			
Rocky Mountain aspen forest and woodland	10110	31	32
Colorado Plateau piñon–juniper woodland	10160	29	14
Rocky Mountain lodgepole pine forest	10500	7	43
Rocky Mountain dry–mesic montane mixed-conifer forest and woodland	10510	6	33
Rocky Mountain mesic montane mixed-conifer forest and woodland	10520	11	64
Southern Rocky Mountain ponderosa pine woodland	10540	19	53
Rocky Mountain subalpine dry–mesic spruce–fir forest and woodland	10550	82	66
Intermountain basins aspen–mixed-conifer forest – low elevation	10611	22	82
Intermountain basins aspen–mixed-conifer forest – high elevation	10612	31	77
Intermountain basins mountain mahogany woodland and shrubland	10620	6	50
EVT			
Rocky Mountain aspen forest and woodland	2011	26	50
Colorado Plateau piñon–juniper woodland and shrubland	2016	43	37
Rocky Mountain lodgepole pine forest	2050	19	63
Rocky Mountain montane mesic mixed-conifer forest and woodland	2052	9	78
Southern Rocky Mountain ponderosa pine woodland	2054	24	46
Rocky Mountain subalpine dry–mesic spruce–fir forest and woodland	2055	53	79
Intermountain basins aspen–mixed-conifer forest and woodland	2061	64	67
Abies concolor forest alliance	2208	14	71

infrequent-fire types such as spruce–fir and piñon–juniper (Fig. 6). BpS reference condition models were determined by managers and scientists familiar with the local ecology of each region during regional workshops. BpS types that are considered to be representative of each region were identified and described based on available historical and ecological data. Some BpS types, such as ponderosa pine and dry mixed-conifer forests, have extensive fire and forest history data with which to parameterize VDDT model runs. Other BpS types are less well studied and their fire and vegetation histories less certain, especially across the range of environmental and community variation within and between regions. The better correspondence between modeled and reconstructed reference conditions in frequent-fire-type models (ponderosa pine and mixed-conifer forest types; Fig. 6) is likely related to the greater amount of fire and forest history research that has been conducted in these forest types. Conversely, fire-infrequent types (spruce, and piñon–juniper woodland types; Fig. 6) have had less fire history research conducted, with the result that their fire regimes and successional patterns are less well documented for input to VDDT modeling. Furthermore, infrequent-fire types generally have fewer observations of historical fires and forest successional changes available for adequate characterization of fire regime parameters for VDDT modeling (e.g. Brown *et al*. 2008*a*).

Another factor that undoubtedly results in varying model and data results is that individual-site fire histories often have experienced contingent historical events that lead to differences from a 'typical' or average fire regime of a particular forest type. Stochastic modeling in FRCC and LANDFIRE generalizes vegetation and its fire regimes into generic types and does not take into account site-specific variability or, more importantly, the history of climate variations or other disturbances that may have affected changes in community structure through time. Variations in site histories undoubtedly contribute to variations in ratios of actual from modeled reference conditions. For example, spruce–fir and lodgepole pine FRCC and LANDFIRE BpS models predict more mid- than late-development stands, but the Utah tree-ring data found the opposite (Fig. 6). This may be due to longer fire intervals in this region than in other areas, leading to generally older stands across landscapes. Many spruce trees found in the tree-ring study were >300 years old at the time of sampling and probably resulted from fires that occurred in the late 1600s, most commonly in 1685 (Heyerdahl *et al*. 2005). However, the current presence of older rather than younger stands does not mean that these forests are outside their historical ranges of variability in either their fire regime or forest structure, but rather that they have not had extensive fires in the intervening period that would have resulted in a larger proportion of mid-successional stands as suggested should be present based on VDDT model results. Without taking into account this history of the forest landscapes, the VDDT models suggest that there is current departure in the landscape proportions of reference conditions in Utah spruce–fir and lodgepole pine forests.

Taking into account differences in fire histories, the trend of model results toward older or younger successional classes in each BpS may be more important to consider in FRCC assessments rather than the absolute proportions of stand structures. This may provide a more realistic perspective for assessing whether a particular BpS should be considered as inside or outside of its historical range of variation. For example, the tree-ring fire data for piñon–juniper (P-J) woodlands show the majority of stands are currently in late-development structural stages (Fig. 6). The FRCC BpS model JUPI2 (Table 4) also predicts more late-development trees than younger, but underpredicts what was found in the tree-ring data. The sensitivity of the VDDT model to fire frequency is critical to the setting of reference conditions. The model inaccuracy may be due to the model's fire

return interval, currently predicted to be ~450 years. If the interval is increased (~1000 years), the model begins to more closely reflect the tree-ring results. A recent assessment of (P-J) ecosystems in the western US concluded that fire was only a minor disturbance in many less productive stands because of lack of both surface and crown fuels with which to carry fire (Romme *et al*. 2009). We believe that many of the Utah stands sampled probably fell into this category of fire regime historically, which means that if the longer intervals had been used in VDDT modeling, the reference conditions would likely be closer to what was found in the tree-ring data. The error may also be due to the definition of a mid-development stand in terms of the age; the mean ages of sampled piñon and juniper were among the highest in the tree-ring study. The mid-definition could be changed for P-J to an older age class by species to define the mid- from late-successional classes in the reference conditions.

Good correspondence between the tree-ring data and models for ponderosa pine (PPIN5), aspen (10110), and mixed-conifer (SPDF, 10510; Fig. 6) suggests that the reference conditions for these BpS were accurately modeled by VDDT parameters, at least in the Utah study sites. However, results of this study suggest that inaccuracy in piñon–juniper and subalpine types makes any decision based on a VDDT output possibly subject to error. For BpS types in which disturbance may not be the major or only factor in tree recruitment, VDDT models may need further evaluation. Additional empirical disturbance and forest history sampling in piñon–juniper, spruce–fir, and lodgepole pine types should increase the available information about these systems to use in VDDT modeling. However, because of generally longer fire intervals in these forests, any departure from historical to present conditions may be less than in frequent-fire BpS such as ponderosa pine and mixed-conifer forests. A possible result of inaccurate estimations of departure and wrong FRCC classification may be the application of incorrect management actions that could lead to even further departure from historical conditions (see also Romme *et al*. 2009).

The only accurate way to establish the age of a stand is to physically sample the trees for ages. We suggest based on the results of our comparison that at least some limited age sampling is needed for FRCC assessments. This sampling probably should include removing cores from the field and crossdating by trained dendrochronologists to most accurately characterize age and successional status of stands. Additional field-sampled fire history and stand establishment data, especially in the less-well-studied ecosystems, should further increase the accuracy of VDDT models through better dynamic estimations of age structures and relationships with fire regimes. However, we also realize that this type of sampling is expensive and – perhaps more critically to the efficient use of FRCC in forest management decisions – more time-consuming than FRCC visual assessment methods as currently practiced. Nevertheless, we suggest that some sort of compromise solution could be found that would provide both the most accurate as well as timely data possible for FRCC assessment needs.

LANDFIRE maps

Zhu *et al*. (2006) used a cross-validation technique to determine that existing vegetation data layer accuracies are between 60 and 89% in LANDFIRE maps. Our study's comparison of LANDFIRE and tree-ring data falls on the lower end of the estimate of Zhu *et al*. (2006) (Table 4). When broken down by BpS and EVT, some types are more accurately represented in LANDFIRE data than others. EVT mapping in LANDFIRE is most accurate for the mixed-conifer and spruce–fir types. These forests generally have the densest and most continuous canopies, and may have been easiest to identify through remote sensing methods because of their continuous canopies and more distinctive NDVI reflectance in Landsat spectral bands (Zhu *et al*. 2006). Conversely, sparser canopy cover may have led to lower accuracy in other types such as piñon–juniper, which is similar to what Zhu *et al*. (2006) found. It should be noted, however, that piñon–juniper plots sampled for the tree-ring study were generally found in ecotonal areas near lower ends of study sites, and may not be wholly representative of piñon–juniper BpS as defined in the LANDFIRE mapping effort.

Conclusion

Historical forest conditions reconstructed from tree-ring data provide opportunities for comparison with FRCC and LANDFIRE modeled vegetation data across multiple forest types. The tree-ring reconstructions we examined suggest that reference conditions are better modeled in frequent-fire forest types but not necessarily in infrequent-fire forest types, at least in Utah forests. Additional studies in fire-infrequent forest types should increase understanding of historical stand compositions, fire histories, and other disturbances with which to better parameterize VDDT reference condition models. The greatest amount of fire history research has been conducted in ponderosa pine and mixed-conifer forests, which likely contributed to the better correspondence between tree-ring data and VDDT model results that we found in this study. We consider this study as only a first step in comparison of empirical vegetation data with vegetation models used in both FRCC assessments and the nationwide LANDFIRE mapping effort. Tree-ring data provide an opportunity to compare site-specific vegetation patterns and fire regime variations that are often not easily accounted for in modeling efforts. Revised methods for assessing FRCC may need to take into greater account both tree ages and stand histories to more accurately compare with model results. We also suggest that ranges of reference conditions be incorporated into the BpS classifications to better take into account fire and forest histories rather than trying to establish average conditions that must be met for a FRCC index to be assigned.

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Land Use Planning and Wildfire: Development Policies Influence Future Probability of Housing Loss

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Abstract

Increasing numbers of homes are being destroyed by wildfire in the wildland-urban interface. With projections of climate change and housing growth potentially exacerbating the threat of wildfire to homes and property, effective fire-risk reduction alternatives are needed as part of a comprehensive fire management plan. Land use planning represents a shift in traditional thinking from trying to eliminate wildfires, or even increasing resilience to them, toward avoiding exposure to them through the informed placement of new residential structures. For land use planning to be effective, it needs to be based on solid understanding of where and how to locate and arrange new homes. We simulated three scenarios of future residential development and projected landscape-level wildfire risk to residential structures in a rapidly urbanizing, fire-prone region in southern California. We based all future development on an econometric subdivision model, but we varied the emphasis of subdivision decision-making based on three broad and common growth types: infill, expansion, and leapfrog. Simulation results showed that decision-making based on these growth types, when applied locally for subdivision of individual parcels, produced substantial landscape-level differences in pattern, location, and extent of development. These differences in development, in turn, affected the area and proportion of structures at risk from burning in wildfires. Scenarios with lower housing density and larger numbers of small, isolated clusters of development, i.e., resulting from leapfrog development, were generally predicted to have the highest predicted fire risk to the largest proportion of structures in the study area, and infill development was predicted to have the lowest risk. These results suggest that land use planning should be considered an important component to fire risk management and that consistently applied policies based on residential pattern may provide substantial benefits for future risk reduction.

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Introduction

The recognition that homes are vulnerable to wildfire in the wildland-urban interface (WUI) has been established for decades [e.g., 1,2]; but with a recent surge in structures burning, this issue is now receiving widespread attention in policy, the media, and the scientific literature. Single fire events, like those in Greece, Australia, southern California, and Colorado have resulted in scores of lost lives, thousands of structures burned, and billions of dollars in expenditures [3–6]. With the potential for increasingly severe fire conditions under climate change [7] and projections of continued housing development [8], it is becoming clear that more effective fire-risk reduction solutions are needed. "Fire risk" here refers to the probability of a structure burning in a wildfire within a given time period.

Traditional fire-risk reduction focuses heavily on fire suppression and manipulation of wildland vegetation to reduce hazardous fuels [9]. Enormous resources are invested in vegetation management [10], but as increasing numbers of homes burn down despite this massive investment, the "business-as-usual" approach to fire management is undergoing reevaluation. One issue is that fuel treatments may not be located in the most strategic positions, i.e., in the wildland-urban interface [11]. Yet, even if treatments surrounded all communities, scattered development patterns are difficult for firefighters to reach [12–14], and fuel treatments do little to protect homes without firefighter access [15–16]. Fuel treatments may also be ineffective against embers or flaming materials that blow ahead of the fire front [17].

One alternative to traditional fire management that is receiving widespread attention is to prepare communities through the use of fire-safe building materials or creating defensible space around structures [17–18]. These actions represent an important shift in emphasis from trying to prevent wildfires in fire-prone areas to better anticipating fires that are ultimately inevitable. Nevertheless, the cost of building and retrofitting homes to be fire-safe can be prohibitive, and these actions do not guarantee immunity from fire [19].

Land use planning is an alternative that represents a further shift in thinking, beyond the preparation of communities to withstand an inevitable fire, to preventing new residential structures from being exposed to fire in the first place. The reason homes are vulnerable to fires at the wildland-urban interface is a function of its very definition: "where homes meet or intermingle with wildland vegetation" [20]. In other words, the location and

pattern of homes influence their fire risk, and past land-use decision-making has allowed homes to be constructed in highly flammable areas [21]. Land use planning for fire safety is beginning to receive some attention in the literature [22–23], and there is growing recognition of the potential benefits of directing development outside of the most hazardous locations [8,19,24].

Despite recent attention in the literature, land use planning for wildfire has yet to gain traction in practice, particularly in the United States. However, fire history has been used to help define land zoning for fire planning in Italy [22], and bushfire hazard maps are integrated into planning policy in Victoria, Australia [25]. Although some inertia inevitably arises from complications with existing policy and plans, a primary impediment to the design and implementation of fire-smart land use planning is lack of guidance about specific locations, patterns of development, or appropriate methodology to direct the placement of new development. Without a solid knowledge base to draw from, planners will be misinformed about which planning decisions may result in the greatest overall reduction of residential landscape risk. Even worse, poor science could result in placement of homes in areas that actually have high fire hazard.

Research on how planning decisions contributed to structures burning in the past provides some guidance about what actions may work in the future. Analysis of hundreds of homes that burned in southern California the last decade showed that housing arrangement and location strongly influence fire risk, particularly through housing density and spacing, location along the perimeter of development, slope, and fire history [26]. Although high-density structure-to-structure loss can occur [27–28], structures in areas with low- to intermediate- housing density were most likely to burn, potentially due to intermingling with wildland vegetation or difficulty of firefighter access. Fire frequency also tends to be highest at low to intermediate housing density, at least in regions where humans are the primary cause of ignitions [29–30].

These results suggest, for example, that placing new residential development within the boundaries of existing high-density developments or in areas of low relief may reduce fire risk. However, it is difficult to know whether broad-scale planning policies would actually result in the intended housing arrangement and pattern at the landscape scale, and whether those patterns would result in lower fire risk. Our objective here was to simulate three scenarios of future residential development, and to project wildfire risk, in a rapidly urbanizing and fire-prone region where we have studied past structure loss [25]. We based all future development on an econometric subdivision model, but we varied the emphasis of subdivision decision-making based on three broad and common growth types.

Although cities vary in extent, fragmentation, and residential density [31–32], urban form typically adheres to a set of common patterns [33–34], and we based our development scenarios on the three primary means by which residential development typically occurs: infill, expansion, or leapfrog [34]. Infill is characterized by development of vacant land surrounded by existing development, typically in built-up areas where public facilities already exist. [35–36], and should result in higher structure density rather than increased urban extent. Expansion growth occurs along the edge of existing development, extends the size of the urban patch to which it is adjacent, and may have variable influence on structure density. Leapfrog growth occurs when development occurs beyond existing urban areas such that the new structure is surrounded by undeveloped land. This type of growth would expand the urban extent and initially result in lower structure density; but these areas may eventually become centers of growth from which infill or expansion can occur. We asked:

1) Do residential development policies reflecting broad growth types affect the resulting pattern and footprint of development across the landscape?
2) Do differences in extent, location, and pattern of residential development translate into differences in wildfire risk, based on the current configuration of structures?
3) Which development process, infill, expansion, or leapfrog, results in the lowest projected fire risk across the landscape?

Methods

Study Area

The study area included all land within the South Coast Ecoregion of San Diego County, California, US, encompassing an area of 8312 km^2. The region is topographically diverse with high levels of biodiversity, and urban development has been the primary cause of natural habitat loss and species extinction [37]. Owing to the Mediterranean climate, with mild, wet winters and long summer droughts, the native shrublands dominating the landscape are extremely fire-prone. San Diego County was the site of major wildfire losses in 2003 and 2007 [38], although large wildfire events have occurred in the county since record-keeping began, and are expected to continue, as fire frequency has steadily increased in recent decades [29,39]. The county is home to more than three million residents, and approximately one million more people are expected by 2030 [40]. Although most residential development has been concentrated along the coast, expansion of housing is expected in the eastern, unincorporated part of the county.

Econometric Subdivision Model

A host of alternative modeling approaches exist to simulate future land use scenarios [41], including a cellular automaton model that we previously applied to the study area [42]. We chose to use an econometric modelling approach for this study because we wanted to capture fine-scale, structure-level patterns and processes that are correlated with housing loss to wildfire [26]; and econometric models may perform better at the scale of individual parcels [43].

Although we based the three development scenarios on generalized planning policies, we also wanted to ensure that the residential projections were realistic and adhered to current planning regulations. The objective of the econometric modeling was to estimate the likelihood that residential parcels will subdivide in the future. Therefore, we used a probit model to estimate the transition probability of each parcel based on a range of potential explanatory variables typically associated with parcel subdivision and housing development [44–45].

To develop the model of subdivision probability, we acquired GIS data of the county's parcel boundaries in years 2005 and 2009 from the San Diego Association of Governments (SANDAG). The dependent variable was equal to 1 if a parcel subdivided between 2005 and 2009, and zero otherwise. Using these data layers we first determined which parcels were legally able to subdivide given current land use regulations. Minimum lot size restrictions are typically considered the most import restriction for determining future land use. We deemed a parcel eligible for subdivision if the current lot size was greater than twice the minimum legal size given the land class. To determine which parcels subdivided between 2005 and 2009, we queried parcel IDs where the total

area was reduced by at least the minimum lot size between the two time periods. Finally, we were able to generate a suite of variables that determine the likelihood of a parcel developing in the future (Table S1).

We overlaid the parcel boundaries over a range of GIS layers representing our explanatory variables. These data are available to download at (http://www.sandag.org/index.asp?subclassid = 100&fuseaction = home.subclasshome). Our explanatory variables included: parcel size, parcel size squared, six dummy variables which capture non-linear effects of parcel size, distance to the coast, distance to the coast squared; distance to city center and its square, current zoning, slope, land use, roads, if the parcel is in a protected area, if the parcel is in a development area, if the parcel is in the redevelopment area (Table 1).

Spatial Model of Future Development under Planning Alternatives

The outcome of the land use change econometric model is the subdivision probability for each parcel for a five-year time step. Based on these probabilities, we developed a GIS spatial simulation model of future land use under three distinct planning scenarios: infill (development in open or low density parcels within already developed areas), expansion (development on the fringe of developed areas), and leapfrog (development in open areas). The model runs in four 5-year time steps from 2010 to 2030, and generates the spatial locations of new housing units in the county.

Although development decisions could feasibly depend on fire risk, we did not model that here. There is no evidence that fire has influenced past regional planning decisions, so it was not used as an explanatory variable in the econometric model. Although we could have evaluated the potential for future development decisions to be based in part on fire risk, this would have required simulation of feedbacks between fires and probability of development. Because our objective in this study was to isolate the effects of the three distinct growth types, we modeled fire risk only as a function of development pattern and not vice versa.

We constructed a complete spatial database of existing residential structures in the study area [26]. These structures and their corresponding parcel boundaries served as the initial conditions for all three scenarios of the spatial simulation model. The current and projected future GIS layers of structures were also subsequently used in the fire risk model (see below). The

Table 1. Variables and results from the probit regression model of parcel subdivision in San Diego County.

Subdivided (1 = yes,0 = no)	Coefficient	Std. Err.	z	P>|z|	[95% Conf. Interval]	
Acres of lot	0.0026342	0.00075	3.51	0	0.001164	0.004105
Acres of lot ^{2}	−3.02E-06	1.29E-06	−2.34	0.019	−5.55E-06	−4.93E-07
Distance to ocean	−7.42E-06	1.33E-06	−5.59	0	−0.00001	−4.82E-06
Distance to ocean ^{2}	2.33E-11	8.28E-12	2.82	0.005	7.11E-12	3.96E-11
Distance to major road	2.17E-07	2.74E-06	0.08	0.937	−5.16E-06	5.59E-06
Distance to major road ^{2}	−1.94E-11	1.70E-11	−1.14	0.252	−5.27E-11	1.38E-11
Distance to nearest city center	−0.0000115	1.70E-06	−6.76	0	−1.5E-05	−8.16E-06
Distance to nearest city center ^{2}	2.89E-11	9.70E-12	2.98	0.003	9.91E-12	4.79E-11
Slope between 0–5%	0.6211289	0.211761	2.93	0.003	0.206085	1.036173
Slope between 5–10%	0.3911427	0.210684	1.86	0.063	−0.02179	0.804076
Slope between 10–25%	0.0716669	0.212725	0.34	0.736	−0.34527	0.4886
Rural Residential	−0.3563149	0.071512	−4.98	0	−0.49648	−0.21615
Single Family	0.1361149	0.068678	1.98	0.047	0.001509	0.270721
Multi-Family	−0.2505093	0.151486	−1.65	0.098	−0.54742	0.046397
Road	0.015329	0.086094	0.18	0.859	−0.15341	0.184069
Open Space	−0.7440933	0.099145	−7.51	0	−0.93841	−0.54977
Orchard/Vineyard	−0.5813305	0.097867	−5.94	0	−0.77315	−0.38951
Agriculture	−0.9785208	0.132734	−7.37	0	−1.23867	−0.71837
Vacant Land	−0.5222501	0.074586	−7	0	−0.66844	−0.37606
Zoned protected	0.253769	0.076881	3.3	0.001	0.103086	0.404452
Area marked for redevelopment	−0.2680261	0.14069	−1.91	0.057	−0.54377	0.007722
Area marked for development	0.5780101	0.064103	9.02	0	0.452371	0.703649
Parcel between 10–20 acres	−0.3379532	0.065899	−5.13	0	−0.46711	−0.20879
Parcel between 5–10 acres	−0.6119036	0.067012	−9.13	0	−0.74325	−0.48056
Parcel between 2–5 acres	−1.16297	0.07062	−16.47	0	−1.30138	−1.02456
Parcel between 1–2 acres	−1.563956	0.090286	−17.32	0	−1.74091	−1.387
Parcel between.5–1 acres	−1.999939	0.099893	−20.02	0	−2.19573	−1.80415
Parcel between.25–.5 acres	−2.178273	0.117101	−18.6	0	−2.40779	−1.94876
Constant	−1.397931	0.227467	−6.15	0	−1.84376	−0.9521

Sample size 113 001, LR Chi^2 1535.23, pro>chi 0, pseudo R^2 0.22. Further description of the variables is provided in Table S1.
doi:10.1371/journal.pone.0071708.t001

dataset of existing housing includes locations of 687,869 structures, of which 4% were located within the perimeter of one of 40 fires that burned since 2001. During these fires, 4315 structures were completely destroyed, and another 935 were damaged.

For future development scenarios, we wanted to allocate an equal number of new structures to the landscape. This was to ensure that any predicted difference in fire risk was a function of the arrangement and location of structures, not the total number of structures. Nevertheless, differences in the total number of structures were simulated with each of the 5-year time steps. We determined the number of housing units to add during the simulations based on projections made by San Diego County [46]. Using factors such as development proposals, general plan densities, and information from jurisdictions, the county estimated that between 331,378 units and 486,336 units could be supported within the developable residential land by 2030. Because the eastern, desert portion of the county was not included in our study area, we used a conservative approach and simulated the addition of 331,378 new dwelling units. We divided this number by four to define the number of new dwelling units to add at each time step, assuming a linear growth rate.

One output of the econometric model was the prediction of the maximum number of new dwelling units that could be added to each parcel. However, dwelling units may consist of apartments as well as single family homes. The mix of single and multifamily units in the region has remained relatively constant over time, and the overall trend has been a mix of roughly 1/3 multifamily and 2/3 single family units. Because the fire risk model is based on points representing structure locations across the landscape, regardless of the number of dwelling units per structure, we needed to generate a conversion factor from dwelling units to structures. We therefore defined a minimum lot size of 0.25 acre on which no more than a single structure could be built, regardless of the number of dwelling units in it (i.e., a single family home or apartment complex). Then, once a parcel was selected for development by the model (see details below), we divided its total area by the maximum number of dwelling units to be added, according to the econometric model. If the result was larger than 0.25, we subdivided parcels according to the result. If not, we quantified how many 0.25 acre parcels fit into the original parcel, and generated the new parcel boundaries accordingly.

Using the initial map of parcels (year 2010), we classified each parcel that was defined as eligible for development (in the previous stage) as suitable for one of the three planning scenarios described above, according to the number of developed parcels in its immediate neighborhood (i.e., those parcels that share a boundary with the focal parcel). We defined 'developed parcels' as ones that had more than one house per 20 acres (8.09 ha). Therefore, according to these density thresholds, we allowed some parcels with nonzero housing density to be considered as 'undeveloped' because these large, rural parcels might contain a single or a handful of houses but they exist within a large open area. In other words, the overall land cover of these parcels was effectively undeveloped, and we therefore assumed that development in adjacent parcels would be akin to development in open areas.

We defined infill parcels as those that were completely surrounded by developed parcels. Expansion parcels had at least one neighboring parcel that was undeveloped; and leapfrog parcels were those with no developed parcels in their immediate surroundings. We reclassified the type of each available parcel in the same manner after each time step, to account for changing dynamics in the development map of the county.

We conducted three simulations, one for each development scenario (infill, expansion, and leapfrog). In each simulation, all parcels were eligible to subdivide, regardless of their class. Therefore, to build a simulation for a specific scenario, we increased the development probability of parcels of the selected scenario by 20%, to favor their development compared to the other types of parcels, without prohibiting development in the other parcel types. This approach was necessary because the projected number of dwelling units was much larger than it would be possible to fit in infill and leapfrog class parcels solely. For example, as the spatial coverage of developed parcel expands, there is less contiguous area that is undevelopable and suitable for leapfrog development. Therefore, the scenarios are not exclusive, but rather a mixture of the three development types. Yet, in each scenario, there is one main type of development, and smaller amounts of development events of the other two types.

Due to the immense computational demand of the simulations, we adopted a deterministic, rather than a stochastic approach to decide on which parcels were subdivided. After enhancing the transition probability according to the corresponding scenario, we ranked and then sorted all parcels according to their probability of subdivision. We then sequentially selected parcels, while simultaneously tallying the number of dwelling units in them, until the development target in that time step (one fourth of the total number of dwelling units to be added: 82,795) was reached. Once the development target was reached, we moved to the next time step. After each time step, the remaining parcels that were still eligible for development were re-classified to development types according to the new spatial configuration of the landscape.

Once a parcel was selected for subdivision, and the number of new parcels to develop in it was calculated (as detailed above), an equal-area spatial splitting model was employed to split the parent parcel to the predefined number of equal-area child parcels. We developed a simple splitting model which is based on iterative splitting of larger parcels into two smaller parcels using a straight line splitting boundary. Once the parcel was fully split into the needed number of sub-parcels, we allocated a new structure inside each new parcel by generating a point at its centroid (center of gravity). The point datasets of all structure locations per time step per scenario were passed over to the fire risk model, which is described below.

Fire Risk Modeling and Analysis

To project the distribution of fire risk under alternative scenarios, we used MaxEnt [47–48], a map-based modeling software used primarily for species distribution modeling [48], but we have used it successfully for ignition modeling [50] and for projecting current fire risk in the study area [26]. For this study, we slightly modified the model from Syphard et al. [26]. The dependent variable was the location of structures destroyed by fire between 2001 and 2010. Although inclusion of damaged structures in the data set does not significantly affect results [26], we only included completely destroyed structures to avoid the introduction of any uncertainty.

The MaxEnt software uses a machine-learning algorithm that iteratively evaluates contrasts among values of predictor values at locations where structures burned versus values distributed across the entire study area. The model assumes that the best approximation of an unknown distribution (i.e., structure destruction) is the one with maximum entropy. The output is an exponential function that assigns a probability to every cell of a map. Thus, the resulting continuous maps of fire risk represented the probability of a structure being destroyed by fire. In these output maps, areas of predicted high fire risk that did not have structures on them represented environmental conditions similar to those in which structures have actually burned.

We based the explanatory variables on those that were significantly related to burned structures in Syphard et al. [26], including maps depicting housing arrangement and pattern, housing location, and biophysical factors. Housing pattern variables reflected individual structure locations as well as the arrangement of structures within housing clusters. We calculated housing clusters, defined as groups of structures located within a maximum of 100 m from each other, by creating 100 m buffers around all structures and dissolving the overlapping boundaries [51].

Because burned structures were significantly related to small housing clusters [26], we calculated the area of every cluster as an attribute, and then created raster grids based on that attribute. Low-to intermediate housing density and distance to the edge of the cluster were also significant explanatory variables relative to housing pattern and location [26], so we also created raster grids for those. GIS buffer measures at 1-km have been found to explain approximately 90% of the variation in rural residential density [52], so we developed density grids using simple density interpolation based on a 1-km search radius, with area determined through square map units. To create grids representing distance to the edge of clusters, we first collapsed the cluster polygons into vector polyline files, and then created grids of interpolated Euclidean Distance to the edge within each cluster.

Because the MaxEnt model randomly selects background samples in the map to compare with locations of destroyed structures, we used a mask to restrict sampling to the developed environment within cluster boundaries; the distance to the edge of the cluster would represent a different relationship inside a cluster boundary versus outside in the wildland. We also modified the grids to ensure that any random sample located within the 100m buffer zone would receive a value of 100m; thus, all points within the buffer were considered "the edge of the development".

After creating the grids representing housing pattern and arrangement of the current configuration of structures, we applied the same algorithms to the maps of simulated future structure locations. We thus generated grids representing future housing pattern and arrangement under alternative development scenarios. The other explanatory variables, including fire history, slope, fuel type, southwest aspect, and distance to coast [26] remained constant through time for current and future scenarios. Although historic fire frequency and fuel type typically change through time, we did not simulate their dynamics here because we wanted to isolate the effect of planning decisions on housing pattern and arrangement while holding everything else constant.

We conditioned the MaxEnt model on present distributions of housing using ten thousand random background points and destroyed structures located no closer than 500-m to minimize any effect of spatial autocorrelation. We used 80% (260 records) of these data for model training, and 20% [66 records) for testing. We repeated the process using cross-validation with five replicates and used the average of these five models for analyses. For smoother functions of the explanatory variables, we used hinge features, linear, and quadratic with an increase in regularization of beta set at 2.5, based on Elith et al. [48]. The smoother response curves minimize over fitting of the model. We conducted jackknife tests of explanatory variable importance.

We first developed the model using mapped explanatory variables derived from the current configuration of structures. To project fire risk under the different time steps of the alternative development scenarios, projected the model conditioned upon current conditions onto maps representing future conditions by substituting the grids representing future housing pattern and arrangement. This is similar to how potential future distributions of species are projected under climate change scenarios [49].

To quantify differences among current and future alternative scenarios, we calculated metrics representing housing density, pattern, and footprint to determine the extent to which the planning policies produced differences in housing pattern and location. We compared the modeled structure fire risk of the scenarios by overlaying all maps of structure locations with their respective mapped output grids from the MaxEnt models and calculating probability of burning for every structure point. We also calculated total area of risk by selecting three threshold criteria [53]. These criteria, at 0.05, 0.25, and 0.5 represented three different degrees of risk, and we calculated the proportion of structures that were located in risk areas for every time step in all scenarios.

Results

The probit econometric model, run on 113 001 observations, showed that larger parcels were most likely to subdivide, although the relationship between parcel size and subdivision probability was non-linear (Table 1). Parcels closer to existing roads, the ocean, those with lower slopes, and those designated as fit for development were all most likely to develop. Parcels designated in redevelopment areas were less likely to develop. Overall, the model had a pseudo r –squared of 0.22.

The land use simulation model, based on a combination of the econometric subdivision model and three different growth policies, resulted in substantial differences in the extent and pattern of housing of the three scenarios. The total area of housing development, or the housing footprint, was largest for simulations where leapfrog growth dominated, followed by expansion-type development, and then infill (Figure 1a). The differences in the housing footprint became larger among the scenarios over time, but the largest difference was between infill and the other two development types. As the housing footprint expanded in the three scenarios, the corresponding housing density declined, so that leapfrog growth resulted in the lowest housing density per 1-km, followed by expansion and then infill (Figure 2b). Despite the near inverse of this relationship, there was generally a larger separation among scenarios with regard to housing density. With larger housing footprints and lower housing density, the number of separate housing clusters increased while their size decreased (Figure 2c).

In the first two time steps of the model (2015 and 2020), the simulated development pattern closely followed the desired pattern in the scenario, although some of the growth in the infill scenario ended up becoming expansion or leapfrog (Table 2). In the last two time steps (2025 and 2030), there were not enough infill parcels left, and thus, the majority of growth in these simulations became expansion, followed by infill, and then leapfrog. In the last time step, there were not enough isolated parcels in the leapfrog scenario and thus, the majority of development became expansion. Thus in general, as more development occurred in the simulations by the year 2030, the majority took the form of expansion.

The area under the curve (AUC) of receiver operating characteristic (ROC) plots, indicating the ability of the MaxEnt model to discriminate between burned and unburned structures, averaged across five cross-validated replicate runs was 0.91. The AUC represents the probability that, for a randomly selected set of observations, the model prediction was higher for a burned structure than for an unburned structure [49].The two most important variables in the model according to the internal jackknife tests in MaxEnt [47] were related to housing pattern:

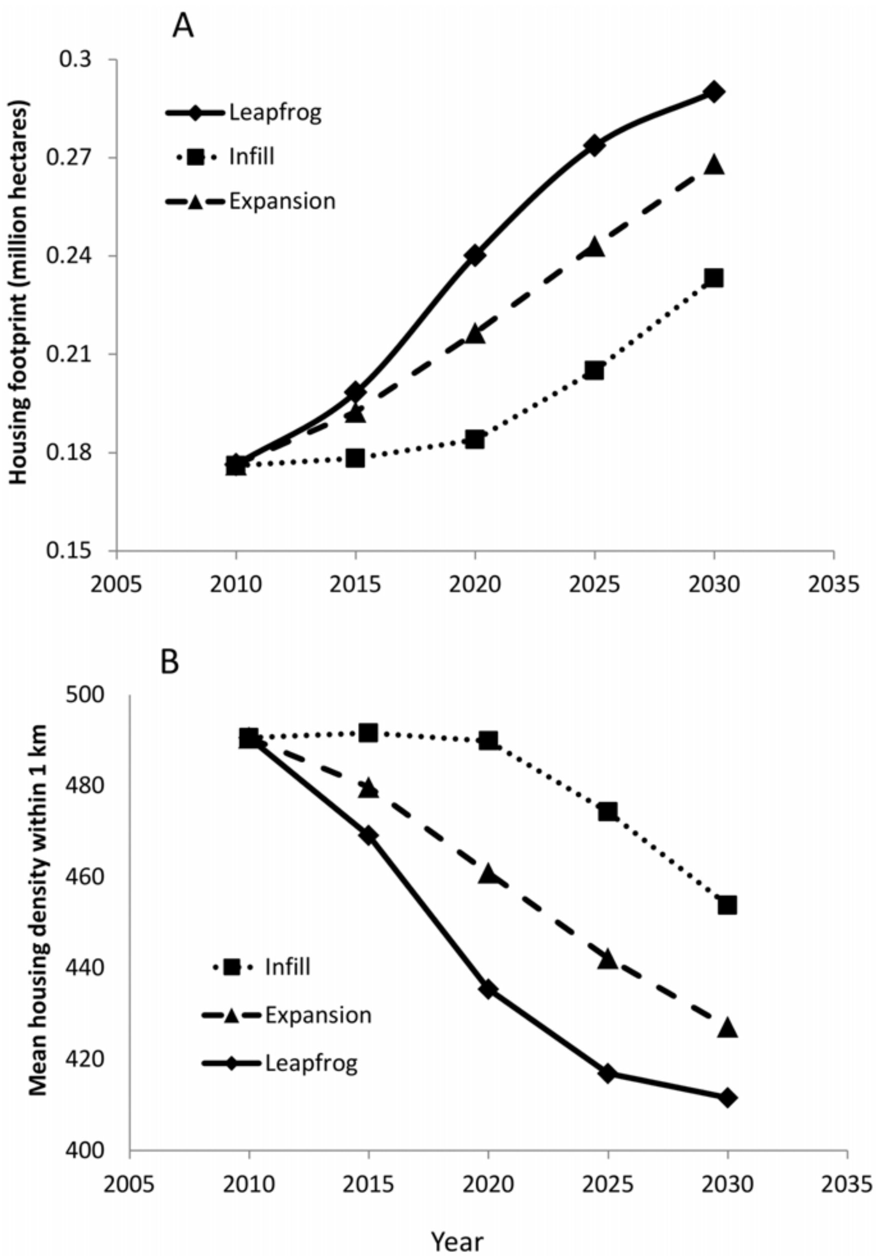

Figure 1. Trends of development extent and pattern for three planning policy simulations from 2010–2030, including A) total housing footprint representing the area of land within all housing clusters, and B) mean housing density averaged across all housing clusters.
doi:10.1371/journal.pone.0071708.g001

low to intermediate housing density and small cluster size and housing density (Figure 3). The distance to the edge of housing cluster was a less important contribution.

Maps showing the probability of a structure being destroyed in a wildfire, displayed as a gradient from low to high risk, show broad agreement relative to the general areas of the landscape that are riskiest, with correlation coefficients ranging from 0.85–0.91 (Figure 4). Nevertheless, subtle differences are apparent in the three development-scenario maps by year 2030, with the highest-risk areas in the expansion scenario located farther east than infill, and the highest-risk areas in leapfrog occupying a wider extent than either of the other two scenarios.

Differences among current housing and the three development scenarios are clearly illustrated through the mean landscape risk, or total probability of all structures burning (Figure 5). All three development scenarios were predicted to experience an increase in mean landscape risk over the duration of the simulations, except for infill at year 2015. The highest landscape risk to structures was predicted for the leapfrog scenario, followed by expansion, and then infill. The increase in risk over time is more gradual for the infill scenario than the other two scenarios.

The ranking of scenarios varied according to the proportion of structures located within different levels of risk defined through binary thresholding (Figure 6). When the continuous risk maps were thresholded at the lowest number of 0.05, a large proportion

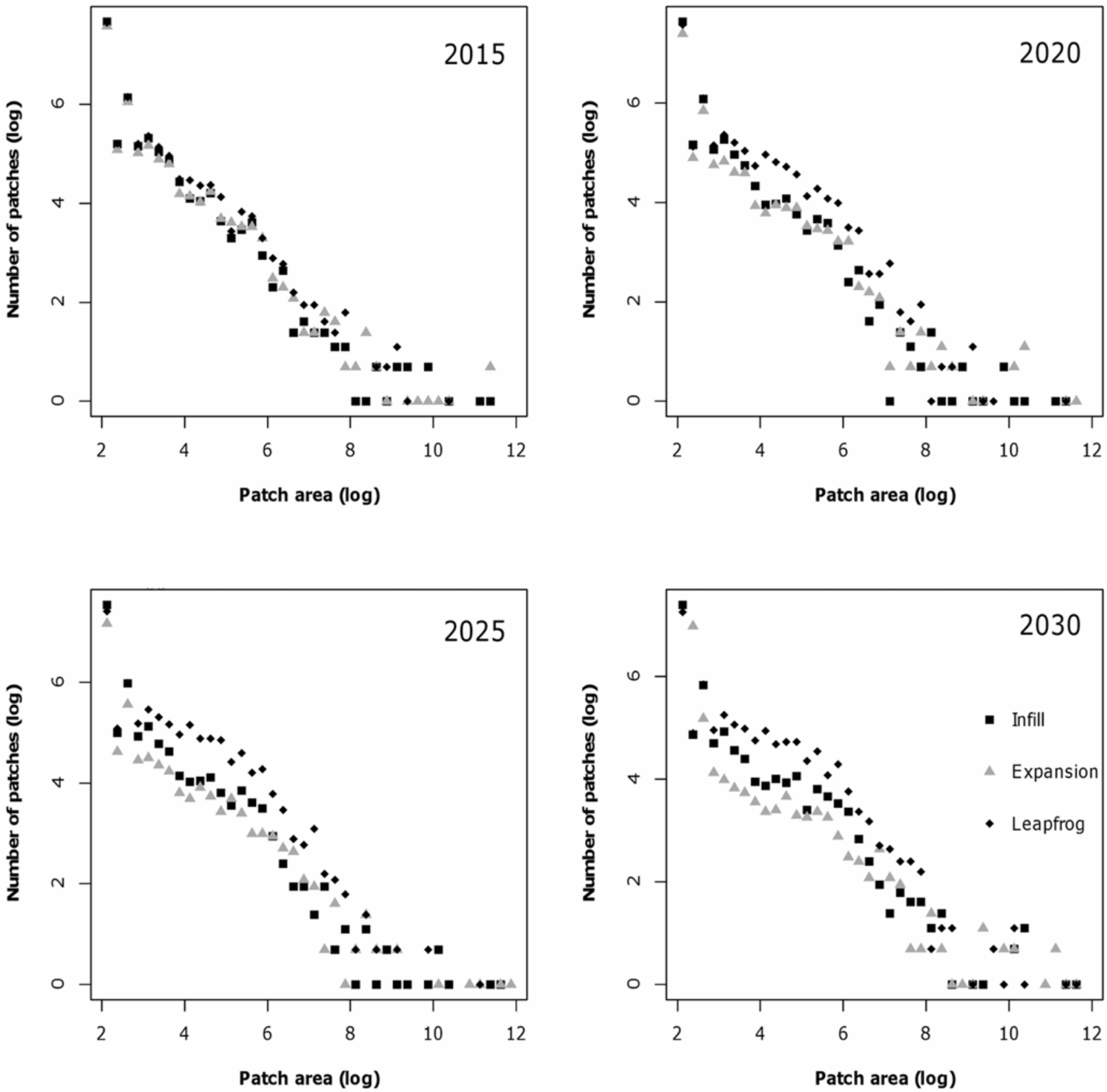

Figure 2. Trends in number of patches and patch area for three planning policy simulations from 2010–2030. Numbers were log-transformed for better visual representation of the scenarios.
doi:10.1371/journal.pone.0071708.g002

of structures in all scenarios fell within areas defined as risky according to this criterion. At this threshold, the proportion of structures in high-risk areas increased linearly for the expansion and leapfrog development scenarios while the proportion of infill homes increased more gradually. When risk was defined more conservatively at 0.25, temporal trends for the leapfrog and infill scenarios were similar to the 0.05 threshold. However, the proportion of structures at risk in the expansion scenario initially increased to 2020, but this proportion leveled off and declined by 2030. When the threshold was highest at 0.50, a very low proportion of structures in any scenario were located in areas at risk. But in these high-risk areas, the expansion scenario switched places with infill to have the lowest proportion of structures at risk in all time steps. Leapfrog had the largest proportion of homes at risk. This proportion of homes located in areas at risk with a threshold at 0.5 declined over time for all three scenarios.

Discussion

Our simulations of residential development showed that planning policies based on different growth types, applied locally for subdivision of individual parcels, will likely produce substantial and cumulative landscape-level differences in pattern, location, and extent of development. These differences in development pattern, in turn, will likely affect the area and proportion of

Table 2. Pattern of simulated development under infill, expansion, and leapfrog growth policies.

		Actual development		
Development scenario	**year**	***Infill***	***Expansion***	***Leapfrog***
Infill	2015	9450	18	6
	2020	11787	153	29
	2025	236	624	144
	2030	325	890	179
Expansion	2015	0	772	0
	2020	0	1243	2
	2025	0	1871	1
	2030	0	2662	0
Leapfrog	2015	0	10	408
	2020	0	5	1132
	2025	1	83	3563
	2030	34	917	0

The numbers in the table denote the numbers of patches of a given development type.
doi:10.1371/journal.pone.0071708.t002

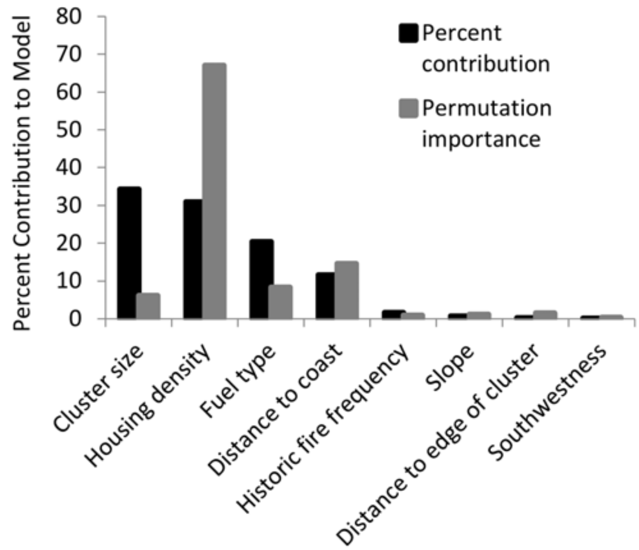

Figure 3. The importance of explanatory variables averaged across five cross-validated replications in the MaxEnt fire risk model. Percent contribution is determined as a function of the information gain from each environmental variable throughout the MaxEnt model iterations. Permutation importance reflects the drop in model accuracy that results from random permutations of each environmental variable, normalized to percentages.
doi:10.1371/journal.pone.0071708.g003

structures at risk from burning in wildfires. In particular, the scenarios with lower housing density and larger numbers of small, isolated clusters of development, i.e., leapfrog followed by expansion and infill, were generally predicted to have the highest predicted fire risk to the largest proportion of structures in the study area. Nevertheless, rankings of scenarios were affected by the definition of risk.

Theoretically, it makes sense that leapfrog development produced fragmented development with larger numbers of small patches, lower housing density, and a larger housing footprint; and that infill resulted in the opposite, with expansion in the middle. By definition, leapfrog development requires open space around all sides of the newly developed parcel, whereas infill requires development on all sides, and expansion requires development on one side and open space on another. Implementing these planning policies on real landscapes, however, can be complex if there are more houses to build than there are parcels that meet the definitions of the three planning rules, and thus not all development conforms strictly to the policy [54]. In our simulations, parcels meeting the definition of each growth type had a higher probability of subdividing; yet, as we were simulating a real landscape, many newly developed parcels did not meet the scenario criteria. That the three scenarios nevertheless produced substantial differences in landscape-level development patterns shows that decision-making at the individual level can lead to meaningful broad-scale effects.

The objective of the econometric model was to provide a baseline probability to predict which parcels were most likely to subdivide; thus, the econometric model itself provides no explanation of how a given policy affects likelihood of subdivision, although it does indicate the correlation between the policy and the outcome. In our setting, which areas are protected, marked for redevelopment, or marked for development may be endogenous to the land owner decision to subdivide. In the case of these variables especially, our results should not be interpreted as causal predictors. Likewise, we use data only from 2005–2009 to predict changes to 2030. If major changes in the land market take place over this time horizon our model will not be able to take this into account.

Although some differences in predicted fire risk among the three scenarios likely stemmed from location of new structures relative to variables such as distance to coast, fuel type, or slope, the most important variables in the fire risk model were housing density and cluster size, with most structure loss historically occurring in areas with low housing density and in small, isolated housing clusters. Thus, leapfrog development was generally the riskiest scenario and infill the least risky. The most surprising result was the variation in predicted risk for the expansion scenario over time and at different thresholds. While leapfrog and infill showed similar trajectories across thresholds, expansion went from being the highest-risk scenario at the low threshold to being the lowest-risk scenario at the highest threshold. Because the threshold is merely a way to group structures into a binary classification, this means that, while the average risk calculated across all homes shows expansion to rank in the middle of infill and leapfrog throughout the simulation (Figure 5), the other two scenarios have a relatively larger proportion of homes that are modeled to be at a very high risk (i.e., 0.25 or 0.5), particularly by the end of the simulations. Because the total number of structures with a risk greater than 0.25 or 0.5 is relatively low in all scenarios, this difference in distribution of homes at the highest risk is not reflected in the mean. Another reason for the shift in rank of expansion over time is that, as more development occupied the landscape, there were fewer parcels remaining to accomplish infill or leapfrog type growth in the other scenarios. Thus, by the end of the simulations in year 2030, the majority of growth in all scenarios was expansion, and there was some convergence between scenarios. Finally, the change in risk of expansion growth over time may reflect that, despite the relatively low importance of distance to edge of cluster as an explanatory variable, expansion growth is characterized as having an initially fragmented landscape pattern that eventually merges into large patches with low edge.

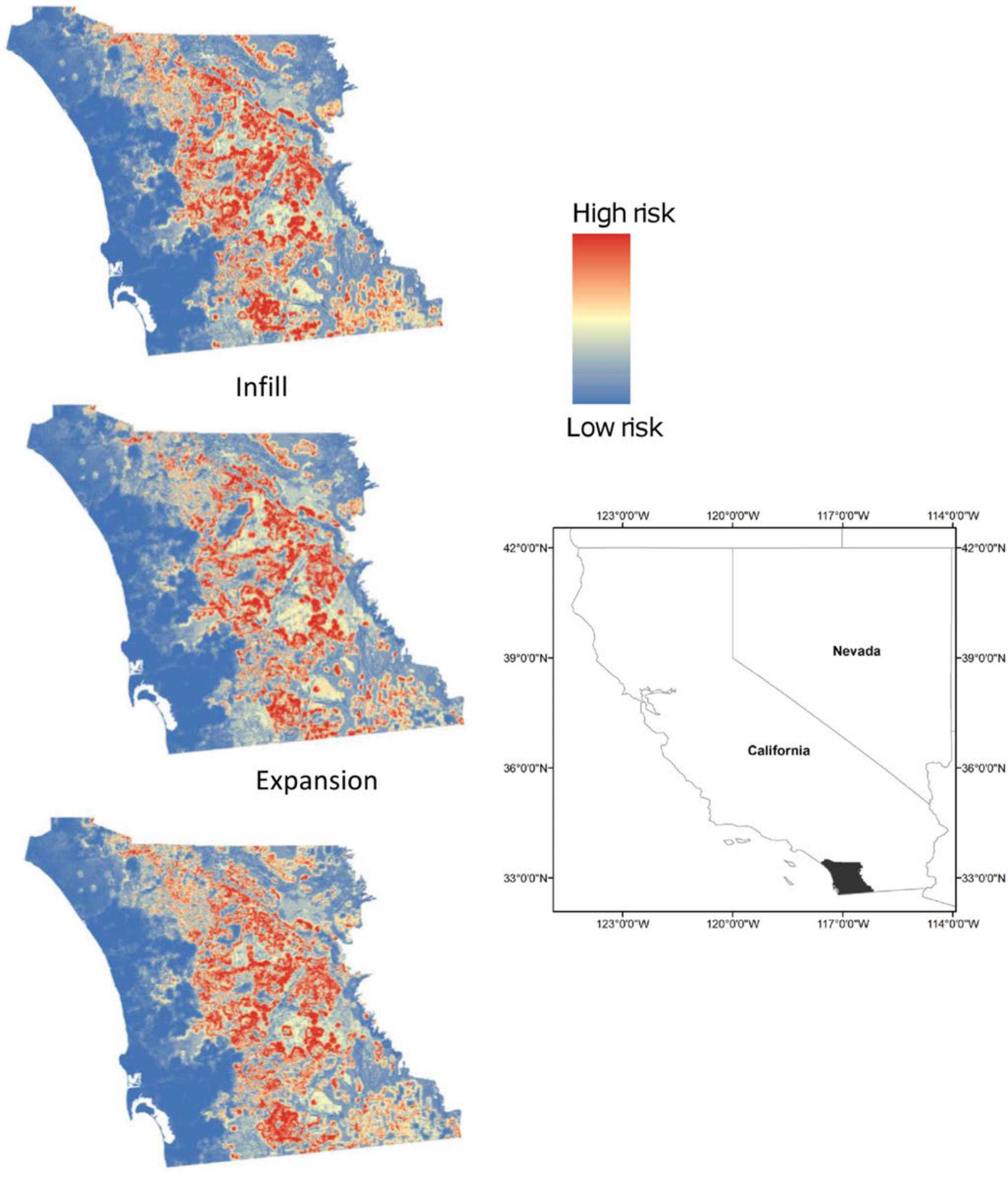

Figure 4. Maps of the study area showing projected wildfire risk at year 2030 for simulations of residential development under policies emphasizing infill, expansion, or leapfrog growth.
doi:10.1371/journal.pone.0071708.g004

Although leapfrog development clearly ranked highest in terms of fire risk, the interpretation of which planning policy is best may depend on fire management objectives and resources, as well as other considerations such as biodiversity or ecological impacts.

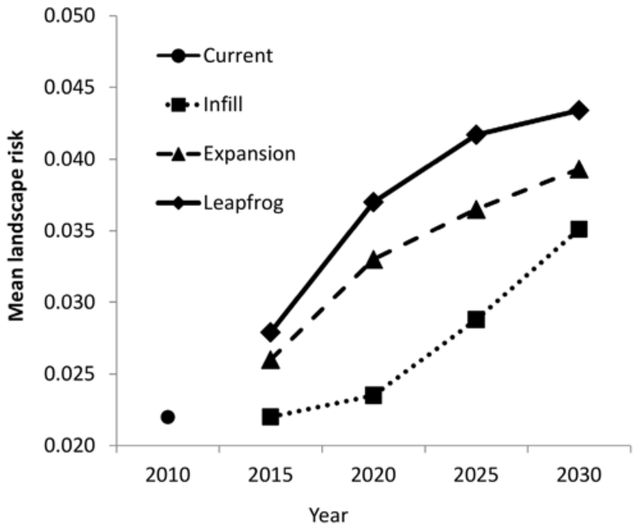

Figure 5. Projected landscape fire risk, reflecting the probability of burning in a wildfire averaged across all residential structures on the current landscape and in three development scenarios of infill, expansion, and leapfrog for year 2030.
doi:10.1371/journal.pone.0071708.g005

The spatial pattern of development affects multiple ecological functions and services [55], with potentially varying conservation implications; both leapfrog and expansion development consumed more land than infill, which would likely lead to more ecological degradation [56]; nevertheless, higher-density clustered development may be dominated by more invasive species [57]. Trade-offs between fire protection and conservation are common, but techniques are available for identifying mutually beneficial solutions [58].

Different perceptions of the fire risk results could also potentially translate into different planning priorities for management. For example, if the priority is to plan for the lowest overall risk to structures, then the mean landscape risk clearly delineates the rankings of options, with infill being the winner. However, if the objective is to reduce the number of structures at the highest risk threshold, i.e., $> = 0.5$, then expansion is the best option, at least by 2030. An important consideration for fire management is the total area that needs to be protected, as well as the length of wildland-urban interface [8,13]. Therefore, despite the lower number of structures at the highest risk thresholds, expansion creates more edge than infill and may translate into greater challenges for firefighter protection.

Although we did not create separate scenarios for high or low growth, the results at different time steps can be substituted to envision the potential outcome of developing more or fewer houses. In the short term, the total fire risk is projected to increase proportionately as more land is developed. However, given the inverse relationship between housing density and fire risk, it is possible that this trend could reverse if housing growth eventually resulted in expansive high-density development.

Land use planning is one of a range of options available for reducing fire risk, and the best outcome will likely be achieved through a combination of strategies that include homeowner actions, improvements in fire-safe building codes, and advanced fire suppression tactics. Although we isolated the effect of land use planning policy in the three development scenarios, the fire risk model nevertheless showed that the pattern and location of structures in this study area were the most important out of a suite of factors influencing structure loss. We used a correlative approach that did not incorporate mechanisms or feedbacks, but our models clearly illustrated differences in the cumulative effects of individual planning decisions. The relationship between spatial pattern of development and fire risk is likely related to the intermixing of development and wildland vegetation [29,59]; thus, these results likely apply to a wide range of fire-prone ecosystems with large proportions of human-caused ignitions. Nevertheless, because fire risk is highly variable over space and time, and due to a range of human and biophysical variables [60], we recommend planners develop their own models for the best understanding of where the most fire-prone areas are in their region [19].

With projections of substantial global change in climate and human development, we recommend that land use planning should be considered as an important component to fire risk management, potentially to become as successful as the prevention of building on flood plains [61]. History has shown us that preventing fires is impossible in areas where large wildfires are a natural ecological process [4,9]. As Roger Kennedy put it, "the

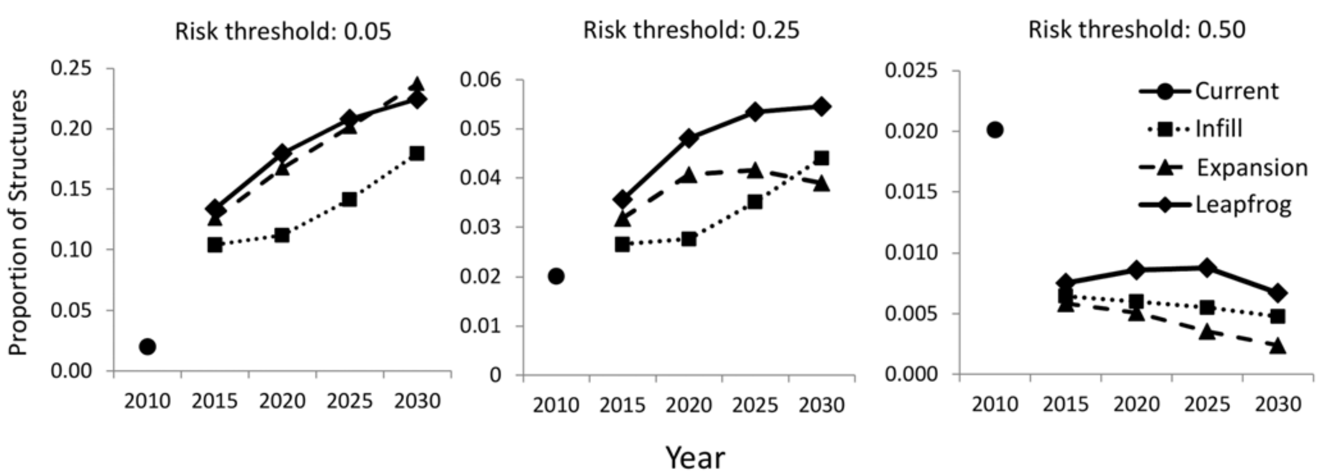

Figure 6. Proportion of residential structures that are located in areas of high fire risk defined using thresholds from the fire risk model of 0.05, 0.25, and 0.5 for current structures and for structures simulated under infill, expansion, and leapfrog growth policies.
doi:10.1371/journal.pone.0071708.g006

problem isn't fires; the problem is people in the wrong places [62]."

Supporting Information

Table S1 Definitions and summary statistics for variables used in the probit model.
(DOCX)

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Author Contributions

Conceived and designed the experiments: ADS JEK ABM VB. Performed the experiments: ADS ABM VB. Analyzed the data: ADS ABM VB. Wrote the paper: ADS JEK ABM VB.

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The role of defensible space for residential structure protection during wildfires

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Abstract. With the potential for worsening fire conditions, discussion is escalating over how to best reduce effects on urban communities. A widely supported strategy is the creation of defensible space immediately surrounding homes and other structures. Although state and local governments publish specific guidelines and requirements, there is little empirical evidence to suggest how much vegetation modification is needed to provide significant benefits. We analysed the role of defensible space by mapping and measuring a suite of variables on modern pre-fire aerial photography for 1000 destroyed and 1000 surviving structures for all fires where homes burned from 2001 to 2010 in San Diego County, CA, USA. Structures were more likely to survive a fire with defensible space immediately adjacent to them. The most effective treatment distance varied between 5 and 20 m (16–58 ft) from the structure, but distances larger than 30 m (100 ft) did not provide additional protection, even for structures located on steep slopes. The most effective actions were reducing woody cover up to 40% immediately adjacent to structures and ensuring that vegetation does not overhang or touch the structure. Multiple-regression models showed landscape-scale factors, including low housing density and distances to major roads, were more important in explaining structure destruction. The best long-term solution will involve a suite of prevention measures that include defensible space as well as building design approach, community education and proactive land use planning that limits exposure to fire.

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Introduction

Across the globe and over recent decades, homes have been destroyed in wildfires at an unprecedented rate. In the last decade, large wildfires across Australia, southern Europe, Russia, the US and Canada have resulted in tens of thousands of properties destroyed, in addition to lost lives and enormous social, economic and ecological effects (Filmon 2004; Boschetti *et al.* 2008; Keeley *et al.* 2009; Blanchi *et al.* 2010; Vasquez 2011). The potential for climate change to worsen fire conditions (Hessl 2011), and the projection of continued housing growth in fire-prone wildlands (Gude *et al.* 2008) suggest that many more communities will face the threat of catastrophic wildfire in the future.

Concern over increasing fire threat has escalated discussion over how to best prepare for wildfires and reduce their effects. Although ideas such as greater focus on fire hazard in land use planning, using fire-resistant building materials and reducing human-caused ignitions (e.g. Cary *et al.* 2009; Quarles *et al.* 2010; Syphard *et al.* 2012) are gaining traction, the traditional strategy of fuels management continues to receive the most attention. Fuels management in the form of prescribed fires or mechanical treatments has historically occurred in remote, wildland locations (Schoennagel *et al.* 2009), but recent studies suggest that treatments located closer to homes and communities may provide greater protection (Witter and Taylor 2005; Stockmann *et al.* 2010; Gibbons *et al.* 2012). In fact, one of the most commonly recommended strategies in terms of fuels and fire protection is to create defensible space immediately around structures (Cohen 2000; Winter *et al.* 2009). Defensible space is an area around a structure where vegetation has been modified, or 'cleared,' to increase the chance of the structure surviving a wildfire. The idea is to mitigate home loss by minimising direct contact with fire, reducing radiative heating, lowering the probability of ignitions from embers and providing a safer place for fire fighters to defend a structure against fire (Gill and Stephens 2009; Cheney *et al.* 2001). Many jurisdictions provide specific guidelines and practices for creating defensible space, including minimum distances that are required among trees and shrubs as well as minimum total distances from the structure. These distances may be enforced through local ordinances or state-wide laws. In California, for example, a state law in 2005 increased the required total distance from 9 m (30 ft) to 30 m (100 ft).

Despite these specific guidelines on how to create defensible space, there is little scientific evidence to support the amount and location of vegetation modification that is actually effective

at providing significant benefits. Most spacing guidelines and laws are based on 'expert opinion' or recommendations from older publications that lack scientific reference or rationale (e.g. Maire 1979; Smith and Adams 1991; Gilmer 1994). However, one study has provided scientific support for, and forms the basis of, most guidelines, policy and laws requiring a minimum of 30 m (100 ft) of defensible space (Cohen 1999, 2000). The modelling and experimental research in that study showed that flames from forest fires located 10–40 m (33–131 ft) away would not scorch or ignite a wooden home; and case studies showed 90% of homes with non-flammable roofs and vegetation clearance of 10–20 m (33–66 ft) could survive wildfires (Cohen 2000). However, the models and experimental research in that study focussed on crown fires in spruce or jack pine forests, and the primary material of home construction was wood. Therefore, it is unknown how well this guideline applies to regions dominated by other forest types, grasslands, or nonforested woody shrublands and in regions where wooden houses are not the norm.

Some older case studies showed that most homes with non-flammable roofs and 10–18 m (33–ft) of defensible space survived the 1961 Bel Air fire in California (Howard *et al.* 1973); most homes with non-flammable roofs and more than 10 m (33 ft) of defensible space also survived the 1990 Painted Cave fire (Foote and Gilless 1996). Also, several fire-behaviour modelling studies have been conducted in chaparral shrublands. One study showed that reducing vegetative cover to 50% at 9–30 m (30–ft) from structures effectively reduced fireline intensity and flame lengths, and that removal of 80% cover would result in unintended consequences such as exotic grass invasion, loss of habitat and increase in highly flammable flashy fuels (A. Fege and D. Pumphrey, unpubl. data). Another showed that separation distances adequate to protect firefighters varied according to fuel model and that wind speeds greater than 23 km h^{-1} negated the effect of slope, and wind speed above 48 km h^{-1} negated any protective effect of defensible space (F. Bilz, E. McCormick and R. Unkovich, unpubl. data, 2009). Results obtained through modelling equations of thermal radiation also found safety distances to vary as a function of fuel type, type of fire, home construction material and protective garments worn by firefighters (Zárate *et al.* 2008).

Although there is no empirical evidence to support the need for more than 30 m (100 ft) of defensible space, there has been a concerted effort in some areas to increase this distance, particularly on steep slopes. In California, a senate bill was introduced in 2008 (SB 1618) to encourage property owners to clear 91 m (300 ft) through the reduction of environmental regulations and permitting needed at that distance. Although this bill was defeated in committee, many local ordinances do require homeowners to clear 91 m (300 ft) or more, and there are reports that some people are unable to get fire insurance without 91 m (300 ft) of defensible space (F. Sproul, pers. comm.). In contrast, homeowner acceptance of and compliance with defensible space policies can be challenging (Winter *et al.* 2009; Absher and Vaske 2011), and in many cases homeowners do not create any defensible space.

It is critically important to develop empirical research that quantifies the amount, location and distance of defensible space that provides significant fire protection benefits so that guidelines and policies are developed with scientific support. Data that are directly applicable to southern California are especially important, as this region experiences the highest annual rate of wildfire-destroyed homes in the US. Not having sufficient defensible space is obviously undesirable because of the hazard to homeowners. However, there are clear trade-offs involved when vegetation reduction is excessive, as it results in the loss of native habitats, potential for increased erosion and invasive species establishment, and it potentially even increases fire risk because of the high flammability of weedy grasslands (Spittler 1995; Keeley *et al.* 2005; Syphard *et al.* 2006).

It is also important to understand the role of defensible space in residential structure protection relative to other factors that explain why some homes are destroyed in fires and some are not. Recent research shows that landscape-scale factors, such as housing arrangement and location, as well as biophysical variables characterising properties and neighbourhoods such as slope and fuel type, were important in explaining which homes burned in two southern California study areas (Syphard *et al.* 2012; 2013). Understanding the relative importance of different variables at different scales may help to identify which combinations of factors are most critical to consider for fire safety.

Our objective was to provide an empirical analysis of the role of defensible space in protecting structures during wildfires in southern California shrublands. Using recent pre-fire aerial photography, we mapped and measured a suite of variables describing defensible space for burned and unburned structures within the perimeters of major fires from 2001 to 2010 in San Diego County to ask the following questions:

1. How much defensible space is needed to provide significant protection to homes during wildfires, and is it beneficial to have more than the legally required 30 m (100 ft)?
2. Does the amount of defensible space needed for protection depend on slope inclination?
3. What is the role of defensible space relative to other factors that influence structure loss, such as terrain, fuel type and housing density?

Methods

Study area

The properties and structures analysed were located in San Diego County, California, USA (Fig. 1) – a topographically diverse region with a Mediterranean climate characterised by cool, wet winters and long summer droughts. Fire typically is a direct threat to structures adjacent to wildland areas. Native shrublands in southern California are extremely flammable during the late summer and fall (autumn) and when ignited, burn in high-intensity, stand-replacing crown fires. Although 500 homes on average have been lost annually since the mid-1900s (Calfire 2000), that rate has doubled since 2000. Most of these homes have burned during extreme fire weather conditions that accompany the autumn Santa Ana winds. The wildland–urban interface here includes more than 5 million homes, covering more than 28 000 km^2 (Hammer *et al.* 2007).

Property data

The data for properties to analyse came from a complete spatial database of existing residential structures and their

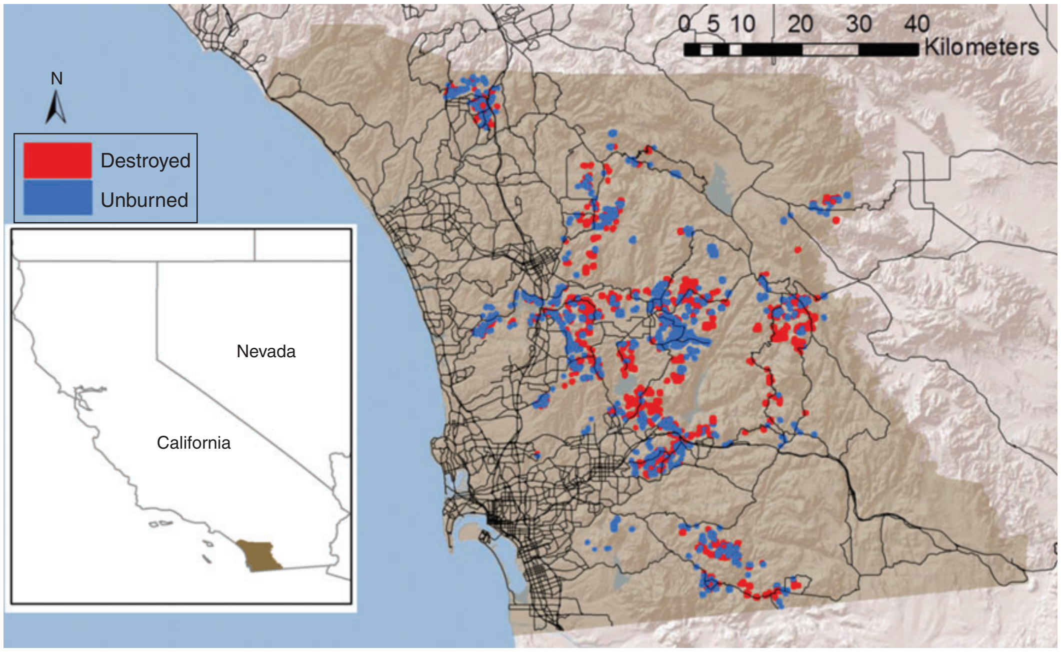

Fig. 1. Location of destroyed and unburned structures within the South Coast ecoregion of San Diego County, California, USA.

corresponding property boundaries developed for San Diego County (Syphard *et al.* 2012). This dataset included 687 869 structures, of which 4315 were completely destroyed by one of 40 major fires that occurred from 2001 to 2010. Our goal was to compare homes that were exposed to wildfire and survived with those that were exposed and destroyed. To determine exposure to fire, we only considered structures located both within a GIS layer of fire perimeters and within areas mapped as having burned at a minimum of low severity through thematic Monitoring Trends in Burn Severity produced by the USA Geological Survey and USDA Forest Service. From these data, we used a random sample algorithm in GIS software to select 1000 destroyed and 1000 unburned homes that were not adjacent to each other, to minimise any potential for spatial autocorrelation. Our final property dataset included structures that burned across eight different fires. More than 97% of these structures burned in Santa Ana wind-driven fire events (Fig. 1).

Calculating defensible space and additional explanatory variables

To estimate defensible space, we developed and explored a suite of variables relative to the distance and amount of defensible space surrounding structures, as well as the proximity of woody vegetation to the structure (Table 1). We measured these variables based on interpretation of Google Earth aerial imagery. We based our measurements on the most recent imagery before the date of the fire. In almost all cases, imagery was available for less than 1 year before the fire.

Our definition of defensible space followed the guidelines published by the California Department of Forestry and Fire Protection (Calfire 2006). 'Clearance' included all areas that were not covered by woody vegetation, including paved areas or grass. Although Google Earth prevents the identification of understorey vegetation, woody trees and shrubs were easily distinguished from grass, and our objective was to measure horizontal distances as required by Calfire rather than assess the relative flammability of different vegetation types. Trees or shrubs were allowed to be within the defensible space zone as long as they were separated by the minimum horizontal required distance, which was 3 m (10 ft) from the edge of one tree canopy to the edge of the next (Fig. 2). Although greater distances between trees or shrubs are recommended on steeper slopes, we followed the same guidelines for all properties. For all structures, we started the distance measurements by drawing lines from the centre of the four orthogonal sides of the structure that ended when they intersected anything that no longer met the requirements in the guidelines. A fair number of structures are not four sided; thus, the start of the centre point was placed at a location that approximated the farthest extent of the structure along each of four orthogonal sides.

We developed two sets of measurements of the distance of defensible space based on what is feasible for homeowners within their properties *v.* the total effective distance of defensible space. We made these two measurements because homeowners are only required to create defensible space within their own property, and this would reflect the effect of individual homeowner compliance. Therefore, even if cleared vegetation extended beyond the property line, the first set of distance measurements ended at the property boundary. The second set of measurements ignored the property boundaries and accounted for the total potential effect of treatment. For all measurements, we recorded the cover types (e.g. structure >3 m (10 ft) long, property boundary, or vegetation type) at which the distance measurements stopped (Table 1). Because property

Table 1. Defensible space variables measured for every structure

Urban veg, landscaping vegetation that was not in compliance with regulations within urban matrix; wildland veg, wildland vegetation that was not in compliance with regulations; orchard, shrub to tree-sized vegetation in rows; urban to wildland, landscaping vegetation that leads into wildland vegetation; structure, any building longer than 3 m (10 ft)

Variable	Definition
Distance defensible space within property	Measure of clearance from side of structure to property boundary calculated for four orthogonal directions from structure and averaged
Total distance defensible space	Measure of clearance from side of structure to end of clearance calculated for four orthogonal directions from structure and averaged
Cover type at end of defensible space	Type of cover encountered at end of measurement (urban veg, wildland veg, orchard, urban to wildland, structure)
Percentage clearance	Percentage of clearance calculated across the entire property
Neighbours' vegetation	Binary indicator of whether neighbours' uncleared vegetation was located within 30 m (100 ft) of the main structure
Vegetation touching structure	Number of sides on which woody vegetation touches main structure (1–4) Structure with more than 4 sides were viewed as a box and given a number between 1 and 4
Vegetation overhanging roof	Was vegetation overhanging the roof? (yes or no)

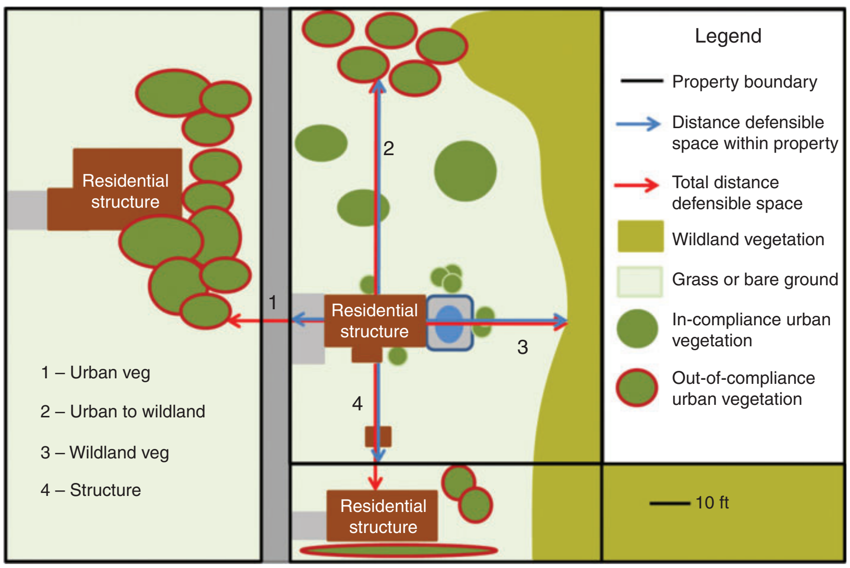

Fig. 2. Illustration of defensible space measurements. See Table 1 for full definition of terms.

owners usually can only clear vegetation on their own land, it is possible that the effectiveness of defensible space partly depends upon the actions of neighbouring homeowners. Therefore, we also recorded whether or not any neighbours' un-cleared vegetation was located within 30 m (100 ft) of the structure.

To assess the total amount of woody vegetation that can safely remain on a property and still receive significant benefits of defensible space, we calculated the total percentage of cleared land, woody vegetation and structure area across every property. This was accomplished by overlaying a grid on each property and determining the proportion of squares falling into each class. Preliminary results showed these three measurements to be highly correlated, so we only retained percentage clearance for further analysis. To evaluate the relative effect of woody vegetation directly adjacent to structures, we also calculated the number of sides of the structure with vegetation touching and recorded whether any trees were overhanging structures' roofs.

In addition to defensible space measurements, we evaluated other factors known to influence the likelihood of housing loss to fire in the region (Syphard *et al.* 2012, 2013). Using the same data as in Syphard *et al.* (2012, 2013), we extracted spatial information from continuous grids of explanatory variables for the locations of all structures in our analysis. Variables included interpolated housing density based on a 1-km search radius; percentage slope derived from a 30-m digital elevation model (DEM); Euclidean distance to nearest major and minor road and fuel type, which was based on a simple classification of US Forest Service data (Syphard *et al.* 2012), including urban, grass, shrubland and forest & woodland.

Analysis

We performed several analyses to determine whether relative differences in home protection are provided by different distances and amounts of defensible space, particularly beyond the legally required 30 m (100 ft), and to identify the effective treatment distance for homes on low and steep slopes.

Categorical analysis

For the first analysis, we divided our data into several groups to identify potential differences among specific categories of defensible space distance around structures located on shallow and steep slopes. We first sorted the full dataset of 2000 structures by slope and then split the data in the middle to create groups of homes with shallow slope and steep slope. We divided the data in half to keep the number of structures even within both groups and to avoid specifying an arbitrary number to define what constitutes shallow or steep slope. The two equal-sized subsets of data ranged from 0 to 9%, with a mean of 8% for shallow slope, and from 9 to 40%, with a mean of 27% for steep slope. Within these data subsets, we next created groups reflecting different mean distances of defensible space around structures. We also performed separate analyses based on whether defensible space measurements were calculated within the property boundary or whether measurements accounted for the total distance of defensible space.

Within all groups, we calculated the proportion of homes that were destroyed by wildfire. We performed Pearson's Chi-square tests of independence to determine whether or not the proportion of destroyed structures within groups was significantly different (Agresti 2007). We based one test on four equal-interval groups within the legally required distance of 30 m (100 ft): 0–7 m (0–25 ft), 8–15 m (26–50 ft), 16–23 m (51–75 ft) and 24–30 m (76–100 ft). A second test was based on three groups (24–30 m (75–100 ft), 31–90 m (101–300 ft) and >90 m (>300 ft) or >60 m (>200 ft)) to evaluate whether groups with mean defensible space distances >30 m (>100 ft) were significantly different from groups with <30 m (<100 ft). When defensible space distances were only measured to the property boundary, few structures had mean defensible space >90 m (>300 ft). Therefore, we used a cut-off of 60 m (200 ft) to increase the sample size in the Chi-square analysis. In addition to the Chi-square analysis, we calculated the relative risk among every successive pair of categories (Sheskin 2004). The relative risk was calculated as the ratio of proportions of burned homes within two groups of homes that had different defensible space distances.

Effective treatment analysis

In addition to comparing the relative effect of defensible space among different groups of mean distances, as described above, we also considered that the protective effect of defensible space for structures exposed to wildfire is conceptually similar to the effect of medication in producing a therapeutic response in people who are sick. In addition to pharmacological applications, treatment–response relationships have been used for radiation, herbicide, drought tolerance and ecotoxicological studies (e.g. Streibig *et al.* 1993; Cedergreen *et al.* 2005; Knezevic *et al.* 2007; Kursar *et al.* 2009). The effect produced by a drug or treatment typically varies according to the concentration or amount, often up to a point at which further increase provides no additional response. The effective treatment (ET50), therefore, is a specific concentration or exposure that produces a therapeutic response or desired effect. Here we considered the treatment to be the distance or amount of defensible space.

Using the software package DRC in R (Knezevic *et al.* 2007; Ritz and Streibig 2013), we evaluated the treatment–response relationship of defensible space in survival of structures during wildfire. To calculate the effective treatment, we fit a log-logistic model with logistic regression because we had a binary dependent variable (burned or unburned). We specified a 2-parameter model where the lower limit was fixed at 0 and the upper limit was fixed at 1. We again performed separate analyses for data subsets reflecting shallow and steep slope, as well as from measurements of defensible space taken within, or regardless of, property boundaries. We also performed analyses to find the effective treatment of percentage clearance of trees and shrubs within the property.

Multiple regression analysis

To evaluate the role of defensible space relative to other variables, we developed multiple generalised linear regression models (GLMs) (Venables and Ripley 1994). We again had a binary dependent variable (burned versus unburned), so we specified a logit link and binomial response. Although the proportion of 0s and 1s in the response may be important to consider for true prediction (King and Zeng 2001; Syphard *et al.* 2008), our objective here was solely to evaluate variable importance. We developed multiple regression models for all possible combinations of the predictor variables and used the corrected Akaike's Information Criterion (AICc) to rank models and select the best ones for each region using package MuMIn in R (R Development Core Team 2012; Burnham and Anderson 2002). We recorded all top-ranked models that had an AICc value within 2 of that of the model with lowest AICc to identify all models with empirical support. To assess variable importance, we calculated the sum of Akaike weights for all models that contained each variable. On a scale of 0–1, this metric represents the weight of evidence that models containing the variable in question are the best model (Burnham and Anderson 2002). The distance of defensible space measured within property boundaries was highly correlated with the distance of defensible space measured beyond property boundaries ($r = 0.82$), so we developed two separate analyses – one using variables measured only within the property boundary and the other using variables that accounted for defensible space outside of the property boundary as well as the potential effect of neighbours having uncleared vegetation within 30 m (100 ft) of the structure. A test to avoid multicollinearity showed all other variables within each multiple regression analysis to be uncorrelated ($r < 0.5$).

Surrounding matrix

To assess whether the proportion of destroyed structures varied according to their surrounding matrix, we summarised the most common cover type at the end of defensible space measurements (descriptions in Table 1) for all structures. These summaries

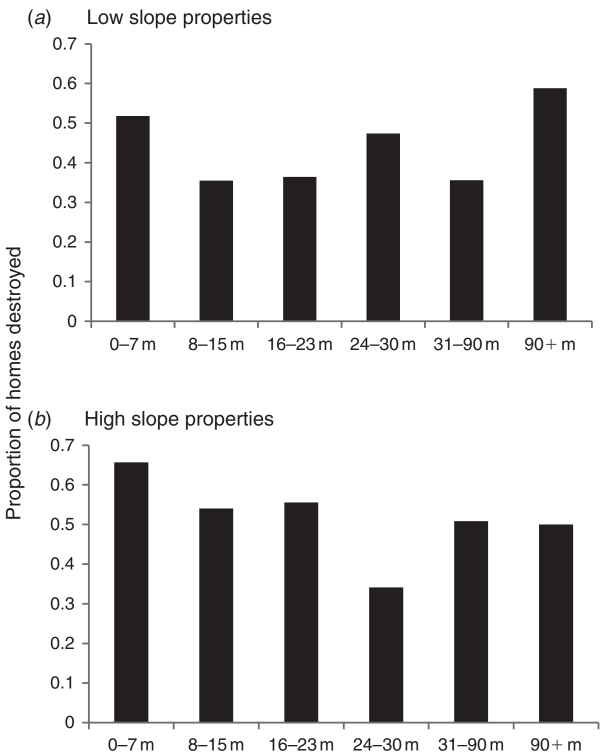

Fig. 3. Proportion of destroyed homes grouped by distances of defensible space based upon total distance of clearance within property boundary, for structures on (*a*) shallow slopes (mean 8%) and (*b*) steep slopes (mean 27%).

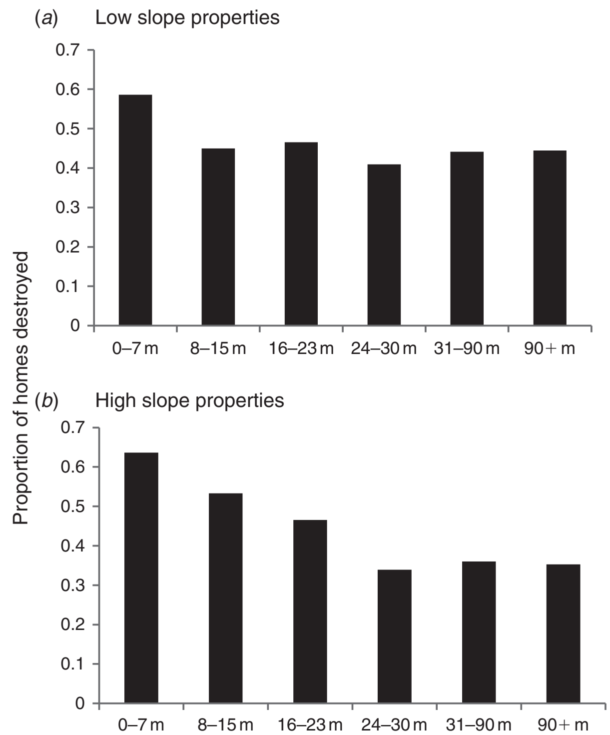

Fig. 4. Proportion of destroyed homes grouped by distances of defensible space based upon total distance of clearance regardless of property boundary, for structures on (*a*) shallow slopes (mean 8%) and (*b*) steep slopes (mean 27%).

were based on the majority surrounding cover type from the four orthogonal sides of the structure. We also noted cases in which there was a tie (e.g. two sides were urban vegetation and two sides were structures).

Results

Categorical analysis

When the distance of defensible space was measured both 'only within property boundaries' (Fig. 3) and 'regardless of property boundaries' (Fig. 4), the Chi-square test showed a significant difference ($P < 0.001$) in the proportion of destroyed structures among the four equal-interval groups of distance ranging from 0 to 30 m (0–100 ft). This relationship was consistent on both shallow-slope and steep-slope properties, although the relative risk analysis showed considerable variation among classes (Table 2) There was a steadily decreasing proportion of destroyed structures at greater distances of defensible space up to 30 m (100 ft) on the steep-slope structures with defensible space measured regardless of property boundaries (Fig. 4*b*). Otherwise, the biggest difference in proportion of destroyed structures occurred between 0 and 7 m (0–25 ft) and 8–15 m (26–50 ft) (Figs 3*a–b*, 4*a*).

When the distance of defensible space was measured in intervals from 24 m (75 ft) and beyond, the Chi-square test showed no significant difference among groups ($P = 0.96$ for shallow-slope properties and $P = 0.74$ for steep-slope properties) (Figs 3, 4), although again, the relative risk analysis showed considerable variation (Table 2).There was a slight increase in the proportion of homes destroyed at longer distance intervals when the defensible space was measured only to the property boundaries (Fig. 3*a–b*). This slight increase is less apparent when distances were measured regardless of boundaries (Fig. 4*a–b*).

The relative risk calculations showed that the ratio of proportions was generally more variable among successive pairs when the distances were measured within property boundaries (Table 2). For these calculations, the risk of a structure being destroyed was significantly lower when the defensible space distance was 8–15 m (25–50 ft) compared to 0–7 m (0–25 ft) on both shallow- and steep-slope properties. On the steep-slope properties, there was an additional reduction of risk when comparing 24–30 m (75–100 ft) to 16–23 m (50–75 ft). However, the risk of a home being destroyed was slightly significantly higher when there was 31–90 m (101–225 ft) compared to 16–23 m (50–75 ft). For distances that were measured regardless of property boundary (total clearance), the only significant differences in risk of burning were a reduction in risk for 8–15 m (25–50 ft) compared to 0–7 m (0–25 ft).

Table 2. Number of burned and unburned structures within defensible space distance categories (m), their relative risk and significance

A relative risk of 1 indicates no difference; <1 means the chance of a structure burning is less than the other group; >1 means the chance is higher than the other group. The relative risk is calculated for pairs that include the existing row and the row above. Confidence intervals are in parentheses

	Distance within property				Total distance			
	Burned	Unburned	Relative risk	*P*	Burned	Unburned	Relative risk	*P*
Shallow slope								
0–7	200	186			162	114		
8–15	109	198	0.69 (0.12)	<0.001	108	132	0.77	0.002
16–23	51	89	1.03 (0.30)	0.850	78	90	1.03	0.770
24–30	36	40	1.30 (0.39)	0.110	50	70	0.90	0.430
31–90	28	47	0.79 (0.24)	0.220	79	99	1.06	0.640
60 or 90+	10	6	1.67 (0.63)	0.040	8	9	1.01	0.830
Steep slope								
0–7	245	128			224	128		
8–15	174	148	0.82 (0.10)	0.001	158	139	0.84	0.008
16–23	85	68	1.03 (0.16)	0.750	73	83	0.87	0.210
24–30	29	56	0.61 (0.17)	0.004	26	50	0.73	0.080
31–	29	28	1.49 (0.48)	0.050	39	68	1.06	0.760
60 or 90+	5	5	0.98 (0.47)	0.950	4	8	0.91	0.830

Table 3. Effective treatment results reflecting the distance (in metres, with feet in parentheses) and percentage clearance within properties that provided significant improvement in structure survival during wildfires

The property mean is the average distance of defensible space or percentage clearance that was calculated on the properties before the wildfires and provides a means to compare the effective treatment result to the actual amount on the properties

	All parcels effective treatment ($n = 2000$)	Parcel mean	Shallow slope (mean 8%) effective treatment ($n = 1000$)	Parcel mean	Steep slope (mean 27%) effective treatment ($n = 1000$)	Parcel mean
Defensible space within parcel	10 (33)	13 (44)	4 (13)	14 (45)	25 (82)	11 (35)
Total distance defensible space	10 (32)	19 (63)	5 (16)	20 (67)	20 (65)	18 (58)
Mean percentage clearance on property	36	48	31	51	37	35

Effective treatment analysis

Analysis of the treatment–response relationships among defensible space and structures that survived wildfire showed that, when all structures are considered together, the mean actual defensible space that existed around structures before the fires was longer than the calculated effective treatment (Table 3). Regardless of whether the defensible space was measured within or beyond property boundaries, the estimated effective treatment of defensible space was nearly the same at 10 m (32–33 ft).

The effective treatment distance was much shorter for structures on shallow slopes (4–5 m (13–16 ft)) than for structures on steep slopes (20–25 m (65–82 ft)), but in all cases was <30 m (<100 ft). Although longer distances of defensible space were calculated as effective on steeper slopes, these structures actually had shorter mean distances of defensible space around their properties than structures on low slopes (Table 3).

The calculated effective treatment of the mean percentage clearance on properties was 36% for all properties, 31% for structures on shallow slopes and 37% for structures on steep slopes (Table 3). In total, the properties all had higher actual percentage clearance on their property than was calculated to be effective. However, this mainly reflects the shallow-slope properties, as those structures on steep slopes had less clearance than the effective treatment.

Multiple regression analysis

When defensible space was measured only to the property boundaries, it was not included in the best model, according to the all-subsets multiple regression analysis (Table 4). However, it was included in the best model when factoring in the distance of defensible space measured beyond property boundaries (Table 5). In both multiple regression analyses, low housing density and shorter distances to major roads were ranked as the most important variables according to their Akaike weights. Slope and surrounding fuel type were also in both of the best models as well as other measures of defensible space, including the percentage clearance on property and whether vegetation was overhanging the structure's roof. The number of sides in which vegetation was touching the structure was included in the best model when defensible space was only measured to the property boundary. The total explained deviance for the multiple regression models was low (12–13%) for both analyses.

Table 4. Results of multiple regression models of destroyed homes using all possible variable combinations and corrected Akaike's Information Criterion (AICc)

Includes variables measured within property boundary only. Top-ranked models include all those ($n = 12$) with AICc within 2 of the model with the lowest AICc. Relative variable importance is the sum of 'Akaike weights' over all models including the explanatory variable

Variable in order of importance	Relative variable importance	Model-averaged coefficient	Number inclusions in top-ranked models
Housing density	1	−0.003	12
Distance to major road	1	−0.0005	12
Percentage clearance	1	−0.02	12
Slope	1	0.03	12
Vegetation overhang roof	1	0.5	12
Fuel type	0.67	Factor	9
Vegetation touch structure	0.49	0.07	6
Distance defensible space within property	0.45	−0.0002	5
South-westness	0.36	−0.0007	3
Distance to minor road	0.28	−0.0002	1
D^2 of top-ranked model			0.123

Table 5. Results of multiple regression models of destroyed homes using all possible variable combinations and corrected Akaike's Information Criterion (AICc)

Includes variables measured beyond property boundary. Top-ranked models include all those ($n = 6$) with AICc within 2 of the model with the lowest AICc. Relative variable importance is the sum of 'Akaike weights' over all models including the explanatory variable

Variable in order of importance	Relative variable importance	Model-averaged coefficient	Number inclusions in top-ranked models
Housing density	1	−0.003	6
Distance to major road	1	−0.0005	6
Total distance defensible space	1	−0.004	6
Percentage clearance	1	−0.01	6
Vegetation overhang roof	0.99	0.4	6
Slope	0.99	0.03	6
Fuel type	0.86	Factor	4
South-westness	0.42	−0.0009	2
Distance to minor road	0.36	−0.0009	2
Neighbours' vegetation	0.27	0.08	1
Vegetation touch structure	0.27	0.18	1
D^2 of top-ranked model			0.125

Surrounding matrix

The cover type that most frequently surrounded the structures at the end of the defensible space measurements was urban vegetation, followed by urban vegetation leading into wildland vegetation, and wildland vegetation (Fig. 5). Many structures were equally surrounded by different cover types. There were no significant differences in the proportion of structures destroyed depending on the surrounding cover type. However, a disproportionately large proportion of structures burned (28 *v.* 9% unburned) when they were surrounded by urban vegetation that extended straight into wildland vegetation.

Discussion

For homes that burned in southern Californian urban areas adjacent to non-forested ecosystems, most burned in high-intensity Santa Ana wind-driven wildfires and defensible space increased the likelihood of structure survival during wildfire. The most effective treatment distance varied between 5 and 20 m (16–58 ft), depending on slope and how the defensible space was measured, but distances longer than 30 m (100 ft) provided no significant additional benefit. Structures on steeper slopes benefited from more defensible space than structures on shallow slopes, but the effective treatment was still less than 30 m (100 ft). The steepest overall decline in destroyed structures occurred when mean defensible space increased from 0–7 m (0–25 ft) to 8–15 m (26–50 ft). That, along with the multiple regression results showing the significance of vegetation touching or overhanging the structure, suggests it is most critical to modify vegetation immediately adjacent to the house, and to move outward from there. Similarly, vegetation overhanging the structure was also strongly correlated with structure loss in Australia (Leonard *et al.* 2009).

In terms of fuel modification, the multiple regression models also showed that the percentage of clearance was just as, or more important than, the linear distance of defensible space.

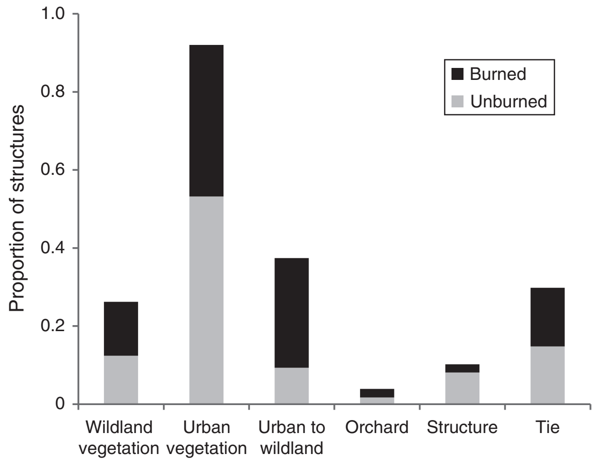

Fig. 5. Proportion of destroyed and unburned structures based on the primary surrounding cover type at the end of defensible space measurements. There were no significant differences in the proportion of burned and unburned structures within cover types ($P = 0.14$). Cover types are defined in Table 1.

However, as with defensible space, percentage clearance did not need to be draconian to be effective. Even on steep slopes, the effective percentage clearance needed on the property was <40%, with no significant advantage beyond that. Although these steep-slope structures benefited more from clearance, they tended to have less clearance than the effective amount, which may be why slope was such an important variable in the multiple regression models. Shallow-slope structures, in contrast, had more clearance on average than was calculated to be effective, suggesting these property owners do not need to modify their behaviours as much relative to people living on steep slopes.

Although the term 'clearance' is often used interchangeably with defensible space, this term is incorrect when misinterpreted to mean clearing all vegetation, and our results underline this difference. The idea behind defensible space is to reduce the continuity of fuels through maintenance of certain distances among trees and shrubs. Although we could not identify the vertical profile of fuels through Google Earth imagery, the fact that at least 60% of the horizontal woody vegetative cover can remain on the property with significant protective effects demonstrates the importance of distinguishing defensible space from complete vegetation removal. Thus, we suggest the term 'clearance' be replaced with 'fuel treatment' as a better way of communicating fire hazard reduction needs to home owners.

The percentage cover of woody shrubs and trees was not evenly distributed across properties, and we did not collect data describing how the cover was distributed. Considering the importance of defensible space and vegetation modification immediately adjacent to the structure, it should follow that actions to reduce cover should also be focussed in close proximity to the structure. The hazard of vegetation near the structure has apparently been recognised for some time (Foote *et al.* 1991; Ramsey and McArthur 1994), but it is not stressed enough, and rarely falls within the scope of defensible space guidelines or ordinances.

In addition to the importance of vegetation overhanging or touching the structure, it is important to understand that ornamental vegetation may be just as, if not more, dangerous than native vegetation in southern California. Although the results showed no significant differences in the cover types in the surrounding matrix, there was a disproportionately large number of structures destroyed (28% burned *v.* 9% unburned) when ornamental vegetation on the property led directly into the wildland. Ornamental vegetation may produce highly flammable litter (Ganteaume *et al.* 2013) or may be particularly dangerous after a drought when it is dry, or has not been maintained, and species of conifer, juniper, cypress, eucalypt, *Acacia* and palm have been present in the properties of many structures that have been destroyed (Franklin 1996). Nevertheless, ornamental vegetation is allowed to be included as defensible space in many codes and ordinances (Haines *et al.* 2008).

One reason that longer defensible space distances did not significantly increase structure protection may be that most homes are not destroyed by the direct ignition of the fire front but rather due to ember-ignited spot fires, sometimes from fire brands carried as far as several km away. Although embers decay with distance, the difference between 30 and 90 m (100 and 300 ft) may be small relative to the distance embers travel under the severe wind conditions that were present at the time of the fires. The ignitability of whatever the embers land on, particularly adjacent to the house, is therefore most critical for propagating the fire within the property or igniting the home (Cohen 1999; Maranghides and Mell 2009).

Aside from roofing or home construction materials and vegetation immediately adjacent to structures (Quarles *et al.* 2010; Keeley *et al.* 2013), the flammability of the vegetation in the property may also play a role. Large, cleared swaths of land are likely occupied at least in part by exotic annual grasses that are highly ignitable for much of the year. Conversion of woody shrubs with higher moisture content into low-fuel-volume grasslands could potentially increase fire risk in some situations by increasing the ignitability of the fuel; and if the vegetation between a structure and a fire is not readily combustible, it could protect the structure by absorbing heat flux and filtering fire brands (Wilson and Ferguson 1986).

The slight increase in proportion of structures destroyed with longer distances of defensible space within parcel boundaries was surprising. However, that increase was not significant in the Chi-square analysis, although there were some significant differences in the pairwise relative risk analysis. Nevertheless, the largest significant effect of defensible space was between the categories of 0–7 m (0–25 ft) to 8–15 m (26–50 ft), and it may be that differences in categories beyond these distances are not highly meaningful or reflect an artefact of the definition of distance categories. These relationships at longer distances are likely also weak compared to the effect of other variables operating at a landscape scale. Although the categorical analysis allowed us to answer questions relative to legal requirements and specific distances, the effective treatment analysis was important for identifying thresholds in the continuous variable.

The multiple regression models showed that landscape factors such as low housing density and longer distances to major roads were more important than distance of defensible space for explaining structure destruction, and the importance of

these variables is consistent with previous studies (Syphard *et al.* 2012, 2013), despite the smaller spatial extent studied here. Whereas this study used an unburned control group exposed to the same fires as the destroyed structures, previous studies accounted for structures across entire landscapes. The likelihood of a fire destroying a home is actually a result of two major components: the first is the likelihood that there will be a fire, and the second is the likelihood that a structure will burn in that fire. In this study, we only focussed on structure loss given the presence of a fire, and the total explained variation for the multiple regression models was quite low at ~12%. However, when the entire landscape was accounted for in the total likelihood of structure destruction, the explained variation of housing density alone was >30% (Syphard *et al.* 2012). One reason for the relationship between low housing density and structure destruction is that structures are embedded within a matrix of wildland fuel that leads to greater overall exposure, which is consistent with Australian research that showed a linear decrease of structure loss with increased distance to forest (Chen and McAneney 2004). That research, however, only focussed on distance to wildland boundaries and did not quantify variability in defensible space or ornamental vegetation immediately surrounding structures. Thus, fire safety is important to consider at multiple scales and for multiple variables, which will ultimately require the cooperation of multiple stakeholders.

Conclusions

Structure loss to wildfire is clearly a complicated function of many biophysical, human and spatial factors (Keeley *et al.* 2009; Syphard *et al.* 2012). For such a large sample size, we were unable to account for home construction materials, but this is also well understood to be a major factor, with older homes and wooden roofs being most vulnerable (Franklin 1996; Cohen 1999, 2000). In terms of actionable measures to reduce fire risk, this study shows a clear role for defensible space up to 30 m (100 ft). Although the effective distances were on average much shorter than 30 m (100 ft), we recognise that additional distance may be necessary to provide sufficient protection to firefighters, which we did not address in this study (Cheney *et al.* 2001). In contrast, the data in this study do not support defensible space beyond 30 m (100 ft), even for structures on steep slopes. In addition to the fact that longer distances did not contribute significant additional benefit, excessive vegetation clearance presents a clear detriment to natural habitat and ecological resources. Results here suggest the best actions a homeowner can take are to reduce percentage cover up to 40% immediately adjacent to the structure and to ensure that vegetation does not overhang or touch the structure.

In addition to defensible space, this study also underlines the potential importance of land use planning to develop communities that are fire safe in the long term, in particular through their reduction to exposure to wildfire in the first place. Localised subdivision decisions emphasising infill-type development patterns may significantly reduce fire risk in the future, in addition to minimising habitat loss and fragmentation (Syphard *et al.* 2013). This study was conducted in southern California, which has some of the worst fire weather in the world and many properties surrounded by large, flammable exotic trees. Therefore, recommendations here should apply to other non-forested ecosystems as well as many forested regions.

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Dead forests burning: the influence of beetle outbreaks on fire severity and legacy structure in sub-boreal forests

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Abstract. Recent regional mountain pine beetle (MPB) outbreaks have generated unprecedented tree mortality across the fire-prone landscapes of western North American forests and could potentially modify fire severity and postfire ecological effects. In 2012, 2013, and 2014, three fires burned through high mortality, gray-phase lodgepole pine-dominated forests in the plateau regions of central interior British Columbia, Canada, providing an opportunity to test for interactions between MPB outbreaks and wildfires. We inventoried 63 plots that spanned gradients of outbreak severity, fire severity, and burning conditions in a wilderness setting. Our objective was to evaluate the influence of outbreak severity on fire severity by assessing typical first-order fire effects as well as legacy structure related to the consumption of woody biomass on snags/trees. We found no evidence of a relationship between outbreak severity and fire severity for six of seven first-order fire effects, with the exception of deep charring. We found evidence that legacy structure in the form of consumed branch structure and deep char development had greater odds of occurrence on MPB-killed snags compared to trees killed during wildfire. Our results indicate two key findings. First, fire severity as it relates to most first-order fire effects measures is not influenced by outbreak severity, instead it is more strongly influenced by the interaction of fuels, weather, and topography during fire events. Second, our results highlight how the interaction between outbreak severity and fire severity alters postfire structural legacies and their functional attributes, which could have important ecosystem implications.

Key words: deep char; *Dendroctonous ponderosae*; legacy structure; mountain pine beetle; *Pinus contorta* var. *latifolia*; short-interval disturbances; snags; sub-boreal; wildfire.

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Introduction

Forest ecosystems across western North America are increasingly experiencing ecological disturbances from wildfires burning through landscapes with abundant tree mortality from insect outbreaks. The recent mountain pine beetle (*Dendroctonous ponderosae*, hereafter MPB) outbreaks circa the 1990s and 2000s are responsible for tree mortality in forests that span over 25 million hectares across the western United States and Canada (Raffa et al. 2008, Bentz et al. 2010, Meddens et al. 2012), and British Columbia (BC) houses nearly 20 million of those hectares (Axelson et al. 2009, Perrakis et al. 2014). The spatial extent and high mortality rates associated with recent outbreaks alter standing woody fuels in affected forests from mostly alive to mostly dead, which changes the composition of the fuel profile and raises concerns for increased fire severity (Hicke et al. 2012, Jenkins et al. 2012). The overlap between MPB outbreak and wildfire disturbances that recur within a short time interval may lead to linked effects in which the first event alters the extent, severity, or probability of occurrence for the second event (Kulakowski and Veblen 2007, Simard et al. 2011). Previous field-based studies have investigated interactions of short-interval MPB-fire disturbances with variable levels of lodgepole pine (*Pinus contorta* var. *latifolia*) mortality in montane regions of the western United States (Harvey et al. 2014*a*, *b*, Agne et al. 2016) and found fire severity to be either weakly linked or unrelated to outbreak severity. However, the magnitude of the MPB outbreaks in BC far exceeds the conditions seen in the western United States (Raffa et al. 2008) and the biophysical environment differs from earlier studies (Harvey et al. 2014*b*, Agne et al. 2016), such that further investigation is required to understand the implications of fire burning through BC's MPB-affected forests.

The changes in fuel profiles from MPB outbreaks and subsequent stand breakdown have raised concerns among land managers for altered fire behavior and potential changes in subsequent fire severity that could generate burning conditions that are more hazardous and more severe than from fire burning through stands of live trees. Severe tree mortality alters the configuration, continuity, and moisture content of fuels over time as stands break down (i.e., needle loss, branch breakage, sloughing bark, snag fall), all of which may influence fire behavior (Page and Jenkins 2007, Hicke et al. 2012). Dry, dead fuels ignite more quickly (Stockstad 1979) and at lower temperatures (Stockstad 1975) compared to live fuels. When these dead fuels are coarse, they are prone to smoldering (Brown et al. 2003), which often extends burning time beyond the initial flaming front (Alexander 1982), thus allowing for dry dead fuels to burn longer and have more biomass consumed (Brown et al. 2003, Hyde et al. 2011) that could alter the structural legacies that persist postfire. The only known empirical study examining the effects of altered fuel profiles on fire behavior found that spread rates increased through red-phase outbreak conditions in lodgepole pine forests in BC (Perrakis et al. 2014). Simulation models posit crown fire to increase during the red phase of the outbreaks, 1–3 years postattack, and then decline as needles are dropped from the canopy and snags transition to the gray-phase of the outbreaks, 3–10 years postattack (Hicke et al. 2012). Alternative models suggest a shift from active crown fire during the red phase to passive crown fire in the gray phase (Klutsch et al. 2011, Simard et al. 2011, Schoennagel et al. 2012) that could result in more biomass consumption and simplification of the legacy structure of snags.

Retrospective data that evaluate fire effects to characterize fire severity provide a complement to measures of fire behavior for understanding interactions between MPB outbreak and wildfire. Fire severity is often characterized by measurements of first-order fire effects (Reinhardt et al. 2001, Ryan and Elliot 2005) and refers to the amount of immediate ecological change associated with vegetation mortality and biomass loss from fire (Keeley 2009). Retrospective studies on MPB-fire interactions are two pronged either using remotely sensed data (e.g., satellite imagery) to quantify the amount of change between prefire and postfire conditions at coarser resolutions, or field studies that measure fire effects on the ground to characterize fire severity at finer scale resolutions. Existing remote sensing studies have shown that outbreak severity does not increase fire likelihood (Meigs et al. 2015), fire severity (Meigs et al. 2016), or area burned (Hart et al. 2015) for forests in the western United States.

Field studies can capture subtleties that may be absent in remote sensing studies and have found that the relationship between outbreak severity and fire severity varies across the western United States. These studies have focused on subalpine lodgepole pine forests (Harvey et al. 2014*a*, Agne et al. 2016) and forests dominated by subalpine fir (*Abies lasiocarpa*) but with substantial basal area of lodgepole pine (Harvey et al. 2014*b*) across topographically complex landscapes. Generally, these studies have suggested that gray-phase outbreak severity results in decreased fire severity (Harvey et al. 2014*a*, Agne et al. 2016), or limited to no change in fire severity (Harvey et al. 2014*b*, Agne et al. 2016)—with the exception of deep char, a metric of fire severity that showed a consistently positive relationship with severity of MPB outbreaks (Harvey et al. 2014*b*). Some measures of fire severity increased under extreme fire weather and were attributed to burning conditions, including deep char (Harvey et al. 2014*b*), suggesting that prefire beetle outbreak and burning conditions contribute to deep charring on wood. Deep char is generated through incomplete combustion of deadwood often from long, smoldering burns (Bird et al. 2015) that result in more biomass consumption and is visually distinct compared to scorch that is generated from flaming combustion and typically occurs on trees that are alive at the time of fire (Campbell et al. 2007). Deep char is distinguished by its iridescent black with patterning like the scales of alligator skin in contrast to the matte black, dusty appearance of scorch. Deep charring on trees changes the structure and function of the postfire landscape (Campbell et al. 2007, Donato et al. 2016) by altering structural legacies and has been clearly recognized as an important severity metric when examined in areas of high-severity reburns (fire + fire; Donato et al. 2016). However, the deep char effect, and the altered structural legacy it contributes to the postfire landscape, has largely been ignored in the context of insect outbreak and wildfire interactions.

Here, we examine the effect of gray-phase outbreak severity on fire severity for lodgepole pine-dominated forests with high prefire mortality rates, in central interior BC. Our objective was to evaluate the influence of outbreak severity on fire severity by assessing first-order fire effects after three recent wildfires that burned in 2012, 2013, and 2014. We wanted to (1) ascertain whether the extensive MPB-induced tree mortality that spans the sub-boreal forests of BC responds similarly in terms of first-order fire effects to forests that have burned and been studied in the western conterminous United States and (2) expand understanding and recognition of how postfire legacies (e.g., snags) can be affected by MPB outbreaks. We anticipated first-order fire effects (e.g., scorch/char height and area on trees, surface char, exposed mineral soil) would be unaffected by the severity of the outbreaks and primarily driven by fire weather, based on previous findings (Harvey et al. 2014*a*, *b*, Agne et al. 2016). In the context of structural legacies, we anticipated that snags killed by the MPB outbreak a decade prior to fire would burn longer, through smoldering combustion that would consume more wood biomass and lead to consistent development of deep char. We also predicted the interaction between outbreak severity and fire would reduce the structural complexity on snags, due to the potential extended duration of smoldering combustion in addition to prefire stand breakdown where MPB-killed trees experience needle loss, branch breakage, and shedding of bark. In contrast, the legacies of trees that were alive at time of fire and then killed by the wildfire (i.e., fire-killed) would have less deep char and retain much more structural complexity.

Methods

Study area

We conducted our field sampling across three fires that burned in 2012, 2013, and 2014 in Tweedsmuir and Entiako Provincial Parks, which are situated in the sub-boreal forests on the southern portion of the Nechako Plateau in BC (Fig. 1). The study area has a mean monthly maximum temperature of 8.5°C (range −3.3 to 19.8°C), a mean monthly nighttime temperature of −2.8°C (range −11.9 to 6.7°C), and total annual precipitation of 507.6 mm with a monthly mean of 42.3 mm (range 22.7–60.8 mm), based on the monthly means from the 1981–2010 climate normals (Abatzoglou et al. 2018). Precipitation accumulates as snow in the winter and rain during the remainder of the year. Although it rains through the summer (Abatzoglou et al. 2018), there are weeks with no rain that are associated with persistent high-pressure ridges (Nash and Johnson 1996). Within the fire perimeters, landscapes are associated with the Sub-Boreal Pine Spruce and Sub-Boreal Spruce biogeoclimatic zones (Meidinger and Pojar 1991), and lodgepole pine is the dominant canopy species (Fig. 1; BCMFLNRO 2012). Moisture gradients dictate composition, structure, and disturbance history, based on historical reconstructions from surrounding areas (Steventon 2001, Francis et al. 2002) and stand age distributions (Delong 1998). Within our fire perimeters, climax lodgepole pine inhabits the driest end of the moisture gradient, seral lodgepole pine persists with mean fire returns of 100–175 years, and climax spruce (*Picea engelmannii* × *glauca*) occupies pockets with high moisture levels such as riparian zones or through succession with long intervals of no fire (Parminter 1992). The landscape is gently rolling with low topographic relief, minimizing the topographic influence on fire behavior. Elevation ranges from 850 to 1300 m, in the region.

Field sampling occurred within three wildfire perimeters (Fig. 1). All fires were lightning ignited and received minimal to no suppression activities due to wilderness management

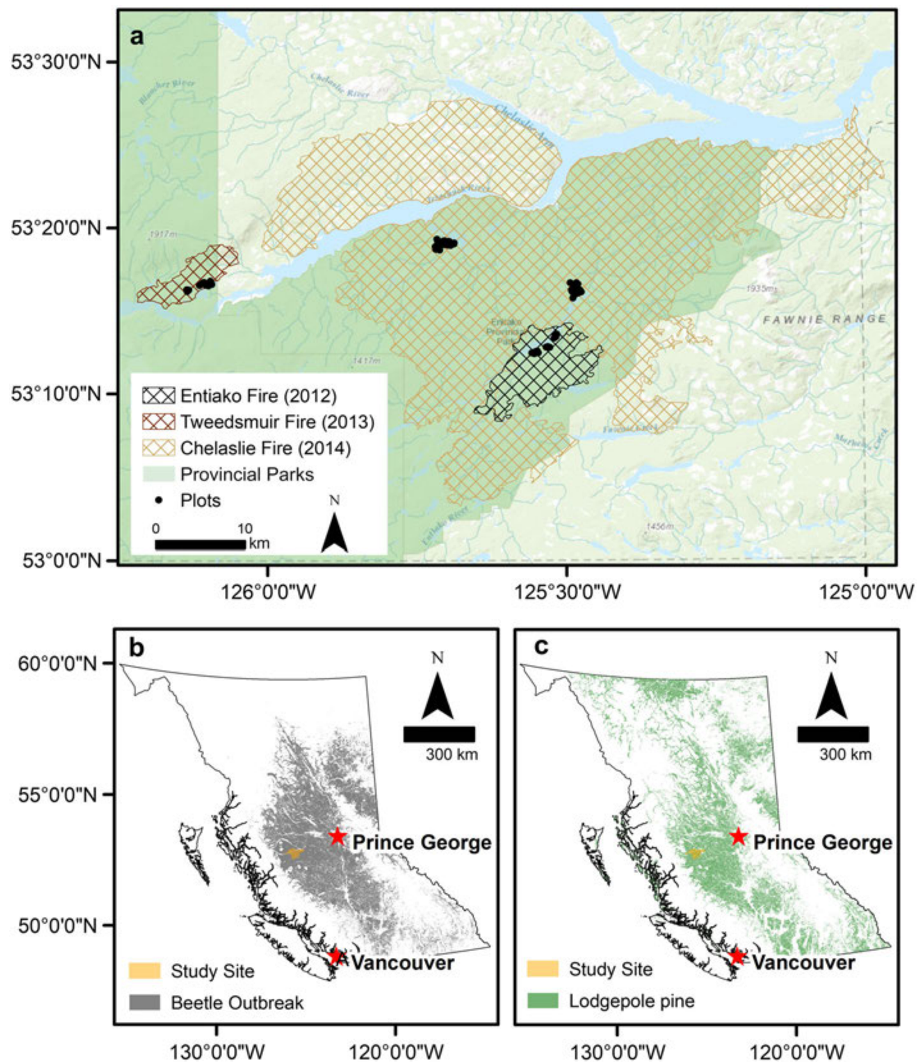

Fig. 1. Maps of the study area, MPB outbreak extent, and lodgepole pine range. Fire perimeters for three study fires that burned in 2012 (Entiako Fire; 7459 ha), 2013 (Tweedsmuir Fire; 3354 ha), and 2014 (Chelaslie Fire; 133,000 ha). (a) Provincial park boundaries are displayed as protected areas and overlaid with fire perimeters. Panel (b) shows the extent of the MPB outbreak across British Columbia based on aerial survey data from 2000 to 2011 (BCMFLNRO 2016). Panel (c) shows the estimated range of lodgepole pine across British Columbia.

objectives for the parks (Rob Krause and Mike Pritchard, *personal communication*). The Entiako Lake fire (R10171) burned 7450 ha from 3 August 2012 to 22 September 2012 (BCWS 2016). The Tweedsmuir fire (R10252) burned 3600 ha from 9 September 2013 to 16 September 2013 (BCWS 2016). The Chelaslie River fire (R10070) burned 133,100 ha from 8 July 2014 to 26 October 2014 (BCWS 2016). All fires burned through gray-phase outbreak conditions of varying severity. The recent MPB outbreaks, circa 1990s and 2000s, peaked in the region around 2003/2004 and have affected forests across much of the province (Fig. 1). MPB activity has declined since 2006 (Wulder et al. 2009). The lag between peak outbreak and the three wildfires was about a decade, with standing dead trees (snags) beginning to transition to coarse woody debris in some outbreak-affected stands. The location of fires within parks provided a rare opportunity to study MPB-fire interactions without the interference of active management (i.e., harvest/salvage activity, fire management/suppression).

Sampling design

Field sampling occurred from late June through August 2016, allowing us to characterize a snapshot of early successional forest communities from two to four years postfire. The study area has no road access, and sampling was limited by hiking and boating distances from three remote cabins within the parks (Fig. 1). Study plots were distributed through forest dominated by lodgepole pine. We selected plots based on a two-pronged approach including an a priori site selection using digital data, followed by verification and final selection in the field. A priori digital data included aerial survey data of MPB outbreak severity (BCMFLNRO 2016), burn severity maps generated from the differenced normalized burn ratio (dNBR; Eidenshink et al. 2007), and vegetation maps from the province's Vegetation Resource Inventory (VRI) to target areas of pure lodgepole pine (BCMFLNRO 2012). Aerial survey data for MPB outbreaks were a coarse resolution (400 m raster data products) and indicated areas within and around our study sites ranged between 50% and 100% canopy mortality (BCMFLNRO 2016). We selected from 12 to 39 plots in each study fire, depending on accessibility, for a total of 63 plots (Appendix S1: Table S1). We distributed plots across a gradient of fire severity within each fire class based on dNBR maps as low (n = 22), moderate (n = 18), and high (n = 23) that equated to light surface, severe surface, and crown fire based on our field measurements.

We visited plots on the ground for data collection. We verified canopy trees were predominantly lodgepole pine and were representative of the fire severity in the surrounding area. From the plot center location, we recorded GPS coordinates using a Garmin hand-held unit (GPSMAP 78s) and established a ten-by-ten-meter (100 m^2; 0.01 ha) survey plot, and divided it into four quadrants along the north–south and east–west axes, identified as NE, SE, SW, and NW. Within each quadrant, we placed a one-by-one-meter (1 m^2) subplot at increasing distances from the center of the plot (NE-1 m, SE-2 m, SW-3 m, and NW-4 m). Within plots, we recorded information for each live or dead tree: species, live/dead, evidence of MPB activity including exit holes and j-shaped galleries, diameter at breast height, and measures of first-order fire effects to characterize fire severity (including legacy structures) described in detail below. Within subplots, we recorded surface fire severity as first-order fire effects including duff depth, exposed mineral soil, terrestrial surface char, and litter. Measured variables at the plot and subplot resolution were used to characterize stand structure, MPB outbreak severity, fire severity and to analyze the relationship between MPB outbreaks and fire severity.

Mortality status of canopy trees and outbreak severity

We identified trees as live or dead at time of sampling and assigned a cause of death to each dead tree. We used these data to quantify MPB outbreak severity, mortality from fire, and cumulative mortality for each plot. We identified a tree's cause of death based on protocols adapted from Harvey et al. (2013, 2014*a*). We attributed each tree as most likely to have been (1) killed prefire by MPB (i.e., MPB-killed), (2) killed prefire by another agent (i.e., other-killed), (3) killed by fire (i.e., fire-killed), or (4) live postfire with no evidence of MPB activity (Table 1). We evaluated snags for MPB activity unless they were alive at time of sampling. We assessed each dead canopy tree for presence or absence of exit holes associated with adult beetles emerging from the tree (Harvey et al. 2013, 2014*a*). Then, we removed bark from each dead tree to identify galleries specific to MPB or other bark beetle species (Harvey et al. 2013, 2014*a*). We classified a tree as MPB-killed if it had the requisite exit holes and j-shaped galleries specific to MPB. While much of the prefire tree mortality present was linked to MPB, we also observed significant Ips beetle (*Ips pini*) activity, which we included as other-killed if there was no evidence of MPB. We classified a tree as other-killed if it was lacking evidence of exit holes and j-shaped galleries, but other evidence suggested death prior to fire such as no needle retention in the canopy, sloughing bark, other insect activity, and decay at the base, which is common in this system due to the moist climate (Table 1); this was a small portion of the total trees sampled (7%). We classified a tree as

Table 1. The criteria and classes used to identify a tree's cause of death for the study region in BC.

Cause of death	Description	Trees sampled (%)
Live tree	Live when sampled; green canopy; no visible beetle activity	4.67
Fire-killed	Dead when sampled; scorched bark, branches, and/or outer sapwood; no evidence of galleries or exit holes from MPB or other bark beetle activity; not highly decayed/weathered particularly at the base and in the canopy	28.95
MPB-killed	Dead when sampled; no needles remaining in the canopy; vacated mountain pine beetle (MPB) galleries in cambium with exit holes in remaining bark	59.38
Other-killed	Dead when sampled; highly decayed/weathered, no bark, missing branches; more advanced decay then MPB-killed trees; full deep char with no identifiable vacated MPB galleries	7.00
Prefire-killed: MPB-killed + Other-killed	All prefire-killed from both MPB-killed and other-killed	66.38

Notes: Methods adapted from Harvey et al. (2014*a*). Trees sampled summarize observed data from field collections.

fire-killed if it had red needles in the canopy or postfire needle drop, and no evidence of prefire MPB or other beetle activity. We estimated a general metric of prefire-killed trees as the combination of MPB-killed and other-killed (Table 1). We calculated plot-level metrics for outbreak severity as the proportion of MPB-killed trees per plot and prefire mortality as the proportion of all prefire-killed trees per plot. Pre-outbreak stand estimates were based on all trees in the plot, regardless of status.

Fire severity recorded as first-order fire effects at the plot level

We characterized fire severity with seven measures of first-order fire effects that were scaled to a plot-level metric. We measured fire effects on standing trees/snags and the terrestrial surface including: height of scorch and/or char on trees, percent cover of scorch and/or char on trees, percent deep charring on trees, litter/duff depth, proportion of remaining litter, proportion of terrestrial surface char, and proportion of exposed mineral soil. Scorch and deep char are visually distinct, scorch with a dusty, matte black appearance and deep char with an iridescent black, scale-like appearance. In some cases, snags had both areas of scorch and deep char. The height of scorch and/or deep char (hereafter scorch/char) was measured to the nearest 0.5 m on each tree with four-meter measuring sticks and converted to mean scorch height per plot. We estimated the percent area covered, as height and circumference, of scorch and/or deep char (hereafter scorch/char) and calculated a mean proportional area per plot. We inverted the mean proportion of area per plot to the proportion of unscorched area per plot for analysis. We recorded deep char for each tree as no deep char, <50% deep char, or 50–100% deep char coverage on the snag and calculated the proportion of snags with deep char for a plot-level variable. The four terrestrial surface fire effects metrics were measured in each subplot in the four quadrants of the plot. Litter/duff depth was measured as the combination of litter plus duff to the nearest millimeter in two opposing corners of each subplot and averaged to a plot-level variable. We recorded the percent of remaining litter, terrestrial surface char, and exposed mineral soil and calculated a mean for each variable from the four subplots to generate plot-level metric. Remaining litter, terrestrial surface char, and exposed mineral soil were converted to proportions for analysis purposes. Because we surveyed plots between two and four years postfire, we captured various early successional stages in postfire litter accumulation and vegetative regrowth.

Fire severity recorded as biomass consumption of legacy structure at the tree level

To characterize the effect of outbreaks and wildfires on postfire legacy structure, we categorized biomass loss on each tree based on the remaining branch structure and deep char. The remaining branch structure refers to the fine, moderate, and coarse branch structure, and it was quantified as presence or absence. A classification of absence meant that there was no remaining branch structure on the tree and no associated branches on the ground in the area of the tree/snag, which indicated that branches were consumed by fire. As described above, deep char was visually distinct from scorch and recorded as absent, <50%, or 50–100% deep char coverage on each snag. We retained these categories to assess the relationship between deep char and remaining branch structure and converted the categories to presence or absence of deep char for each tree/snag to evaluate the relationship between a tree's cause of death and deep char development.

Fire weather and topography

The Entiako, Tweedsmuir, and Chelaslie fires that provided the footprint for our study burned during three different fire seasons (2012, 2013, and 2014) across a landscape with low topographic complexity. Fires burned over a relatively long duration within each season, which allowed us to account for variability in fire weather and day-of-burn conditions (Appendix S1: Table S1). Each plot was assigned a day of burn from day-of-burn progression maps estimated from MODIS hotspot data (Parks 2014), which allowed us to assign the daily fire weather index (FWI) to each plot that was generated from the nearest weather station (Appendix S1: Table S1). The calculated FWI is a metric from the Canadian Forest Fire Weather Index System (Van Wagner 1987) that integrates temperature, relative humidity, and wind speed. We used the FWI to assign each plot a burning condition category of moderate (>13–29) or extreme (>29), based on breakpoints outlined by Alexander and De Groot (1988). All fires experienced moderate burning conditions within a portion of their perimeter; however, extreme burning conditions (FWI ≥ 29) only occurred in two of the three fires (2012 Entiako and 2014 Chelaslie fires; Appendix S1: Table S1). For plots, elevation fluctuated between 873 and 1043 m. Plots were relatively flat with a mean slope of 2.6° (range 0–20°). We did not pursue topography as an explanatory variable of fire severity due to the low topographic variability at our study plots.

Statistical analysis

We tested for relationships between each of the seven fire effects metrics and MPB outbreak severity at the plot level, while accounting for burning conditions. Our seven fire effects metrics served as response variables: average scorch/char height, average proportion of unscorched/uncharred area on trees, proportion of trees with deep char, litter/duff depth, proportion of remaining litter, proportion of terrestrial surface char, and proportion of exposed mineral soil. We tested the relationship of each response variable against the proportion of MPB-killed trees (our index of MPB severity) and burning conditions, which was included as a categorical variable of moderate or extreme FWI (Appendix S1: Table S1). An interaction term between burning conditions and proportion of MPB-killed trees was included in all models to assess whether observed relationships changed under different fire weather conditions. Relationships with scorch/char height and litter/duff depth were fit using linear models. The proportion of terrestrial surface char was logit-transformed and fit with a linear model. All other models in which the response variable was a proportion were analyzed with generalized linear models, and each response variable was fit with a distribution appropriate for the type and distribution of the response variable (see Appendix S1: Table S3 for distributions associated with each analysis). We also ran each model and replaced the proportion of MPB-killed trees with the proportion of prefire-killed trees, since dead trees would all be similar in terms of conditions and moisture content regardless of what killed them. The models with the proportion of prefire-killed trees demonstrated similar statistical relations to the proportion of MPB-killed trees. We report all models that were statistically significant, and we kept all fire effects models that were run with the proportion of MPB-killed trees as an explanatory variable, since these models were a more conservative estimate of MPB caused mortality.

We evaluated the effect of outbreak severity and wildfire on postfire legacy structure from tree-level fire effects of deep char and branch structure loss. We analyzed data at the tree level using two different response variables: (1) presence/absence (1/0) of branch structure on individual trees and (2) presence/absence (1/0) of deep char on individual trees. We accounted for burning conditions as a categorical variable of moderate or extreme FWI (Appendix S1: Table S1). An interaction term between burning conditions and cause of death was included in all models. We used generalized linear mixed models with a binomial distribution for presence/absence data using a logit link, and each model included the plot as a random effect and the interaction term between cause of death and burning conditions. Results are reported as probability of occurrence, and the comparison between mortality types (e.g., MPB-killed versus fire-killed) is reported as the odds ratio. Additionally, we determined whether the presence/absence of branch structure was related to the coverage of deep char on the tree. Our explanatory variable of deep char was treated as a three-level categorical variable of no deep char, <50% coverage of deep char, or 50–100% coverage of deep char on the tree, while accounting for burning conditions.

We assessed fit for all models by visually inspecting the residuals, which appeared to be adequately met. We evaluated and corrected for overdispersion in all generalized linear models and generalized linear mixed models when necessary. For our two linear models, assumptions of normality and constant variance of the residuals were checked graphically and appeared to be adequately met. We assessed the interaction term with a drop-in-deviance test. The interaction term was retained in each model regardless of statistical significance, because of the known interaction between fire weather and fuels. All statistical analyses were conducted in R statistical computing software version 3.4.4 with the stats package (R Development Team 2018). For generalized linear models, we used the function glm in the MASS package (Venables and Ripley 2002). For generalized linear mixed models, we used the function glmer in the lme4 package (Bates et al. 2015). We considered $p < 0.05$ as convincing evidence of a relationship and $P < 0.10$ as suggestive of a relationship to minimize the potential of a type II error. Data and code for analyses are available online (Talucci 2019).

Results

We collected data from 943 trees across 63 field plots with 910 lodgepole pine trees/snags and 33 spruce trees/snags. Canopy tree species were predominantly lodgepole pine with a plot mean of 96% (range across plots 63–100%; Appendix S1: Table S2). Estimated mortality from MPB was 59% of all trees sampled, and estimated prefire mortality (i.e., MPB-killed plus other-killed) was 66% of all trees sampled (Table 1). When we evaluated just lodgepole pine mortality across all 63 plots, the estimated mean for lodgepole pine killed by MPB was 63% and the estimated mean for lodgepole pine killed by all agents prior to fire (all prefire) was 70% (Appendix S1: Table S2). Cumulative mortality was estimated at 93% for lodgepole pine as a combination of prefire and fire mortality (Appendix S1: Table S2).

Effect of outbreak severity on first-order fire effects at the plot level

The effect of outbreak severity on fire severity was limited, with six of seven fire effects showing no evidence of an effect (Fig. 2, Appendix S1: Table S3). average scorch/char height, average proportion of unscorched/uncharred area on trees, litter/duff depth, proportion of remaining litter, proportion of terrestrial surface char, and proportion of exposed mineral soil showed no evidence of an effect of outbreak severity (Fig. 2, Appendix S1: Table S3). Outbreak severity did show evidence of an effect on proportion of trees with deep char. Under moderate burning conditions, the proportion of trees with deep char increased with increasing outbreak severity (Fig. 2, Appendix S1: Table S3), which held true when we substituted the proportion of prefire-killed trees for MPB-killed trees (Fig. 2, Appendix S1: Table S3). Under extreme burning conditions, the relationship between the proportion of MPB-killed trees and deep char was not statistically significant, however when we substituted the proportion of prefire-killed trees for MPB-killed trees that relationship was statistically significant (Fig. 2, Appendix S1: Table S3).

Effect of outbreak and wildfire on legacy structure

Outbreak severity and wildfire showed distinct evidence of an effect on the legacy structure of the forest, measured by biomass consumption as deep char and branch structure loss on individual trees. Both deep char development and branch structure loss had greater odds of occurrence when a tree was dead prior to fire (i.e., MPB-killed or prefire-killed) compared to being

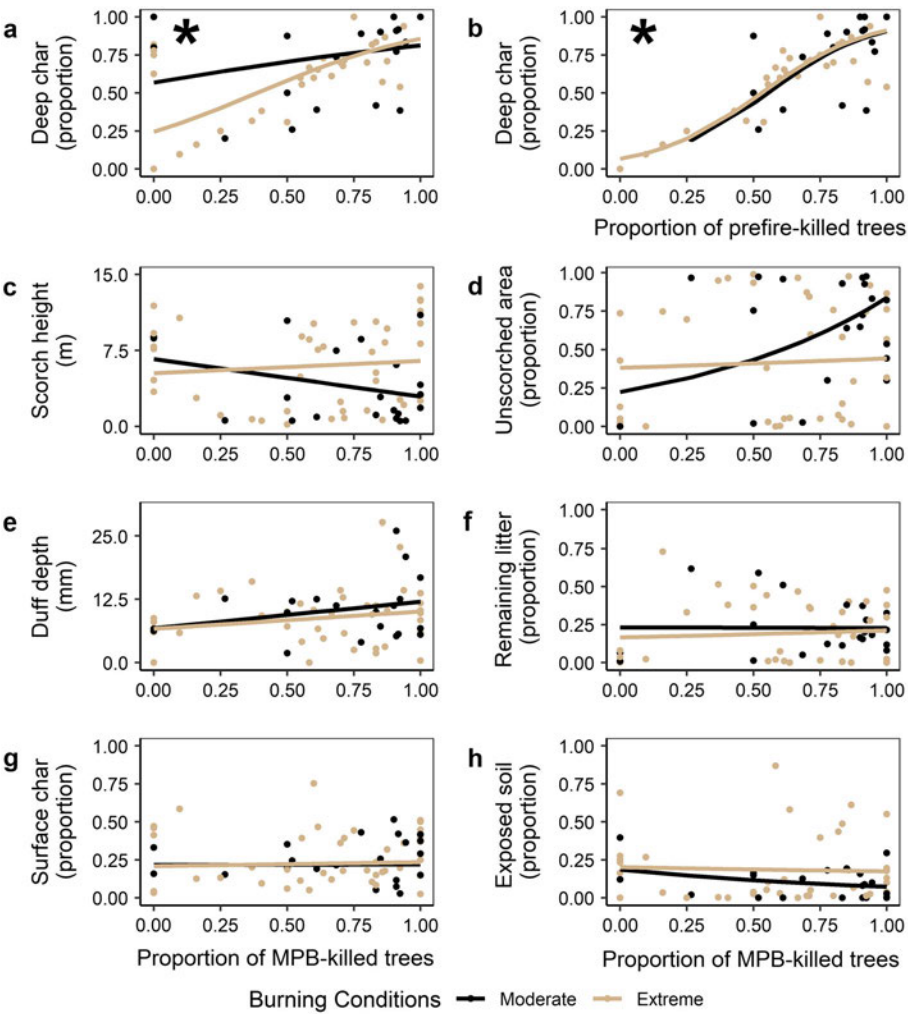

Fig. 2. The relationships between outbreak severity and seven first-order fire effects measured at the plot level: proportion of trees with deep char (a–b), average scorch/char height (c), average proportion of unscorched/uncharred area on trees (d), litter/duff depth (e), proportion of remaining litter (f), proportion of terrestrial surface char (g), and proportion of exposed mineral soil (h). Response variables are along the *y*-axis with the explanatory variable of the proportion of mountain pine beetle (MPB) killed trees or prefire-killed trees (only panel b) along the *x*-axis. Points are the raw data (n = 63 plots), and fitted lines show the estimated statistical relationship. The response variable of deep char is shown in panels a and b, and they were the only two models that indicated a strong statistical relationship (*). Response variables c–h were unrelated to outbreak severity. See Appendix S1: Table S3 for model estimates and confidence intervals.

alive at time of fire, which was consistent across both moderate and extreme burning conditions (Figs. 3, 4, Appendix S1: Table S4). Under both moderate and extreme burning conditions, a MPB-killed and prefire-killed snag had greater odds of developing deep char compared to a fire-killed tree (Figs. 3, 4, Appendix S1: Table S4). There were greater odds of branch structure being consumed on a MPB-killed and prefire-killed snag compared to a fire-killed tree under moderate conditions, and the size of that effect was slightly smaller under extreme conditions but still significant (Figs. 3, 4, Appendix S1: Table S4). We found that branch structure had greater odds of being consumed when deep char exceeded 50% coverage on the tree for both

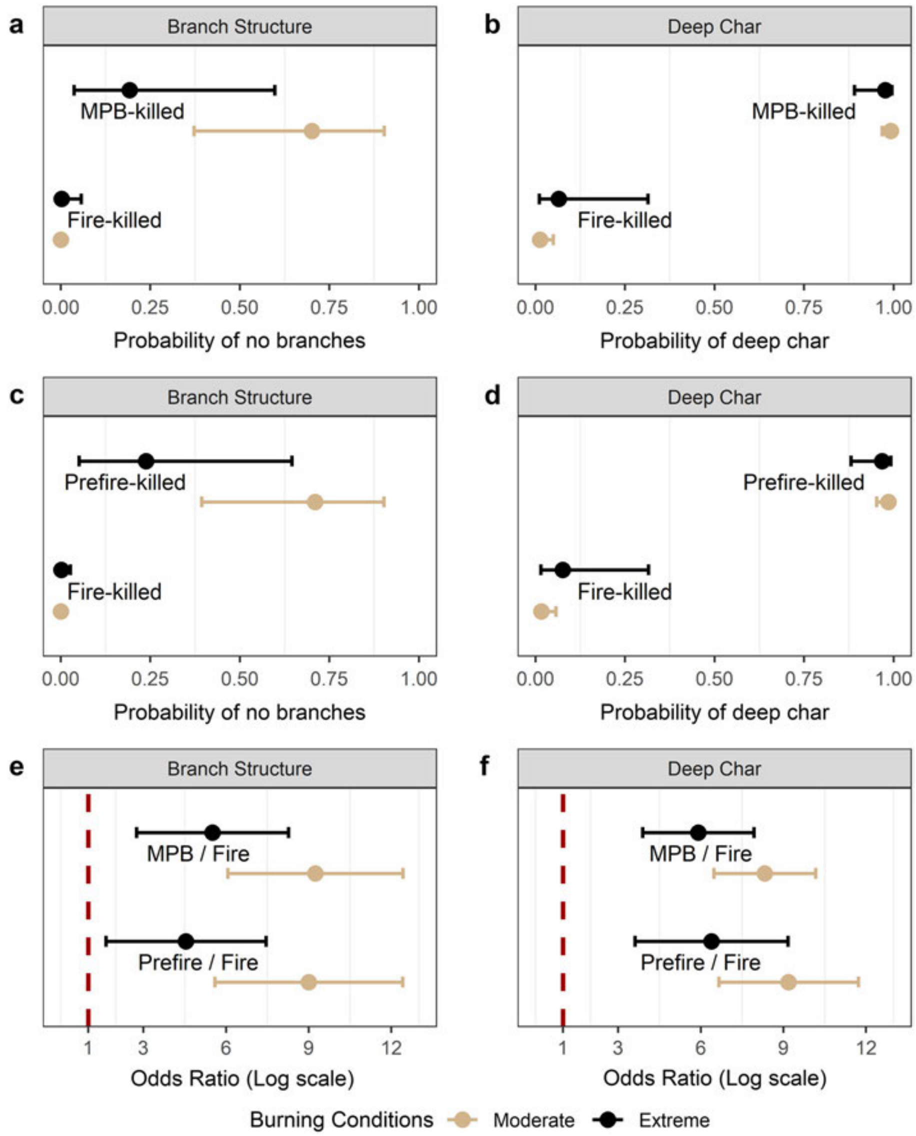

Fig. 3. The effect of outbreak severity and wildfire on legacy structure measured at the tree level. Tree-level fire effects show the role of mountain pine beetle (MPB) outbreak severity (MPB-killed and prefire-killed) on consumption of woody material and simplification of structural legacies in the form of branch loss and deep char development (a–d). Comparison between groups (i.e., MPB-killed versus fire-killed) is shown in (e) and (f) as odds ratios with the red dashed line marking no difference at one. The model estimates are listed in Appendix S1: Table S4.

moderate and extreme burning conditions (Fig. 5, Appendix S1: Table S5).

Discussion

We found that fire severity as measured by scorch/char height and area, and surface fire metrics, is not influenced by MPB outbreak severity but that fire severity measured as biomass loss and legacy structure was consistently influenced by the outbreak history. These findings from BC align with previous field research that evaluated the influence of outbreak severity on fire severity in the western United States

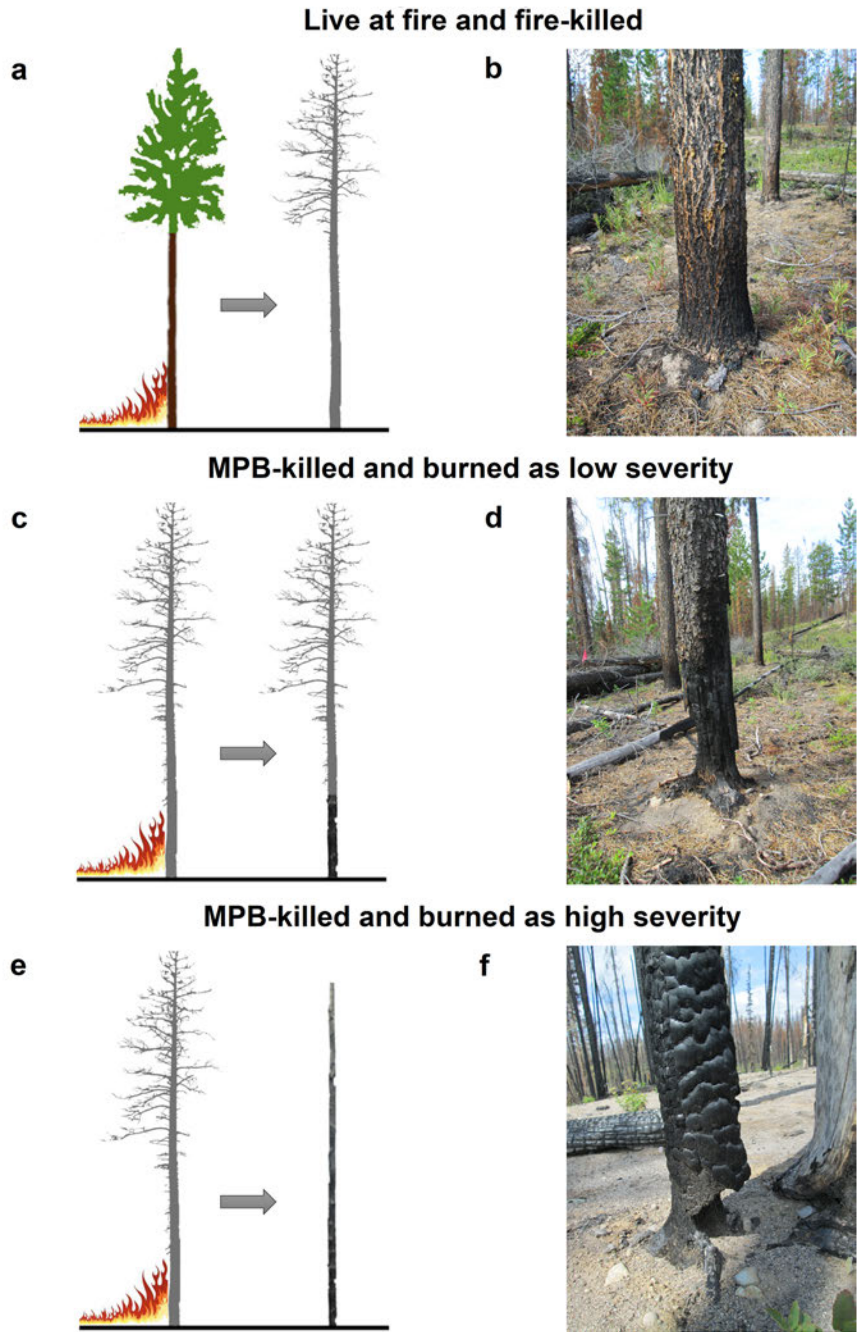

Fig. 4. Tree-level fire effects are dependent on whether a tree is alive or dead at time of fire. Panel (a) illustrates a tree that is live at time of fire and killed by fire, with the adjacent panel (b) showing a photo of a tree live at time of fire and killed by fire with scorched bark but no consumption of the tree. Panel (c) illustrates a MPB-killed tree that burns under low severity conditions, with the adjacent panel (d) showing a photo of deep char development and consumption at the base of the tree, which is attributed to fungal development (Donato et al. 2009). Panel (e) illustrates a MPB-killed tree that burns under high-severity conditions, with the adjacent panel (f) showing a photo of deep char that covers the entire tree in a plot that burned as high severity.

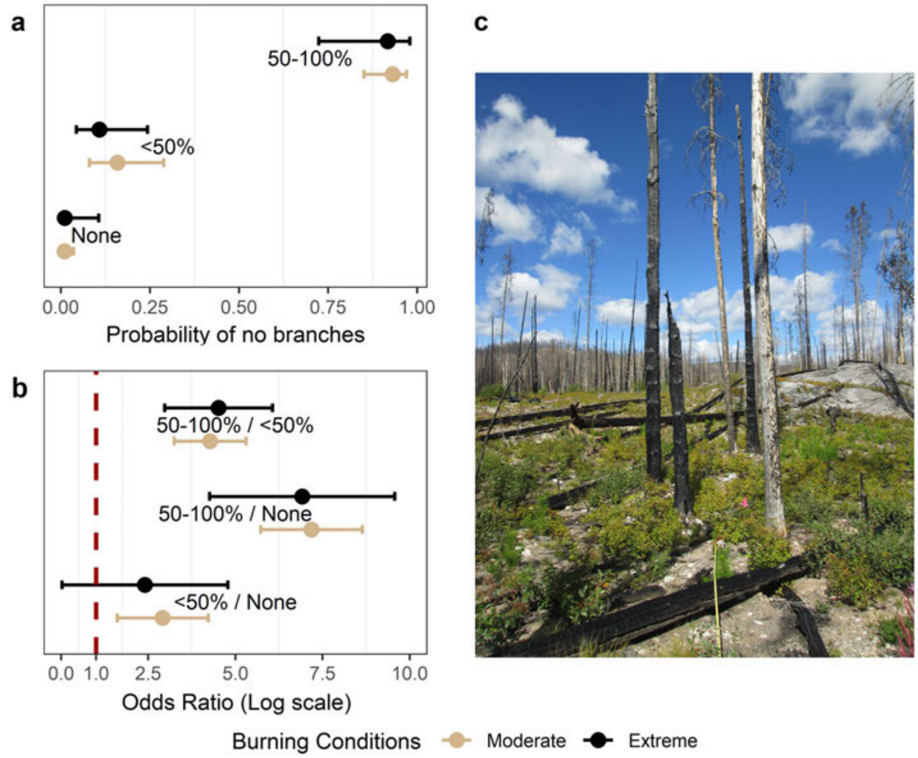

Fig. 5. Deep char coverage influences the consumption of branch structure on a tree. The probability of branch loss from deep char development is shown in (a). Comparisons between groups are shown in (b) with a red dashed line demarcating no difference at one. In (c), a photo of a snag with deep char and no branches adjacent to a snag with branches still intact and no deep char. Model estimates are list in Appendix S1: Table S5.

(Harvey et al. 2014*a*, *b*, Agne et al. 2016), but extends our understanding of these short-interval disturbances by highlighting the synergistic effect on postfire structural legacies. Prefire mortality had a greater likelihood for increased biomass consumption and deep char, which aligns with findings on reburns, where wildfires recur in short intervals (Donato et al. 2016). Prefire mortality, regardless of the mechanism of death, results in an altered legacy structure that is more simplistic and has more deep char. While this effect on structural legacies is generally accepted in the field, it has been broadly overlooked and unquantified in assessments of how outbreak affects fire severity. When asking the question "does MPB outbreak affect fire severity, are they linked disturbances," the answer is yes—specifically through the deadwood structure that remains in these ecosystems.

Effect of outbreak severity on first-order fire effects at the plot level

Outbreak severity did not show evidence of an effect on fire severity for six out of seven measured first-order fire effects; the exception was deep char. This reflects similar findings in gray-phase outbreak conditions found by Harvey et al. (2014*b*) and extends our understanding to the geography of BC's sub-boreal forests. Our finding indicates some inherent noise and uncertainty in our data as well as the influence of fire weather. The six fire effects—scorch/char height and area, duff depth, litter, surface char, and exposed soil, are likely controlled by the combined factors of the fire environment, that is, the interaction of fuels, weather, and topography (Countryman 1972, Krawchuk and Moritz 2011, Whitman et al. 2015) but without a strong signal from outbreak fuel structure, which aligns with previous research evaluating interactions between outbreak severity and fire severity (Harvey et al. 2014*a*, *b*, Agne et al. 2016). Scorch on trees is naturally variable and can be driven by multiple factors including the composition of fuel structures, crown and/or surface fire spread, burning conditions, slope steepness, and ignition patterns (Alexander and Cruz 2012*a*). Our results show no evidence of a relationship between terrestrial surface fire effects and outbreak severity, which was also consistent with findings in previous retrospective studies with gray-phase outbreak conditions (Harvey et al. 2014*b*, Agne et al. 2016). The lag time between needle drop and our study fires would have allowed for the decomposition of fine fuels (Simard et al. 2011, Harvey et al. 2013) thus minimizing the effect of outbreak on surface fuels. Most snags were still standing at time of fire, so that the concern of increased surface fire severity from abundant coarse woody debris was not observed. These findings support the general narrative that low-frequency and high-severity fire regimes associated with lodgepole pine in sub-boreal forests are strongly driven by climate systems of high-pressure, creating dry-hot conditions conducive for burning such that variability in fuel structure/vegetation plays a secondary role (Bessie and Johnson 1995, Nash and Johnson 1996, Whitman et al. 2015).

Effect of outbreak and wildfire on structural legacies

Our findings support the notion that dead wood, which in our landscapes is predominantly snags generated by MPB outbreaks, burns differently than live wood and indicates an important MPB-fire connection. Live trees rarely experience significant combustion, and therefore, little to no consumption occurs on the tree (Campbell et al. 2007), which is attributed to higher moisture content compared to their dead counterparts (Brown et al. 1985). Extended periods of smoldering and glowing combustion (Brown et al. 1985, Page and Jenkins 2007, Hyde et al. 2011) are facilitated by lower moisture content in snags and coarse wood (Stockstad 1979). Lower moisture content in snags could enable passive crown fire or torching of snags (Wenger 1984), which may be the primary mechanism for consumption of branch structure. Some simplification of branch structure may also occur on gray-phase MPB-killed trees prior to fire. The torching of snags and extended periods of smoldering have been demonstrated in areas that experience reburn, wildfires that recur in short intervals (Donato et al. 2016). High-severity reburns have shown there is an eightfold increase in deep char development on snags and the retention of woody biomass is half the amount of once burned areas, in the Klamath Mountains of southwestern Oregon (Donato et al. 2016). Where wind-throw is followed by wildfire in short intervals, snags and coarse wood have been shown to be reduced with a marginal increase in charred material (Buma et al. 2014). In lodgepole pine/Douglas-fir (*Pseudotsuga menziesii*) forests on the Chilcotin Plateau of BC south of our study sites, areas of high prefire mortality from MPB outbreak experienced 13% more consumption of dead wood and the variability in canopy consumption was attributed to mortality status with dead prefire snags having more of their branch structure consumed (Brad Hawkes, *personal communication*). This evidence indicates that it is not necessarily the mechanism of prefire mortality, for example, MPB outbreak, wind-throw, or prior wildfire, but the fact that there is an abundance of deadwood with altered moisture levels and fuel structures compared to live wood, which alters postfire ecological and structural legacies as they relate to standing snags and coarse woody debris.

The consumption of branches and deep char development on snags alters the structural legacies that endure through fire. These altered legacies may introduce long-term implications for ecosystem structure and function including availability of canopy seedbank, accumulation of coarse woody debris, and early seral structure and resources for early seral species (Franklin et al. 2000, Swanson et al. 2011, Johnstone et al. 2016). After MPB outbreak, lodgepole pine snags continue to retain some of their aerial seedbank in the canopy postmortem while some cones fall to the forest floor (Teste et al. 2011). Cones in snags or on the forest floor can be exposed to extended heating from a snag smoldering or slower moving surface fire (Alexander and Cruz 2012*b*), which could reduce seedbanks and influence postfire resilience (Johnstone et al. 2016). The loss in snag biomass and branch structure alters the accumulation of coarse wood that may influence short- and long-term carbon and nutrient cycles (Harmon 2001), structure of habitat for wildlife (Fontaine et al. 2009, House 2014) including nesting and perching habitat, and both structure and function of early seral ecosystems (Swanson et al. 2011). More charring on trees reduces the quality of the snag for saproxylic insects thereby affecting foraging woodpeckers (Saint-Germain et al. 2004, Nappi et al. 2010), which could influence trophic webs. Deep char development can encapsulate the remaining wood, which may limit decomposition, slow decay, and extend long-term carbon storage (Preston 2009, Bird et al. 2015). Together, these changes to dead wood that may alter the long-term structure and function in the postfire forest are considered compound disturbance effects, where the outbreak severity and fire severity work in combination to create unique post-disturbance conditions that are different than the outcomes of the singular disturbance of wildfire (Paine et al. 1998). Further research is needed to determine the long-term implications of compound disturbance effects related to legacy structure, coarse wood recruitment, carbon storage, pyrogenic carbon, habitat structures, trophic webs, and early seral ecosystems in forests where fires are increasingly burning through stands with high volumes of snags from insects, windthrow, drought, and prior fire.

Conclusion

Sub-boreal forest ecosystems of BC have experienced widespread tree mortality from the MPB outbreaks, generating a fuel structure characterized by an abundance of deadwood that is now interacting with wildfires. The contiguous landscape of lodgepole pine-dominated forests situated at the epicenter of the outbreak in western North America allowed us to assess interacting, or linked, effects between outbreak and fire severity. Our results suggest that while many first-order fire effects are not influenced by outbreak severity, legacy structure related to the degree of biomass consumption is strongly influenced by the interaction of outbreak severity and fire severity. These findings are especially important to consider after the 2017 and 2018 fire seasons in BC wherein a record number of hectares burned, with many of the fires burning through snag forests affected by MPB outbreaks.

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Supporting Information

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecs2.2744/full

Perspectives on LANDFIRE Prototype Project Accuracy Assessment

Chapter 13

Perspectives on LANDFIRE Prototype Project Accuracy Assessment

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Introduction

The purpose of this chapter is to provide a general overview of the many aspects of accuracy assessment pertinent to the Landscape Fire and Resource Management Planning Tools Prototype Project (LANDFIRE Prototype Project). The LANDFIRE Prototype formed a large and complex research and development project with many broad-scale data sets and products developed throughout its various stages. The scope of the project was defined as mapping and modeling vegetation, wildland fuel, and fire regime characteristics (Rollins and others, Ch. 2). Because of the breadth of the investigation, it is important to base our expectations for accuracy on a clear understanding of the intricacies, interdependencies, and scope of mapping and modeling LANDFIRE products. Our goals in this chapter are to: 1) provide relevant background information regarding accuracies and what was realistically achievable in the LANDFIRE Prototype, 2) provide background regarding our strategies for LANDFIRE National, 3) describe our actual LANDFIRE Prototype accuracy results in broad terms, and 4) provide recommendations for the national implementation of LANDFIRE. This chapter is not intended to provide an exhaustive list and description of all of the various accuracy-related issues and conclusions resulting from the LANDFIRE Prototype (for specific details, the reader will be referred to the appropriate chapters). Rather, this chapter is intended to be broad in scope and to place the many accuracy components within the context of the LANDFIRE Prototype and LANDFIRE National projects. Please note that Lunetta and Lyon (2004) provide an in-depth discussion of the current state of accuracy assessment within the science community.

Background

General Accuracy Tenets and Philosophy

First we will provide the reader with several broad tenets used in defining accuracy assessment for the LANDFIRE Prototype Project and thereby lay the foundation for the more in-depth discussion following.

Tenet 1: Assuming that thematic detail and spatial scale are constant, product accuracy is generally inversely correlated with the size of the region being assessed.

Within the remote sensing literature, there are many references to accuracy levels, and many of the reported values are quite high. These high levels may lead to inflated expectations regarding what types of accuracies will be achievable from LANDFIRE. Many previous studies were conducted within relatively small study

In: Rollins, M.G.; Frame, C.K., tech. eds. 2006. The LANDFIRE Prototype Project: nationally consistent and locally relevant geospatial data for wildland fire management. Gen. Tech. Rep. RMRS-GTR-175. Fort Collins: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

areas, often aided by high levels of "hand crafting" during the mapping process and/or in-depth knowledge of the particular study area. We do not have the luxury of spending a great amount of time and effort on any one particular region mapped through the LANDFIRE Project, and the mapping and modeling tasks need to be accomplished through largely automated processes. These limitations do not by any means reduce the value of the products being created through LANDFIRE; however, it should be stated that LANDFIRE products will likely have lower overall accuracies than do data sets derived from more localized studies characterized by large amounts of field data, increased processing effort that may include on-screen digitizing and recoding, and/or iterative refinement of modeled results.

Tenet 2: The higher the thematic detail, the lower the accuracy.

A relatively large number of vegetation classes were mapped for the LANDFIRE Prototype (Long and others, Ch. 6). While the chosen map unit classification system made sense on many levels for the LANDFIRE Prototype, it must be recognized that the proliferation of classes in this or similarly complex systems will imply a relative decrease in accuracy levels. This does not in any way diminish the value of the vegetation products, but is rather simply a result of a more complex map unit classification design. For example, a two-category classification of water and uplands is likely to result in high accuracy, with expected accuracies above 99 percent. This high accuracy does not mean that the *value* of the product is particularly high, but simply reflects that the accuracy for depicting these two classes is high. Additionally, there are difficulties that arise when categorizing continuous phenomena into rigid and discrete classes. For instance, a more detailed map unit classification system might treat juniper and pinyon – juniper ecosystems as several discrete classes even though the boundaries between them are relatively arbitrary and difficult to delineate both in the field as well as within the imagery. With complex vegetation map unit legends, such as that used in the LANDFIRE Prototype, vegetation class accuracy levels can be expected to drop. Nevertheless, LANDFIRE products reliably and consistently describe the distribution of vegetation composition, condition, and structure and associated wildland fuel and fire regimes across broad landscapes. These mapped data are useful for hazardous fuel reduction projects, for a variety of resource management projects, and for both strategic and tactical wildland fire management.

Tenet 3: Field information used for assessing accuracy is not perfect.

As mentioned under Tenet 2, the LANDFIRE Prototype vegetation map unit legends are relatively complex (Long and others, Ch. 6). The map unit classifications are developed using large quantities of field data, and all of the field plots are assigned to one of the many possible classes. Most of these plots are used to generate maps, but some are reserved for use in the accuracy assessment phase of the investigation. We recognize four major potential sources of error associated with field plot data:

- Errors occur frequently in the identification of species and measurement of vegetation structure in the field (for example, in the data for one prototype field plot, a misplaced decimal point indicated a shrub height of 60 feet).
- The vegetation on some field plots has undoubtedly changed between the time the field data were collected and when the imagery was acquired.
- Geo-location errors in plot and imagery data result in inaccurate characterization of some imagery pixels.
- The assignment of plots to specific vegetation classes will have errors associated with the wide array of opinions among professional field ecologists regarding the field classification of any given field plot.

Tenet 4: The modeled results of complex ecological systems will be characterized by ambiguity and controversy.

The products generated from the LANDFIRE Prototype represent our best approximations in depicting the current status of very complex natural phenomena. The information used in our modeling efforts is based on the best available input data and assumptions. However, although our output products represent reasonable and robust depictions of current conditions, we recognize that, due to lack of baseline research, our knowledge of certain ecological systems is imprecise. Use of such information in the modeling process may result in potential flaws in the products, and hence not all of the core LANDFIRE deliverables will be free of error and ambiguity. Nevertheless, the LANDFIRE Project represents an integration of the best available science in remote sensing, ecosystem simulation, landscape fire and succession modeling, predictive landscape mapping, and wildland fire behavior and effects prediction.

We are therefore confident that the products generated represent the best current assessments of the status of these ecosystems with regard to wildland fire and will be of great value to natural resource managers.

Accuracy Assessment Considerations for LANDFIRE

The need for conducting accuracy assessments of the spatial products created from mapping projects has been well documented (Congalton 1991; Foody 2002). Factors that influence map accuracy include (but are not limited to) the remote sensing platform, the quality of ancillary sources of information, the quality of field data, the floristic complexity of the map unit classification system used, and the sampling design. Traditional first-order map accuracy estimates involve generating an error matrix, computing overall accuracy, and estimating "producer's accuracy" and "user's accuracy" (Congalton 1991). In the past, assessment of map accuracy has involved much post-mapping fieldwork in order to develop error matrices. These formal, traditional accuracy assessments involving field campaigns can be labor-intensive, time-consuming, and cost-prohibitive, especially when dealing with projects that cover large regions of diverse and overlapping vegetation composition and conditions (Stehman and others 2000). For this reason, only a few efforts have conducted accuracy assessments across broad expanses such as the entire United States (Stehman and others 2003; Wickham and others 2004).

Techniques that worked well in assessing mapping accuracy across large regions for the 1990s National Land Cover Database (NLCD; Vogelmann and others 2001) employed modifications of traditional accuracy assessment methodologies (Stehman and others 2003; Wickham and others 2004). As background, the 1990s NLCD database was developed using Landsat satellite imagery acquired for the Multi-Resolution Land Characteristics (MRLC) 2001 consortium using methods previously described (Vogelmann and others 1998). During development of the database, it was determined that an accuracy assessment for the large area product was required, and that such an effort would have to be modified from more traditional assessments. The modifications were necessary in part due to the scarcity of field data across the mapped regions, the large size of the area being assessed (and associated high costs of collecting data from a statistically valid number of field locations across the entire conterminous United States), difficulties in assigning unambiguous map unit labels to many field plots, and geolocational errors associated with field plot and satellite-derived mapping information.

Three important lessons learned from the accuracy assessments of the 1990s NLCD effort pertain directly to the accuracy assessment methods used during the LANDFIRE Prototype Project:

- Collecting data for and compiling custom field databases is time consuming and expensive. Similarly, combining data from disparate sources and distilling them into a training database for mapping purposes is time consuming, expensive, and can result in data inconsistencies unless special effort is made to crosswalk and/or standardize input data. On the other hand, using existing field data, rather than collecting custom field data, saves both time and money. In short, for large-area projects, it makes sense to use existing field data for conducting accuracy assessments.
- Determining accuracy values for different subregions is acceptable when mapping large regions. Accuracies are likely to vary across large mapped areas due to region-specific heterogeneity in landscape composition and structure, and it was advantageous to derive an understanding of the geographic variability of accuracies of the products developed for LANDFIRE. To this end, use of a systematic random sampling design can provide optimal results. Such a design ensures that all geographic regions are adequately sampled and thereby ensures that at least some estimates of accuracies exist throughout the entire study region.
- Some errors are more "wrong" than others. For instance, for the LANDFIRE effort, misclassification of a pinyon – juniper stand as a riparian woodland stand will likely have a greater negative impact on the predicted fire behavior than misclassification of a pinyon – juniper stand as a juniper stand. Furthermore, some vegetation types are spectrally and biogeographically very similar to other vegetation types, and even with "perfect" source material, it is difficult to adequately distinguish some of these classes. For example, Douglas-fir and white fir are spectrally very close (fig. 1), and both species inhabit similar ecological niches. In regions where both Douglas-fir and white fir occur, we can expect significant confusion between the two classes. For instance, in central Utah, cross validation accuracies for these two classes were quite low, as anticipated. Nonetheless, we suspect that the errors related to misclassifying similar vegetation types will only minimally impact predicted fire behavior, whereas

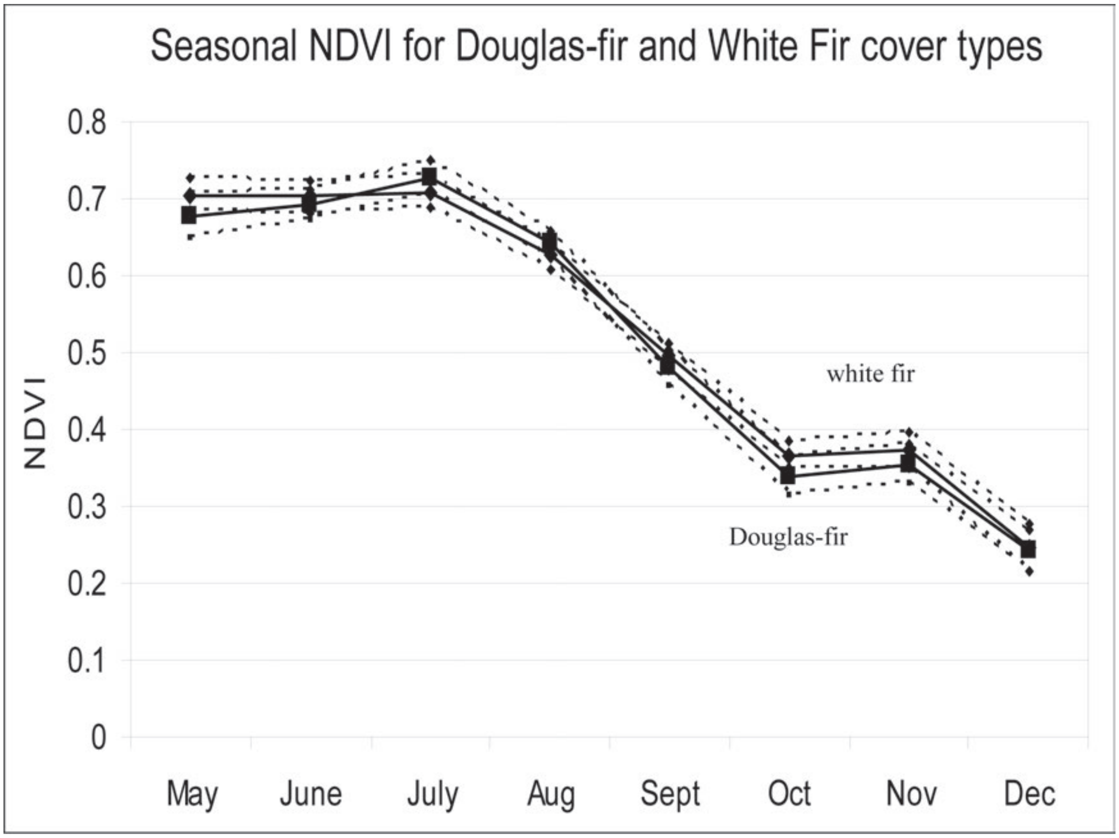

Figure 1—Seasonal normalized difference vegetation index (NDVI) spectral profiles for Douglas-fir and White Fir cover types.

errors related to misclassifications of more dissimilar vegetation types lead to greater negative impact. For this reason, both ecologists and image analysts need to critically analyze error matrices in order to fully understand and characterize the ways in which product errors may affect project objectives.

We took these lessons into consideration in the design of our LANDFIRE accuracy assessment protocol:

- Because LANDFIRE is a large-region project, we tapped into a variety of data sources and made use of existing field data to assess the accuracy of LANDFIRE Prototype products (rather than wasting time and money collecting data for and compiling a custom field database). See Caratti, Ch. 4 for details on the acquisition of data for and compilation of the LANDFIRE reference database.
- Cross-validation error matrices were generated and examined separately for both LANDFIRE Prototype regions.
- For the LANDFIRE Prototype, mappers, ecologists, and wildland fire scientists critically evaluated errors at several stages in prototype product development. These evaluations resulted in aggregation and disaggregation of classes based on the "mappability" and "model-ability" of the vegetation classes. See Keane and Rollins, Ch. 3 and Long and others, Ch. 6 for detailed descriptions of the creation of the final vegetation legends for the LANDFIRE Prototype. This expert-based process for map unit classification refinement is built into the accuracy assessment system for LANDFIRE National.

Overview of Accuracy Assessment Conducted for the LANDFIRE Prototype Project

The LANDFIRE Prototype Project involved many sequential steps, intermediate products, and interdependent processes, each involving evaluations of the accuracy of intermediate and final products. Please see appendix 2-A in Rollins and others, Ch. 2 for a detailed outline of the procedures followed to create the entire suite of LANDFIRE Prototype products.

Role of Input Data

Field data accuracy issues—Field data played a critical role in many stages of the LANDFIRE Prototype. These data were essential inputs for developing the vegetation products, percent canopy cover and height data layers, and potential vegetation data layers. See Caratti, Ch. 4 for detailed information on data acquisition for and compilation of the LANDFIRE reference database.

Described below are a number of data quality issues that needed to be addressed in the LANDFIRE Prototype.

- *Number of field plots*: For the LANDFIRE Prototype accuracy assessment, we used all field plot data that met the stringent quality-control criteria (Caratti, Ch. 4) and represented the large number of classes mapped during the vegetation mapping tasks (for details about the vegetation mapping procedures, see Frescino and Rollins, Ch. 7 and Zhu and others, Ch. 8) We used literally thousands of points for each of the two prototype regions. During this process, we recognized that some vegetation classes had limited numbers of field plots. Short of gathering additional plot information (see Keane and Rollins, Ch. 3 for LANDFIRE Prototype design criteria), there was no obvious solution to this problem. We attempted to map these rarely sampled vegetation types, even when we had limited numbers of field plots for those classes. We believe that most of these rare classes were under-represented in the resultant products.

- *Field plot geolocational accuracy*: Field plots must have accurate geolocational coordinates to geographically rectify with the many spatial databases involved in the LANDFIRE process. This was especially important during the vegetation cover and structure characterization phase of the LANDFIRE Prototype, wherein each field plot was matched with a single Landsat pixel and used in the mapping process. Any significant error in the field location coordinates has the potential to match the wrong spectral information with that particular field plot, thereby resulting in mapping error. For the prototype effort, we overlaid plot locations onto satellite imagery to determine whether there were plots that obviously did not match the imagery. While most plot locations appeared to be reasonable, we observed that many plots representing natural vegetation were actually located on major roads. When plot information was originally acquired for these sites, the actual Global Positioning System (GPS) measurements were apparently made at the road locations adjacent to the field plots, rather than within the field plots. Thus, the GPS locations did not exactly match the locations where the field measurements were made. For these sites in the LANDFIRE Prototype, a new set of geolocations was derived to better represent actual field plot locations.

  In another case, we noted (also based upon imagery assessment) that many putative shrub sites were located in obviously forested areas. We later discovered that those plots corresponded to a particular project in which the main focus was to describe shrub vegetation regardless of whether or not it represented the dominant vegetation type. These plots were consequently discarded from the prototype accuracy assessment. Both cases illustrate the need for assessing field plot information in conjunction with satellite imagery to ensure that the field information is accurately recorded.

  Moreover, it should be recognized that satellite imagery can have georeferencing errors as well. As a general rule, the coordinates of most pixels in the imagery used for the LANDFIRE Prototype are within 30 meters of the actual location – but exceptions occur. Even in the case where a pixel has slightly greater than a 15-meter error associated with it, this may be large enough to create a slight yet definite mismatch between the imagery and field information. While there is little that we can do about this problem, we at least need to recognize that some of the error term associated with the products generated will be attributable to this issue.

- *Assignment of field data into discrete vegetation classes*: One of the challenges in generating land cover maps is the stratification into discrete classes of a very complex natural world composed of multiple continuums. Regardless of which vegetation map unit system is used, many vegetation plots will represent elements of two or even more classes, and thus some plots will defy unambiguous categorization. As an example of one such problem, we mapped Juniper and Pinyon – Juniper (PJ) as two distinct classes. In nature, pinyon pine and juniper often coexist, but sometimes juniper occurs as more-or-less pure stands. We used 25 percent juniper composition as the threshold separating Juniper from Pinyon – Juniper (in other words, if a stand had 75 percent or greater basal area juniper in a stand comprised of both pinyon pine and juniper, it was called "Juniper"; whereas, if it had less than 75 percent juniper, it was called "PJ"). Analysis of seasonal spectral data indicated that many juniper stands were spectrally distinct from many of the PJ stands (fig. 2); however, significant spectral overlap existed between the two classes, as well. After decision tree classification, cross-validation accuracies indicated significant error in the classification of these two cover types (fig. 3). We believe that much of this error is attributable to the artificial boundaries imposed by the classification of a continuum.

- *Temporal correlation between field data and satellite imagery*: Disturbance such as that caused by fire, insects, or logging can alter the sites enough to cause the temporal mismatches between field data and satellite imagery that result in classification problems. For the prototype, we made use of a large volume of existing field data acquired from disparate sources (Caratti, Ch. 4), and much of the field information was acquired over a long period of time. Although information from many plots was relatively old (for example, field data acquired over a 10-year time period prior to imagery acquisition), we determined that many of these plots still contained information that was useful and relevant to the LANDFIRE Prototype. For example, plots located within reasonably intact and undisturbed forests or sagebrush lands, under normal circumstances, do not change much over a 10-year span. After completing the first prototype study in Utah, we recognized the importance of using a change-detection approach and employed such an approach in the northern Rockies prototype region to discard plot information derived from areas that changed between the times when the field information was obtained and when the imagery was acquired.

Geospatial data issues—Landsat imagery data from the MRLC 2001 consortium served as the primary source of spatial data for developing the vegetation and structure products (Homer and others 2004) (refer to

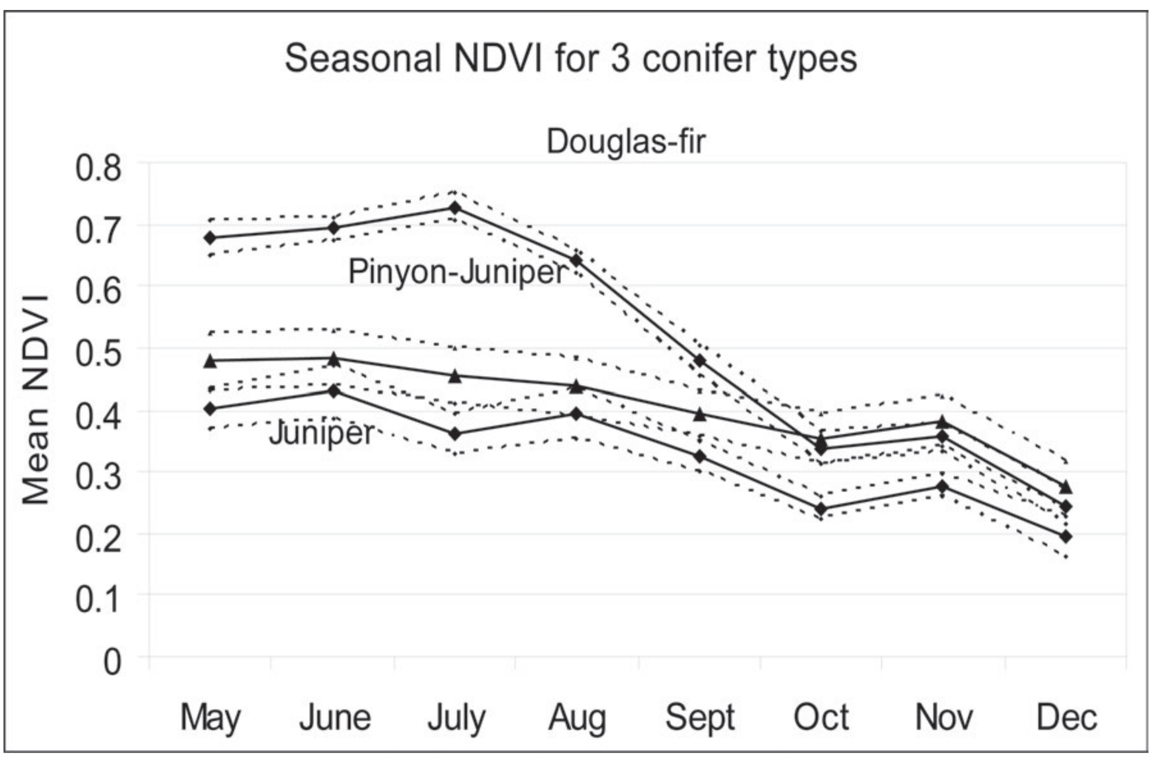

Figure 2—Seasonal normalized difference vegetation index (NDVI) spectral profiles for Douglas-fir, Pinyon – Juniper, and Juniper cover types.

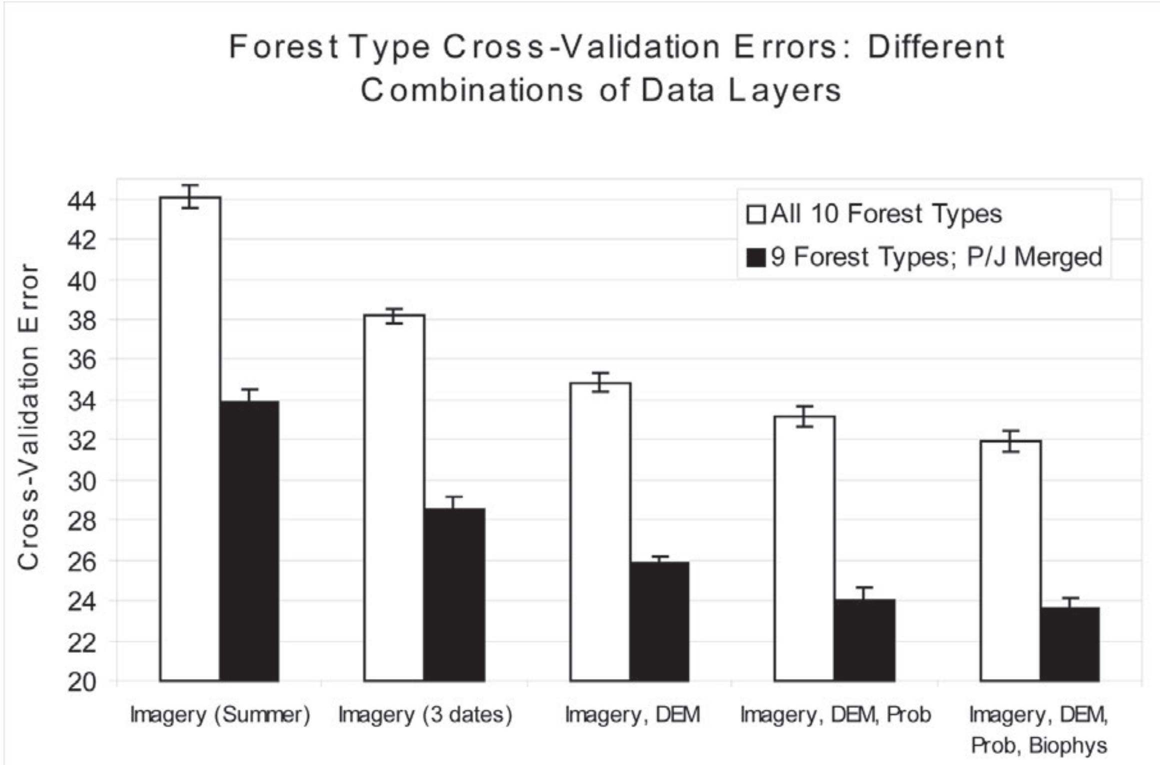

Figure 3—Cross-validation errors for forest types in the Zone 16 prototype study area as a function of different amounts of input source material. Black bars depict the effects of merging the Pinyon – Juniper and Juniper classes.

Zhu and others, Ch. 8 for further discussion regarding the imagery and ancillary data sources used for vegetation mapping in the LANDFIRE Prototype). In general, the images used for the prototype effort were the best data available during the LANDFIRE Prototype and represented three seasonal time periods (leaf-off spring, leaf-on summer, and leaf-on fall). Although the MRLC 2001 data used are of high quality, problems can arise when using any source of remotely sensed information. The foremost imagery-related problems affecting the LANDFIRE Prototype included atmospheric issues, disparate imagery acquisition dates, and geolocational problems.

- *Atmospheric issues*: Most of the acquired image scenes used in the prototype effort were of excellent quality. Even the best scenes, however, have occasional cloud and/or haze problems, which can either totally obstruct the view of portions of landscape or change the digital values enough to impact the mapping process. While not a large problem in the prototype areas, there were a few locations for which imagery quality was sub-par. These issues are inevitable and are likely to be a bigger problem in cloudier locations of the country such as the eastern United States and the upper Midwest.
- *Disparate imagery acquisition dates*: We attempted to use imagery from similar time periods as much as possible; however, due to cloud issues, optimal imagery data were not always available. Using scenes from different dates of the same year, such as using July and September data in the same "leaf-on" mosaic, resulted in problems resulting from phenological differences. Using scenes from different years, such as using one scene from 2002 and an adjacent scene from 2003, resulted in problems related to different weather patterns (for example, vegetation spectral response can be very different during wet versus dry years) and to occasional land cover changes that occurred between years. For the LANDFIRE Prototype, we attempted to minimize these problems through careful selection of scenes and use of spatial "date of acquisition" information in our decision tree and regression tree classifications.

- *Geolocational problems*: Images used in this investigation were processed using the National Landsat Archive Production System methods (USGS Landsat Website 2004). Data were corrected for terrain and projected to a standard projection (Albers Equal Area) using automated software processing. Individual pixel coordinate information was approximately 30 meters from actuality. Thus, even when field information had precise GPS coordinates, the field data were sometimes linked to the wrong pixel due to imagery registration errors. Because of technological, time, and budget constraints, we could not circumvent this problem. Registration methods needed to be consistent and automated to ensure that the process was feasible for application over the entire United States. We simply had to assume that the field data adequately characterized an area broader than the precise location of the plot and that the image pixel used was spectrally representative of its surrounding pixels. Note that in many cases, the quality-control checks performed on the field data mitigated some of these problems.

Ancillary data issues—Other sources of input information for the LANDFIRE Prototype included Digital Elevation Model (DEM) data and derivative products, 1990s NLCD land cover data (Vogelmann and others 2001), 2000s NLCD land cover data (Homer and others 2004), a suite of biophysical gradient data layers (Holsinger and others, Ch. 11; Keane and others 2001; Rollins and others 2004), and potential vegetation information (Frescino and Rollins, Ch. 7). Error terms are associated with each data type. While it is beyond the scope of this chapter to describe in detail all of the sources of errors associated with the many data layers, a few specific points should be made:

- Although not flawless, each data source used in the LANDFIRE Prototype represented the best available science and data quality.
- The source of the DEM data was the National Elevation Dataset (NED) (Gesch and others 2002). Although NED is an excellent source of digital elevation data, it came to our attention during the final stages of the prototype effort that another data source would have been more appropriate: the Elevation Derivatives for National Applications (EDNA) data set (http://edna.usgs.gov). The EDNA data represent a set of data layers derived from an earlier version of the NED. To create the EDNA data layers, the NED data were "smoothed" so that they would be better suited for hydrological modeling purposes. It should also be noted that, regardless of the source of the digital elevation model information, there are horizontal and vertical error terms associated with these data sets tracing back to the original source material. These digital elevation model data sets are regularly improved and updated.
- The 1990s and 2000s NLCD data sets were used for stratification purposes at various stages in the prototype effort, and both data sets have known error terms associated with them. See Yang and others (2001) and Homer and others (2004) for details regarding the accuracies of these products.

Accuracy of Thematic Maps

Cross-validation and points for independent validation—Accuracy assessment is an integral component of land cover mapping work. When a large number of field points are available, a reasonable alternative to generating traditional first-order accuracy estimates (see the above section *Accuracy Assessment Considerations for LANDFIRE*) is cross-validation. To create the LANDFIRE vegetation products, we employed decision tree analysis implemented within the See5 program (Quinlan 1993) using Landsat, DEM, slope, aspect, biophysical gradient, and potential vegetation data layers. The program enables cross-validation, which consists of repeated experiments in which a subset of the sample is used to train a classification model and an unseen subset is used to evaluate the model. In model runs for the prototype effort, we found that a five-fold cross-validation was appropriate. In each model run, the original field point data sets were divided into five subsets of equal size, and each subset was used to evaluate the algorithm trained using the remaining four subsets. Theoretically, this approach is not as thorough as a rigorous, statistically designed post-mapping field accuracy assessment campaign. It has been shown, however, that cross-validation can provide accuracy estimates comparable to these time-consuming and expensive methods (Huang and others 2003). See Frescino and Rollins, Ch. 7 and Zhu and others, Ch. 8 for actual accuracy results and cross-validation error matrices for the vegetation products derived for the LANDFIRE Prototype. For LANDFIRE National, we recommend reserving a set percentage of plots from the decision and regression tree analyses for independent accuracy assessment. See the *Recommendations for National Implementation* section below for details.

Field verification—Although it is not always feasible to conduct a detailed field verification and validation campaign, when possible, field visits at various stages of product development can be highly useful. Field visits, both during and after the product generation phase, provide the technical teams conducting the mapping work with a good basic understanding of the natural vegetation and ecology of the regions in which they are working. Further, field checks of particular sites to determine if they match the modeled results can be very instructive and useful for improving mapping accuracies. For the LANDFIRE Prototype, we made three separate field visits of approximately five days each. We traveled to the central Utah highlands region twice (once before mapping and once after the products were created), and we traveled once to the western Montana region (post-mapping). In all cases, images and/or maps were evaluated in the field, and actual plot measurements were made. Although not statistically rigorous, such efforts provided a better understanding of potential problem areas for future methods improvement. For example, an area of western hemlock was overestimated in the map products, and we were able to trace the overestimation back to problems in the original field sampling methods used to help generate the training data in the mapping process. Although no obvious solution to the problem was apparent, the case illustrates the importance of field visits in methods improvement. In another field activity, spectral measurements of shrub and herbaceous vegetation density were made by one team in the western Montana region to help refine shrub and herbaceous canopy cover methodology. This activity was undertaken in an attempt to improve canopy cover mapping and is being considered for the National Implementation of LANDFIRE.

Consistency checks with data from other sources—Related data sets, generated by other projects and for other applications, are often available and can be used for comparison purposes. The USGS Gap Analysis Program (GAP), for example, generates detailed vegetation maps for conservation management and planning (http://www.gap.uidaho.edu). We compared the GAP products created for the central Utah highlands prototype area with the cover type maps created for the LANDFIRE Prototype. The two sources of data compared reasonably well in some cases and less so in others (see figs. 4 and 5). It should be noted that the GAP products were created using different field databases than those used for the LANDFIRE Prototype. In addition, the vegetation map unit classification systems used were different, which limited the utility of direct, parallel comparison between the GAP products and LANDFIRE products. Although such comparisons may lack statistical rigor, they indicate where major qualitative similarities and differences exist between products and in turn may indicate which classes and regions are the most suspect. In addition, vegetation and structure products should be reviewed by regional experts whenever possible to determine whether noteworthy mapping problems exist and whether additional work is warranted. Such a review is recommended for national implementation of LANDFIRE.

Accuracy of Potential Vegetation Type and Canopy Fuel Maps

We generated potential vegetation type (PVT) data sets using decision tree software and cross-validation routines very similar to those used for generating vegetation maps. We also produced coinciding maps of confidence, which depict the relative prediction errors representing a spatial and visual representation of PVT map accuracy. See Frescino and Rollins, Ch. 7 for detailed descriptions and results of these activities. We estimated the accuracy of canopy fuel layers using regression tree procedures in which correlation coefficients were generated to measure the agreement between the predicted values and actual values. Additionally, we compared with predicted values a set of points randomly selected from the LANDFIRE reference database from each prototype zone. As in the case of PVT, we also produced coinciding maps of confidence. See Keane and others, Ch. 12 for a detailed description of canopy fuel accuracy.

Accuracy of Maps Based on Landscape Simulation Models

Accuracy evaluation of vegetation maps created from satellite imagery and ancillary data is straightforward and is based on a foundation of scientific literature (Foody 2002; Lunetta and Lyon 2004). In contrast, it is often conceptually very difficult to ascertain the quantitative accuracy of many of the products that are generated through complex modeling efforts, such as those employed to create the historical reference conditions for quantifying ecological departure in LANDFIRE. Moreover, it is difficult — if not impossible — to assign an absolute measure of accuracy to an ecological departure product because such a product represents deviation from conditions modeled under a variety of limitations in terms of baseline ecological data. Modeling assumptions, while based on the best available disturbance ecology science, may or may not be completely valid. Without the luxury of time-travel, it is very difficult to validate what the "normal" or historical vegetation condition actually was.

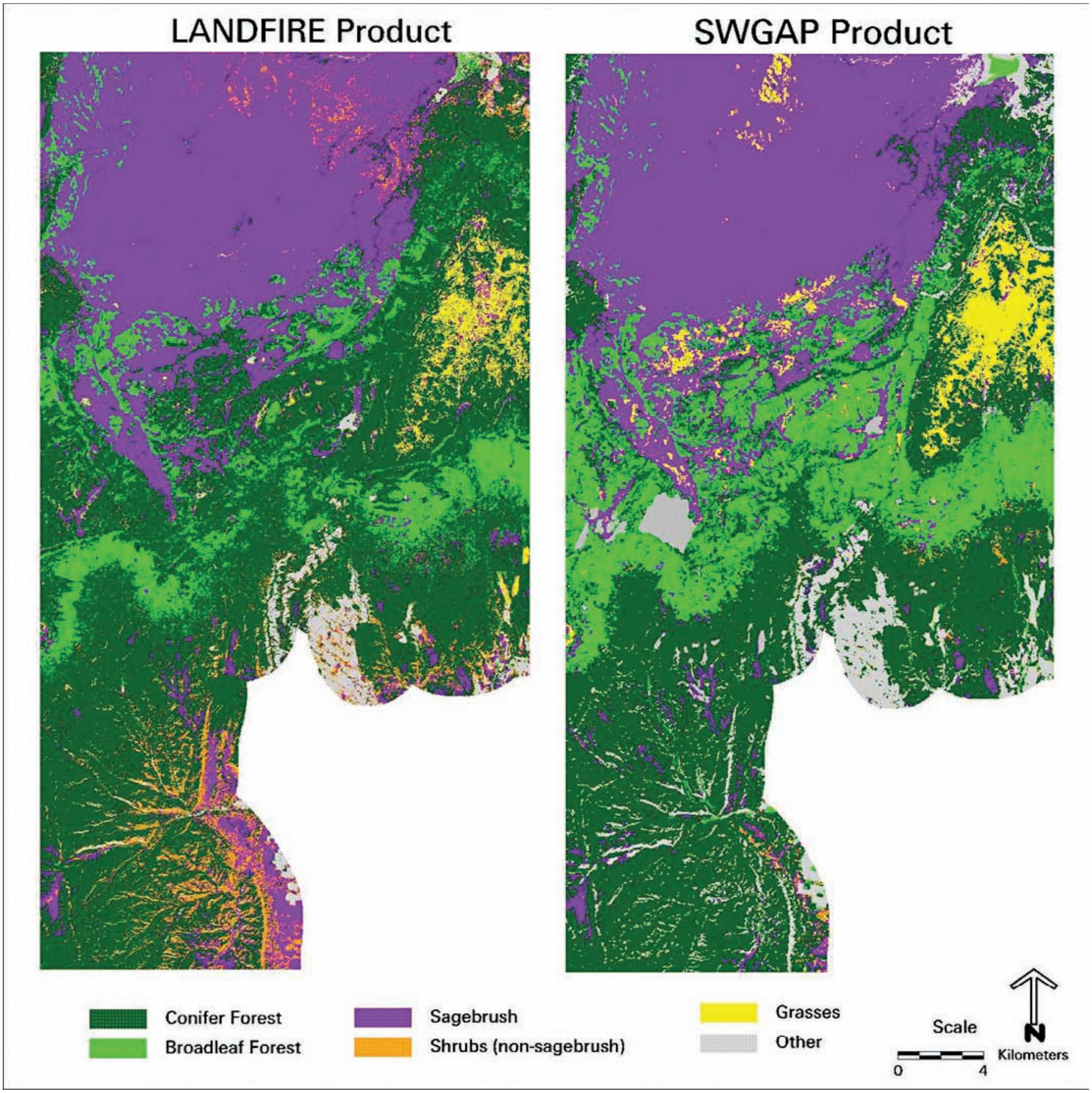

Figure 4—Comparison between a LANDFIRE vegetation type product and a product developed by the Southwest GAP Project in southern Utah. Multiple thematic classes have been combined to facilitate visual comparisons.

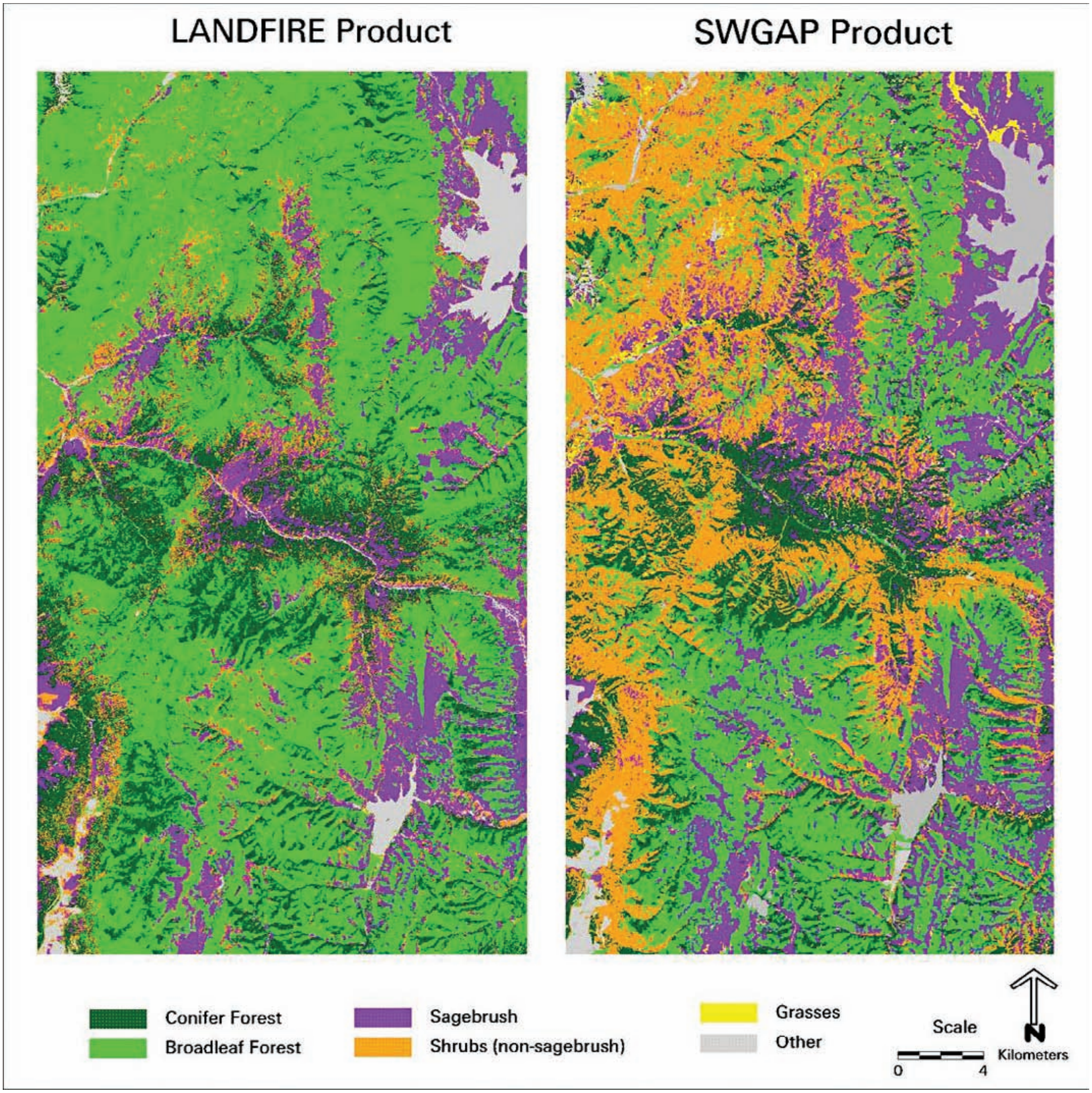

Figure 5—Additional comparison between a LANDFIRE vegetation type product and a product developed by the Southwest GAP Project in southern Utah. Multiple thematic classes have been combined to facilitate visual comparisons. Major differences between shrub and broadleaf forest classes can be traced back to differences in classification systems (Gambel oak and bigtooth maple were categorized as trees in the LANDFIRE map unit classification and as shrubs by GAP).

For accuracy assessment approaches used to evaluate LANDFIRE products based on landscape simulation models, see Pratt and others, Ch. 10 and Holsinger and others, Ch. 11. In addition, see the *Recommendations for National Implementation* section below for suggestions on improving the accuracy assessment of LANDFIRE products based on landscape simulation models.

Recommendations for National Implementation

Source Data

All source data need to be inspected carefully. This is especially true for field data and imagery, which form important foundations for much of the ensuing LANDFIRE tasks. As a matter of course, if field data used for training are inaccurate, then the resulting products will likely have lower levels of accuracy. Imagery quality can also greatly affect accuracy levels of derived products. Although optimal imagery data sets are not always available for a given location, there are usually several excellent options. It is important to ensure that the best possible imagery data sets are used. Below are some specific recommendations regarding the selection of source data.

Number of field plots—As general rule, the more field reference plots, the better. For each LANDFIRE National mapping zone, we anticipate using literally thousands of field plots in order to develop adequate characterizations. These must represent the entire range of conditions that occur throughout the mapping zones. For vegetation map unit classification development, for example, we have a target number of at least 100 plots per class. Fewer plots per class would diminish our confidence in our ability to map that class accurately and would likely result in the inadequate mapping of that particular feature. Rare classes (land cover features limited in occurrence across the landscape) are notoriously difficult to map accurately, largely because there are relatively few field plots representing these classes that can be used for training data. For national implementation of LANDFIRE, we recommend 1) generating vegetation products using all plots, 2) evaluating results, 3) determining which vegetation classes were represented by too few plots, and 4) re-running the map unit classification without these rare classes.

Field plot geolocational accuracy—Field plots with inaccurate coordinates have the potential to cause significant error in mapping results. We recommend that field plot locations be overlaid onto the imagery and that the plot locations be visually inspected to determine if attribute data for each plot are consistent with the imagery. Points located on roads or other locations clearly not characterized by the reference plot should be either omitted or shifted to the appropriate location.

Field data temporal issues—Much of the field information available for the national implementation of LANDFIRE is likely to have been acquired by various organizations over a relatively long period of time. As discussed above, inclusion of plots located in areas where the vegetation has changed between the time the field information was collected and when the imagery was acquired can cause significant mapping problems. The ideal situation is for field data and imagery to be acquired at approximately the same time, but this is impractical due to the large volume of field data necessary for product generation. One option is to discard plots with relatively old information (by imposing an arbitrary cutoff of five or more years); however, including as many plots as possible, even if some include older information, is preferable because even old plots can contain useful information. For this reason, for national implementation, we recommend using the change-detection approach developed for the western Montana prototype area. We recommend using normalized difference vegetation index (NDVI) change between 1990s and 2000s NLCD imagery to locate and isolate plots that have changed markedly over the last 10 years. If a plot is located within a region of high spectral change (based upon imagery analysis) and if the change appears to be related to a land cover change event (such as fire, logging, or insect disease) as opposed to a cloud or cloud shadow, the plot should be flagged and omitted from further analyses.

Imagery data—Imagery acquired by Landsat will likely continue to be the primary source of spatial data for developing vegetation and structure products for LANDFIRE National. The MRLC 2001 consortium, of which the LANDFIRE Project is a partner, is the best source for imagery in part because it is readily obtained and has been consistently pre-processed. Although this imagery represents the best data available, we do anticipate some issues that will need to be addressed. As with the prototype effort, we anticipate the primary imagery-related problems impacting LANDFIRE National to include atmospheric issues, disparate imagery acquisition dates, and geolocational issues (see above section *Geospatial data issues*). It is anticipated that haze and cloud problems will be especially prevalent in the eastern U.S., upper Midwest, and in the Pacific Northwest. Imagery differences related to phenological variables are also likely to impact mapping on a grander scale than was experienced in the prototype effort. When current MRLC data are deemed insufficient for LANDFIRE purposes (based upon visual inspection), additional scenes should be purchased and processed and incorporated into the mosaicking process.

Ancillary data—LANDFIRE will continue to use the best available source data for national implementation. One change that we recommend is using the EDNA data set (USGS EDNA website 2004) as the primary source of digital elevation data. These data are more refined than the data used in the prototype effort. The 1990s and 2000s NLCD data sets will continue to be used for stratification purposes at various stages of LANDFIRE National.

Accuracy of Output Products

Output product inspection—All LANDFIRE products must initially undergo an inspection phase during which the following question is asked: "Do these products make sense?" Although admittedly subjective, many errors will be caught early in the process through such inspections. If performed properly, such an initial evaluation provides a valuable safeguard that can save time and prevent the need to recreate the products.

Cross-validation and error matrices—As in the LANDFIRE Prototype, we recommend the use of cross-validation for approximating accuracies, especially for existing vegetation type and potential vegetation type. Correlation coefficients derived from regression tree analyses should be used when generating continuous variable data sets. Error matrices should be evaluated to facilitate better understanding of the strengths and weaknesses of the vegetation products. Regarding creation of the mapping models, we recommend using 5- or 10-fold cross-validation for each of the individual LANDFIRE mapping zones.

Points for independent validation—For national implementation of LANDFIRE, we recommend reserving a set percentage of plots from the decision tree and regression tree analyses solely for assessing accuracy. Note, however, that the field-referenced data used as input are collected from various projects and agencies, and thus the original source of field data cannot be considered a "random" sample of plots. Any sample of plots selected from a non-random set of points cannot be considered statistically random. Nonetheless, we have determined that withholding a limited number of points for validation purposes provides worthwhile accuracy information.

Nevertheless, we determined that it's better to produce a more accurate set of products with imperfect accuracy information than a less accurate set of products with better known accuracy estimates. We do not want to withhold plots that would best be used for model and product development. As a compromise, we recommend that two percent of the plots be withheld from the modeling activities. These plots will then be used to estimate accuracies for aggregations of LANDFIRE mapping zones or "superzones". We plan to merge data sets from three to four adjacent mapping zones and conduct validation activities for these regions. A target of at least 50 plots for each vegetation class per superzone provides useful information for estimating accuracies.

Stratification of accuracy assessment—In addition to providing general accuracy information at the superzone and individual mapping zone levels, we recommend providing more local estimates of accuracy nested within these other levels. This will be accomplished through spatial stratification of broad areas using biophysical gradient modeling information and other sources of spatial data and through thematic aggregation of similar vegetation types for localized regions. The process of stratifying mapping zones into zones based on the biophysical gradient layers developed for LANDFIRE (see Holsinger and others, Ch. 5) will be used as a basis to further our understanding of product errors, which in turn will enable refinement of future mapping procedures. This stratification process may facilitate the discrimination of different vegetation types with similar spectral signatures that occupy sites having very different environmental characteristics.

Field verification—As discussed above, we recommend conducting a modest level of field verification throughout LANDFIRE National. Field visits provide the technical teams with a basic understanding of the natural vegetation and ecology of the regions in which they are working, and field visits to particular sites serve to verify (or invalidate) the modeled results. Ideally, a field visit should take place at the beginning of each zone's mapping activities for familiarization purposes, and an additional field visit should occur near the end of the mapping process to verify and refine the mapping process.

Consistency checks with data from other sources—Whenever possible, products should be compared with existing independently produced data sets. In some cases, products unrelated to LANDFIRE have been generated for certain local areas, and these can be used to help assess accuracies of LANDFIRE products. Spatial and tabular data potentially provide good general information. In addition, we recommend that LANDFIRE support the generation of local validation data sets, where appropriate.

Accuracy of Maps Based on Landscape Simulation Models

As discussed above, it is generally very difficult to ascertain the quantitative accuracy of products generated through complex landscape modeling efforts. Even so, there are some approaches suitable for assessing the validity of certain LANDFIRE modeled products, such as modeled historical fire regimes.

Although as of yet there are no examples of complete data sets representing historical vegetation conditions for the entire United States at the spatial grain of the LANDFIRE products, there are local historical data sets that can be used to "spot check" the validity of the products generated. For instance, historical aerial photographs and field-based data sets may provide useful information for assessing modeled historical fire regime products. Although not a true quantitative analysis, comparisons with historical data will likely provide information regarding the validity of the products.

As described above, it is important that the outputs from complex modeling activities be scrutinized carefully and checked for obvious flaws or deviations from expected results. As obvious as this seems, we are aware of numerous investigations in which this avenue has been neglected and in which spatial products were produced but not carefully examined. Although this type of evaluation does not yield quantitative error estimates, it can provide valuable insight regarding probable accuracies.

Finally, users of the LANDFIRE data sets should recognize that the inputs to the modeling process, while not always perfect, reflect the most accurate and current information available and are based upon ecologically sound assumptions. For these reasons, LANDFIRE products represent state-of-the-art modeling and technology and thus a significant improvement over other current options.

Conclusion

There is no single recommended procedure for deriving accuracy estimates for LANDFIRE products. Because time- and cost-related constraints, it will not be possible to conduct traditional accuracy assessments for the LANDFIRE mapping region (the entire U.S.). Yet at the same time, we recognize that evaluations of quality and accuracy increase the credibility of the final LANDFIRE products. Additionally, we can learn much by assessing error terms in the products, and this knowledge can be invaluable for future mapping and modeling endeavors. We suggest conducting a suite of accuracy assessment methods for LANDFIRE National, ranging from mostly qualitative assessments (such as the critical inspection of products, consultation with regional experts, and comparisons with existing data sets) to more quantitative analyses (such as cross-validation assessments, traditional accuracy assessments at the superzone level, and select evaluations at local levels). These combined approaches will provide LANDFIRE data users with the accuracy information necessary to facilitate the appropriate use of the data.

For further project information, please visit the LANDFIRE website at www.landfire.gov.

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Chapter 8

Mapping Existing Vegetation Composition and Structure for the LANDFIRE Prototype Project

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Introduction

Overview

The Landscape Fire and Resource Management Planning Tools Prototype Project, or LANDFIRE Prototype Project, required the mapping of existing vegetation composition (cover type) and structural stages at a 30-m spatial resolution to provide baseline vegetation data for the development of wildland fuel maps and for comparison to simulated historical vegetation reference conditions to develop indices of ecological departure. For the LANDFIRE Prototype Project, research was conducted to develop a vegetation mapping methodology that could meet the following general requirements:

- Cover types (species composition) must be characterized at a scale suitable for subsequent mapping of wildland fuel and fire regime condition class (FRCC). The vegetation map unit classification used for mapping cover types must be based on existing national systems, such as the United States National Vegetation Classification System (NVCS; Grossman and others 1998). The alliance (a community with multiple dominant species) or association (a community with a single dominant species) levels of this standard must provide a clearly defined list of map units that can be used as a basis for mapping vegetation classes that are both scaleable and representative of suitable units for modeling historical fire regimes (see Long and others, Ch. 6 for details on the LANDFIRE vegetation map units).
- The mapping of existing vegetation structure must be based on the relative composition of forest, shrub, and herbaceous canopy cover and average forest, shrub, and herbaceous canopy height. Although structural stages are discrete map units describing unique combinations of canopy cover and canopy height by life form, mapping individual canopy cover and height variables as continuous variables is desired to provide additional information for mapping and modeling vegetation and flexibility for setting threshold values.

The task of mapping existing vegetation is interconnected with several major tasks performed in the LANDFIRE Prototype Project. The mapping of existing vegetation requires attribute tables developed from the LANDFIRE reference database (LFRDB) (Caratti, Ch. 4), satellite imagery acquisition and processing, the development of a vegetation map unit classification system (see Long and others, Ch. 6), the development of a biophysical settings stratification (Frescino and Rollins, Ch. 7), and the modeling of environmental gradient layers (Holsinger and others, Ch. 5). The design and testing of the vegetation mapping methodology have substantial influences on the outcome of the overall project because accuracies of subsequent products (such as maps of wildland fuel) are a function of the accuracy of mapped vegetation types and structure. In this chapter, we discuss the design features of the existing vegetation mapping component of LANDFIRE and present

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results of the prototype. We conclude the chapter with recommendations for the national implementation of a consistent vegetation mapping effort.

Technical Problems

Significant technical limitations exist regarding achieving desired accuracies in the mapping of vegetation types and structure variables over broad areas. In the LANDFIRE Prototype, accuracies were affected by the spatial resolution, geographic extent, and information content defined by the project's objectives. The U.S. Geological Survey (USGS) Gap Analysis Program demonstrated the feasibility of mapping many existing vegetation cover types at the regional scale; however, methodologies have been inconsistent between regions (Eve and Merchant 1998). In addition, the mapping of forest canopy cover using imagery and regression techniques has been routinely performed for the operational mapping of vegetation structure variables (Huang and others 2001). Beyond that, however, literature reporting success stories regarding the mapping of vegetation structure using imagery is scant.

We conducted a prototype study to test a methodology for mapping vegetation cover types and structure variables. The three central objectives of the study were to:

- test an adaptable approach for mapping existing vegetation types and canopy structure at a 30-m resolution for the entire prototype area;
- develop digital maps of existing vegetation types and structural stages and conduct an accuracy assessment for the vegetation deliverables; and
- document research findings and limitations to the consistent mapping of existing vegetation composition and structure.

Specifically, this study tested a vegetation mapping protocol that met the design criteria and guidelines of the LANDFIRE Project (Keane and Rollins, Ch. 3). Further, this study investigated the limitations of using data contained within the LANDFIRE reference database (Carrati, Ch. 4) as training data and the applicability of satellite and ancillary data in meeting LANDFIRE's objectives. For vegetation modeling and wildland fuel mapping, the LANDFIRE Prototype Project required a structural stage map classified on the basis of mapped canopy cover (closed and open) and canopy height (high and low) by forest, shrub, and herbaceous life forms. We attempted to generate continuous maps of vegetation height and cover to maximize the utility of these products in a variety of applications.

As described in Rollins and others (Ch. 2), the LANDFIRE Prototype Project was conducted in two mapping zones: Zone 16, located in the central highlands of Utah and covering approximately 4 million ha of forest ecosystems (57 percent of the total land cover) and 2.5 million ha of shrub and herbaceous ecosystems (35 percent of the total land cover); and Zone 19, located in the northern Rocky Mountains of western Montana and northern Idaho and covering approximately 5.4 million ha of forest ecosystems (47 percent of the total land cover) and 5 million ha of shrub and herbaceous ecosystems (44 percent of the total land cover).

Literature Review of Vegetation Mapping

Similar to other natural science problems, the regional-scale mapping of vegetation types and structure variables carries unique technical and organizational challenges (Gemmell 1995). Spatial variations of vegetation types and structure are generally not characterized by unique spectral signatures, as captured by conventional broadband optical sensors (Kalliola and Syrjanen 1991; Keane and others 2001). Although significant improvements can be made by using specialized sensors, such as hyperspectral spectrometer or canopy lidar, data from such sensors having desired spatial resolutions are not available at national or regional scales. The associated enormous data volumes and high costs (in time and labor) make these technologies impractical for large-area applications at the present time.

Various techniques exist for modeling and estimating vegetation type and canopy structure (particularly percent forest cover); these include physics-based canopy reflectance models, empirical models linking ground-referenced data to satellite imagery, spectral mixture analysis, neural networks, and direct measurement using lidar and interferometric synthetic aperture radar. Each of these approaches has limitations in large-area applications, such as those related to cost and consistency. However, recent applications using the classification and regression tree (CART) approach (Breiman and others 1984) have been found to overcome many such limitations, provided sufficient amounts of field and geospatial data are available. Recent studies (Friedl and others 2002; Huang and Townshend 2003; Mahesh and Mather 2003; Yang and others 2003) have demonstrated the utility of CART techniques in mapping land cover, estimating species distribution, modeling percent forest canopy cover, and computing imperviousness at a 30-m grid resolution for large areas and even for the United States. Although CART techniques require relatively little human decision-making during algorithm executions, it is important to note that, ultimately, the knowledge scientists have acquired through studying vegetation patterns and attributes enhances the development mapping models to produce the most accurate results possible. Computer classifiers, regardless of their sophistication, are no substitute for scientists' understanding of the patterns, attributes, and conditions of existing vegetation and associated ecological processes.

Environmental data layers (such as elevation) are important predictor variables for characterizing vegetation patterns and attributes and for stratifying the distribution of vegetation along environmental gradient lines (Balice and others 2000). The use of spectral bands in combination with topographic data (for example, digital elevation models (DEM), slope, and aspect) is common in many land cover and vegetation mapping applications. However, topographic data capture only a part of the overall environmental factors that determine the establishment, growth, distribution, and succession of plant species and associations. The incorporation of a more complete set of environmental gradient layers into the mapping of existing vegetation should lead to increased predictive power and thematic accuracy (Keane and others 2002; Rollins and others 2004). Keane and others (2002) discuss techniques for deriving an entire set of climate, soil, and ecological gradient layers using interpolated weather observations in conjunction with topographic and soil databases and also describe the advantages of using such biophysical gradients in combination with remote sensing and field data to map vegetation, wildland fuel, and general ecosystem conditions.

In addition to the development and use of gradient variables, Keane and others (2001, 2002), Keane and Rollins, Ch. 3, and Rollins and others (2004) also suggest an approach for developing site-specific biophysical settings maps by mapping stable, late-seral communities as a function of certain climate, topographic, soil, and ecological gradients. This mapped "potential" vegetation can be used as a stratification tool in mapping actual vegetation distribution by constraining the distribution of cover types to those geographic strata where growth of the cover types' dominant species is ecologically possible.

Methods

The LANDFIRE Prototype Project involved many sequential steps, intermediate products, and interdependent processes. Please see appendix 2-A in Rollins and others, Ch. 2 for a detailed outline of the procedures followed to create the entire suite of LANDFIRE Prototype products. This chapter focuses specifically on maps of vegetation composition and structure, which served as important precursors to maps of wildland fuel and ecological departure in the LANDFIRE Prototype Project. Figure 1 outlines the technical approach used in LANDFIRE Prototype vegetation mapping and illustrates the data flow between several technically challenging tasks. Details of these tasks are described below.

Satellite Data Acquisition and Processing

The LANDFIRE Project partnered with the Multi-Resolution Land Characterization (MRLC) Consortium (Homer and others 2004) to facilitate the acquisition and processing of Landsat imagery. The consortium has completed the acquisition and processing of a full set of Landsat imagery for the United States with a minimum of three cloud-cover dates (circa 2001) for each pixel corresponding to phenological cycles of leaf-on, leaf-off, and spring green-up. Huang and others (2002) describe the steps involved in processing the MRLC satellite imagery, including terrain-corrected geometric registration and radiometric calibration using at-satellite reflectance models, calculations of normalized difference of vegetation index (NDVI), and tasseled cap transformations. The MRLC Consortium-sponsored development of the National Land Cover Dataset (NLCD) includes general land cover map units such as forest, agriculture, water, and urban areas mapped at a 30-m resolution (Homer and others 2004). The acquisition and processing of satellite imagery and the mapping of NLCD land cover map units were conducted for mapping zones, which were loosely delineated along major ecological regions. The LANDFIRE central Utah highlands and northern Rockies prototype areas were examples of these MRLC map zones.

The LANDFIRE Prototype Project had access to the following data layers from the MRLC catalogue for the Utah and northern Rockies prototype areas: 10 spectral bands for each of the 3 Landsat seasonal acquisitions (6 original spectral bands excluding the thermal band, 3 tasseled cap transformation bands, and 1 NDVI band) and land cover classes mapped to Anderson's Level 1 land cover classification (Anderson and others 1976). Using these data as a starting point, we mapped forest, shrub, and herbaceous cover types and structure attributes. These maps formed the foundation for mapping wildland fuel and fire regime characteristics (Holsinger and others, Ch. 11; Keane and others Ch. 12).

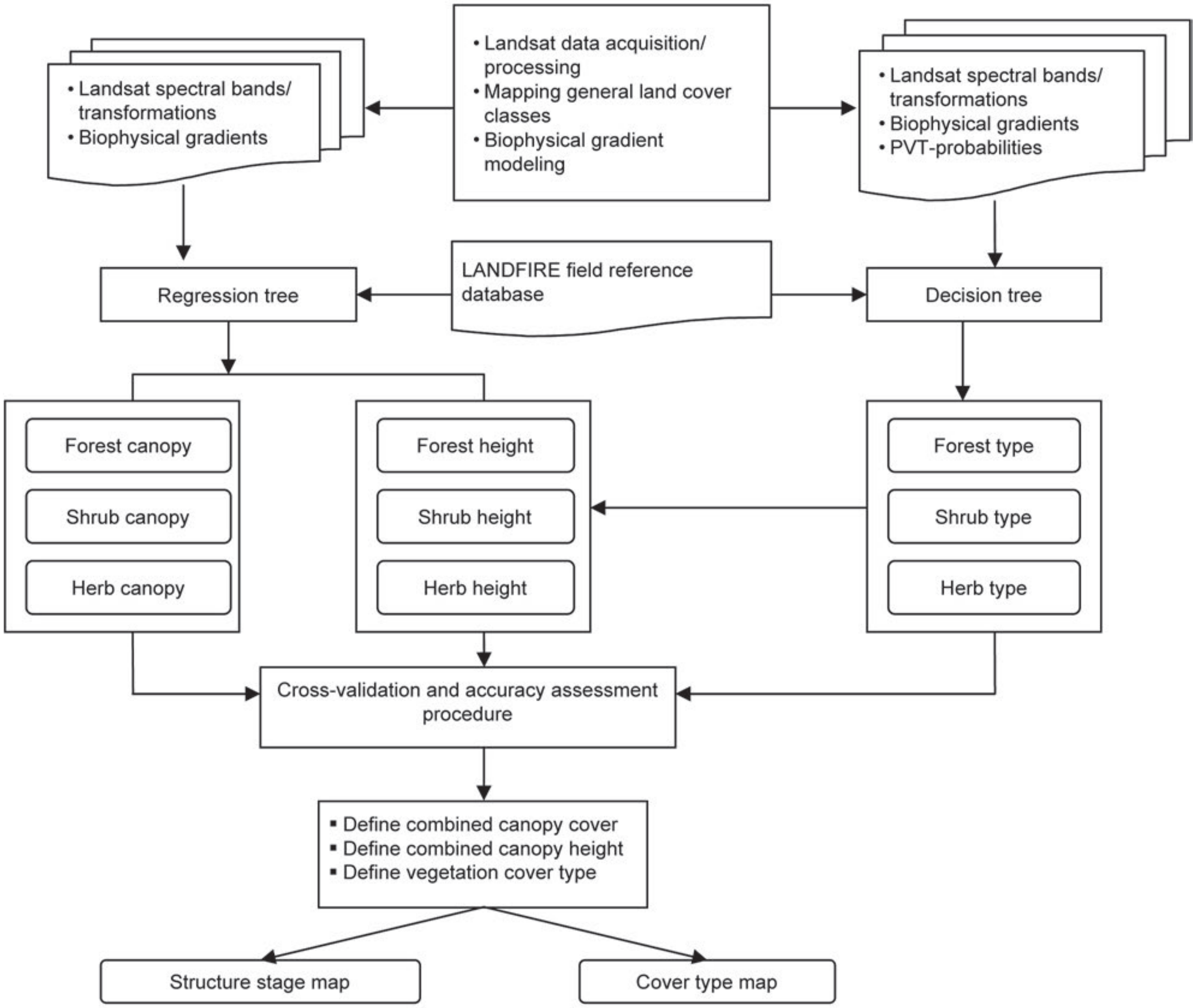

Figure 1—Flow diagram of the methodology used for mapping cover type and vegetation structure in the LANDFIRE Prototype Project.

Use of Biophysical Gradient Variables and Potential Vegetation Maps

In addition to the spectral predictor variables discussed above, the LANDFIRE existing vegetation mapping task incorporated two ancillary data sets that functioned differently in the mapping process. One was a suite of biophysical gradient layers developed as a set of intermediate LANDFIRE products with input from weather, topographic, and soil databases (Holsinger and others, Ch. 5: table 6). Table 1 lists the biophysical gradient variables used in the prototype for mapping existing vegetation; these represent a winnowed set of the entire suite of variables produced for the LANDFIRE Prototype. Biophysical gradients were used in the mapping process to provide a geographic context for the ecological processes that control establishment, growth, and distribution of vegetation communities.

The second data set was a potential vegetation type (PVT) map with attributes describing the probability of specific cover types existing in each PVT. This database was derived by calculating the distribution of cover types within individual PVTs by intersecting the plots contained in the LFRDB with the PVT map (Keane and Rollins, Ch. 3; Frescino and Rollins, Ch. 7). Conceptually, by using the PVT and cover type probability

Table 1—Biophysical and topographic layers used in the LANDFIRE vegetation mapping process.

Symbol	Description	Unit	Source data
SRAD	Daily solar radiation flux	$KW/m^2/Day$	Weather and topographic data
Tmin	Daily minimum temperature	C°	Weather and topographic data
Tmax	Daily maximum temperature	C°	Weather and topographic data
Tnight	Daily average nighttime temperature	C°	Weather and topographic data
Dday	Degree days	C°	Weather and topographic data
PPT	Daily precipitation	cm	Weather and topographic data
RH	Relative humidity	%	Weather and topographic data
PET	Potential evapotranspiration	kgH_2O/yr	Weather and topographic data
AET	Actual evapotranspiration	kgH_2O/yr	Weather, topographic, and soil data
GSWS	Growing season water stress	-Mpa	Weather, topographic, and soil data
PSI	Soil water potential	-Mpa	Weather, topographic, and soil data
KDBI	Keetch-Byram drought index	Index	Weather database
SWF	Soil water fraction	%	Weather, topographic, and soil data
Sdepth	Soil depth to bedrock	cm	Soil and topographic data
LAI	Potential leaf area index	Index	Landsat spectral data
DEM	Digital elevation model	m	National Elevation Database
Slope	Slope	%	National Elevation Database
Aspect	Aspect	Azimuth	National Elevation Database
POSIDX	Topographic position index	Index	National Elevation Database

information in the mapping of vegetation cover types, we implemented a stratification that constrained cover types to the geographic areas where cover types were ecologically possible. Sites (pixels) where certain cover types were not likely to occur would have low probabilities; therefore, these cover types were less likely to be predicted for these pixels. Each cover type was associated with a probability distribution map. The probability layers were implemented in the mapping process much in the same way as the biophysical gradient layers and satellite imagery.

Vegetation Map Unit Classification

Two different approaches were used in the development of the vegetation map unit classification systems for the prototype mapping zones. For the central Utah mapping zone, we formulated the map unit classificaton based on an overall understanding of the presence of vegetation alliances and associations (Long and others, Ch. 6). For the northern Rocky Mountains prototype area, we examined and summarized the LFRDB to form the basis for the vegetation map unit classification. Brohman and Bryant (2005) have described these approaches as the "top-down" and the "bottom-up" approaches, respectively. Long and others (Ch. 6) discuss the criteria and factors used in developing the LANDFIRE vegetation map unit classification systems, the lessons learned in applying them, and recommendations for a national approach to vegetation map unit development.

We were concerned with two technical issues when evaluating the map unit classifications of existing cover types for the prototype: 1) whether each cover type was sufficiently represented by an adequate number of field-referenced data from the LFRDB and, if not, how such "rare map units" should be treated and 2) whether some cover types (such as the Juniper cover type versus the Pinyon – Juniper cover type) would be floristically or ecologically difficult to separate in spectral, biophysical, and geographical domains. The technical issues were considered in the context of four guidelines defined at the beginning of the LANDFIRE Prototype Project: a map unit, whether it is a cover type or a fuel model, must be identifiable, scalable, mappable, and modelable (Keane and Rollins, Ch. 3). Because the prototype study areas were the first mapping zones to be mapped under the LANDFIRE design criteria and guidelines, we were unsure whether the map unit classification systems could perform consistently across different geographic areas.

Reference Data

Caratti (Ch. 4) describes in detail the compilation of the LFRDB for the prototype. The compilation of the LANDFIRE reference database relied on the coordination of three separate and independent efforts: 1) the cooperation and support from the U.S. Forest Service (USFS) Forest Inventory and Analysis (FIA) database collected nationwide on permanent inventory plots (Smith 2002); 2) the collection and processing of existing field data from all land management units such as Bureau of Land Management districts or national parks; and 3) the acquisition of new, supplementary field data from areas where there were no or not enough existing data (for example, various western rangelands in the United States do not currently have adequate field data collection programs).

Because the LFRDB was compiled from various sources collected for different purposes, information gleaned from the LFRDB was highly variable in terms of sampling design. The FIA data represented the most consistent information for forest cover types and canopy height. Rangeland field data usually contained cover type labels, but structure information was rare. In addition, reference data for mapping forest canopy cover were generated by calculating the number of forest cells within a 30-m cell using either high-resolution satellite data (spatial resolution of 1-m or better) or digital orthophotographs (Homer and others 2004).

Quality-control procedures were conducted as a part of the existing vegetation mapping process to detect problems and errors inherent in field-referenced data derived from disparate sources. We assumed that these procedures would identify most existing data problems but would not identify and eliminate all problems. These procedures were as follows:

Detecting outdated field data—Many field plots measured in years past were considered useful if the dominant species had not changed. A substantial number of plots, however, had undergone major disturbances such as fire or logging. We therefore computed the differences between the 1992 and 2001 Landsat NDVI values to flag field plots with conditions that had potentially changed during that 10-year period.

Detecting field data with erroneous geographic coordinates—We identified major geo-coding problems such as coordinates located on roads or located out of mapping areas. We visually examined plot locations overlaid with road networks and general land cover maps (such as NLCD maps).

Detecting field data with major coding errors—We detected such problems by overlaying field data on raw satellite imagery and by sorting variables according to major cover types. For example, if a field plot coded as *sagebrush* was located in the center of an otherwise intact *forest* polygon, or if a shrub plot had a height value taller than that of forest plots, such plots were flagged.

Reducing spatially clumped field plots—The LFRDB contains field data that come from different sources and are collected with different objectives, which occasionally results in spatially clumped plot information. In order to produce a spatially well-distributed and balanced data sample, we sub-sampled clumps of the available data to result in a more even distribution of field data.

The use of these quality-control procedures resulted in the exclusion of a number of available field plots from either the mapping or validation processes. This led to a total of 6,177 field plots (1,809 FIA forest plots and 4,368 non-FIA forest and rangeland plots) for Zone 16 and 7,735 field plots (1,993 FIA forest plots and 5,742 non-FIA forest and rangeland plots) for Zone 19 to be used for subsequent training or accuracy assessment. These numbers differ slightly from other applications of the LFRDB in LANDFRIE mapping because, based on objectives, each mapping effort implemented its own quality control procedure. Although all of the plots contained LANDFIRE cover type labels, only subsets of plots from the LFRDB had attributes of canopy height and canopy cover (table 2). In addition, ten percent of the field data points available for each of the cover type and structure mapping tasks were withheld from the mapping process for the purpose of accuracy assessment (Vogelmann and others, Ch. 13).

Mapping Algorithms

Classification and regression tree algorithms have demonstrated robust and consistent performance and advantages in integrating field data with geospatial data layers (Brown de Colstoun and others 2003; Friedl and Brodley 1997; Hansen and others 2000; Joy and others 2003; Moisen and others 2003, Moore and others 1991; Rollins and others 2004). Nonparametric CART approaches recursively divide feature space into many subsets in a hierarchical fashion to achieve the best overall model performance (lowest error and highest R^2, derived using a cross-validation technique). For this study, we adopted the classification tree algorithm to map vegetation types as discrete map units and the regression tree algorithm to map canopy cover and canopy height as continuous variables using two related

Table 2—Numbers of field reference plots in each mapping zone used in either mapping or accuracy assessment and corresponding to various map products. Forest canopy cover mapping relied on imagery of high spatial resolution instead of field reference plots.

	Mapping zone	Number of cover types	Cover type plots	Canopy cover plots	Canopy height plots
Forest	16	10	1,809	N/A	1,809
	19	14	1,993	N/A	1993
Shrub	16	14	1,595	2,120	1,698
	19	15	1,788	1,788	989
Herbaceous	16	7	300	2,263	1,311
	19	8	597	597	282

commercial applications: See5 (classification trees) and Cubist (regression trees) developed by Quinlan (1993). The mapping models were trained on the compiled data set of spectral bands and biophysical ancillary variables listed in table 1 and cover type and structure variables from the LFRDB.

Vegetation Database Development

Training vegetation mapping models—The creation of the CART-based algorithms for mapping existing vegetation involved several steps: 1) exploration of general data such as correlation analyses and plotting of cover types from the LFRDB against predictor layers, 2) iterations of CART algorithm runs to determine the adequacy of training data and other biophysical layers, 3) visual evaluation of classification and regression trees and final output maps, 4) generation of cross-validation statistics as an initial indicator of map accuracies, and 5) development of vegetation maps by applying the final mapping models. As mentioned above, we withheld data from 10 percent of available field reference plots for accuracy assessment and used the rest of the field plots for training the CART algorithms. We ran classification tree or regression tree classifiers, depending on whether the mapped theme was categorical or continuous, and generated 10-fold cross-validation statistics. Results of the cross-validation were used to determine the quality of training data and the performance of the predictor layers, but not to assess the final accuracy of resulting maps.

Determination of rare and similar map units—Although the LANDFIRE Prototype Project vegetation map unit classifications were developed to meet specific design criteria and guidelines (Keane and Rollins, Ch. 3; Long and others, Ch. 6), two technical questions arose during the mapping of existing vegetation: how to treat 1) rare cover types and 2) spectrally and biophysically similar cover types. We considered a cover type to be rare if it was supported with fewer than 30 reference plots, and those plots were not concentrated in one general location. We retained a rare map unit in the overall mapping process if the resulting spatial pattern made sense (such as when a riparian cover type followed river patterns) and if retaining the map unit did not result in a significant drop in accuracy. Otherwise, the rare map unit would be omitted. Additionally, we decided, based on differences in historical disturbance regimes, to keep cover types that were biophysically and spectrally similar (such as Pinyon – Juniper) separate, even though merging the cover types would significantly improve overall map accuracy.

Stratifications by life form—During the mapping of these vegetation attributes, the question arose as to whether the cover types and structural stages should be constrained by their respective forest, shrub, and herbaceous life forms; that is, we questioned whether a given pixel could be assigned more than one life form for cover type, height, and canopy designations. Multiple life form assignments provided flexibility for the characterization of wildland fuel. Such flexibility would also benefit other potential applications of LANDFIRE data, such as insect and disease or biomass studies. In the process of LANDFIRE vegetation mapping, we therefore modeled each pixel independently for each of the three life forms (forest, shrub, and herbaceous; fig. 1).

Product Validation Plan and Accuracy Assessment

The LANDFIRE accuracy assessment is described in detail in Vogelmann and others (Ch. 13). We tested the approach in which ten percent of the field data points available for cover type mapping were withheld from the mapping process for the purpose of accuracy assessment but found that the approach did not work well because of the uneven availability of field data in support of different cover types in the map unit classification. For several cover types in each of the mapping zones, the amount of data withheld in the 10 percent sample was too low to be statistically meaningful. As the result, we reported overall accuracies for cover types using the results of 10-fold cross-validations. For structure variables, we used a set of independent plots to assess statistical accuracy using regression techniques. This afforded us the opportunity to examine the behaviors of mapping structure variables versus those of categorical variables. Forest canopy cover, mapped with fine-resolution imagery as training data, would be assessed with both a sample of withheld reference points generated from the fine-resolution imagery as well as field estimates obtained from the use of digital cameras equipped with fisheye lenses.

Results

Maps of Cover Type and Structural Stage

We applied the vegetation mapping approach described above to the central Utah and northern Rockies prototype areas. Spectral imagery, biophysical gradients, PVTs, and probabilities were used together with field plot data to produce maps of forest, shrub, and herbaceous cover types, as well as canopy cover and canopy height by life form.

Accuracy of LANDFIRE Prototype Vegetation Mapping

We reported accuracy assessments using a cross-validation approach for cover types by life form (table 3) and by withholding field data for the structure variables by life form (table 4). For cover types, only overall accuracies were reported. For structural stages, R^2 values were variable and ranged from relatively consistent (for forest canopy cover and height) to relatively inconsistent (for shrub and herbaceous canopy cover and height). This variability indicates that forest structure may be mapped reasonably as a continuous variable, whereas consistency and accuracy would be questionable when mapping shrub and herbaceous structure as continuous variables. However, when evaluated as two-class variables (either as closed and open canopy cover or high and low canopy height), results showed that the same shrub and herbaceous structure can perform as consistently and accurately as categorical variables.

Discussion

Analysis of Mapping Consistency for Vegetation Types and Structure

In general, we found that the approach described above for mapping existing vegetation characteristics effectively met LANDFIRE requirements, which was a difficult objective to achieve due to the large number of vegetation map units, reliance on existing field-referenced data, the task of characterizing vegetation structure, and the requirement for a nationally consistent methodology. For the moderately detailed vegetation map unit classification, mapping accuracies of 60 percent or better were achieved at a 30-m spatial resolution.

We explored the mapping of more than two map units for structure variables. For example, we mapped herbaceous height to three map units (0 to 0.5 m, >0.5 to 1 m, and >1 m), shrub height to four map units (0 to 0.5 m, >0.5 to 1 m, >1 to 3 m, and >3 m), and forest height to four map units (0 to 5 m, >5 to 10 m, >10 to 25 m, and >25 m). The tests yielded independent overall accuracies of 73, 61, and 82 percent for herbaceous, shrub, and forest height, respectively. From these results, we concluded that grouping continuous values of the structure variables into several discrete map units would be an acceptable and rational alternative methodology for national implementation of the LANDFIRE methods. Use of this alternative methodology would require the development of a consistent national structural stage map unit classification.

Table 3—Cross validations (10 percent withheld, ten-fold repetitions) conducted separately by mapping zones and by forest, shrub, and herbaceous life forms.

Life form	Mapping zone	Number of classes	Cross validation
Forest	16	10	67
	19	14	64
Shrub	16	14	62
	19	15	68
Herbaceous	16	7	60
	19	8	56

Table 4—Accuracy assessments conducted separately for two structure variables by life forms and map zones. Overall accuracy (OA) was obtained by using holdout withheld field plots (n) that were set aside based on quality and distribution of the total available field plot data (N). Structure variables are treated as both continuous variables measured with the R2 statistic and two-class categorical variables for overall accuracy (OA). The Ttwo canopy cover classes of canopy cover are closed (≥40%) and open (<40%); for canopy height they classes are high (≥10m, 1m, 0.24m) and low (<10m, 1m, 0.24m) for forest, shrub, and herbaceous life forms, respectively.

	Map	Canopy cover			Canopy height		
Life form	zone	n/N	R^2	Overall accuracy	n/N	R^2	Overall accuracy
Forest	16	1,272/20,000	0.78	0.92	220/2204	0.58	0.88
	19	1,200/20,000	0.88	0.89	127/5,541	0.56	0.78
Shrub	16	125/1,253	0.41	0.74	107/1,073	0.36	0.85
	19	119/1,788	0.59	0.79	81/989	0.65	0.86
Herbaceous	16	18/182	0.37	0.71	15/280	0.04	0.86
	19	126/597	0.58	0.69	75/182	0.63	0.70

Consistency in field sampling and data collection affects the consistency of mapping vegetation characteristics. Of the three types of reference data used in mapping existing vegetation, cover type and canopy height values can generally be identified or measured consistently in the field. Canopy cover, on the other hand, can be difficult to measure in the field. This issue does not affect the measurement of forest canopy cover values because training data are derived from high-resolution (1 m or better) imagery by calculating numbers of high-resolution forest pixels within each 30-m Landsat pixel. The use of inconsistently estimated canopy cover values as training data, however, can potentially affect the mapping of shrub and herbaceous canopy percent cover (as happened during the prototype). Shrub and herbaceous canopy results from the two prototype mapping zones were reasonable (table 3), but difficulties in consistently estimating canopy cover in the field indicated that we needed to further research new or alternative methods for mapping shrub and herbaceous canopy cover.

The results of this study may be attributed, in part, to the use of ecologically significant ancillary data layers, which accounts for a moderate but nonetheless significant increase in accuracy (ranging from 1 to 9 percent). The development of biophysical gradient layers and PVT probabilities follows a standardized process for all mapping zones. However, for any given area, satellite reflectance can vary significantly for the same cover type with different canopy cover percentages (either due to land management practices or regeneration stages) or appear similar for different vegetation types or different structural stages during certain seasonal periods. Different cover types or structural stages, however, should respond consistently to the effects of biophysical gradient variables such as soil depth or potential evapotranspiration (PET); this addition of information from the biophysical gradient variables increases the likelihood that these map units will be discriminated by mapping algorithms. For example, one might expect Engelmann spruce (*picea engelmannii*) to grow in relatively deep soil on cool, north-facing sites with low PET, regardless of whether it is found in Zone 16 or Zone 19. Therefore, the incorporation of biophysical and PVT data in the mapping process should contribute to enhanced consistency and thematic accuracy in mapped existing vegetation across the United States.

Even though the existing vegetation maps shown in figures 2 and 3 characterize the vegetation composition of all life forms, it should be noted that each life form was mapped independently, by design, for cover type and structure. Modeling life forms independently preserves the possibility of more than one mapped life form per pixel (in other words, allows for probabilities of multiple canopy layers within a pixel) to improve fuel mapping and enhance the range of the data's ecological applications. However, mapping approaches should be carefully considered when comparing or merging these separate data sets. For example, a final map of cover types may look different depending on the order of precedence between forest, shrub, and herbaceous cover and the threshold values used in defining the life forms (for example, a pixel with 10 percent or greater forest canopy cover may be considered as forested land). It is important that precedence and thresholds be applied uniformly between mapping zones for consistency.

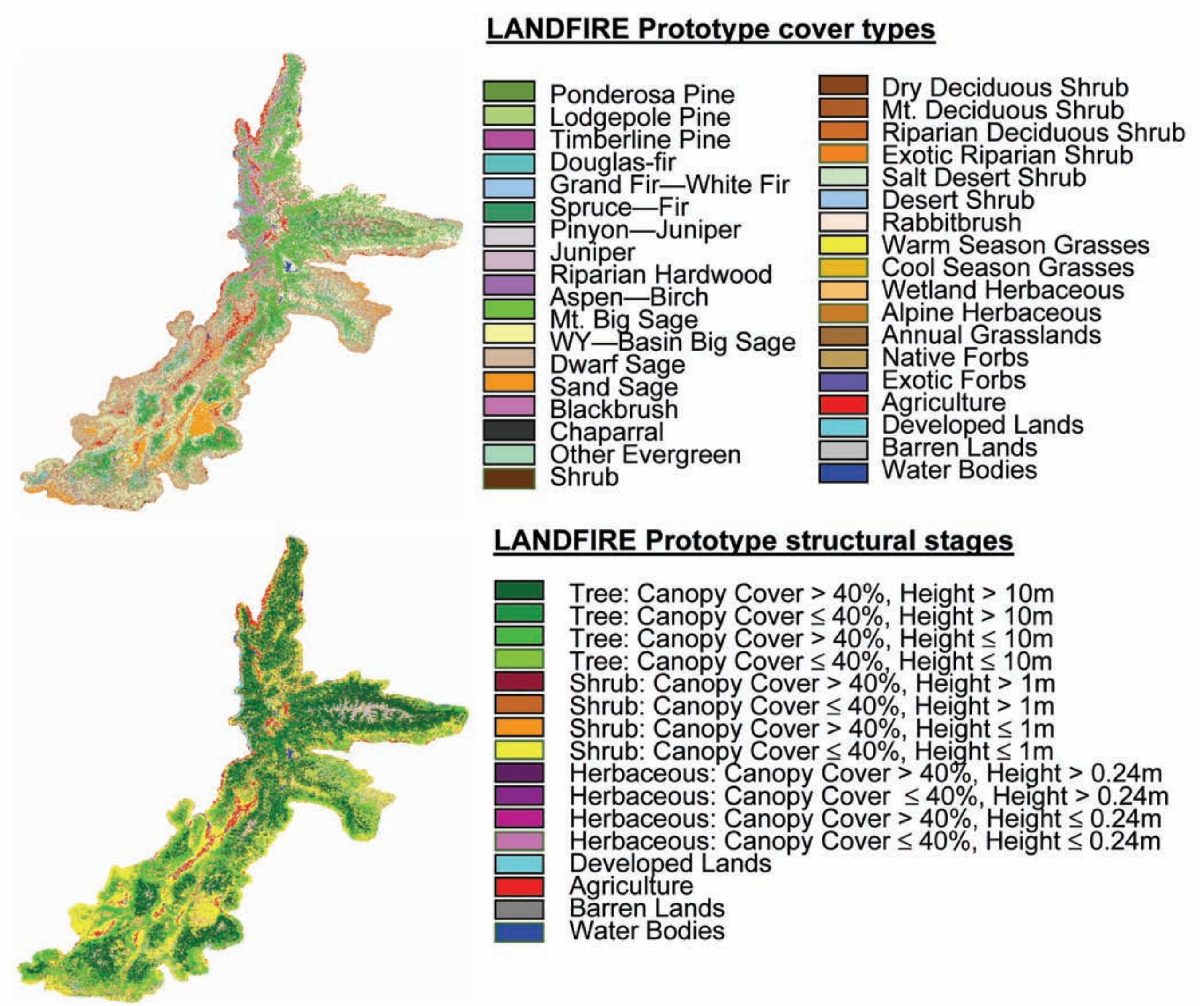

Figure 2—LANDFIRE Prototype cover type (top) and structural stage (bottom) maps for Zone 16. The cover type map is compiled from separate forest, shrub, and herbaceous cover type maps, whereas the structural stage map is grouped from continuous maps of height and cover for display purposes.

Factors that Affect Mapping Accuracies

Several factors should be considered when examining the accuracy estimates for maps of cover types and structure. First, the mapping and accuracy assessment of cover type and structure variables by life form were conducted based on field-referenced databases of different sizes and data collected throughout the study areas using a variety of sampling strategies. As would be expected, vegetation mapping was sensitive to the availability of field data. Test results showed that the number of field-referenced plots used for mapping and accuracy assessment affected not only the level but also the consistency of mapping accuracies, with fewer plots related to greater variability in accuracy estimates and more plots to more robust accuracy estimates (fig. 4). Data for herbaceous vegetation were limited in availability relative to the overall size of the field-referenced data set and hence affected herbaceous mapping accuracy. To improve uncertainties related to shrub and herbaceous cover and height, we determined that these variables should be mapped as categorical map units.

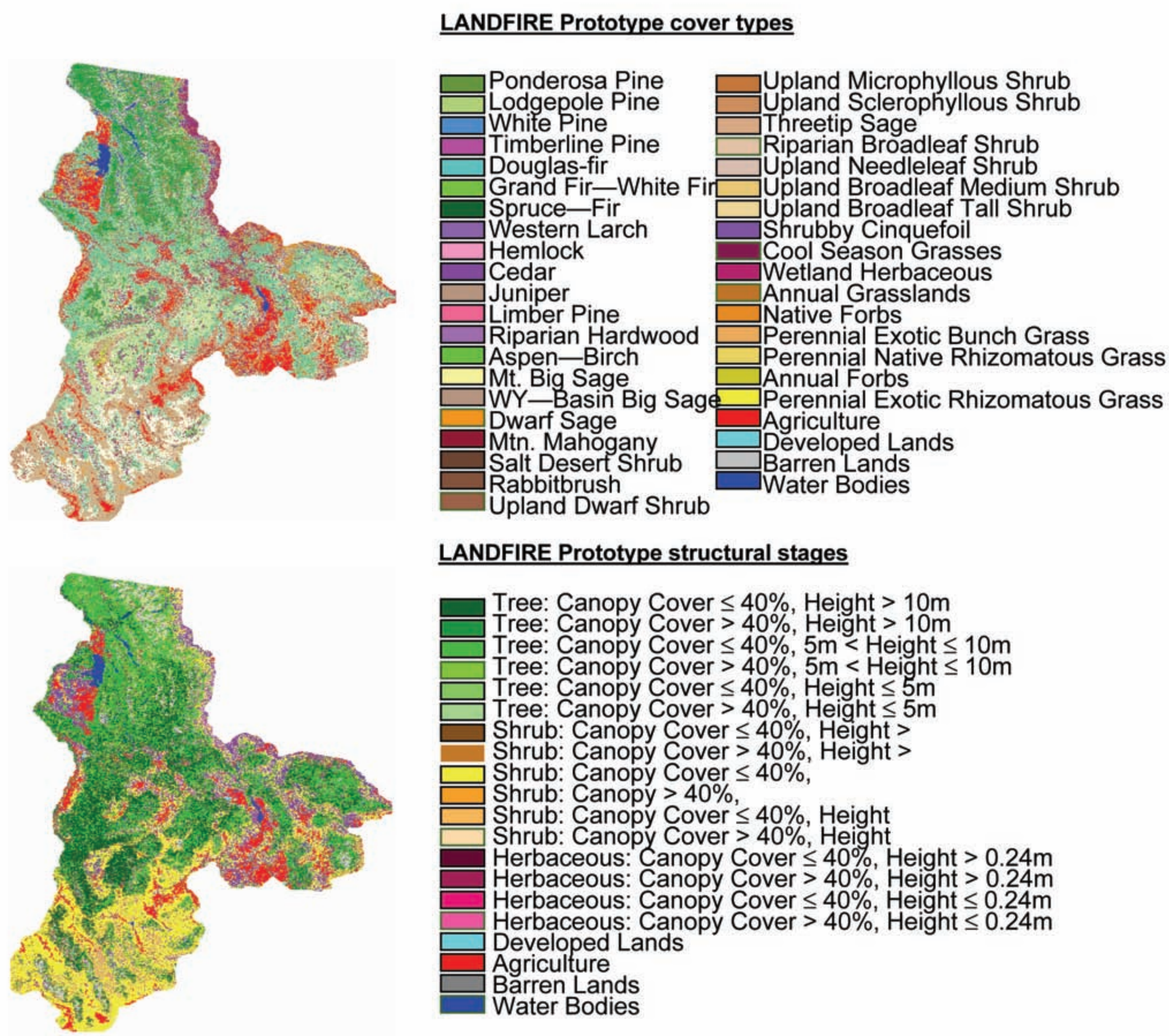

Figure 3—LANDFIRE Prototype cover type (top) and structural stage (bottom) maps for Zone 19. The cover type map is compiled from separate forest, shrub, and herbaceous cover type maps, whereas the structural stage map is grouped from continuous maps of height and cover for display purposes.

Second, field-referenced data, with which mapping models were trained and accuracy assessed, were collected from different sources, for different objectives, and with different techniques. Even though these plot data were quality-screened and standardized through an extensive effort (Caratti and others, Ch. 4), it was inevitable that the differences and errors in field data carried over into map quality and accuracy assessment. For example, certain reference data for forest canopy cover were derived using digital ortho-photographs, viewing forest cover synoptically from above the canopy. On the other hand, field estimates for shrub and herbaceous canopy cover were made using visual estimation from close-range, oblique positions that limited objectivity and consistency. We did not experience these problems when determining forest, shrub, and herbaceous *height*, which was usually directly measured and had a high degree of user-confidence.

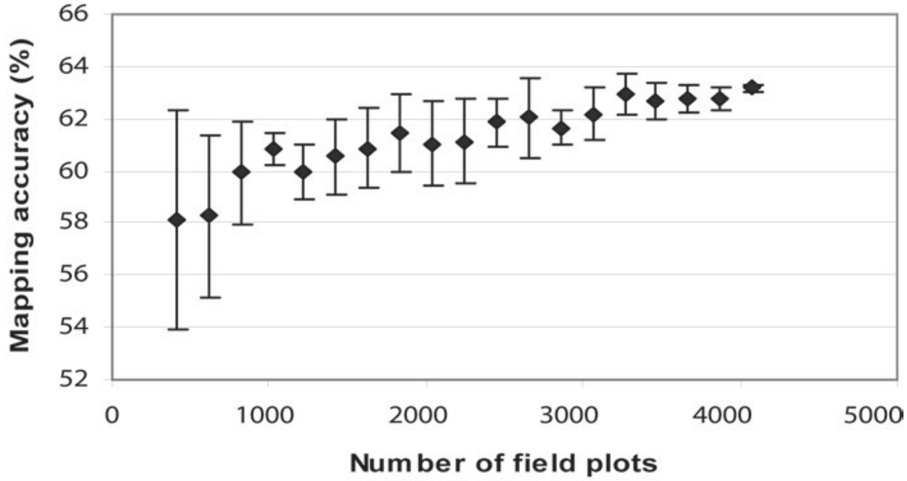

Figure 4—Cross-validation accuracy estimates obtained for the mapping of forest cover types as a function of the number of forest field plots. More plots contributed to better accuracy and consistency (smaller standard deviation) to a certain point, after which the relationship became flat.

Third, as discussed above, rare map units and ecologically and biophysically similar map units affected mapping accuracies. For example, if the Juniper cover type was merged with the combined Pinyon – Juniper cover type, forest cover type accuracy increased by more than 10 percent. The rationale for keeping such similar cover types separate is that, even though they occupy similar ecological niches and have similar site characteristics, separating them increases the utility of the LANDFIRE wildland fuel and fire regime products.

Utility of Biophysical Gradient Data for Vegetation Mapping

Although the use of DEM data for improving mapping results has been widely documented, the effects of a whole host of biophysical gradient layers and PVT-probability data layers is largely untested at the scale and scope of this study. These data layers provide information that supplements satellite imagery. Plant distribution patterns and conditions are strongly linked to a multitude of environmental factors (for example, temperature, soil, weather patterns, day length, soil properties, and rainfall), and the accurate characterization of these variables should, at least in theory, improve mapping results. In addition, spatial information that indicates where particular vegetation types can and cannot exist across a wide region (that is, PVT-probability data layers) should be similarly useful. Figure 5 compares cross-validation results using mapping models with and without the additional biophysical gradients listed in table 1 and using PVT-probabilities as predictor variables. Figure 6 displays mean and standard deviation values of a subset of the biophysical variables intersected with vegetation cover types from field plot data collected in the central Utah prototype area. These figures show that the incorporation of certain biophysical gradients and PVT-probabilities in mapping models contributes to increased mapping accuracy and consistency. These results are consistent with the findings of Keane and others (2002) and Rollins and others (2004).

Vegetation Patterns in Areas of Major Disturbances

Wildfires, insect and disease outbreaks, and forest clear cuts are some of the major disturbances to ecosystems captured by the satellite sensor in terms of their spectral properties. How well did our mapping capture and reflect these changes in vegetation conditions? We evaluated our mapping methods' effectiveness in this regard by looking at known areas of wildland fire, bark beetle infestation, and clear-cuts in the prototype mapping zones.

We evaluated two wildland fires areas that burned in Bryce (summer 2001) and Zion (fall 2001) national parks to determine what differences might exist between pre-fire and post-fire vegetation maps when mapped with the same pre-fire models. Pre- and post-fire map comparisons showed distinct differences between both

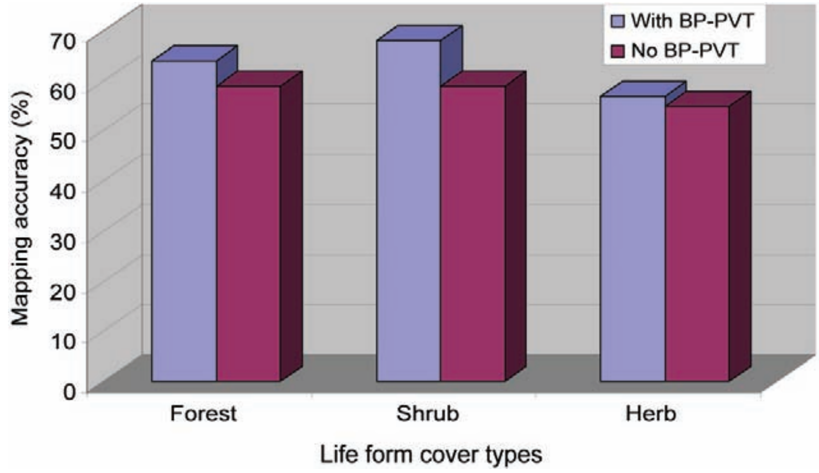

Figure 5—Cross-validation accuracy estimates obtained in the Zone 16 prototype area, by life form, with and without the 15 biophysical gradients and PVT-probabilities in the mapping models. An average of 8 percent increase in cross-validation accuracy was obtained by incorporating the selected biophysical gradients and PVT-probabilities that together describe the habitats of the cover types to be mapped.

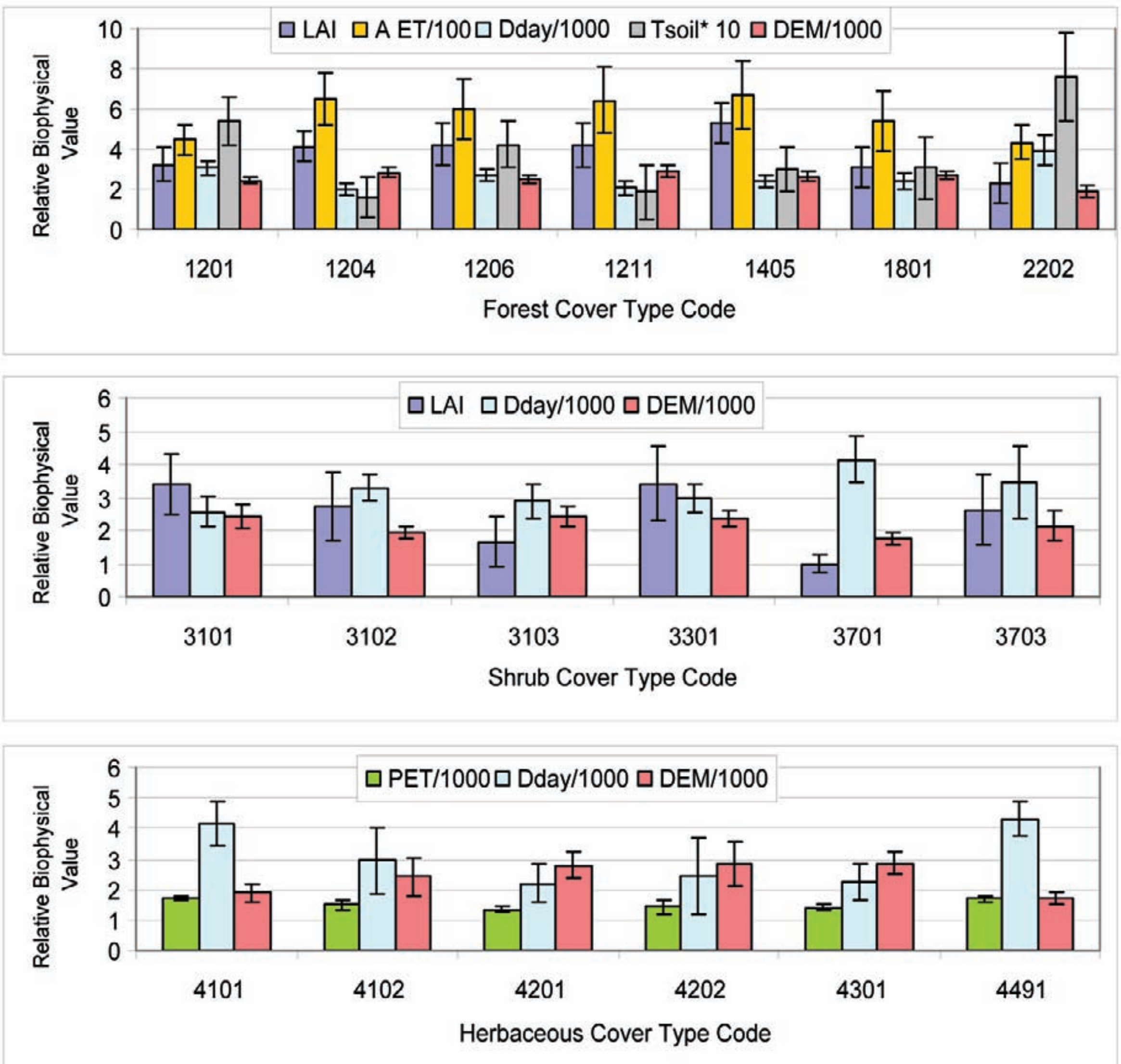

Figure 6—Mean and standard deviation values of selected biophysical variables found effective in mapping cover type against various forest (top), shrub (middle), and herbaceous (bottom) cover types of field-reference data. Most of the biophysical variables were divided or multiplied by a constant for display purposes. Refer to table 1 for definitions and descriptions of the biophysical variables. Refer to Long and others, Ch. 6 appendix 6-A for vegetation cover type coding protocol.

vegetation cover types and structural stages. The Bryce fire, a prescribed fire, showed general shifts from forest to shrub map units and, regarding structure, showed a shift toward increased low-height shrubs. The Zion fire, a wildfire, revealed a shift from predominately deciduous forest types to low shrubs.

Using bark beetle survey data obtained from the Dixie National Forest, we conducted simple zonal statistical analyses. Results indicated that the mapped species composition corresponded fairly well to that of those species identified in the survey data for the years 1998-2000. (Note that the level of actual disturbance varied within the survey data and was not differentiated in this study.) Structure information was not available in the survey data, but mapped structure data indicated that most bark beetle infestations occurred in areas identified as high forest cover (greater than 40 percent canopy cover) and height (greater than 10 m), indicating old-growth forest.

Similarly, we compared clear-cut areas, identified using modeling and masking methods, with mapped vegetation cover type and structure variables. Shrubs and a high percentage of grasses were dominant in clear-cut areas. Structural stages indicated a trend from forests with high canopy cover and canopy height to a high percentage of low cover (less than 40 percent), low height (less than 1 m) shrubs. Herbaceous cover was identified as being high cover (greater than 40 percent) with mixed heights.

Field Data Quality and Quantity Requirements

The acquisition of field-referenced data posed a significant challenge to the LANDFIRE Prototype effort, both logistically and technically. Caratti (Ch. 4) describes the logistical efforts and complications associated with conducting a national field data campaign. Specifically, technical challenges encountered during the mapping process, such as uneven amounts and disparate quality of field data used to meet various vegetation mapping objectives, were tied to the fact that the LFRDB was based on data from varying sources and collected with different objectives. As discussed above, such issues necessitated the careful implementation of a quality-control and quality-assurance (QA/QC) process prior to the training of the mapping algorithms for existing vegetation types and structure. “Lessons learned” from the QA/QC process follow:

- Accuracy and consistency are a function of the amount of available field-referenced data. Greater amounts of field-referenced data contribute to enhanced confidence in mapping accuracy (fig. 4), whereas limited field-referenced data are correlated to reduced confidence in mapping accuracies of affected cover types.
- The use of data from different sources requires that special attention be given to those cover type map units that are not supported with sufficient numbers of field plots. Both prototype mapping zones had map units with only a few field reference data points for training. As discussed above, the question of how to define and treat rare map units arose during the prototype, and we defined rare map units as those having less than 30 field reference plots scattered spatially within a mapping zone. Options for the treatment of these rare map units included keeping the map units in maps, omitting them, or omitting them and then “burning” the few field plots to the map in a post-process and merging them with floristically similar cover types. For the prototype, we chose to retain the rare map units in the models and resulting map products to inform the development of the LANDFIRE vegetation map unit classification system. For national implementation, rare map units that cannot be supported with a sufficient number of field plots will not attain target-level accuracies. We recommend omitting such map units from the mapping of existing vegetation cover types.
- Spatial distribution and a valid probability-based sampling design increase the consistency and accuracy of the map products. Compared with field-referenced data from various agency sources, the use of FIA forest inventory plots for mapping forest cover types and structure produced more consistent and accurate mapping results because the sampling design for FIA data produced training data that were spatially well-distributed across the landscape. Further, FIA data required very little additional processing time and were easy to use; in contrast, non-FIA field data required extensive processing time, related to QA/QC and re-selecting/re-sampling, to derive suitable data sets (in terms of spatial distribution and data quality) from available data points. For example, in Zone 19, a Bureau of Land Management study produced more than 4,800 field plots, mostly describing sagebrush, Douglas-fir, and lodgepole pine vegetation communities, in a relatively small area of approximately 1,152 km^2, near Salmon, ID. Spatially, this data set equated to approximately one plot for every 24 ha, versus a mapping zone average of one plot for every 835 ha. The inclusion of this data set in the

training process overwhelmed the mapping models and overrode areas with sparse plot coverage of different cover types. We therefore determined that the application of locally limited or concentrated data collected using various sampling designs to an entire mapping zone could have adverse effects on the accuracy of final products. For this reason, forest mapping in LANDFIRE National should employ FIA data exclusively. Rangeland mapping in LANDFIRE National, however, will require extensive QA/QC processing steps to transform available field-referenced data to a more suitable data set.

- As noted above, the following critical steps should be taken prior to the development of the mapping models: 1) examine field-referenced data, 2) conduct QA/QC procedures to detect spatial errors as well as information content-related errors, 3) correct these errors if necessary, and 4) derive a final, refined, error-free data set for training and accuracy assessment. This is a time-consuming yet necessary process that will contribute to increased consistency and confidence of map products.

Effects of the Vegetation Map Unit Classification System

Determining accuracy objectives and the appropriate extent of mapping areas are among the factors that need to be considered when defining a workable national vegetation map unit classification system. If floristically or ecologically overlapping cover types (such as Juniper and Pinyon -- Juniper or Upland Microphyllous and Upland Sclerophyllous) are to be mapped for LANDFIRE National, then guidelines must be developed for defining how the mapping accuracy of such overlapping map units is to be assessed.

Next, although our use of the NVCS was a reasonable starting point for vegetation map unit classification and the approach worked fine for each individual mapping zone, vegetation cover types were not always comparable between the two prototype mapping areas, however, as is evidenced by the legends in figures 2 and 3. As a result, accuracy estimates for the two prototype mapping zones could not be compared in a straightforward fashion, particularly for shrub cover types.

As discussed above, another challenge encountered during the application of the two vegetation map unit classification approaches (as discussed above in the *Vegetation Map Unit Classification* section) was answering the question of how to treat rare map units. There were no guidelines for consistently defining and treating rare map units. Moreover, there was no answer as to whether dropping rare map units, instead of using the alternative options discussed above, might affect the utility of LANDFIRE vegetation maps in other future natural resource management projects.

Recommendations for National Implementation

Because of the size and complexity of this research effort, many questions concerning LANDFIRE's national implementation are as of yet unanswered. The field data compilation effort will be an expensive and time consuming task, and a pressing need exists regarding the study of links between mapping performance, resource expenditure, and methods of field data collection. Ecological relationships between mapped potential vegetation and existing vegetation need to be investigated. Further research must be conducted to quantify the relative contributions of the different approaches and data sets used in the prototype. Performance consistency must be tested between adjacent western mapping zones, as well as in one or more prototype areas located in the eastern United States. Repeatability of the methods used in the prototype, both temporally and spatially, must also be evaluated. Furthermore, it is not clear whether the LANDFIRE Prototype methodology will suffice for other vegetation metrics, such as quantifying woody or non-woody biomass; a study in this area could yield information leading to enhanced applications of LANDFIRE vegetation maps. Nevertheless, the LANDFIRE Prototype Project provides sufficient information on which to base several recommendations regarding the national implementation of LANDFIRE.

Ways to Ensure Consistent National Vegetation Mapping

As noted above, several tasks related to existing vegetation mapping for the prototype effort may be standardized and potentially automated to facilitate LANDFIRE's national implementation. These tasks include: 1) the creation of a national vegetation map unit classification system that is mappable using spectral and biophysical/ecological data and is supported with adequate field-referenced data; 2) the consistent acquisition and processing of a multi-seasonal Landsat database; 3) the application of QA/QC procedures to the LFRDB to ensure a robust field-referenced database that can be used for a wide variety of applications; 4) the consistent modeling of biophysical data layers and probabilities of existing vegetation species or types associated with potential vegetation types; and 5) the continued application of CART as the primary mapping algorithms to ensure objectivity and flexibility when using high volumes of field data and predictor variables. We discuss these points in detail below.

Need for a Mappable Vegetation Map Unit Classification System

The vegetation map unit classification system used for the national implementation of LANDFIRE must meet a number of key criteria including the following: 1) the system must be nationally consistent, ecologically logical and hierarchical, acceptable to a wide array of users and groups, and must meet existing Federal Geographic Data Committee (FGDC) standards; 2) vegetation map units must be mappable using operational methodology to achieve reasonable accuracies; and 3) the map unit classification system must include vegetation map units that have high relevance with respect to the core LANDFIRE products. The Ecological Systems classification (Comer and others 2003) developed by NatureServe meets these objectives. This system represents the hierarchical merging of NVCS alliances into a nationally available suite of vegetation map units. Unlike alliances, which have proved exceedingly difficult to map accurately, most Ecological Systems classes are mappable, assuming an adequate number of field plots exist for training purposes. In addition, the Ecological Systems classification was developed by plant ecologists, lending credibility to the approach and resulting in a greater level of acceptance throughout the user community. We anticipate that a few additional "target alliance" map units will be added to the LANDFIRE National map unit classification legend on a case-by-case basis. These will be added only when it is determined that a particular map unit not specifically identified by the Ecological Systems classification has special relevance to LANDFIRE.

Need for National Field-referenced Data Collection and Processing

Many LANDFIRE tasks rely on a comprehensive, consistent, and extensive field-referenced database. The database serves as a reference for the development, testing, and accuracy assessment of all LANDFIRE vegetation, biophysical settings, and wildland fuel data layers and of all vegetation and fire regime simulation models. Field data from existing projects should be incorporated into this database whenever available and should include but not be limited to data sets such as FIREMON fire monitoring databases, USFS Landscape Ecosystem Inventory Systems databases, and the National Park Service fire monitoring databases. In addition, the USFS FIA Program's forest inventory plot database proved a useful source for the majority of forest data. Where data are lacking, supplemental field data collection is required to fill informational needs on rangeland map units. This assortment of field-referenced data should be collectively scrutinized for quality assurance, regularly updated, and maintained as a comprehensive LANDFIRE field-referenced database.

Need for Nationally Consistent Imagery Database

The availability of a quality Landsat imagery catalog is a key prerequisite for national implementation of the approaches developed for the LANDFIRE Prototype Project. Among all predictor variables, it is satellite imagery that usually captures the most current vegetation conditions, and, when used repeatedly over time, identifies changes in vegetation conditions and distributions. Thus, we recommend that LANDFIRE National continue to play an active role in the MRLC Consortium. This membership ensures the continued development of suitable multi-seasonal Landsat image catalogs, optimal levels of image processing (geometric, radiometric, and atmospheric rectification and calibration) for the rest of the country, and mapping zone-based image compilation for national vegetation mapping. In addition, LANDFIRE National should support studies that examine and compare the characteristics of other mid-resolution sensors with those of Landsat. Even though the LANDFIRE Project does not currently require any additional Landsat imagery, the potential benefits of using different satellite data for future updating should be considered.

Need for Nationally Consistent Set of Biophysical Gradient Layers

Biophysical gradients have effects similar to that of Landsat imagery on the spatial and information integrity of existing vegetation maps. Many of the biophysical layers are physiologically and ecologically related to the establishment, distribution, and conditions of plant species, and the incorporation of these gradient layers into the mapping process contributes to increased accuracies. For the national implementation of LANDFIRE, we recommend that a set of biophysical gradient layers similar to those listed in table 1 be used to map vegetation in all mapping zones. In addition, we recommend that further research be conducted to quantify the contribution of the individual biophysical variables to mapping accuracy. Furthermore, research should be conducted to minimize residual coarse-resolution imprints in 30-m biophysical data resulting from the coarser resolution weather and soil databases used to produce these data. The development of standard minimum mapping units in modeling simulations has shown promise in standardizing the process and eliminating coarse imprints.

The Need to Continue with Research and Improvements

Although results of the LANDFIRE Prototype Project indicate that the general approach should effectively meet target accuracy and consistency requirements for national implementation, there are areas where continued research and improvements are needed. One ongoing research effort involves the development of a new and more consistent approach to mapping shrub and herbaceous canopy cover. Current research is testing ways to effectively correlate calibrated Landsat-based NDVI to shrub and herbaceous canopy cover (Liu and others 2004). Other research areas include more efficient use of the individual biophysical gradient layers, more effective mapping of riparian vegetation, and a national accuracy assessment strategy.

Conclusion

The mapping of existing vegetation with complete national coverage at a 30-m spatial resolution is a core requirement of the LANDFIRE Project. National data at this 30-m resolution do not currently exist. As a result, the prototype research was needed to answer questions related to the mapping and characterizing of cover types and structure variables. LANDFIRE's existing vegetation products are expected to provide data not only for use in wildland fire management, but also for use in many other natural resource and environmental applications. Findings from the LANDFIRE Prototype effort are summarized as follows:

If supported with an adequate amount of field-referenced data, target accuracies of 60 percent or better are achievable for a mid-level vegetation map unit classification at the regional scale. The addition or subtraction of floristically or ecologically similar cover types has significant effects on resulting accuracies. Of the three major life forms, herbaceous cover types are the most difficult to map because these species adapt to many general biophysical characteristics and have few unique spectral signatures. Relationships between the floristic complexity of the vegetation map unit classification and mapping accuracies indicate that the national vegetation map unit classification will need to be designed carefully to include adequate flexibility.

For LANDIFRE, vegetation structure is defined by canopy cover and canopy height of forest, shrub, and herbaceous life forms. These structure attributes can be mapped consistently as categorical variables. Mapping these attributes as continuous variables, particularly for shrub and herbaceous height and cover, is inconsistent and, thus, is not recommended for national implementation of the LANDFIRE prototype methods.

Field data collection and processing are the most critical factors in ensuring that LANDFIRE maps of existing vegetation are objective and accurate. The detection and correction of errors existing in field-referenced data are time-consuming but absolutely necessary tasks, particularly for field data from sources other than FIA (as these other data sets tend to be locally limited and have various sampling designs). The objective of repeated field data processing and quality control is to derive a refined, high-quality field data set.

The incorporation of LANDFIRE biophysical gradient layers and cover-type probabilities associated with potential vegetation types into the mapping models contributes to a significant increase in mapping accuracy. In addition, the use of the biophysical and ecological stratifications that describe the environmental effects on species establishment and growth also contributes to enhanced mapping consistency.

For further project information, please visit the LANDFIRE website at www.landfire.gov.

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We Want a Healthy Forest
Create jobs
Healthy Water Sheds

Utilize Material as soon as possible so it doesn't Rot & go to waste

Have a Bench Mark so all contractors have to do quality work.

Leroy Cordova

D Elk herds - what's is the balance (Ted)
between FS and State

- FS doesn't manage animals, we manage habitat for success

- State manages population

- VCP - have we talked or (Chris L)
worked w/ them. They run the large herd. In the spring, they come to VCP and eat forage for allotments

- Fencing is an issue, but VCP puts in on permittees.

- Referenced incident 2 years ago w/ 300+ head of cattle

More logging to clean up the forest, to help with fire more jobs for the community. & grazing for animal, too much forest canopy that doesn't allow sun to reach ~~forest~~ floor, ~~for~~ healthy understory - trees are too dense ~~ity~~ & make all trees unhealthy

Roads need improvement.

Manuel Garcia

We need more logging

We opportunity to get out

Material before burning for

fuelwood. We can clean up

area before burning. We can

help out with slash ~~buyth~~ piling.

Walon Garcia

- More logging & forest treatment for health forest.

From Earl Voss

THANKS – VERY HELPFUL AT
DISPELLING RUMORS, ADDING GOOD
INFORMATION.. VERY REASSURING

Moises - Concerned about the Rx burning and visual quality. also killing remaining trees after thinning through burning

i.e., look @ Cebolla & Cangilon

Doesn't want to see that in Canones w/s.

+ Losing latillas and other wood products through Rx

More development
of recreation to gain
more resourses through tourism
& economic advantages for the
Community

①
- Why are some areas being rx (Mr. Smith)
treated prior to Rx and some
are just Rx.

- 5-10 miles temp Roads.
  - Have they been determined when located.
    Not yet.

- Goats peaked Mr. Smith's Interest
  - helping maintain as another ~~rather than~~ tool beyond burning.

2 - Goats help w/ gambel oak maintenance

- Goats are browsers, they go for nm locust, oak, etc - all that comes back after Rx